Understanding the development and characteristics of conservation area networks

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Declaration

This thesis is the result of my own work and does not include anything which is the outcome of work done by or in collaboration with others.

Rachel Sykes, February 2020

Abstract

Protected areas are an essential component of efforts to halt biodiversity loss and they are widely used to protect species and habitats, and maintain essential ecosystem services that underpin human society and wellbeing. This is reflected by the Convention on Biological Diversity's Aichi Target 11, which commits signatory nations by 2020 to protect at least 17% of the terrestrial realm through various state, private and community conservation designations, which are placed in areas important for biodiversity, representative, well-managed and integrated into the wider landscape. This target will be revisited at the 2020 Conference of the Parties, where it is expected that the target may be raised to perhaps as much as 30%. This is an ambitious commitment, and while there is a substantial body of research on protected areas, gaps remain in our understanding of how to rapidly build a global protected area network that covers a significant proportion of the Earth's surface and is effective in maintaining its conservation value and supporting neighbouring people.

In this thesis I aim to address some of these gaps. Chapter 2 addresses the problem of expanding a protected area network in the context of a densely populated and highly transformed country, in which remaining habitats exist only in small, scattered fragments. I examine the trade-offs involved, between the area of land necessary to meet representation targets and minimum protected area size thresholds, and the opportunity costs that may be incurred due to lost agricultural land. Chapter 3 examines the characteristics of a conservation area network comprising state-owned and managed protected areas, and other conservation areas owned and managed by private individuals and communities. I study the different contributions that conservation areas of differing governance types could make to the overall extent and representativeness of a network. Chapter 4 presents a conceptual framework in which I explore what drives the establishment of conservation areas across the globe. I highlight many frequently overlooked socio-economic and political factors that help explain why conservation area network

extent differs so greatly between countries, and describe what conditions may be necessary to create an enabling environment for the growth of networks in the future. Chapter 5 presents a new methodology developed to improve the accuracy of estimates of global conservation area coverage. I produce a sample of the terrestrial realm that is representative of 10 key biogeographical and socio-economic factors that can be used as the focus for data collection efforts.

Table of contents

List of tables

List of figures

List of supplementary tables

List of supplementary figures

Chapter 1. Introduction

Over recent decades it has become clear that biodiversity is in decline across the world, with species extinction rates up to 1,000 times greater than the natural background rate (Pimm et al. 2014; Butchart et al. 2010). This decline is accompanied by degradation and fragmentation of habitats, deteriorating ecosystem functioning and rising global temperatures. Protected areas – defined by the IUCN as *"a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values"* (Dudley 2008) – have been proposed as a tool to tackle all these issues (Woodley et al. 2012; MacKinnon et al. 2011; Ricketts et al. 2010; Millennium Ecosystem Assessment 2005; Rodrigues et al. 2004). By preventing harmful activities, such as clearing for intensive agriculture or unsustainable resource extraction, species populations are allowed 'breathing space' to recover or increase, while habitats are preserved and ecosystem services maintained.

Humanity has a long history of protecting particular areas that are considered special against disturbance and development. Some are sites of spiritual or religious significance, such as sacred forest groves in India managed by local people who believe deities reside there (Ramakrishnan et al. 1998). There are believed to be between 100,000 and 150,000 such sacred groves in India alone (Ormsby & Bhagwat 2010). Some were prime hunting grounds reserved for the sole use of royalty or nobility, like the New Forest in southern England which was designated as a Royal Forest by King William I in 1079 (Newton 2011). Others were concerned with sustainability and the preservation of valuable resources, such as traditional land management regimes in the Middle East that set aside rangelands to prevent overgrazing (Chape et al. 2008).

However, the establishment of protected areas (PAs) whose key aim is the preservation of nature both for its own sake and for the benefit of the general public is largely a recent phenomenon. Yellowstone National Park in the United States, created in 1872, is generally regarded as the first modern PA (Watson et al. 2014). Over the course of the 20th century, PAs grew steadily in number and extent, and into the early decades of the 21st century saw a dramatic increase (Juffe-Bignoli et al. 2014). Between 1990 and 2015, PA coverage increased by 92% for terrestrial and 513% for marine environments (Butchart et al. 2015). There are over 1,000 different PA designation types (Chape et al. 2008) and it is estimated that PAs now cover approximately 15% of the terrestrial realm and 8% of the marine realm (IUCN 2019), making them one of the most important land use allocations in the world (Chape et al. 2005).

1.1 The effectiveness of protected areas

1.1.1 Species

Alongside this rapid and substantial increase in PAs, biodiversity continues to decline in variety and abundance. WWF's Living Planet Report in 2014 found that the total number of wild animals, on land and in the sea, more than halved between 1970 and 2010. Populations of freshwater species fell by as much as 76%, while Latin America showed the greatest overall drop of 83% (WWF 2014). Although many species that have suffered decline are not yet at imminent risk of extinction, their greatly reduced population is still cause for concern. Even relatively small declines in common species can have significant ecological consequences (Gaston & Fuller 2008), while declines of this magnitude may wreak even greater ones (Mace et al. 2014).

The shrinking populations observed by WWF did not occur only on unprotected land. There is an assumption that simply protecting an area in law or making damaging activities illegal will necessarily produce a positive outcome, yet Craigie et al. (2010) found an average 59% decline in population abundance of large mammals in 78 PAs across Africa since 1970. Western et al. (2009) and Ottichilo et al. (2000) found similarly sharp declines in their studies of Kenyan

PAs. There is evidence that PAs have failed to prevent the population decline or extinction of primates in Indonesia (Meijard & Nijman 2000), birds in Spain (Sergio et al. 2005; Suárez et al. 1993), butterflies and amphibians in the USA (Schlicht et al. 2009; Fellers & Drost 1993) and migratory ungulates in South Africa (Tambling & du Toit 2005). These declines were due to, among other factors, over-hunting, poaching, persecution, invasive species and natural disasters, indicating that legal protection was unable to prevent or mitigate the effects of human activities and natural stochastic factors.

On the other hand, PAs appear to have helped preserve populations of large herbivores in Tanzania (Stoner et al. 2007), raptors in Botswana (Herremans & Herremans-Tonnoeyr 2000), tigers and deer in Nepal (Wegge et al. 2009), 45 bird species in France (Devictor et al. 2007), cranes in China (Ma et al. 2009), banteng in Vietnam (Pedrono et al. 2009) and a range of tropical rainforest species across 11 African countries (Struhsaker et al. 2005). They have been shown to be more effective than other conservation measures in both terrestrial (Taylor et al. 2011) and marine environments (Sciberras et al. 2013). Thus, the evidence that PAs are an effective strategy in preventing the decline or extinction of vulnerable species populations is somewhat mixed.

1.1.2 Habitats

The evidence to suggest PAs are effective at reducing habitat loss, which is a concern for both wildlife conservation and climate change mitigation (Nelson & Chomitz 2011), appears a little more convincing. Studies looking into the effects of protection on habitats are more numerous, as changes in land cover (particularly deforestation) are generally quicker and easier to detect than the population trends of wildlife. In their review of studies investigating habitat change in PAs, Geldmann et al. (2013) found that 82% of the 76 studies used indicated that habitat loss was greater outside PAs than inside, while 12% found the reverse. The remaining 6% could not discern any effect. Despite local failures, the overall global picture is good: in a study of nearly 200 PAs, DeFries et al.

(2005) found that rates of habitat loss were 2.6 times lower inside than outside; Joppa and Pfaff (2010) found that in 75% of 147 countries analysed, protection reduced conversion of natural land cover; Bruner et al. (2001) studied 93 PAs in 22 tropical countries, finding that the majority were effective at stopping land clearing while also – to a lesser extent – effective at mitigating logging, hunting, fire and grazing; and Scharlemann et al. (2010) found that globally, unprotected forests lose twice as much carbon into the atmosphere as PAs. Further studies also support these findings (Françoso et al. 2015; Barber et al. 2014; Pfaff et al. 2014).

However, it is worth exercising caution in the interpretation of these results. Andam et al. (2008) point out that many such studies may be subject to various biases that are skewing the results in favour of the conclusion that PAs prevent habitat loss. They argue that researchers typically fail to account for biases in deforestation and protection, and consequently overestimate reduction in deforestation in PAs by as much as 65%. For example, researchers often use unprotected lands as their control to compare against the trends observed in PAs. But it is well-known that protection is not applied randomly across a landscape; rather, the areas that are chosen for protection are more likely to be in more inaccessible and less economically valuable land (Butchart et al. 2012; Millennium Ecosystem Assessment 2005; Pressey 1995). It is often 'the land that nobody wants' so governments and landowners lose little or nothing by designating these areas as PAs. As such, by virtue of their inherent characteristics, these areas are already at lower risk of deforestation or other habitat fragmentation and degradation than other areas nearby might be (Joppa & Pfaff 2009).

Even when PAs and their adjacent unprotected lands are similar enough for apparently accurate comparison, it has also been found that the establishment of a PA can merely displace damaging human activities into surrounding areas, rather than stopping them altogether. This might include displacement of development, hunting or agricultural pressures, exploitation to meet demand from tourism or even pre-emptive clearing by local landowners to prevent further regulation (Andam et al. 2008). Thus, habitat loss decreases within the PA but increases outside it, giving the impression that the PA has been effective; but while protection may have reduced habitat loss within the area of the PA itself, it has not reduced habitat loss overall. Instead, it has merely 'spilled over' into neighbouring areas, which may have been equally important for wildlife (Armsworth et al. 2006). However, there can also be positive spillover effects of PA establishment such as improved productivity, law enforcement and additional protection in neighbouring areas. For example, Roberts et al. (2001) show that a small PA network of five reserves in the marine territories of the Caribbean island of St Lucia increased the catches of adjacent fisheries by up to 90%, despite the 35% decrease in fishing area. Determining the extent of spillover effects is difficult, but in their study of Costa Rican PAs between 1960 and 1997, Andam et al. (2008) estimate that, overall, spillover effects were negligible and when they did occur they were usually positive.

1.1.3 Ecosystem services

In addition to their benefits for biodiversity and habitats, it is likely that PAs will become ever more vital to the wellbeing of humanity too. In recent years, scientists and economists have been working to understand and quantify the benefits of the multitude of 'services' that nature provides to us for free, upon which we all depend. These ecosystem services include, for example, climate regulation, pollination of vital food crops, provision of potable water, disease control, waste treatment, flood mitigation, opportunities for recreation and so on (Braat & de Groot 2012). These services are tricky to value with complete accuracy (Daily et al. 2000), but Costanza et al. (2014) suggest that in 2011 the value of the biosphere to the global economy through these ecosystem services was between US$125-145 trillion/year. By comparison, global GDP in 2011 was around US$73 trillion/year (The World Bank 2016). This was an update of their earlier estimate of US$16-54 trillion/year compared to a global GDP of US$18 trillion/year (Costanza et al. 1997). Despite these benefits, natural systems continue to be degraded at an alarming rate, threatening their capacity to provide for us (Boisvert & Vivien 2012; Cardinale et al. 2012; Alcamo et al. 2005).

PAs contribute significantly to the provision of ecosystem services: a third of the world's largest cities rely on PAs for their drinking water (Dudley & Stolton 2003), PAs reduce the effects of natural disasters such as floods and droughts, and maintain supplies of fish and other wild foods (MacKinnon et al. 2011), protected forests and wetlands sequester huge quantities of carbon, helping to mitigate climate change (MacKinnon et al. 2011; Scharlemann et al. 2010; Soares-Filho et al. 2010) and PAs are estimated to receive 8 billion recreational visits by the public each year (Balmford et al. 2015). Research into the specific contributions of PAs to different ecosystem services is still in its early days. However, we know that the cost of losing natural areas and the biodiversity they support is substantial, and that many PAs could easily pay for themselves if the revenue they generate were reinvested. Bruner et al. (2004) estimate that the cost of establishing and managing an expanded global PA system would cost US$4 billion per year over the next decade. This figure is not insubstantial but it is well within the means of the international community (and dwarfed by, for example, global spending on the military). In comparison, Balmford et al. (2002) estimated that the habitat conversion, from a natural to artificial state, which occurs globally in a single year has a net cost to humanity of US$250 billion that year, and every year thereafter. Meanwhile, they calculated that an effective global conservation program to protect our remaining wild nature would have a benefit to cost ratio of at least 100:1. The goods and services provided by an adequate and effective PA network would have an annual value of between US$4,400 billion and US$5,200 billion, depending on the level of resource use permitted within PAs.

Thus, the costs of establishing and managing PAs is, it seems, vastly outweighed by the economic benefits provided by healthy natural systems. This is true at national and regional scales, as well as global. For example, in 2009 the Canadian government spent Can$800 million on their national network of PAs, which generated Can$4.6 billion for the Canadian economy and supported 64,000 full time jobs, many in rural and remote areas (CPAWS 2012). In Australia, the budget for the Great Barrier Reef Marine Park Authority was Aus$50 million in 2012-13, while tourism to the reef contributes over Aus$5.2 billion each year to the Australian economy (GBRMPA 2014). Ongoing degradation to the reef may jeopardise the future of this income (Watson et al. 2014). Green et al. (2012) report that in the biodiversity hotspot of the Eastern Arc Mountains, Tanzania, the reinvestment of just 13% of the revenue generated by tourism to Tanzania's PAs would cover the management costs of effective conservation across the region. Further research on the contribution of PAs to the provision of ecosystem services and the revenue they generate is crucial to help persuade national governments to take PAs seriously and invest in their long-term sustainability.

1.2 The future of protected areas

Recognising the essential contribution of PAs to conservation efforts and human wellbeing, the push for further expansion of the global network continues under the auspices of the Convention on Biological Diversity (CBD). The CBD, signed and ratified by almost every nation on Earth, came into force in 1993 with the aim of promoting the conservation and sustainable use of the natural environment and its species (Chandra & Idrisova 2011). In 2010, the tenth meeting of the Conference of the Parties established the 20 Aichi Biodiversity Targets to be achieved by 2020, which are grouped under five strategic goals:

1. Mainstreaming of biodiversity across government and society
2. Reducing pressures on biodiversity and promoting sustainable use
3. Safeguarding ecosystem, species and genetic diversity
4. Enhancing benefits to people from biodiversity and ecosystem services
5. Enhancing implementation through participatory planning, knowledge management and capacity building

These targets set out aspirations for each nation and a framework within which to work towards them, with the ultimate aim of significantly reducing biodiversity loss. Aichi Target 11, under the goal of safeguarding diversity, concerns the establishment of PAs:

"By 2020, at least 17 per cent of terrestrial and inland water areas, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes" (CBD 2010).

The objective of protecting at least 17% of the global terrestrial area has led to the majority of Parties adopting a national target of 17% also. However, the Aichi targets are global in focus and nations are permitted to set their own targets, allowing them to take into account national needs and priorities as well as considering their contribution to global goals (CBD 2010). One third of Parties have therefore adjusted their own national target higher or lower than 17% (ranging between 3% and 50%) according to what they consider appropriate (Butchart et al. 2015). In 2015, 40% of countries had met the 17% terrestrial target, while just 13% had met the 10% marine target (Butchart et al. 2015). The latest data suggest more countries have met their targets but overall we still collectively falling short (IUCN 2019).

1.2.1 Communities

The wording of Target 11 is significant, reflecting an appreciation of the failures of PAs to date both ecologically and socially. Although the target is fundamentally one of increasing total area under protection, it recognises the importance of targeting these areas effectively and managing them with concern for biodiversity outcomes and social justice. In some cases the latter may be as vital to conservation success as scientific research (Pretty & Smith 2004). PAs and the rural communities living in and around them have a troubled history, but the expectations and demands placed on PAs have undergone dramatic change in recent decades. Although, as the IUCN's definition clearly states, the primary purpose of PAs is still to ensure the long-term persistence of nature, there is now also a strong focus on the importance of nature for humans.

It has been recognised that in the past the needs and lives of local and indigenous peoples were often at best disregarded, at worst actively dismantled and destroyed by European colonists who established national parks across the world. These parks were often for the benefit of the wealthy, while local people were forcibly removed from their homes and cut off from the resources they relied on. Similar attempts at exclusionary, 'fortress'-style conservation continue to the present day (Adams & Sutton 2007). But there is a growing understanding that the rights of those who co-exist with biodiversity cannot be ignored and, indeed, that conservation may be more successful and sustainable when local communities are involved (Ostrom & Nagendra 2006; Brockington & Schmidt-Soltau 2004). PAs are often now expected to provide social and economic benefits to support local people, principally through sustainable use of natural resources, opportunities for employment and income generated by tourists. Contrary to the beliefs of some, PAs are not 'poverty traps' but can actually contribute to poverty alleviation (Ferraro et al. 2011; Wittemyer et al. 2008). Thus they are no longer seen as isolated 'islands' of wilderness, but are integrated into the wider landscapes and their human communities (Chape et al. 2008).

Successfully reconciling the goals of biodiversity conservation with social and economic justice is a difficult task, and progress towards a harmonious relationship between PAs and their human inhabitants is slow (DeFries et al. 2007). Yet conservationists cannot shy away from tackling social and economic issues, because the deprivation and eviction of local people from PAs is not only unjust, but can harm the chances of conservation success. Regulations in PAs are sometimes deliberately ignored or even sabotaged in acts of resistance by those whose livelihoods have been adversely affected by their imposition (Di Ciommo & Schiavetti 2012; Brechin et al. 2002). These acts of resistance can include overexploitation of natural resources, setting fire to forests and retaliatory killing of wildlife (Watts & Faasen 2009; Hamilton et al. 2000). However, most often they involve simply continuing everyday livelihood activities – because there is no alternative, and because locals consider the prohibition of such activities illegitimate (Holmes 2007). Further studies suggest mitigating conflict with humans can be more important to conservation success than ecological or demographic factors (Woodroffe & Ginsberg 1998).

Andrade and Rhodes (2012) conducted a meta-analysis of 55 case studies from developing countries in an attempt to identify what factors are related to compliance with PA regulations. They found that the only variable that was significantly related to compliance was local community participation. Indeed, the higher the level of participation, the higher the compliance. The aim of increasing local participation is to give people a collective sense of ownership of and responsibility for the natural resources in their PA. Having their knowledge, opinions and needs taken into account in decision-making makes their commitment to long-term conservation considerably more likely (Pretty & Smith 2004). The importance of local participation and social capital to successful PA and natural resource management is emphasised by researchers working across the world in both terrestrial and marine environments (Oldekop et al. 2015; Gutiérrez et al. 2011; McClanahan et al. 2006; Wells & McShane 2004; Pollnac et al. 2001).

However, permitting resource use as part of PA management strategies does not necessarily imply compromising on biodiversity outcomes; rather, different management strategies ranging from multi-use to strict protection are more appropriate and therefore more effective in different contexts (Pfaff et al. 2014; Porter-Bolland et al. 2011). Multi-use management may be more politically feasible or socially acceptable than strict protection in accessible areas with higher human population density, and therefore have a greater likelihood of success due to higher rates of compliance. In a study of the effectiveness of strict and multi-use PAs in preventing tropical forest fires (with associated deforestation and carbon release) in Latin America, Asia and Africa, Nelson and Chomitz (2011) found that all PAs substantially reduce incidence of fire, even after controlling for terrain, climate and remoteness. However, while strict PAs did reduce incidence of fire, multi-use PAs performed even better, particularly in less remote areas with high agricultural conversion and timber extraction pressures.

Other studies (Porter-Bolland et al. 2011; Hayes 2006) similarly find that forests that are managed and utilised by local communities can be just as effective, or even more so, at reducing deforestation than PAs. On the other hand, some researchers have found that PAs with stricter protection perform consistently better in terms of sustaining or increasing wildlife populations than areas with lower levels of protection (in Tanzania; Stoner et al. 2007; Caro 1999), and that conservation success can still be achieved in the absence of local participation, education or compensation (Struhsaker et al. 2005; Brockington 2004; although the former did find that strong public support was beneficial). Therefore consideration of the unique ecological, political and socio-economic context of each PA is required when producing a management strategy to allow the best chance of success.

1.2.2 Management

Achieving compliance through positive community relations and sound planning obviously requires that PA managers have a management strategy at all. Yet, although the World Database on Protected Areas (WDPA) now contains nearly 250,000 entries (IUCN/UNEP-WCMC 2020), many of these PAs are actually just ‘paper parks’. That is, although they exist as PAs in law, on the ground there is little or no management, such that they may be no more effective at conserving biodiversity than unprotected areas (Visconti et al. 2019; Bertzky et al. 2012; Hockings et al. 2004). Leverington et al. (2010) and Bruner et al. (2001) demonstrate that PA effectiveness is correlated with basic management activities such as clear boundary demarcation, enforcement of regulations and compensation to nearby communities. Where these are lacking, often due to insufficient funding, PAs may fail to prevent biodiversity and habitat loss. Various case studies support this finding, particularly with regard to enforcement (Hilborn 2006; Carrillo et al. 2000).

As well as paper parks, many more PAs will have management regimes in place, but there is little or no monitoring and evaluation so it is not known whether the management undertaken is effective, ineffective, or even harmful. As a result, in recent years there has been a push for more PA management effectiveness (PAME) assessments to be carried out, with some success (Coad et al. 2013). Leverington et al. (2010) analysed the results of more than 8,000 PAME assessments covering 6,200 PAs across the world, and found that 42% had major deficiencies. The assessments analysed showed that many PAs scored poorly on planning and adaptive management, law enforcement, monitoring, communication and community relations, and funding indicators. Yet many also scored relatively well on PA establishment indicators (publication in official government documents, design, boundary marking, tenure resolution and adequacy of legislation) and on conservation outcomes, indicating that PAs were generally preserving their biodiversity values despite serious deficiencies in funding and management.

1.2.3 PADDD

A further threat to the effectiveness of PAs in tackling biodiversity loss is the phenomenon of PA downgrading, downsizing and degazettement (PADDD). It is often assumed that, once established, PAs are essentially permanent, but Mascia and Pailler (2011) report that there have been 89 instances of PADDD over the past century which occurred in at least 36 countries (they suggest their findings are likely to be a considerable underestimate). Zimmerer et al. (2004) examined trends in PA systems globally between 1980 and 2000 and found a scaling back of PA networks in some countries and a shift in many others towards 'utilised environments' rather than strict protection in both new and existing PAs. The latter may not in itself be a cause for concern, given that we know such areas can still be effective for conservation while also accommodating human needs. However, downsizing and degazettement events are concerning, as well as downgrading for purposes other than sustainable use. Mascia and Pailler (2011) detail 16 recent instances of PADDD, plus a further 16 proposed instances, occurring across the world on every continent other than Antarctica, including highly biodiverse countries such as Madagascar, Indonesia, India and South Africa. Mascia et al. (2014) found that, in the absence of PADDD, four additional countries would have met their CBD targets for PA coverage. The majority of these PADDD events are due to demand for natural resources (e.g. oil and minerals) or land for industry, development or cultivation of lucrative crops (e.g. oil palm and rubber) on a large scale (Golden Kroner et al. 2019). Only a small proportion of PADDD events appear to be undertaken to improve conservation regimes (Mascia et al. 2014).

1.2.4 Planning

Finally, one of the most important issues to resolve as governments and conservationists expand the world's PA network in the coming years is where these new PAs should be located. As previously mentioned, the biases in PA location are well known (Joppa & Pfaff 2009), with many PAs placed where they may be least able to prevent land conversion. Some argue that this bias may not always be problematic: PAs in areas that have fewer pressures are more likely to be successful, and will be so with only minimal financial input which, given the often woefully inadequate funding available to conservation bodies, is an important consideration. Alternatively, siting PAs in areas with higher pressures might result in much higher avoided clearance, thus representing better value for money (Joppa & Pfaff 2010). However, given the bias towards certain land characteristics (e.g. mountainous, inhospitable), PAs are therefore also biased towards certain habitat types and species, leaving others that occupy land more valuable to humans significantly under-protected (Margules & Pressey 2000). Consequently, amount of area protected is only loosely correlated with amount of biodiversity protected in reality (Venter et al. 2014; Millenium Ecosystem Assessment 2005).

Thus, in addition to considering the likelihood and impacts of success when siting new PAs, it is vitally important also to ensure that the network is representative of biodiversity. The current global PA network, as it stands, is far from achieving perfect representation. In a global gap analysis of PAs, Venter et al. (2014) found that 17% of over 4,000 threatened vertebrates analysed are not found in a single PA. Overall, 85% are inadequately covered by PAs. They found that if new PAs covering up to 17% of global land area were targeted at the cheapest land, the number of threatened vertebrates covered would increase by just 6%, even if the areas are ecoregionally representative. This suggests that targeting PAs towards areas that are cheapest or easiest to protect will not achieve the persistence of biodiversity. However the authors found that, if threatened vertebrates are incorporated into decision-making as well as cost-efficiency, five times more threatened vertebrates could be adequately covered for only 1.5 times the cost of the cheapest solution. Similarly, McCreless et al. (2013) warn against prioritising countries for conservation action on the basis of cheapness alone, because low costs are often correlated with other factors that can reduce the effectiveness of PAs, such as political instability, poor civil engagement and corruption. However, their study was conducted at national level, and so does not reveal whether cheaper areas within a country are disproportionately affected by these issues.

Navigating the complex trade-offs between the costs and benefits of protecting different areas is the task of conservation planners, who increasingly employ the techniques of systematic conservation planning (SCP) (Margules & Pressey 2000). SCP is a method for prioritising sites for conservation in a systematic way, using defined and objective criteria, with a view to maximising the effectiveness of a PA network while minimising associated costs. This method can be used to design whole new networks, add to an existing network, or evaluate the effectiveness of an existing one. The aim is to design and produce PA networks that, at regional, national and global scales, adhere to the following basic principles (Kukkala and Moilanen 2013):

- Representation: networks should encompass a representative sample of the diversity of species.
- Efficiency: networks should achieve specified conservation goals and do so with minimum cost. Cost may refer to, e.g., cost of protection (such as land purchase) or cost to people (such as the opportunity costs of forgone profits from agriculture or development). Achieving efficiency means new reserves should complement existing ones, and possibly degazetting existing PAs if they do not add value to the network.
- Adequacy: PAs should be sufficient (in terms of size, connectivity, habitat types or resources covered) to ensure the long-term persistence of species populations within them. This includes a level of redundancy, whereby networks include multiple species populations in different PAs, to guard against loss in the event of local extinctions due to, for example, natural

disasters. This requirement must be balanced against the need for efficiency, according to the vulnerability of the species.

Margules and Pressey (2000) lay out six key stages through which this aim can be achieved (other authors have suggested differing approaches, Levin et al. 2013; Groves et al. 2002):

1. Compile data on the biodiversity of the planning region: this involves identifying suitable biodiversity surrogates (due to the impossibility of having fully comprehensive data on every species) and gathering data on the most rare and endangered species.
2. Identify conservation goals for the planning region: these are specific quantitative targets that the desired PA network would meet, for example, coverage of 20% of a species' range, or 1,000ha of a vegetation type.
3. Review existing PAs: some targets may already be met by existing PAs, while new PAs need to be targeted to achieve others that fall short.
4. Select additional conservation areas: using GIS and SCP software such as Marxan (Ball et al. 2009), new PAs can be identified that complement the existing network and achieve conservation goals. The set of areas identified is only preliminary, and should be used in consultation with stakeholders to decide on the new PAs, subject to real-world constraints.
5. Implement conservation actions: decide on the most appropriate form of management for the new areas and put the management plan in place.
6. Maintain the required biodiversity values: ensure the achievement of defined conservation goals (such as the persistence of a population at a given level) through monitoring and adaptive management.

The increasing use of SCP in conservation planning will hopefully begin to remedy some of the deficiencies in current networks caused by the 20th century's largely *ad hoc* approach to PA establishment and management.

1.2.5 Targets

As conservationists try to persuade the international community to agree a more ambitious target post-2020, it is worth also pausing to consider what we are actually aiming for. The Aichi targets are merely interim, and say nothing about what amount of protected area will be sufficient to achieve global conservation goals (Woodley et al. 2019). All CBD targets are negotiated and agreed upon by the Parties, and therefore the outcomes of these negotiations will tend to be compromises, dictated by perceived feasibility and political expediency (Rodrigues & Gaston 2001). It is widely agreed in the scientific community that the current targets are essentially arbitrary, have no basis in biology and will be wholly inadequate to prevent significant biodiversity loss (Svancara et al. 2005). It is expected that at 10-17% protection, up to half of species may still be in danger of extinction in the near future (Soulé & Sanjayan 1998).

Thus some criticise the idea of having targets at all, concerned that they may become 'ceilings of protection' whereby governments interpret the recommended percentages as end goals and fail to appreciate (deliberately or not) that long-term conservation of biodiversity will require much more (Soulé & Sanjayan 1998). Rodrigues and Gaston (2001) also argue that a single universal target, however high, will always be inappropriate because it does not take into account the patterns of diversity, abundance and endemism across the world. Tropical rainforests, for example, support many more species than Arctic tundra – including many specialists that are confined to narrow ecological niches – and so more of their area will require protection to safeguard the future of these species. Consequently, those countries with the greatest shortfall in required protection may not necessarily be those falling short of official targets, rather they are simply those with the most biodiversity. Elsewhere, larger areas need to be protected to ensure coverage of migration routes (Berger 2004), wide-ranging species (Svancara et al. 2005) and, increasingly in the 21st century, habitat corridors to allow movement of species affected by climate change (Noss et al. 2012).

Nonetheless, it is useful to have objective, quantifiable targets against which we can measure progress. They must be scientifically defensible though, if they are to carry any real weight, but calculating such targets is extremely difficult. Many researchers have made attempts over the years using various methods (Rondinini & Chiozza, 2010) and suggested a range of possible figures. Fjeldså and Rahbek (1999) report that representing every known plant species at least once in the global PA network would require protection of 74.3% of the global terrestrial area and 92.7% of tropical rainforests, given their exceptionally high rates of endemism. Soulé and Sanjayan (1998) report regional estimates ranging between 33% and 75%, while a meta-analysis by Svancara et al. (2005) found average estimates around 25% to 50%. For marine PAs, some authors have suggested protection requirements of 50% to 90%, due to the vulnerability of MPAs resulting from their lack of functional boundaries (Boersma and Parrish 1999). E.O. Wilson, in his book Half-Earth, proposes a global target of 50% protection which he argues will ensure the persistence of over 80% of species, while also providing a clear, biologically meaningful goal for conservationists to rally around (Wilson 2016). Wilson's proposal has gained support from those encouraging conservationists to be bolder and accept fewer compromises (Noss et al. 2012).

To assess the current shortfall in required protection, and thus help estimate what amount might be sufficient, some researchers use a 'gap analysis'. This method identifies what proportion of a species' range is overlapped by PAs and then compares that against a coverage target for the species to give an estimate of the shortfall in protection (Rodrigues et al. 2004). However, it requires that we have quantitative targets for each species, but producing accurate coverage targets is impossible to achieve for the vast majority because we have no or only very minimal range data for them. Rodrigues et al. (2004) produced a broad-brush method for calculating targets for species with basic range data available, based on range size. A uniform target of, say, 20% coverage of all species ranges will not produce helpful results – 20% for a wide-ranging species could be an unnecessarily large area, while 20% of a range-restricted species may not be enough to sustain the population. Taking that into account, Rodrigues et al. (2004) assigned a target of 100% coverage to species with a range of 1,000 km^2 or less, and 10% to those with a range of 250,000 km^2 or more, and targets for the remaining species are linearly interpolated on a log-linear scale between those two extremes. This method is not foolproof, due to differences in species' ecology which make area-based targets inappropriate, and to the inevitable errors of commission and omission found in species range data (Rondinini et al. 2006). Nonetheless, it is useful when improving the accuracy of available data in unfeasible.

In addition to uncertainty regarding appropriate targets, there are also issues with measurement towards them. Butchart et al. (2015) estimate that a further 3.3 million km^2 would be needed to meet the global 17% target, but suggest that this shortfall could be reduced simply through improved reporting of existing PAs. A significant proportion of PAs listed in the WDPA are represented only as a point (sometimes with the area given as a value), rather than a polygon with defined boundaries. This makes accurate gap analysis difficult, as it is not clear exactly which areas are protected. Significant progress has been made on some of these issues (Jenkins & Joppa 2009), but there is room for further improvement. In particular, the WDPA largely relies on national governments to report on PAs, resulting in a bias towards state-owned and managed PAs (Lopoukhine & de Souza Dias 2012). Thus for many countries we are lacking comprehensive and accurate data on the numerous PAs that are owned and managed by private individuals or organisations, local communities and indigenous peoples (Dudley et al. 2014). This is a significant omission, as these types of non-state PAs are known to often be as effective, if not more so, than their state-run counterparts in both terrestrial (Cousins et al. 2008; Hayes 2006) and marine environments (McClanahan et al. 2006).

Even with these other types of privately- and communally managed PAs, the shortfall in protection required to conserve Earth's biodiversity is immense (Butchart et al. 2015). Target 11 also recommends increasing the use of 'other effective area-based conservation measures' (OECMs), which may help fill gaps in protection where the establishment of PAs is unfeasible (Donald et al. 2019), and provide connectivity between PAs. OECMs are defined as *"a geographically defined area other than a Protected Area, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the in situ conservation of biodiversity, with associated ecosystem functions and services and where applicable, cultural, spiritual, socioeconomic, and other locally relevant values"* and can cover a variety of land uses (Donald et al. 2019). Some OECMs could also be included in the WDPA; however the IUCN makes clear that for any area to be considered a PA, it should have biodiversity conservation as its primary objective. This goal must take precedence over all other considerations (Dudley 2008). Thus, land uses such as sustainable forestry, organic agriculture, de-militarised zones and so on cannot, although suggested by some, be considered PAs because conservation is not their top priority. However, that is not to say that such areas are not valuable: even as the global PA network expands, most biodiversity will continue to exist outside it and so a move towards sustainable land uses outside PAs is also vital to the conservation of biodiversity (Lopoukhine & de Souza Dias 2012).

1.3 Thesis structure

There is an extensive literature on protected areas and other types of conservation area, reflecting the importance of this approach for biodiversity conservation. However, there are still important gaps, both on understanding the fundamental drivers and specific characteristics of conservation area networks. In this thesis I address a number of these important issues through the following research:

Chapter 2: An analysis aiming to identify priority areas for the expansion of England's NNR network. We improve upon previous work to better account for the highly fragmented nature of habitats in England, and use MinPatch to produce portfolios of priority areas that meet conservation targets while also being concentrated in patches of a manageable size. We then examine the trade-off between area selected and the opportunity costs that may be incurred through agricultural land lost.

Chapter 3: A gap analysis of conservation areas in the province of KwaZulu-Natal, South Africa. We examine the contribution made by conservation areas with different governance types to the overall extent and representativeness of the network, with an emphasis on highlighting the role of private and community conservation areas. To do so, we use protection equality analysis in a new, target-based way that takes into account the differing levels of vulnerability of priority habitats.

Chapter 4: A conceptual framework which examines the drivers of conservation area establishment across the world and thus why conservation area networks differ so significantly between countries. These drivers are divided into those that motivate, and those that influence the capacity to act upon that motivation, as it applies to national governments, private individuals, and communities.

Chapter 5: Development of a new methodology for producing a representative set of areas across the globe, which can be used to determine global patterns in conservation area coverage. Including ten factors that both represent global biodiversity and drive patterns and extent of conservation area coverage, we use the systematic conservation planning software Marxan to produce a sample of land covering 10% of the terrestrial area which will serve as the basis for data collection.

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Chapter 2. Bigger, better, more expensive? Investigating trade-offs between protected area network representativeness, patch size and opportunity costs in England

2.1 Abstract

Natural habitats in England are highly transformed, leaving many species confined to small fragments surrounded by agriculture. Protected areas are a key component of conservation strategies nationally and internationally, as they have been shown to be effective for many species and habitats in different contexts. In 2010 the Lawton Review urged the creation of 'more, bigger, better, joined' protected areas in the UK to stem biodiversity loss, and they are central to the Government's 25 Year Environment Plan. Systematic conservation planning is a widely-used approach for developing such ecological networks. However, it has rarely been used in England, partly because the country's natural habitats are so fragmented that standard analyses based on coarse-scale data are insufficient. Here we overcome this problem by presenting a spatial prioritisation analysis to inform the expansion of England's network of National Nature Reserves (NNRs). We used the Marxan spatial prioritisation software to identify where new NNRs should be located to meet low, medium and high targets for 29 priority habitats, based on planning units that grouped patches of natural vegetation to account for habitat fragmentation. We also used the MinPatch software to investigate how ensuring each NNR is above a specified area threshold influences the opportunity cost to agriculture. We found the area selected varied between 2.9% of the country when conserving up to 50 km^2 of each priority habitat in NNRs with a minimum patch size of 5 km^2, and 31% when conserving up to 150 km^2 of each priority habitat in NNRs with a minimum patch size of 20 km^2. We also found that increasing targets increased the proportion of high quality agricultural land needed, whereas increasing patch size did not, although the total area of land needed increased with both. Our analysis shows the benefits of

using a systematic conservation planning approach to understanding trade-offs between targets, protected area size and opportunity costs, because it ensures that objectives are made explicit and quantifiable. It also demonstrates the scope for applying this approach in even the most ecologically fragmented countries.

2.2 Introduction

Biodiversity is under threat across the world due to habitat loss, unsustainable agriculture, over-harvesting and climate change (Maxwell et al. 2016). In line with global trends, many species in England are in decline, with 39% decreasing in abundance over the past decade and 13% at risk of extinction from the UK (Hayhow et al. 2019). England has already experienced extensive habitat loss and conversion due to its long history of human habitation, high population density, and industrialisation which has resulted in substantial modification of landscapes across the country (Shwartz et al. 2017). Thus, few if any remaining areas could be accurately described as truly “wild” or “natural”. Nonetheless, England still contains some globally significant habitats such as ancient woodlands, chalk grasslands, lowland heathlands and blanket bogs, as well as cliffs and islands that support internationally important populations of seabirds (Hayhow et al. 2019). However, many of these habitats that remain exist only in tiny fragments scattered across the country, separated by large swathes of urban and agricultural land. Protected areas are a key tool in the fight against the loss of species and habitats in the UK and globally (Watson et al. 2014). Studies show that they can be effective (Geldmann et al. 2013; Gray et al. 2016), but it is essential to continue expanding PA networks as well as improving their planning and ongoing management to ensure their efficacy in protecting biodiversity over the long term.

As the UK prepares to leave the European Union, there is a great deal of uncertainty about the future direction of the country, including on the environment. However, in the absence of any specific statement suggesting otherwise, in the short term it appears likely that not much will change. The Lawton review urged the expansion of England's conservation area network with the mantra 'bigger, better, more, joined' (Lawton et al. 2010), while the Department for Environment, Food and Rural Affairs' (Defra) 25 Year Environment Plan includes a commitment to create and restore 500,000 ha of natural habitat as part of the Nature Recovery Network project (UK Government 2018a). National Nature Reserves (NNRs) will play a crucial role in conserving particularly valuable areas within this habitat network, in addition to Sites of Special Scientific Interest (SSSIs) and improved land management practices outside formally protected sites (Isaac et al. 2018). Thus Natural England, the government agency responsible for conservation of the natural environment, is looking at how and where to expand England's NNR network (Natural England 2016). NNRs are established to protect nationally important habitats, species and geology, and provide opportunities for public engagement with the natural world (UK Government 2020a), while SSSIs are intended to protect the very best natural sites which support charismatic, rare and endangered species (UK Government 2020b).

It is vital that new PAs are planned and placed carefully to ensure they are effective in protecting biodiversity and habitats. It is well known that state PAs globally tend to be biased toward land that is remote, inhospitable and of low economic value (Joppa & Pfaff 2009) and away from land that is valuable for food production (Venter et al. 2014). The same is true of England (Oldfield et al. 2004; Shwartz et al. 2017). As a result, protection is often biased towards habitats and species present in these 'lands that nobody wants' (Shands & Healy 1977) and thus away from others that exist in lowland and economically valuable areas. Systematic conservation planning is an approach to PA planning that is designed to ensure that networks adequately cover biodiversity and habitats of conservation concern, within PAs that are sufficiently large and connected to be ecologically viable, while also minimising associated costs to people (Margules & Pressey 2000).

This approach has been adopted by many countries across the world (Sinclair et al. 2018), as it is adaptable for use in almost any landscape or context. However, its use in England has largely been restricted to datasets focusing on single taxonomic groups (Moilanen et al. 2005; Early & Thomas 2007) or recorded at spatial scales that are much larger than the typical size of English PAs (Oldfield et al. 2004; Thomas et al. 2013). This means that until recently, systematic conservation planning has not been used by conservation agencies to inform action on the ground, although it has been used to help design Marine Conservation Zones (JNCC & Natural England 2010; Lieberknecht & Jones 2016). This changed in 2018 with a pilot study developed for Natural England that investigated whether the approach is suitable for informing the planning of England's terrestrial ecological networks (Pett et al. *in press*). The report concluded that systematic conservation planning has great potential for informing decisions at the national and regional scale, but the extreme fragmentation of habitats in England meant that standard approaches based on dividing the planning region into regular-shaped hexagonal planning units produced inefficient results.

The reason why the previous study (Pett et al. *in press*) from England produced inefficient results is that most systematic conservation planning software is designed to select planning units that meet conservation targets, whilst minimising costs and maintaining connectivity (Moilanen et al. 2009). Many of England's priority habitats have a limited extent, while also being highly fragmented and scattered across the country. This meant that Marxan (Ball et al. 2009), the software used in the analysis, had to select many planning units to meet the targets and identify priority areas that were large enough to be viable protected areas, even though most of the land within these planning units was transformed. Fortunately, there are two ways to resolve this problem. The first is to modify the planning units so that they better reflect the distribution of the biodiversity within them, so that some contain all transformed land and some contain the fragments of priority habitats (Holmes & Pugnalin 2016). The second is to modify the outputs using MinPatch, a software package developed to design conservation area networks in which every area meets a specified minimum size threshold (Smith et al. 2010). This will ensure that the potential NNRs identified in the analysis are large enough to be viable. Such an approach is important because it also allows an analysis of the trade-offs involved between protected area network extent and impacts on agriculture (Adams et al. 2010; Venter et al. 2014), and between the size of individual protected areas and the number of protected areas in which each priority habitat is found, a key part of developing resilient ecological networks found (Margules & Pressey 2000; Pressey et al. 2007).

In this study we build on this earlier work (Pett et al. *in press*) and used Marxan to identify priority areas for creating new NNRs in England based on ecologically coherent planning units and three sets of targets for conserving priority habitats. We also use MinPatch and CLUZ, a QGIS plug-in for conservation planning (Smith 2019b), to examine the trade-off between (a) area protected and the opportunity costs incurred through the quality of agricultural land lost, and (b) protected area size and conservation feature replication, i.e. the number of protected areas in which each priority habitat type is found.

2.3 Methods

2.3.1 Conservation features

Our conservation features were 29 priority habitats (Figure 2.1; Table 2.1). We used Natural England's Priority Habitat Inventory raster map, which displays the location and extent of Natural Environment and Rural Communities Act (2006) Section 41 Habitats of Principal Importance (Natural England 2019a).

2.3.2 *Producing the planning system*

We defined the study area using the Land Cover Map 2015 (UK Centre for Ecology and Hydrology 2015), from which we removed saltwater and littoral habitats, and any land that lies below the high water line. We produced a grid of 2 km^2 hexagonal planning units in QGIS (QGIS 2009), clipped them to the study area and combined them with National Parks, National Nature Reserves (NNRs), Sites of Special Scientific Interest (SSSIs) and Areas of Outstanding Natural Beauty (AONBs) using the Union tool in ArcMap 10.5 (ESRI 2011). We then also unioned them with ancient woodland, open mosaic habitat, wood pasture and parkland and habitat network vector layers. The latter describes habitat networks for 18 priority habitats, which includes their existing extent as well as additional areas where these habitats could be created or restored, and areas which could be used to connect existing habitat patches and reduce fragmentation. These habitat layers are derived from a variety of data sources including local record centres, local councils, Ordnance Survey and aerial photography (Edwards et al. 2019; Natural England 2019b). This provided our planning unit layer of hexagons and subsections of hexagons delineating conservation areas and areas of valuable habitat, allowing for these areas to be represented more precisely in the Marxan analysis (Figure 2.2). We then calculated the area of each conservation feature contained within each planning unit using the Tabulate Areas tool.

Figure 2.1. Natural Environment and Rural Communities Act (2006) Section 41 Habitats of Principal Importance.

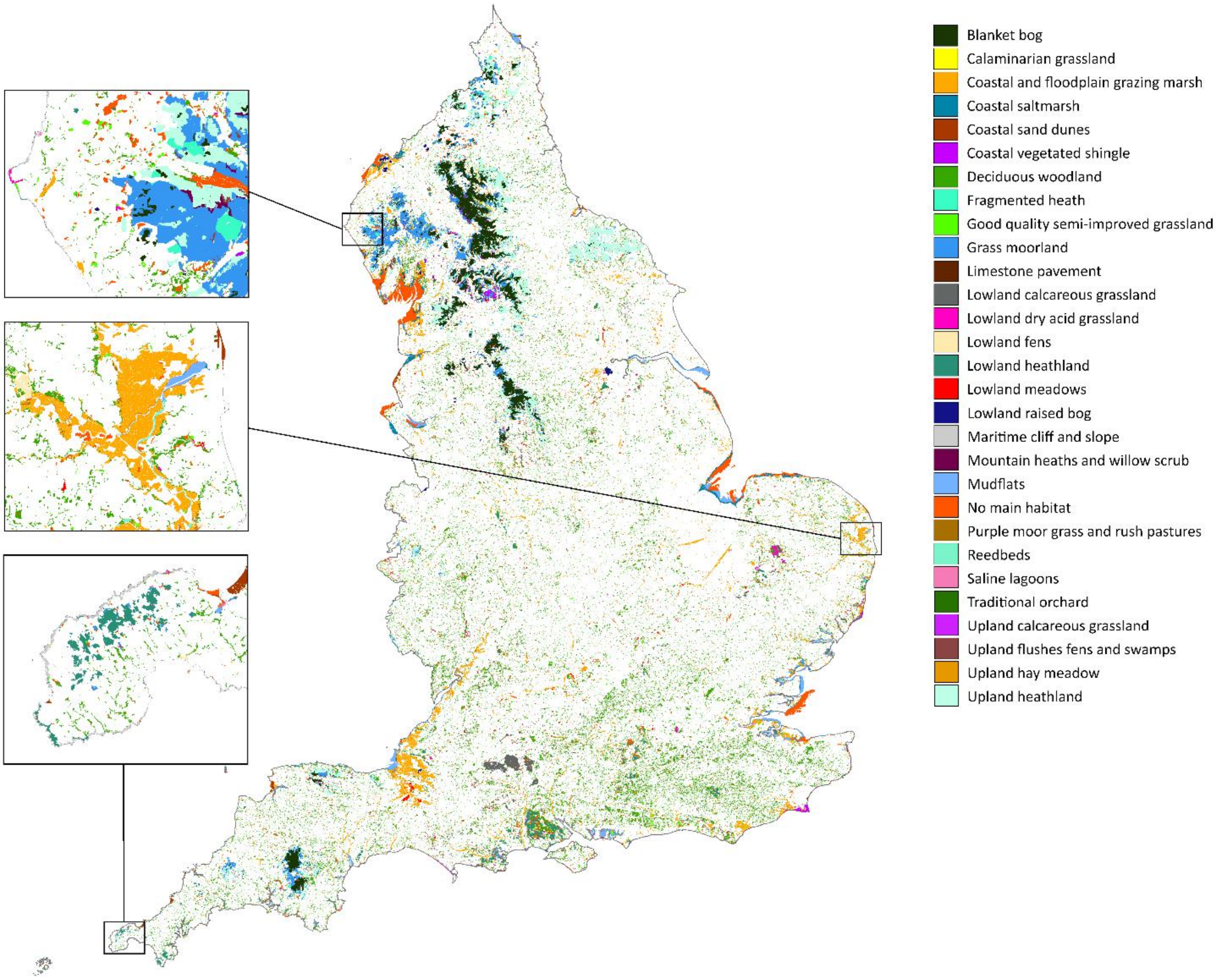


Table 2.1. Natural Environment and Rural Communities Act (2006) Section 41 Habitats of Principal Importance, with a description of each type (JNCC 2011).

Habitat	**Description**
Blanket bog	Rain-fed peatland habitat confined to cool, wet, oceanic climates. Supports a wide range of species including red-throated diver (*Gavia stellata*) and Eurasian golden plover (*Pluvialis apricaria*).
Calaminarian grassland	Semi-natural and anthropogenic, sparsely vegetated habitats on substrates containing heavy metals (e.g. lead, chromium and copper) or other unusual minerals. Characterised by open-structured plant communities of lichens, bryophytes and vascular plants, such as spring sandwort (*Minuartia verna*) and alpine pennycress (*Thlaspi arvense*).

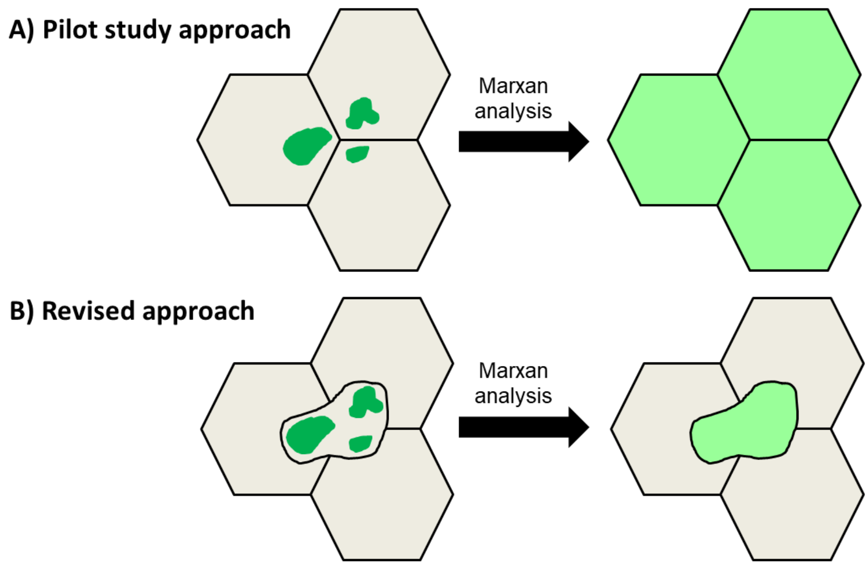
Coastal and floodplain grazing marsh	Periodically inundated pasture or meadows with ditches which maintain water levels, containing standing brackish or fresh water. Most areas are grazed or cut for hay and silage. Rich in plants and invertebrates, and important for breeding waders such as snipe (*Gallinago gallinago*), lapwing (*Vanellus vanellus*) and curlew (*Numenius arquata*). Also supports internationally important populations of Bewick swans (*Cygnus bewickii*) and whooper swans (*Cygnus cygnus*).
Coastal saltmarsh	The upper, vegetated portions of intertidal mudflats. Restricted to sheltered locations in estuaries, saline lagoons, sea lochs, beach plains and behind barrier islands. An important resource for wading birds, wildfowl and passerine birds.
Coastal sand dunes	Develops in exposed locations where there is sufficient sand in the intertidal zone and onshore winds are prevalent. Supports many invertebrates including butterflies, moths, burrowing bees and wasps, as well as flowering plants such as orchids.
Coastal vegetated shingle	A globally restricted coastal sediment type but widely distributed in the UK. Species such as sea kale (*Crambe maritima*), sea pea (*Lathyrus japonicas*), Babington's orache (*Atriplex glabriuscula*), sea beet (*Beta vulgaris*), and sea campion (*Silene uniflora*) are found at the shore, while mixed communities of grassland, heath, moss, lichen and scrub characterise areas behind. Supports breeding birds and diverse invertebrate communities.
Deciduous woodland	Semi-natural woodland growing on a range of soil conditions, characteristed by oak (*Quercus robur*) and other locally native tree species.
Fragmented heath	Lowland or highland heath with fragmented plant communities, vulnerable to growth of invasive bracken.
Good quality semi-improved grassland	Grasslands that have undergone cultivation, but retain potential for restoration.
Grass moorland	Moorland dominated by grass species.
Limestone pavement	Exposed by the action of ice sheets during the last Ice Age, of both geological and biological importance. Rich in vascular plants, bryophytes and lichens.
Lowland calcareous grassland	Develop on shallow lime-rich soils overlying limestone rocks, including chalk. Typically managed as components of pastoral or mixed farming systems. Supports a rich community of plants and invertebrates, including rare species such as monkey orchid (*Orchis simia)*, hoary rockrose (*Helianthemum canum)*, pasque flower (*Pulsatilla vulgaris),* adonis blue *(Lysandra bellargus)* and silver-spotted skipper *(Hesperia comma)*.
Lowland dry acid grassland	Occurs on nutrient-poor, free-draining soils overlying acid rocks or deposits such as sand and gravel. Supports woodlark (*Lullula arborea*), stonecurlew (*Burhinus oedicnemus*), nightjar (*Caprimulgus europaeus*), lapwing (*Vanellus vanellus*), skylark (*Alauda arvensis*), chough (*Pyrrhocorax pyrrhocorax*), green woodpecker (*Picus viridis*), hen harrier (*Circus cyaneus*) and merlin (*Falco columbarius*).

Lowland fens	Peatlands which receive water and nutrients from the soil, rock and ground water as well as rainfall. Supports diverse plant and animal communities, including dragonflies and aquatic beetles.
Lowland heathland	Open landscape on impoverished, acidic mineral and shallow peat soil, characterised by heathers and dwarf gorses.
Lowland meadows	Includes most forms of unimproved neutral grassland across enclosed lowland landscapes, used for hay cropping and grazing. Also found in non-agricultural settings such as recreational sites, churchyards and roadside verges. Supports species such as fritillary (*Fritillaria meleagris*), Dyer`s greenweed (*Genista tinctoria*), green-winged orchid (*Orchis morio*), greater butterfly orchid (*Platanthera chlorantha*), pepper saxifrage (*Silaum silaus*) and wood bitter vetch (*Vicia orobus*).
Lowland raised bog	Peatlands which develop in lowland areas such as the head of estuaries, along river flood-plains and in topographic depressions. Includes flora such as *sphagnum* mosses, bog rosemary (*Andromeda polifolia*), great sundew (*Drosera anglica*) and cranberry (*Vaccinium oxycoccos*). Supports a distinctive range of fauna including waders, wildfowl and invertebrates.
Maritime cliff and slope	Sloping vertical faces on the coast, including soft and hard cliffs. Supports internationally important populations of breeding seabirds, including gannet (*Morus bassanus*), shag (*Phalacrocorax artistotelis*), razorbill (*Alca torda*) and guillemot (*Uria aalge*), as well as rich assemblages of invertebrates including solitary bees and wasps.
Mountain heaths and willow scrub	Natural vegetation in the montane zone, lying above the tree-line, supporting diverse communities of plants and invertebrates including beetles *Stenus glacialis* and *Phyllodecta polaris*, flies *Alliopsis atronitens* and *Rhamphomyia hirtula*, and the spider *Micaria alpina*.
Mudflats	Intertidal mudflats.
No main habitat but additional habitats present	Areas containing candidate priority habitats, but no main habitat type is identified.
Purple moor grass and rush pastures	Occur on poorly drained, acidic soils in lowland areas of high rainfall, with distinct vegetation characterised by purple moor grass (*Molinia caerulea*) and sharp-flowered rush (*Juncus acutiflorus*). Supports species such as greater butterfly orchid (*Platanthera chlorantha*), lesser butterfly orchid (*Platanthera bifolia*), marsh fritillary (*Eurodryas aurinia*), brown hairstreak (*Thecla betulae*), narrow-bordered bee hawkmoth (*Hermaris tityus*), curlew (*Numenius arquata*), snipe (*Gallinago gallinago*) and barn owl (*Tyto alba*).
Reedbeds	Wetlands dominated by stand of common reed (*Phragmites australis*), incorporating open water, ditches, wet grassland and carr woodland. Among the most important UK habitats for birds including the bittern (*Botaurus stellaris*), marsh harrier (*Circus aeruginosus*), crane (*Grus grus*), Cetti`s warbler (*Cettia cetti*), Savi`s warbler (*Locustella luscinioides*), bearded tit (*Panurus biarmicus*) and aquatic warbler (*Acrocephalus paludicola*).

Saline lagoons	Natural or artificial bodies of saline water partially separated from the sea which retain seawater at low tide. Important habitat for invertebrates, waterfowl, marshland birds and sea birds.
Traditional orchard	Orchards managed in a low-intensity way. Structurally and ecologically similar to wood-pasture and parkland, with trees set in herbaceous vegetation.
Upland calcareous grassland	Occur on lime-rich soils above the upper limit of agricultural enclosure. Occur as components of habitat mosaics, generally managed as rough livestock grazing. Supports a wide range of uncommon species.
Upland flushes fens and swamps	Peat or mineral-based upland wetlands, above the limit of agricultural enclosure. Supports a rich flora of vascular plants, including many rare species, and provides nesting habitat for wading birds.
Upland hay meadow	Grasslands characterised by a striking variety and abundance of grasses and dicotyledons such as wood crane`s-bill (*Geranium sylvaticum*), pignut (*Conopodium majus*), great burnet (*Sanguisorba officinalis*) and lady`s mantles (*Alchemilla* species).
Upland heathland	Occurs widely on mineral soils and thin peats on uplands and moorlands beyond the limit of agricultural enclosure. Supports a range of birds, reptiles and invertebrates.

We used the 'Provisional Agricultural Land Classification' vector layer to produce the cost layer (Figure 2.3). It is based on climate, soil and site factors such as rainfall, gradient and stoniness to assess the quality of the land for agriculture (Natural England 2019c). We assigned cost values increasing from the lowest cost for the lowest quality agricultural land, to the highest cost for the highest quality agricultural land. This was to ensure that areas suitable for agriculture were avoided where possible. We also assigned the lowest cost value to land that is already within an existing conservation designation – National Parks, AONBs and SSSIs. Thus, where possible, Marxan would choose these areas to help meet targets. Using the Zonal Statistics tool, we calculated the sum of the cost value of all pixels within each planning unit.

Figure 2.2. Schematic of the planning units selected by Marxan based on (A) the approach used in the pilot study (Pett et al. in press) and (B) the revised approach outlined above. In the pilot approach all three of the hexagonal planning units (in grey) have to be selected (in light green) to meet the target for the priority habitat type (in dark green). In the revised approach these priority habitat patches fall within a planning unit that is derived from habitat network and broader habitat layers which include their distribution, so only this new planning unit is selected (from Pett et al, in press and provided with permission).

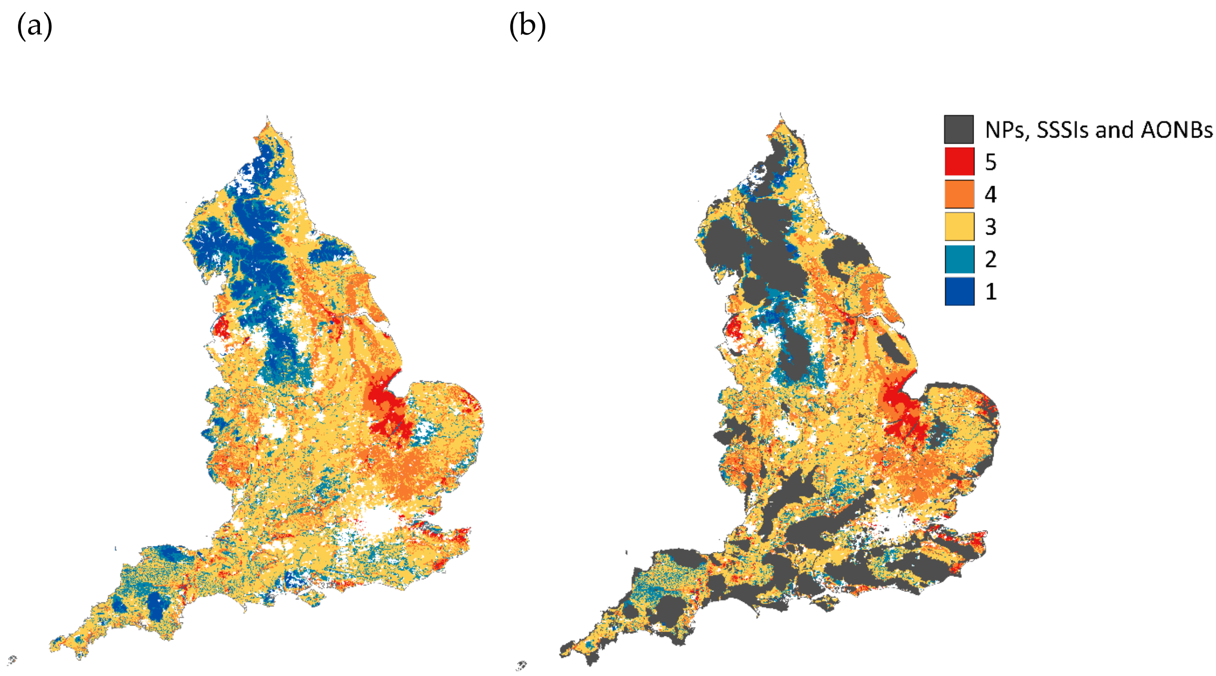


Finally, we classified 2,455 planning units as 'Conserved' because they fall within the boundaries of existing NNRs and 11,792 planning units as 'excluded' because 50% or more of their area is urban. This ensured that Marxan automatically selected the conserved planning units and never selected the excluded planning units. The threshold of no more than 50% urban was decided in consultation with experts from Natural England, with the intention to allow the selection of suburban parks and gardens which may be valuable for wildlife, while excluding highly built up areas that are unlikely to support wildlife.

2.3.3 *Marxan analysis*

To identify priority areas for creating new NNRs or extending existing ones, we used the systematic conservation planning software Marxan (Watts et al. 2017), through the QGIS (QGIS 2009) plug-in CLUZ (Smith 2019b). Marxan is a freely-available spatial prioritisation software package that uses simulated annealing to identify near-optimal sets of planning units that meet targets, whilst minimising planning unit costs and maintaining connectivity (Ball et al. 2009). CLUZ is a free user-friendly interface for Marxan which allows on-screen planning and easy display of selection results.

Figure 2.3. Maps of our cost layer based on the Provisional Agricultural Land Classification (Natural England 2019c), where 1 is the lowest cost and 5 is the highest cost; (a) shows the base cost layer and (b) shows an additional overlay of the conservation areas in which land was assigned a cost of 1 in the analysis regardless of its agricultural quality (National Parks, Sites of Special Scientific Interest and Areas of Outstanding Natural Beauty). Urban land is not assigned a value and these areas were not included in the Marxan analyses.



Each analysis involves running Marxan a specified number of times, producing a near-optimal but different solution each time. Each run consists of a specified number of iterations and increasing the iterations identifies more efficient solutions, although there are diminishing returns (Ball et al. 2009). The user can also influence the extent to which the solution consists of patches of planning units by specifying a Boundary Length Modifier (BLM) value, where setting a higher value produces larger patches. Marxan then identifies the "best" solution, which is the one with the lowest cost, and the selection frequency output, which counts the number of times each planning unit was selected in the solution produced by each run.

We conducted three Marxan analyses, with 100 runs of 100 million iterations each, a BLM of 1.5, and either low, medium or high conservation targets. These targets were developed during a workshop that was attended by experts from Natural England, the RSPB and the Universities of Kent and Liverpool (Pett et al, *in press*). The low, medium and high targets were 50 km^2, 100 km^2 and 150 km^2, respectively, of each feature, unless the total area of the feature was less than the target amount, in which case the target was set to their total area.

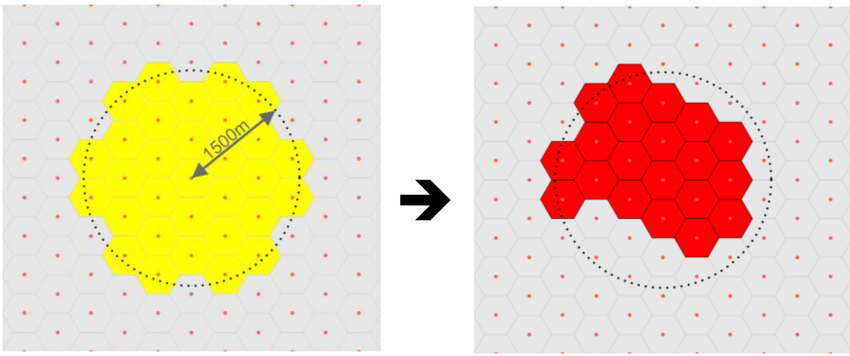
2.3.4 MinPatch analysis

MinPatch is a software package designed to modify Marxan outputs and produce networks of planning units that are grouped together to form patches of at least a specified minimum size (Smith et al. 2010). The process involves four steps: (i) a Marxan portfolio is imported into Minpatch, (ii) patches of planning units in the portfolio that do not meet the minimum size requirement are removed, (iii) new patches, larger than the minimum size requirement are added to meet targets in a cost-efficient way, and (iv) these new patches are 'whittled down' as planning units are removed from the edges one by one, provided that doing so does not increase the overall cost of the portfolio, or prevent the targets or minimum size threshold from being met (Smith 2019a). The user specifies the minimum patch size but also the size of the patches added in step (ii), based on setting the radius of each patch. This radius must be large enough to ensure each added patch is viable but it can be set much larger, producing larger raw patches from which MinPatch can remove more planning units and so identify protected areas that can be more irregular in shape (Figure 2.4).

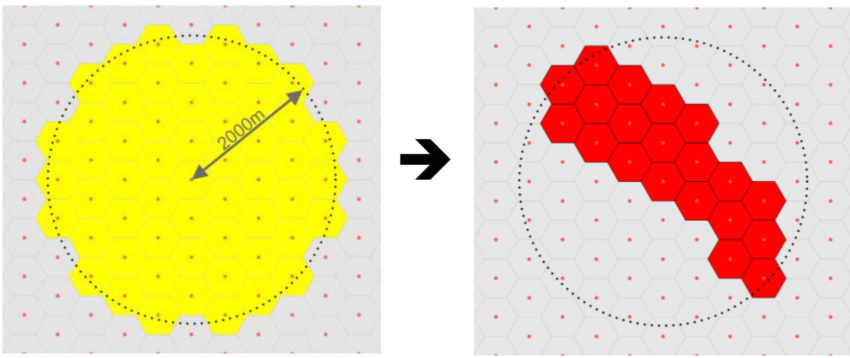
We ran the results of our three Marxan analyses through MinPatch using small, medium and large minimum patch sizes for our low, medium and high target scenarios, thus producing nine sets of outputs based on 100 Marxan portfolios for each output. These patches sizes were defined based on expert opinion as having: an area of 5 km^2 within a radius of 1,800 m; an area of 10 km^2 within a radius of 3,200 m; or an area of 20 km^2 within a radius of 4,800 m. In each analysis MinPatch identified the 'best' portfolio, based on which of the 100 modified portfolios had the lowest combined planning unit and boundary cost, and the selection frequency output based on the number of times each planning unit was selected in the 100 modified portfolios.

Figure 2.4: Illustration of the results of using different new patch radius values on the shape of patches identified using MinPatch. The example is based on a set of hexagonal 25 ha planning units with a minimum patch size threshold of 500 ha and a new patch radius value of (A) 1500 m and (B) 2000 m (replicated with permission from Smith 2019a).

A) Setting the new patch radius of 1500m means that each new patch contains up to 31 planning units. With a minimum patch size of 500 ha, simulated whittling can remove up to 11 planning units (assuming they are not needed to meet any targets) leaving a more compact final patch.



B) Setting the new patch radius of 2000m means that each new patch contains up to 55 planning units. With a minimum patch size of 500 ha, simulated whittling can remove up to 35 planning units so the final patch can have a more irregular or elongated shape.



2.4 Results

2.4.1 Marxan analysis

The total area of the planning system is 130,671 km^2, which we divided into 189,710 planning units in total, with a median size of 0.3 km^2. For the low targets analysis, Marxan selected 10,444 planning units with a total area of 3,844 km^2 (Fig 2.5a). They are dispersed widely across the country, with moderate concentrations in the north in the Yorkshire Dales, Lake District and North Pennines, in the west around Worcestershire and Gloucestershire, on the Suffolk coast and Norfolk Broads, north-west Kent, and the south coast. The selection frequency results show that 149,944 planning units were never selected across the 100 runs, meaning that they are never necessary for meeting the targets, while 5,334 planning units were selected every time, meaning that they are 'irreplaceable' and always necessary for meeting the targets (Fig 2.5b).

For our second analysis using medium targets, Marxan selected 19,034 planning units with a total area of 9,149 km^2 (Fig 2.5c). These show increased concentrations in the north-west, the Peak District, Shropshire, the south-west particularly on Exmoor and Dartmoor, and along the south coast. The selection frequency results show that 145,977 planning units were never selected across the 100 runs, while 12,554 planning units were selected every time (Fig 2.5d).

For our third analysis using high targets, Marxan selected 50,051 planning units with a total area of 26,691 km^2 (Fig 2.5e). These show significant concentrations in the north-west, the Peak District, the west between Birmingham and Bristol, across the south-west, south coast, east coast and west Kent, as well as scattered areas west of London and across East Anglia. The selection frequency results show that 105,822 planning units were never selected across the 100 runs, while 36,335 PUs were selected every time (Fig 2.5f).

The average amounts by which current levels of protection fall short of our targets for each feature are 24.7, 56.5 and 82.9 km², respectively, for the 5, 10 and 20 km² targets. Two features – calaminarian grassland and fragmented heath – are not currently found in any existing NNRs in England (Table 2.2).

Table 2.2. The total extent of each conservation feature in the study region, the area currently contained with the NNR network, and the shortfall in area identified by each of the different target scenarios.

Habitat	Total area (km²)	Area in NNRs (km²)	Low target shortfall (km²)	Medium target shortfall (km²)	High target shortfall (km²)
Blanket bog	2,308.8	87.5	0.0	12.5	62.5
Calaminarian grassland	2.9	0.0	2.8	2.8	2.9
Coastal and floodplain grazing marsh	2,172.6	35.0	15.0	65.0	115.0
Coastal saltmarsh	311.3	68.8	0.0	31.2	81.2
Coastal sand dunes	101.6	16.9	33.1	82.9	84.7
Coastal vegetated shingle	39.2	12.4	25.6	25.6	26.8
Deciduous woodland	7,347.5	110.8	0.0	0.0	39.2
Fragmented heath	90.1	0.0	50.0	89.9	90.0
Good quality semi-improved grassland	739.8	4.3	45.7	95.7	145.7
Grass moorland	1,471.8	21.1	28.9	78.9	128.9
Limestone pavement	12.6	2.0	10.6	10.6	10.7

Lowland calcareous grassland	615.7	26.3	23.7	73.7	123.7
Lowland dry acid grassland	151.4	7.2	42.8	92.8	142.8
Lowland fens	201.7	23.8	26.2	76.2	126.2
Lowland heathland	562.1	63.5	0.0	36.5	86.5
Lowland meadows	210.3	10.2	39.8	89.8	139.8
Lowland raised bog	77.8	32.1	17.9	45.6	45.7
Maritime cliff and slope	131.2	5.6	44.4	94.4	125.5
Mountain heaths and willow scrub	14.1	1.0	13.0	13.0	13.0
Mudflats	141.1	22.2	27.8	77.8	118.9
No main habitat but additional habitats present	1,372.1	29.9	20.1	70.1	120.1
Purple moor grass and rush pastures	90.7	3.1	46.9	87.0	87.6
Reedbeds	31.2	6.0	24.1	24.1	25.2
Saline lagoons	13.1	0.8	11.8	11.8	12.3
Traditional orchard	160.2	0.1	49.9	99.9	149.9
Upland calcareous grassland	92.0	5.0	45.0	86.8	87.0
Upland flushes fens and swamps	100.0	3.4	46.6	96.2	96.6
Upland hay meadow	24.2	0.7	23.3	23.3	23.5
Upland heathland	2,274.8	56.7	0.0	43.3	93.3

Figure 2.5. 'Best' portfolios and selection frequency outputs, respectively, produced by Marxan for (a,b) low, (c,d) medium, and (e,f) high targets. The 'best' portfolios show the set of selected units which met the targets for the lowest cost out of each analysis consisting of 100 runs. The selection frequency outputs show how often the planning units were selected over the 100 runs.

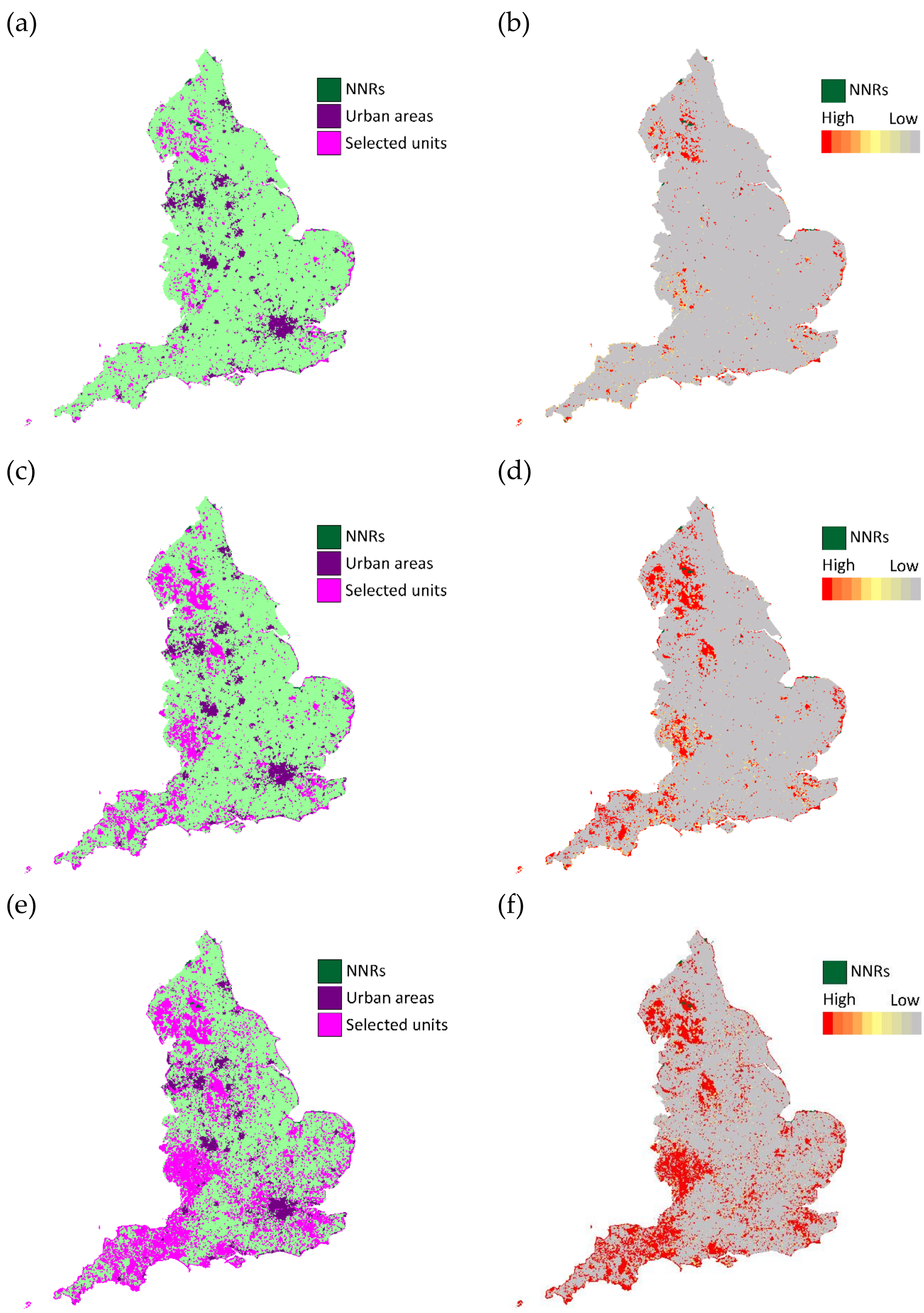


Figure 2.6. Marxan 'best' output for the medium target scenario.

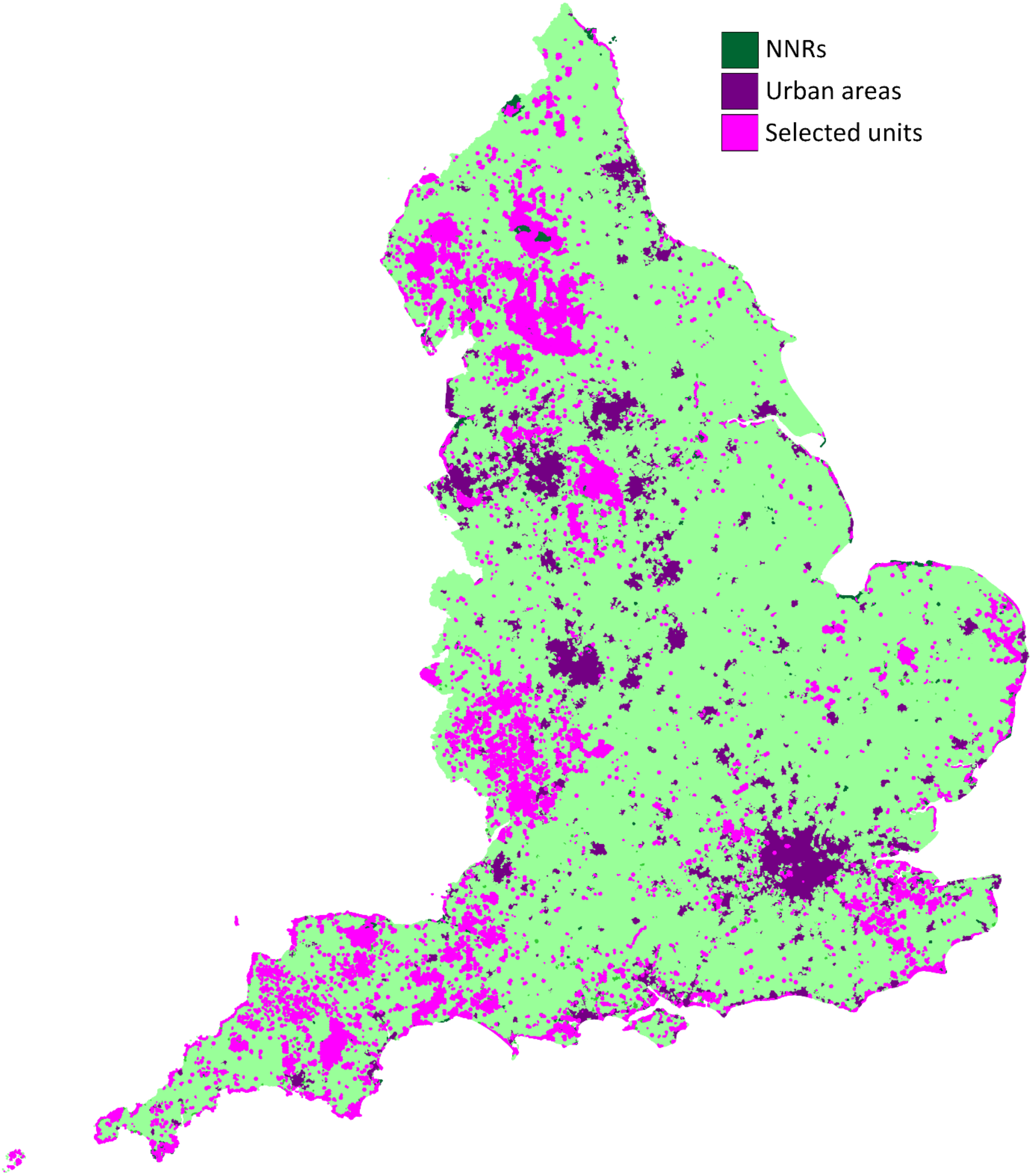
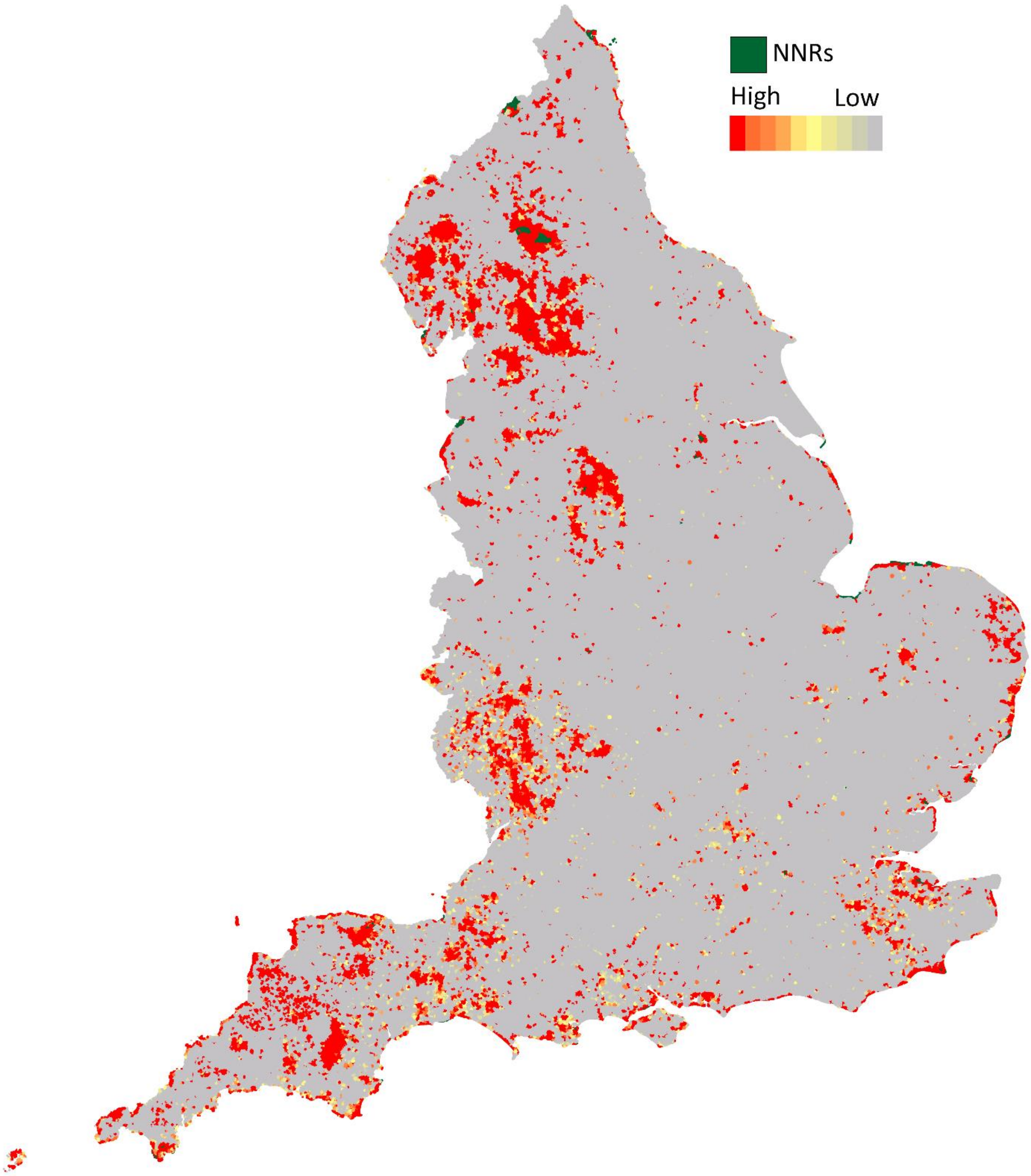


Figure 2.7. Marxan selection frequency output for the medium target scenario.



2.4.2 MinPatch analysis

Using low targets, MinPatch selected between 5,250 and 9,472 km^2 under the different patch size constraints, with the area increasing as the minimum patch size increased. Using medium targets, between 13,649 and 21,184 km^2 was selected, and with high targets between 27,534 and 40,522 km^2 was selected, again increasing as the patch size increased (Figure 2.8a-i). The areas selected are broadly similar to those selected by Marxan for the respective target scenarios, with a focus on the north-west, west and south-west of the country, west Kent and the coast of East Anglia. The total amount selected is greater with these patch size constraints, and although there is significant overlap, the areas selected by MinPatch are also more dispersed than those selected by Marxan (Figure 2.9a-i).

2.4.3 Portfolio extent and costs

The low target / 0 km^2 patch size scenario resulted in just 2.9% of the study region being selected, while the high target / 20ha patch size scenario selected 31.0% of the study region. The land with the lowest cost was selected the most across all scenarios, making up 78.4% of the low target / 0 km^2 scenario (Table 2.3).

Table 2.3. The percentage of England selected in the best portfolio for the low, medium and high target Marxan analysis (minimum patch size = 0 km^2) and under the three MinPatch patch size scenarios.

	Minimum protected area size threshold			
	0 km^2	**5 km^2**	**10 km^2**	**20 km^2**
Low targets	2.9	4	5.1	7.2
Medium targets	7	10.4	13	16.2
High targets	20.4	21.1	26.9	31

Figure 2.8. MinPatch 'best' outputs for (left to right) 5, 10 and 20 km² patch sizes for (a,b,c) low, (d,e,f) medium, and (g,h,i) high target scenarios.

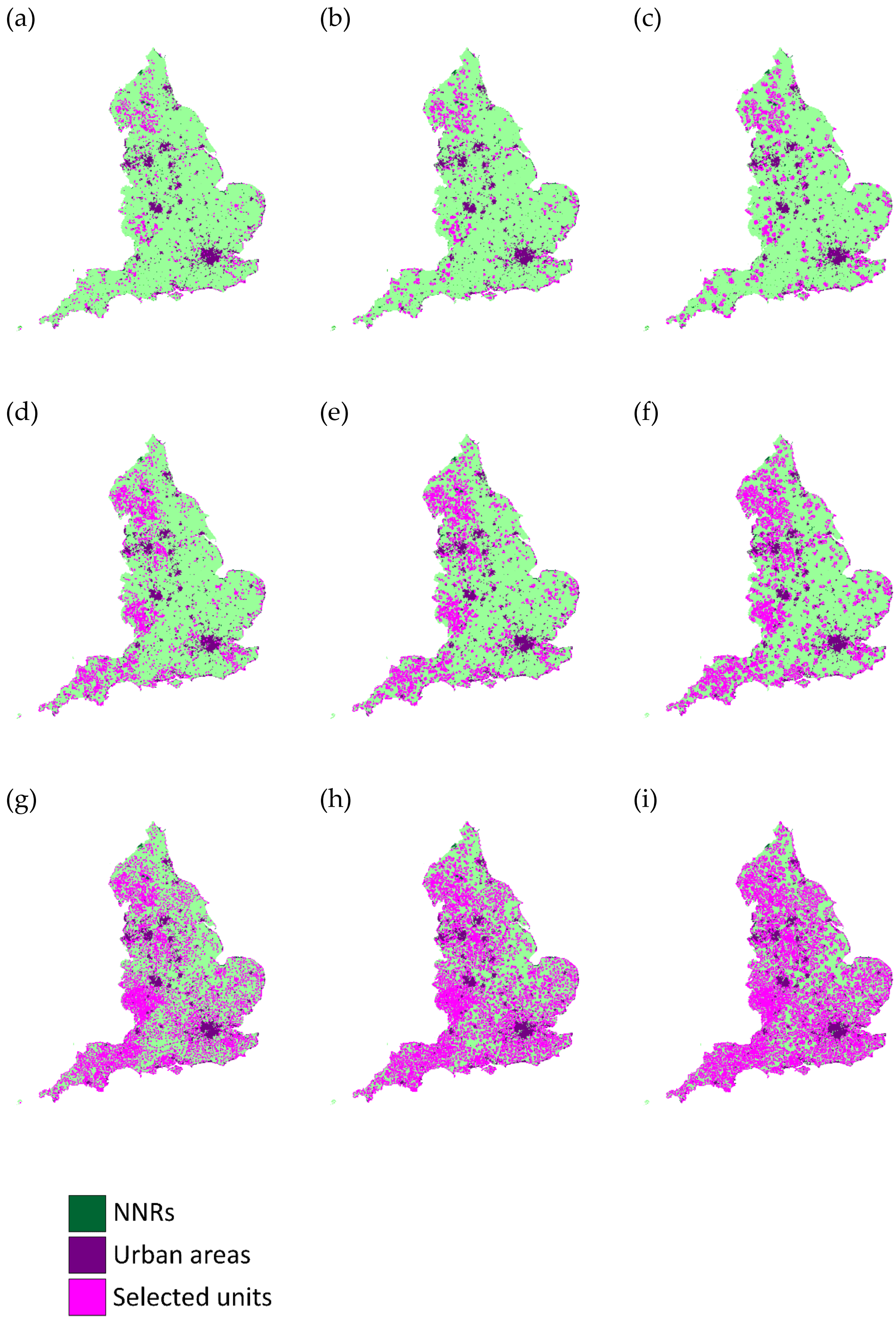


Figure 2.9. MinPatch selection frequency outputs for (left to right) 5, 10 and 20 km² patch sizes for (a,b,c) low, (d,e,f) medium, and (g,h,i) high target scenarios.

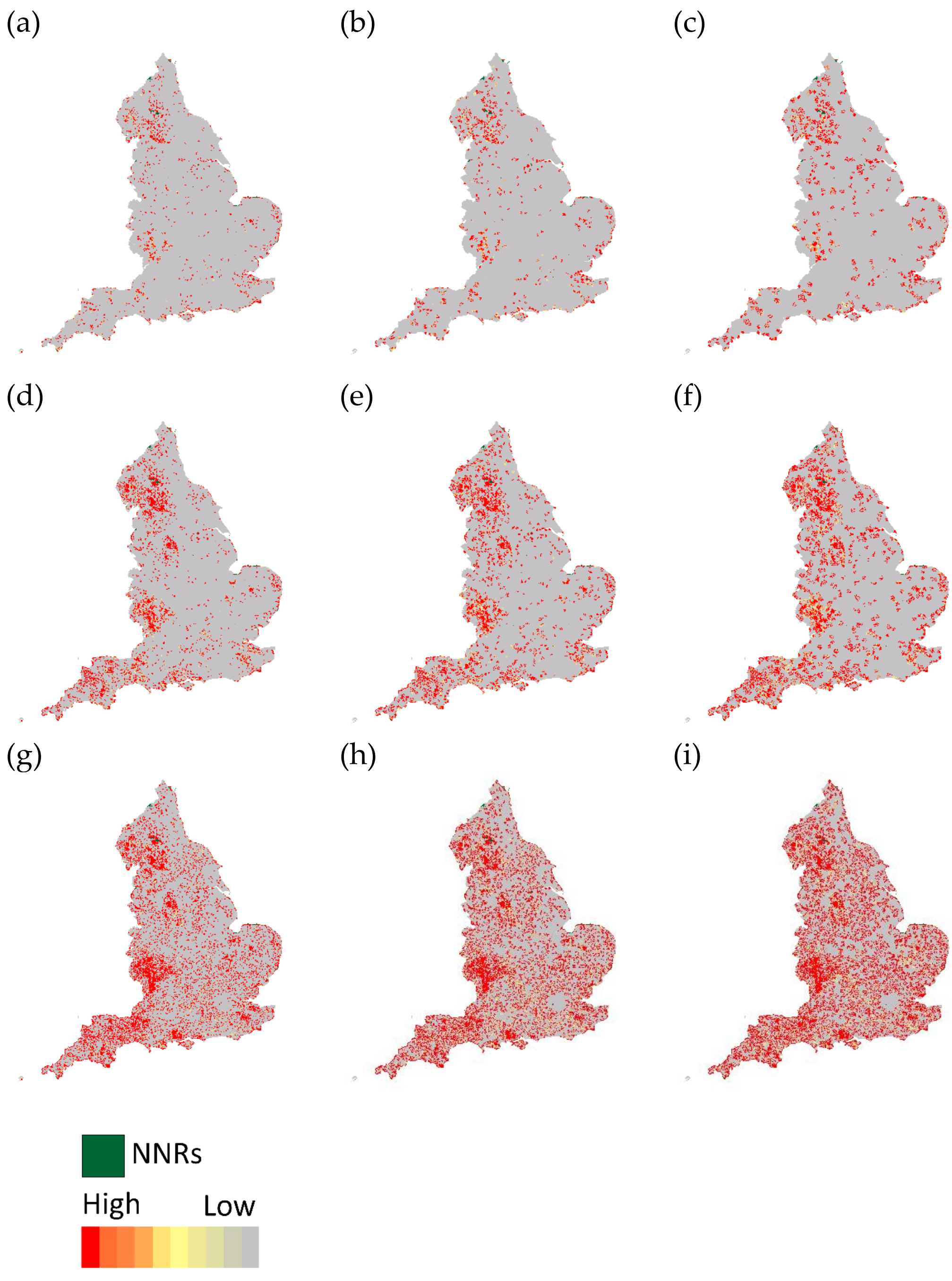


Figure 2.10. MinPatch 'best' output for the 10 km² patch size and medium target scenario.

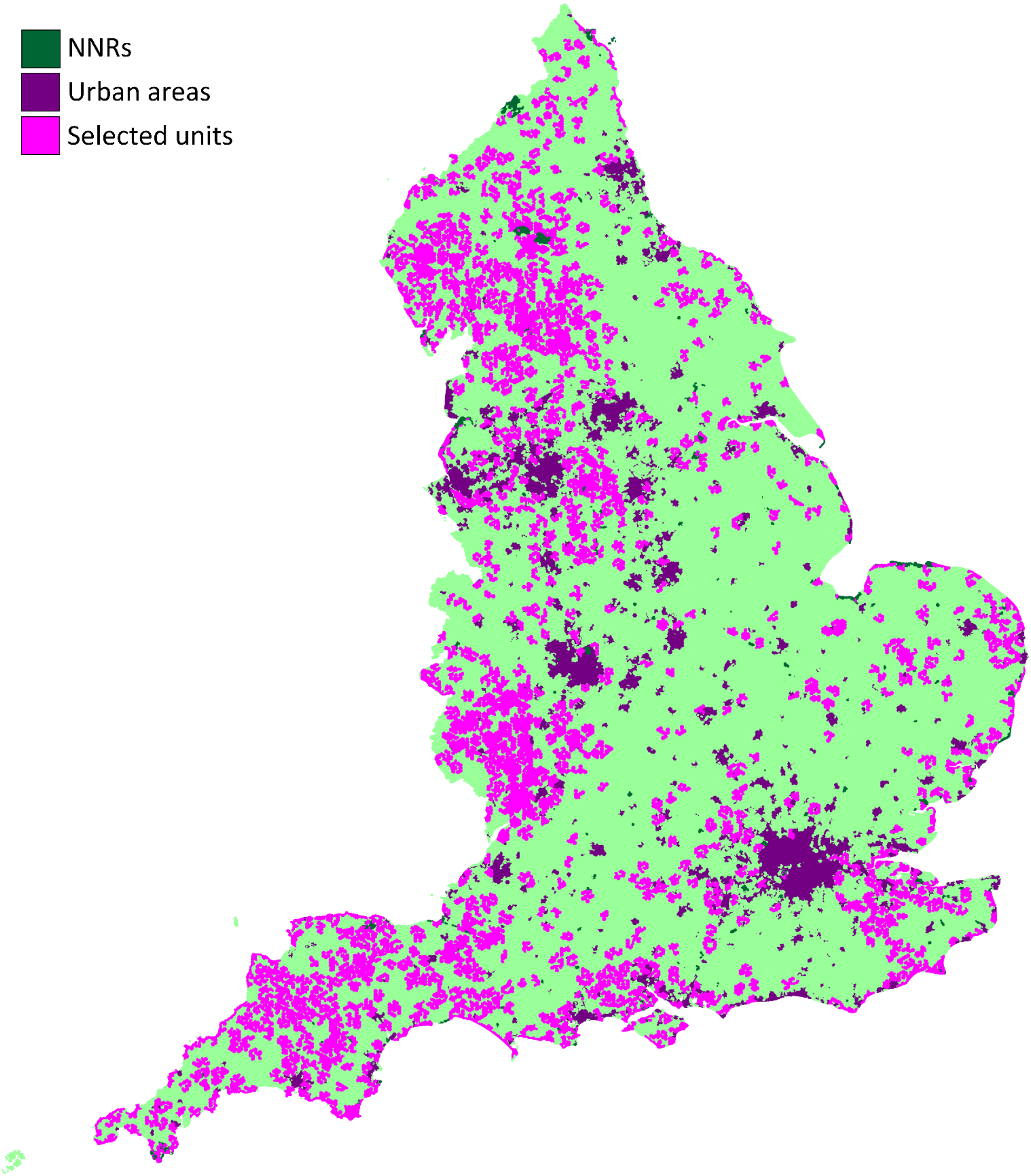


Figure 2.11. MinPatch selection frequency output for the 10 km² patch size and medium target scenario.

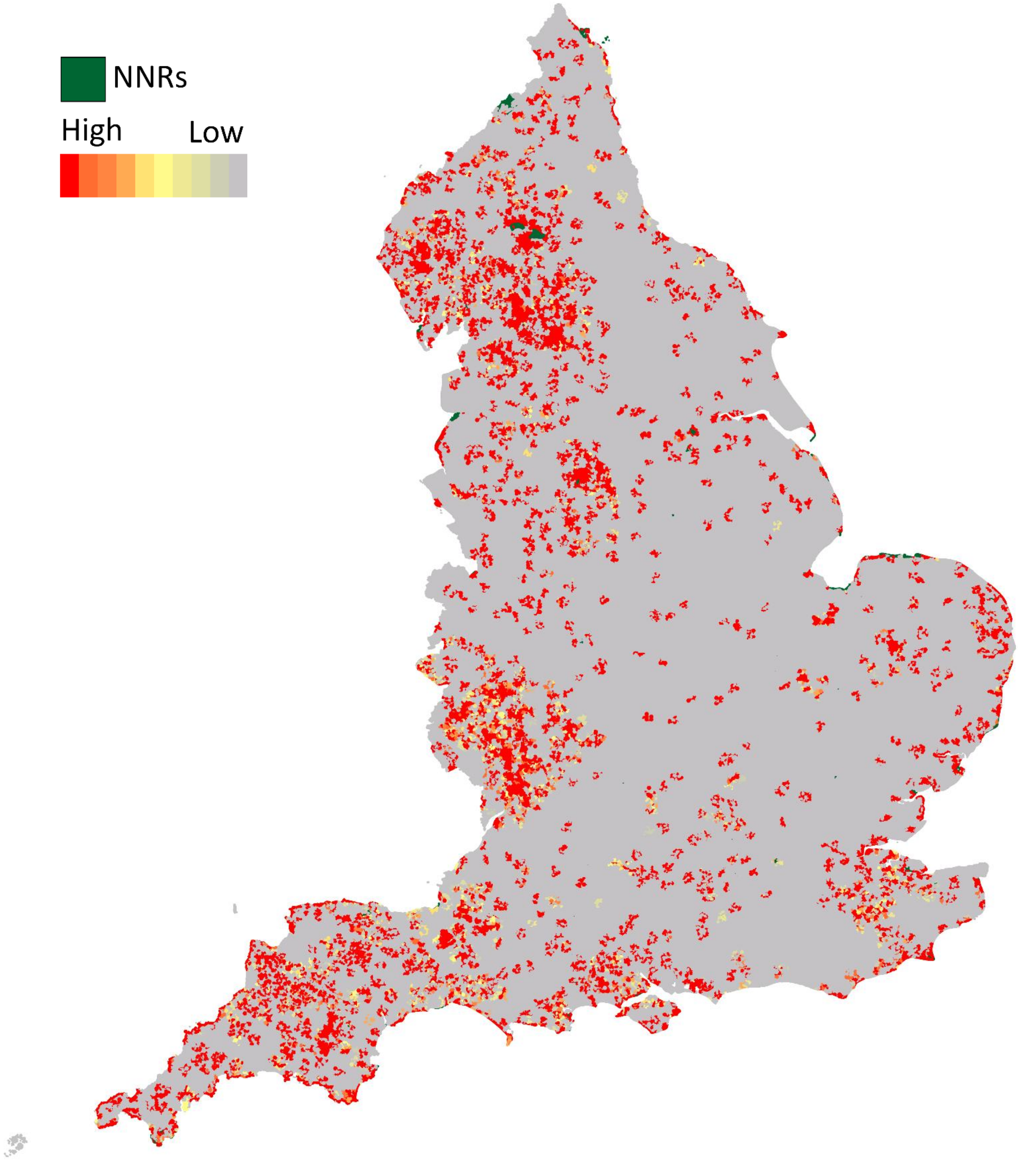


Table 2.4. The percentage of land in the 'best' Marxan solution for each target/patch size combination belonging to each agricultural land quality category; 1 is the lowest quality, 5 is the highest. Column (a) is based on the cost data used by Marxan, in which all land contained within existing conservation areas is assigned a value of 1 regardless of its agricultural quality, while column (b) does not reclassify land within existing conservation areas.

Targets	Minimum patch size (km^2)	(a) PAs given low agricultural quality cost					(b) Original agricultural quality cost				
		1	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**
Low	0	78.4	5.8	11.4	3.7	0.8	33.4	26.9	30.1	8.2	1.4
	5	67.5	7.9	17.1	5.9	1.6	27.1	24.9	35.3	10.5	2.2
	10	64.5	8.5	19.1	6.5	1.5	26.0	25.1	36.5	10.3	2.0
	20	60.5	9.0	21.9	7.2	1.3	24.7	23.3	39.3	10.8	1.8
Medium	0	68.5	7.2	16.8	6.1	1.5	34.7	21.7	32.3	9.4	2.0
	5	62.9	9.1	20.0	6.5	1.5	28.3	22.8	37.2	9.7	2.0
	10	56.5	10.2	24.4	7.4	1.5	25.4	22.4	40.0	10.4	1.9
	20	53.2	10.5	27.1	7.6	1.6	23.0	21.9	42.6	10.4	2.0
High	0	50.6	8.1	29.3	9.9	2.1	17.7	19.9	46.9	12.9	2.6
	5	42.2	9.4	34.6	11.4	2.3	14.9	18.7	49.3	14.4	2.7
	10	40.7	9.7	36.1	11.3	2.1	13.7	18.6	51.0	14.1	2.5
	20	39.2	9.7	37.4	11.5	2.1	13.1	18.1	52.0	14.3	2.5

Table 2.5. The number of patches in which each conservation feature is represented at least once, in the 'best' solutions for each target/patch size combination.

	Low targets (50 km²)				Medium targets (100 km²)				High targets (150 km²)			
Conservation features	**0 km²**	**5 km²**	**10 km²**	**20 km²**	**0 km²**	**5 km²**	**10 km²**	**20 km²**	**0 km²**	**5 km²**	**10 km²**	**20 km²**
Blanket bog	85	111	90	82	152	145	125	115	93	159	141	113
Calaminarian grassland	46	27	28	20	52	30	28	25	39	33	30	25
Coastal and floodplain grazing marsh	321	318	232	223	578	463	407	370	742	871	776	653
Coastal saltmarsh	219	179	138	129	286	168	159	135	280	220	176	147
Coastal sand dunes	64	64	48	44	101	60	57	50	85	73	56	53
Coastal vegetated shingle	68	53	38	36	70	41	38	38	62	49	41	34
Deciduous woodland	1231	936	660	567	2844	1364	1019	768	3543	2861	1890	1222
Fragmented heath	38	35	42	41	96	88	80	70	67	91	79	75
Good quality semi-improved grassland	479	464	388	345	952	762	713	587	1093	1455	1318	1000
Grass moorland	149	155	135	107	284	250	205	171	207	296	238	195
Limestone pavement	29	25	25	22	26	25	20	18	17	23	20	19
Lowland calcareous grassland	170	182	170	177	355	251	249	248	327	445	428	357
Lowland dry acid grassland	134	139	134	134	363	276	280	250	545	536	475	385
Lowland fens	334	315	260	259	604	424	425	393	680	680	655	581
Lowland heathland	193	203	206	206	379	351	335	304	361	469	443	397
Lowland meadows	295	237	225	236	686	410	406	388	643	708	681	593
Lowland raised bog	25	28	30	33	80	64	61	53	72	75	65	57
Maritime cliff and slope	181	111	86	77	225	108	102	101	190	157	127	112

Mountain heaths and willow scrub	11	11	9	9	6	7	6	6	3	7	6	6
Mudflats	231	206	143	130	337	194	168	143	375	268	190	161
No main habitat but additional habitats present	1009	845	593	499	1952	1222	960	734	2293	2435	1786	1213
Purple moor grass and rush pastures	294	156	140	130	902	520	457	368	593	562	479	389
Reedbeds	376	310	206	196	390	225	221	217	305	259	254	227
Saline lagoons	93	80	51	50	94	58	57	52	75	68	64	54
Traditional orchard	491	324	259	258	1471	685	610	531	2630	2222	1597	1092
Upland calcareous grassland	32	40	40	35	87	71	66	54	51	71	67	56
Upland flushes, fens and swamps	68	84	78	70	205	149	127	109	109	152	131	108
Upland hay meadow	187	137	112	87	169	112	100	81	116	117	103	83
Upland heathland	111	136	121	101	227	222	189	155	174	258	217	169

The percentage of the best Marxan portfolios consisting of the lowest agricultural cost land (based on the planning unit cost data in Marxan, which assigned the lowest cost to all land found within National Parks, AONBs and SSSIs) varied between 78.4% for the low targets and 50.6% for the high targets (Table 2.4a). However, the percentage based on the unmodified cost data varied between 33.4% and 13.1% (Table 2.4b). The MinPatch results showed that as the patch size threshold increased, then the amount of higher quality land in the best portfolios increased too, although the largest changes were in the amount of land within an agricultural quality value of 3 (Table 2.4).

2.4.4 Numbers of patches

The number of patches selected per conservation feature (Table 2.5) in each scenario ranged from 3 for mountain heaths and willow scrub to 3,543 for deciduous woodland. The median number of patches per feature selected in each scenario ranged from 107 for the high target / 20 km^2 patch size scenario, to 284 for the medium target / 0 km^2 patch size scenario.

2.5 Discussion

Systematic conservation planning software tools have been used for freshwater (Davies et al. 2009), marine (JNCC & Natural England 2010; Reecht et al. 2015) and terrestrial (Prendergast 1993; Hopkinson et al. 2000; Franco et al. 2009; Thomas et al. 2013) studies in England. However, the terrestrial analyses have mostly been coarse scale and based on representing a small number of taxonomic groups, and used to investigate conservation theory, such as the role of species surrogates in selecting priority areas, or broad conservation policy, such as the overlap between important areas for biodiversity and carbon, rather than produce national plans of priority areas for conservation to inform action on the ground. This study is the first fine-scale spatial prioritisation analysis for England designed to inform the expansion of the NNR network, as well as understand the trade-offs involved in achieving some of the components of the call for 'bigger, better, more, joined' (Lawton et al. 2010). In this section we first discuss the Marxan and MinPatch results, followed by sections on the trade-offs with patch size, opportunity cost and replication and finishing with a discussion of the wider conservation relevance.

2.5.1 Identifying priority areas with Marxan

The Marxan selection frequency maps indicate that there is relatively little flexibility in where protection should be placed in order to meet the specified targets. Planning units appear to be either selected not at all or most of the time, with little in between, thus suggesting inflexibility in the system. This is not surprising, given the relatively little natural habitat remaining in England, and its highly fragmented condition (Ball et al. 2009).

Many of the areas highlighted by the Marxan outputs overlap with existing designations of National Parks, AONBs and SSSIs, particularly the latter. We deliberately encouraged this overlap by assigning the lowest cost to these existing designations, because it may be more feasible to make these areas into NNRs than other areas not currently under any conservation management (Naidoo et al. 2006). Nonetheless, it suggests that current protection may be at least partly focussed in the right areas, and that the level of protection afforded to these places needs to be strengthened. This finding supports the UK government's recent interest in improving the conservation value of National Parks and AONBs (DEFRA 2019). Currently, development is permitted within virtually all conservation designations in England, with some restrictions (UK Government 2018b). Of the thousands of designated sites in England recorded in the WDPA, all but two are IUCN Protected Area Management Category V or VI, or have no reported category (UNEP-WCMC & IUCN 2018). These classifications are low-restriction, such that a variety of for-profit activities may be permitted within them including tourism, construction, agriculture and forestry (Dudley 2008). Strengthening restrictions on damaging human activity and robust enforcement of the law should be a priority in addition to expanding the network.

2.5.2 *Modifying results with MinPatch*

Our MinPatch analyses demonstrate that it is possible to meet all target and patch size constraints, although flexibility is restricted yet further due to the addition of an extra constraint into the optimisation process (Smith et al. 2010; Metcalfe et al. 2015). The areas selected are similar to the original Marxan outputs, focussing largely on the west towards Wales, south-west, far north-west, the coast of East Anglia and west Kent. However, despite these broad similarities, the units selected by MinPatch are much more dispersed. This is due to how fragmented and scattered habitats are across England. Whereas Marxan simply selected only the areas necessary for meeting the targets, when MinPatch selected patches necessary to meet targets for less widespread habitat types, in doing so, other more widespread habitats with more flexibility were also captured within these patches. Thus MinPatch was prevented from only meeting targets in the easiest places, as Marxan did, producing less spatially concentrated results. While the MinPatch results may look less feasible, given the greater area they cover, their dispersed nature may have positive implications. As the effects of climate change increase, it will be crucial to have accessible pockets of habitat spread throughout the country to allow species to move between them in response to changing conditions (Hodgson et al. 2011). In addition, having conservation areas across the country, rather than focussed in a few particular areas, should be beneficial for people's access to nature, which will also become ever more important as we live increasingly urbanised lives, isolated from nature (Soga & Gaston 2016).

Future work using MinPatch could improve upon our analyses by adjusting the patch size to radius ratio. Our analyses, particularly those with a patch size threshold of 20 km^2, tended to produce narrow and elongated patches (Figure 2.4). NNRs this shape may not be a problem if they are fully incorporated into an effective Nature Recovery Network, such that they are surrounded by other well-managed natural areas. But in general, it is preferable to have conservation areas that are more compact in shape with less edge, because they are likely to be more ecologically resilient and easier to manage (Diamond 1975).

2.5.3 Trade offs with area and cost

Our analyses show that imposing a patch size constraint has a considerable effect. Again this is due to the highly fragmented nature of habitats in England. Many habitats do not exist in large enough patches anywhere to easily satisfy our constraints. Thus, for the low and medium targets, adding a patch size constraint of 5 km^2 increases the area selected by around 50%, while a patch size of 20 km^2 more than doubles it. This effect is less pronounced for the high target scenario, where a patch size of 5 km^2 makes little difference but 20 km^2 increases the area by a third, which is similar to results from other studies (Metcalfe et al. 2015). This is because when the targets are high, large clumps of fragments are already selected which can then be joined by MinPatch without the need to add much extra land.

Marxan consistently selected land with the lowest cost across all scenarios, as this was a major constraint in the optimisation process (Ball et al. 2009). The cost data inputted to Marxan was based on agricultural quality which had been adjusted such that all land within existing conservation areas had the lowest cost regardless of its agricultural quality, an approach that has previously been used to encourage Marxan to select these preferred areas to meet targets where possible (Göke et al. 2018). This result suggests that there is considerable scope for expanding the NNR network within areas that are already protected and on land that is relatively poor quality. The highest proportion of low cost land selected occurs with a patch size of 0 km^2, and increases as the patch size threshold is increased, because adding a new constraint typically leads to greater areas needing to be selected. Thus there is a trade-off to be made between connectivity and resilience of habitat patches, and the amount of potentially valuable agricultural land that would need to be sacrificed (Smith et al. 2010).

We also analysed what proportion of the 'best' portfolios were made up of the different agricultural land quality categories without the adjustment to give conservation areas the lowest value. Here we find that the middle category is selected the most across all the scenarios. This is partly explained by the fact that this is the largest category, containing almost half the area of the study region. It may also be due to existing bias in the NNR network. Low agricultural quality land in England is typically characterised by high elevation, rough terrain and poor soils. These areas are already more likely to be protected (Shwartz et al. 2017), and so Marxan will have less need to select them to meet targets for associated conservation features, e.g. upland bogs, heathlands, and mountainous habitats. In addition, Marxan is strongly discouraged from selecting land that is high agricultural quality because of its high cost. Thus, the selection is pushed towards the middle categories, which contain less well protected features while avoiding the most costly areas. The highest quality land never accounts for more than 2.7% of the area selected in any of the scenarios, though this is also partly explained by its limited extent – the highest category covers only 2.7% of the study region as a whole. Thus the proportion of the area selected that is high quality land is never greater than the proportion of England overall that is high quality land. Increasing the patch size did not significantly increase the proportion of high quality agricultural land selected, while increasing the targets did, although the total area of land required increased with both.

While the greatest proportion of the selected areas are in the middle and lower agricultural quality categories rather than the highest, this still represents a large amount of potentially active farmland, given that the higher categories have a relatively small total extent. However, this is not necessarily as great a problem as it may appear. Overall, farming in England is not especially profitable and relies hugely on government subsidies for survival, while a minority of farms produce the majority of the country's food output on a modest proportion of its agricultural lands (UK Government 2018c). Thus there is real potential for rewilding on a significant scale in England, if agricultural subsidies are reformed, meat consumption is reduced and unsustainable farmland turned over to wildlife management and climate change mitigation activities (Pettorelli et al. 2018; Boyd 2019).

2.5.4 Number of patches for each feature

An essential feature of effective conservation area networks is resilience, so that they are able to withstand natural disasters and maintain their conservation value despite local extinctions (Kukkala & Moilanen 2013). This will only become more crucial in the coming decades as climate change causes more frequent extreme weather events (Araújo & Rahbek 2006). Resilience is achieved through incorporating a degree of redundancy into the system; where there are multiple occurrences of species and habitats distributed throughout a network, they are better able to recover and recolonise following a decline in other areas (Hodgson et al. 2011). The greater the number of patches in which a conservation feature is found, the better its chance of persistence. Despite the importance of redundancy and replication in a network and its inclusion in recommendations for network design (JNCC, 2010) it is rarely included explicitly in conservation planning studies because it is difficult to account for in spatial prioritisation analyses (Moilanen et al. 2009). However, recent software development (Smith 2019b) means there is now scope for *post hoc* assessment of the number of priority areas per conservation feature.

Our analysis was the first to investigate these patterns and, as expected, our results show that higher targets result in conservation features being found in more patches. Similarly, the number of patches generally drops as patch size increases, because when patches are larger, they are more likely to meet and merge into a single patch. However, this effect is a less pronounced for restricted range features; because they are only present in a small number of patches, the number in which they are present in the results remains fairly static. For example, the habitat type 'mountain heath and willow scrub' has a range of just 14.1 km^2 and is found in between 3 and 11 patches across all scenarios. In contrast, the widespread 'deciduous woodland' type has a range of 7,347 km^2 and is found in between 567 and 3,543 patches. Such results put issues related to the destruction of specific habitat patches into perspective, as some habitat types are still very common despite being conservation priorities.

2.5.5 *Wider conservation relevance*

This study is the first fine scale systematic conservation planning analysis designed to inform action on the ground in England, but further refinement is necessary to produce a robust and achievable plan. We were only able to use one set of conservation features and uniform targets in our analyses; future analyses would benefit hugely from availability of useable distribution data and feature-specific targets. In particular, we need fine-scale modelled distribution maps for priority species, ecological processes and ecosystem services (Pett et al, *in press*), and appropriate targets for each based on their remaining extent, life history and vulnerability (Carwardine et al. 2009). However, including more features will almost certainly result in larger areas being selected by the software, because priority habitats are unlikely to be perfect surrogates for priority species (Rodrigues & Brooks 2007).

This study is also the first to use MinPatch in a terrestrial English context and thus to look at the trade-offs between patch size, area selected, opportunity costs, and species representation. The high target and large patch size scenarios result in a significant proportion of the country being selected as a priority for conservation. Realistically, the NNR network is not going to be expanded to cover up to a third of the country in the near future. However, these more ambitious scenarios have potential to inform plans for Nature Recovery Networks (Isaac et al. 2018), whereby conservation activities and improved land management practices are carried out on larger areas of habitat linking core protected sites (Newton et al. 2012). Spatial prioritisation analyses that aim to produce national plans for the expansion of conservation areas need to take into account the size and connectivity of the constituent patches, and how many times features are represented in separate areas in addition to the total extent included in the network. For England in particular, these considerations are crucial for the implementation of the Lawton recommendation of 'more, bigger, better, joined' conservation areas.

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Chapter 3. The contribution of state and non-state conservation areas to meeting biodiversity targets in KwaZulu-Natal, South Africa

3.1 Abstract

The international community is committed to conserving 17% of the terrestrial realm by 2020, but it is widely accepted that this is insufficient and future targets will be higher. However, substantial increase will be impossible through the expansion of state PA networks alone; private and community-owned and managed conservation areas will be vital to help meet more ambitious targets. In addition, these non-state conservation areas may be important for conserving species and habitats that are under-represented in state PAs. Here we present a case study which examines the contribution made by conservation areas with different governance types to the overall network in KwaZulu-Natal province, South Africa. We find that state PAs account for 8%, non-state PAs 2.5%, and voluntary conservation areas 15.5% of the province, and that the non-state PAs and voluntary conservation areas contain more natural vegetation than the state PAs, despite also containing a greater proportion of degraded land. They help meet conservation targets for 15 priority habitats in addition to the 24 whose targets are met by state PAs alone, and represent six priority habitat types that are absent from the state PA system. By developing a new version of the Protection Equality metric that accounts for conservation targets, we also show that adding these non-state PAs and voluntary conservation areas improves Protection Equality scores from 0.5 to 0.7. Thus, our work supports calls to increase support for non-state PAs and land under other effective conservation measures (OECMs) and shows the value of this new approach for measuring Protection Equality that accounts for conservation value.

3.2 Introduction

Protected areas (PAs) are the backbone of biodiversity conservation efforts. In recent years there has been a large increase in the area of land and sea protected (IUCN 2016) as nations work towards their Aichi Target 11 commitments, under which 17% of the terrestrial realm and 10% of the marine realm globally must be protected by 2020 (CBD 2010). However, it is also widely accepted that 17% protection for land will be wholly inadequate to ensure the long-term persistence of biodiversity (Maron et al. 2018; Woodley et al. 2012). Indeed, some conservationists argue that 50% of the Earth should be protected (Kopnina et al. 2018; Wilson 2016; Noss et al. 2012), although others support a more modest expansion (Büscher et al. 2017).

Despite these differing visions for the expansion of the future PA estate, there is general agreement that it cannot happen only on public land (Bingham et al. 2017; Butchart et al. 2015). Target 11 explicitly allows for land to be protected in privately and community PAs and other effective area-based conservation measures (OECMs), in addition to traditional PAs owned and managed by governments. Thus, it leaves plenty of scope for different models of ownership, governance and management to be employed in conservation areas for the benefit of people and nature (Oldekop et al. 2016). Moreover, the potential for non-state PAs is clear: indigenous peoples and communities manage or control over 38 million km^2 of land across the world, much of which has high biodiversity value (Garnett et al. 2018), and while data on private land is much harder to come by, the diversity of governance types for existing private PAs ranging from private individuals through NGOs to corporations suggests that there are many workable options available for private lands too (Bingham et al. 2017).

Current figures on the amount of protected land under non-state governance types are believed to be an underestimate, due in part to poor reporting of these types by many national governments (Protected Planet Report 2016; Langholz &

Krug 2004). Of the 217,155 PAs recorded in the World Database on Protected Areas (WDPA), 84% have state governance, 4.5% private governance, 1.8% shared governance, and 0.6% indigenous and local community governance. Thus, it is possible that the different types of conservation area that exist outside state control could contribute significantly to maintaining biodiversity, but we lack data to understand their current and potential role. Here, we help address this by using a case study from a biodiversity hotspot in South Africa to determine the extent to which different types of non-state conservation areas complement and extend the state PA system.

There may be many benefits to protecting private and communal land, as recent work has shown that non-state conservation areas can be as effective at producing positive conservation outcomes as traditional PAs (Schleicher et al. 2017; Porter-Bolland et al. 2011; Hayes 2006). In addition, non-state conservation areas may help protect neglected species and habitats (Fitzsimons & Wescott 2008). This is because state PAs are often biased in their location, being generally found in more remote and less economically productive areas. This means many species and habitats that occur on valuable, lowland areas are highly threatened (Joppa & Pfaff 2009). Much of this more productive land is not managed by the state, so non-state conservation areas may help protect this neglected biodiversity (Fitzsimons & Wescott 2008). Thus, they may be able to not only increase the global coverage of conservation, but also protect species and habitats that are under-represented in state protected area networks.

Measuring the extent to which a conservation area network represents biodiversity is vital for measuring its conservation value (Kukkala & Moilanen 2013). If a network contains all the key species, habitats and ecological processes (known collectively as "conservation features") found in a region then it is considered representative, whereas many PA networks are unrepresentative because of the biases in their locations (Joppa & Pfaff 2009). Measuring representativeness is often done by simply calculating the percentage of each conservation feature contained in the conservation area network. In addition, these data can be used to measure the protection equality of a conservation area network. Protection equality is based on the Gini co-efficient which measures the inequality of values in a frequency distribution (Barr et al. 2011). The Gini co-efficient was originally developed to measure the distribution of income among a population. When applied to conservation areas, this method shows how equally coverage is spread across different conservation features, producing low Protection Equality scores when some features have a small and some have a high proportion of their area protected, and high scores when the proportions of features' areas protected are similar (Chauvenet et al. 2017).

Measuring representation based on percentage protection is particularly important for identifying conservation features that are completely unprotected or conservation area networks that are highly biased (Rodrigues et al. 2004). However, it is less informative when dealing with less extreme example, as the approach assumes that each conservation feature is equally important and should be equally protected. This is rarely the case, so there is a need to adopt methods that account for differences in their conservation value. One of the most widely used approaches is based on setting targets, which involves quantifying how much of a feature should be conserved to ensure its long-term persistence (Carwardine et al. 2009). Conservation planners use ecological data and expert judgement to set measurable targets for how much of the conservation feature should be found within the network, based on extent for most features but also population size for species with available data (Pressey et al. 2003). Thus, it is better to measure a conservation area network's representativeness based on how equally it meets the targets for each conservation feature, rather than simple percentage conserved, so in this study we use an updated version of the Protection Equality methodology that is target-based.

In this study, we focus on conservation areas in KwaZulu-Natal, a province in South Africa. This is because South Africa has a legislative framework that supports various types of non-state conservation area (Paterson 2010), which has encouraged the development of a number of privately and communally managed conservation areas. These vary in the strictness of land use limitations, the length of time for which they are proclaimed, and the incentives that are provided to landowners to facilitate and encourage protection. In addition, there is comprehensive and up-to-date information on conservation areas in KwaZulu-Natal, as well as fine-scale data on vegetation and land use. Here we: (a) measure the extent to which conservation areas under different management types cover KwaZulu-Natal's different elevation zones and vegetation types, and (b) measure the protection equality of these different conservation area networks using the standard calculations which do not account for conservation targets and a new approach that takes into account existing targets for vegetation type conservation in the province.

3.3 Methods

3.3.1 Study area

KwaZulu-Natal has an area of 95,053 km^2 and is the easternmost province of South Africa, sharing borders with Lesotho, Mozambique and Eswatini (formerly Swaziland), as well as three other South African provinces (Figure 3.1). Its elevation ranges from >3,000 m in the uKhahlamba-Drakensberg Mountains in the west, to sea-level where it meets the Indian Ocean coast in the east. These elevation gradients partly drive its high level of species richness and endemism (Lomolino 2001), which is reflected by its inclusion in the Maputaland-Pondoland-Albany biodiversity hotspot.

The province is largely covered by grassland, savanna woodland, bush thicket and forest, and supports populations of charismatic megafauna such as African bush elephant (*Loxodonta africana*), lion (*Panthera leo*), cheetah (*Acinonyx jubatus*), black rhinoceros (*Diceros bicornis*), white rhinoceros (*Ceratotherium simum)* and African wild dog (*Lycaon pictus*) (Di Minin et al. 2013) as well as a large number of endemics, particularly in its Important Bird and Biodiversity Areas (IBAs). For example, Maloti Drakensberg, on the western edge of the province, is part of the Lesotho highland Endemic Bird Area and holds populations of many endangered and range-restricted bird species such as the Cape vulture (*Gyps coprotheres*), Drakensberg rockjumper (*Chaetops aurantius*) and Drakensberg siskin (*Serinus symonsi*). Hluhluwe-iMfolozi, another IBA, is one of the most important sites for mammal conservation in South Africa due to its large populations of ungulates and their predators, and also supports more than 400 bird species (BirdLife 2020).

There is a strong emphasis on conservation tourism in the province's parks and reserves, but it has also undergone significant development and transformation for commercial agriculture and forestry plantations (Fairbanks and Benn 2000).

3.3.2 Conservation area types

We considered three broad types of conservation area in our analysis (Figure 3.1):

1. State PAs: these are legally gazetted conservation areas that are owned and managed by government authorities with a long-term commitment to conservation.
2. Non-state PAs: this comprises three different stewardship designations that can be applied to private and communal land parcels. The first, 'nature reserve', has the same legal status, safeguards and potential access to funding as state PAs. Generally, they also involve a long-term commitment, being protected for a period of 99 years or in perpetuity. The second, 'protected environment', is also legally gazetted, but can be protected for any length of time and has fewer financial incentives available. The third, 'biodiversity management agreement', has no legal safeguards or financial incentives. It is simply an agreement made by the

landowner with the relevant provincial authority to manage their land for conservation.

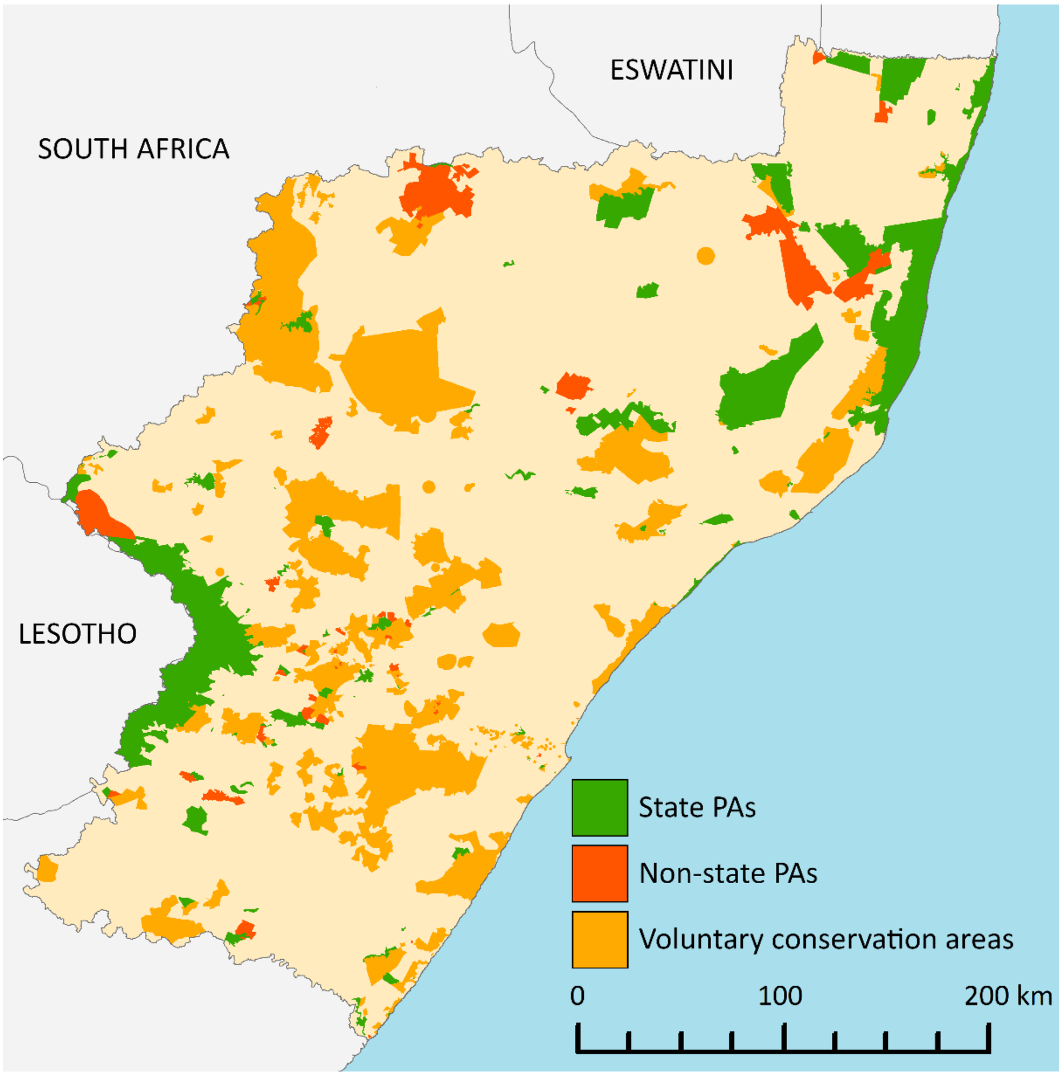
3. Voluntary conservation areas: this comprises conservancies and community conservation areas (CCAs), both of which are voluntary and have no legal safeguards or access to government funding.

We excluded game ranches because of their small extent in the province and their uncertain conservation value (Cousins 2008). State PAs were considered the highest designation, then non-state PAs, then voluntary conservation areas. This is a value judgement based on the fact that state PAs tend to be the most secure and potentially most effective for conservation, with non-state PAs somewhat less so, while voluntary conservation areas have no long-term security (Paterson 2009). Where different conservation designations overlapped, we assigned the higher designation.

3.3.3 Data collection and preparation

Our analysis was based on measuring the area conserved of each of 4 elevation zones and 101 vegetation types (Table 3.1), referred to hereafter as "conservation features". The conservation area boundary polygons were provided by the provincial conservation authority Ezemvelo KwaZulu-Natal Wildlife (EKZNW) and we used ArcGIS 10.5 (ESRI 2016) for all the initial data preparation. Our elevation zone map was produced by clipping the Shuttle Radar Topography Mission's 1 km elevation dataset to the KwaZulu-Natal province boundary and then reclassifying it into four zones of 0-500 m, >500-1000 m, >1000-1500 m and >1500 m.

Figure 3.1. Map of conservation areas in KwaZulu-Natal.



The vegetation type data was based on South Africa's National Vegetation Map (Government of South Africa 2009) a vector GIS layer that divides KwaZulu-Natal into 101 potential vegetation types. To show the actual distribution of these vegetation types, thereby accounting for the land clearance that has removed natural vegetation in many parts of the province, we used data from the EKZNW landcover map, a 20 m resolution raster GIS layer that divides the province into 46 types (EKZNW 2013). We did this based on two scenarios: one which included only land that is in a natural state, and one that also included land degraded by human action that has the potential to recover. Land that has been entirely transformed by human action was excluded from both scenarios. We assigned each landcover type to one of these three categories based on the following: (1) 'natural' landcover types that are intact and undisturbed, although not necessarily pristine, e.g. grasslands and woodlands; (2) 'degraded' types that have experienced some human interference but subsequently recovered, or have the potential to do so in future, e.g. old plantations and rehabilitated mines, and (3) 'transformed' types that have undergone major development and can no longer be considered in any way natural, e.g. settlements and agricultural land (Scott-Shaw & Morris 2015). We intersected these three landcover categories with the map of vegetation types using the ArcGIS Raster Calculator and created two different layers for our scenarios: one 'natural only' and one 'natural + degraded'. The entire extent of one vegetation type was completely transformed and therefore it was not included in the analyses as a conservation feature.

Table 3.1: The 101 vegetation types included in this study, based on the Government of South Africa's National Vegetation Map, with each type's total extent, conservation target, and threat status.

Vegetation type	**Total area (km^2)**	**Target (%)**	**Target area (km^2)**	**Conservation status**
Alluvial Wetlands : Subtropical Alluvial Vegetation : Lowveld Floodplain Grasslands	229.3	31	71.1	Critically Endangered
Alluvial Wetlands : Temperate Alluvial Vegetation : Midland Alluvial Woodland & Thicket	2.1	24	0.5	Critically Endangered
Delagoa Lowveld	87.7	19	16.7	Critically Endangered
Eastern Scarp Forests : Ngome-Nkandla Scarp Forest	85.9	61.61	52.9	Critically Endangered

Eastern Scarp Forests : Northern Coastal Scarp Forest	56.3	61.61	34.7	Critically Endangered
Freshwater Wetlands : Subtropical Freshwater Wetlands : Coastal Lakes & Pans : Lacustrine	0.0	24	0.0	Critically Endangered
KwaZulu-Natal Coastal Belt Grassland	4,115.5	25	1,028.9	Critically Endangered
KwaZulu-Natal Coastal Forests : Dukuduku Moist Coastal Lowlands Forest	84.7	71.69	60.7	Critically Endangered
KwaZulu-Natal Coastal Forests : Southern Mesic Coastal Lowlands Forest	107.1	71.69	76.8	Critically Endangered
KwaZulu-Natal Coastal Forests : Southern Moist Coastal Lowlands Forest	31.7	71.69	22.8	Critically Endangered
KwaZulu-Natal Dune Forests : East Coast Dune Forest	25.0	69.2	17.3	Critically Endangered
KwaZulu-Natal Sandstone Sourveld	1,797.4	25	449.3	Critically Endangered
Lowveld Riverine Forest	100.7	100	100.7	Critically Endangered
Mangrove Forests	25.2	100	25.2	Critically Endangered
Pondoland-Ugu Sandstone Coastal Sourveld	372.8	30.31	113.0	Critically Endangered
Swamp Forests : Barringtonia Swamp Forest	0.9	100	0.9	Critically Endangered
Swamp Forests : Ficus trichopoda Swamp Forest	77.2	100	77.2	Critically Endangered
Swamp Forests : Raphia Swamp Forest	3.7	100	3.7	Critically Endangered

Swamp Forests : Voacanga thouarsii Swamp Forest	4.6	100	4.6	Critically Endangered
Zululand Coastal Thornveld	670.8	19	127.5	Critically Endangered
Alluvial Wetlands : Subtropical Alluvial Vegetation	170.8	31	53.0	Endangered
Alluvial Wetlands : Subtropical Alluvial Vegetation : Lowveld Floodplain Grasslands : Short Grass/ Sedge Wetlands	76.0	31	23.6	Endangered
Eastern Mistbelt Forests	445.3	66.5	296.1	Endangered
Granite Lowveld	36.5	19	6.9	Endangered
KaNgwane Montane Grassland	82.3	24	19.8	Endangered
KwaZulu-Natal Coastal Forests : Maputaland Dry Coastal Lowlands Forest	24.0	71.69	17.2	Endangered
KwaZulu-Natal Coastal Forests : Maputaland Mesic Coastal Lowlands Forest	89.6	71.69	64.2	Endangered
KwaZulu-Natal Coastal Forests : Maputaland Moist Coastal Lowlands Forest	136.4	71.69	97.8	Endangered
KwaZulu-Natal Dune Forests : Maputaland Dune Forest	164.4	69.2	113.8	Endangered
Lebombo Summit Sourveld	117.5	24	28.2	Endangered
Mabela Sandy Grassland	231.7	23	53.3	Endangered
Maputaland Coastal Belt	2,209.5	25	552.4	Endangered
Marine Saline Wetlands	17.6	24	4.2	Endangered
Midlands Mistbelt Grassland	5,478.2	23	1,260.0	Endangered

Moist Coast Hinterland Grassland	4,377.1	25	1,094.3	Endangered
Alluvial Wetlands : Subtropical Alluvial Vegetation : Lowveld Floodplain Grasslands : Tall Reed Wetland	25.7	31	8.0	Vulnerable
Alluvial Wetlands : Temperate Alluvial Vegetation	1,498.2	24	359.6	Vulnerable
Dry Coast Hinterland Grassland	2,765.8	25	691.4	Vulnerable
East Griqualand Grassland	2,152.4	23	495.0	Vulnerable
Freshwater Wetlands : Eastern Temperate Wetlands	548.0	24	131.5	Vulnerable
Freshwater Wetlands : Subtropical Freshwater Wetlands	140.7	24	33.8	Vulnerable
Freshwater Wetlands : Subtropical Freshwater Wetlands : Short Grass/ Sedge Wetlands : Dune Slack	2.8	24	0.7	Vulnerable
Income Sandy Grassland	4,381.1	23	1,007.6	Vulnerable
KwaZulu-Natal Coastal Belt Thornveld	1,119.6	25	279.9	Vulnerable
Mooi River Highland Grassland	2,672.3	23	614.6	Vulnerable
Northern KwaZulu-Natal Moist Grassland	6,977.2	24	1,674.5	Vulnerable
Northern Zululand Mistbelt Grassland	529.0	23	121.7	Vulnerable
Paulpietersburg Moist Grassland	2,841.2	24	681.9	Vulnerable
Southern KwaZulu-Natal Moist Grassland	2,321.1	23	533.8	Vulnerable
Western Maputaland Clay Bushveld	1,525.5	19	289.8	Vulnerable

Zululand Lowveld	6,655.3	19	1,264.5	Vulnerable
Alluvial Wetlands : Temperate Alluvial Vegetation : Midland Floodplain Grasslands	17.9	24	4.3	Least Threatened
Amersfoort Highveld Clay Grassland	131.9	27	35.6	Least Threatened
Basotho Montane Shrubland	26.4	28	7.4	Least Threatened
Drakensberg Afroalpine Heathland	62.6	27	16.9	Least Threatened
Drakensberg Foothill Moist Grassland	3,849.5	23	885.4	Least Threatened
Drakensberg Montane Forests	64.2	63.5	40.8	Least Threatened
Drakensberg-Amathole Afromontane Fynbos	14.3	27	3.9	Least Threatened
Eastern Free State Sandy Grassland	39.0	24	9.4	Least Threatened
Eastern Scarp Forests : Northern Zululand Lebombo Scarp Forest	76.5	61.61	47.1	Least Threatened
Eastern Scarp Forests : Southern Coastal Scarp Forest	113.8	61.61	70.1	Least Threatened
Eastern Valley Bushveld	3,138.8	25	784.7	Least Threatened
Freshwater Wetlands : Drakensberg Wetlands	57.7	24	13.8	Least Threatened
Freshwater Wetlands : Eastern Temperate Wetlands : Lakes & Pans	0.5	24	0.1	Least Threatened
Freshwater Wetlands : Lesotho Mires	0.0	24	0.0	Least Threatened
Freshwater Wetlands : Subtropical Freshwater Wetlands : Coastal Lakes & Pans	75.9	24	18.2	Least Threatened
Freshwater Wetlands : Subtropical Freshwater	69.9	24	16.8	Least Threatened

Wetlands : Coastal Lakes & Pans : Endorheic				
Freshwater Wetlands : Subtropical Freshwater Wetlands : Short Grass/ Sedge Wetlands	469.7	24	112.7	Least Threatened
Freshwater Wetlands : Subtropical Freshwater Wetlands : Short Grass/ Sedge Wetlands : Coastal Plain Depression	7.8	24	1.9	Least Threatened
Freshwater Wetlands : Subtropical Freshwater Wetlands : Tall Grassland/ Sedge/ Reed Wetlands	147.9	24	35.5	Least Threatened
Inland Saline Wetlands : Subtropical Salt Pans	25.6	24	6.1	Least Threatened
Inland Saline Wetlands : Subtropical Salt Pans : Floodplain Pans (Open)	21.0	24	5.0	Least Threatened
Inland Saline Wetlands : Subtropical Salt Pans : Rain fed (Endorheic) Pans (Closed)	5.4	24	1.3	Least Threatened
Ithala Quartzite Sourveld	820.1	27	221.4	Least Threatened
KwaZulu-Natal Highland Thornveld	5,009.4	23	1,152.2	Least Threatened
KwaZulu-Natal Hinterland Thornveld	1,526.1	25	381.5	Least Threatened
Lesotho Highland Basalt Grassland	9.5	27	2.6	Least Threatened
Licuati Sand Forests : Eastern Sand Forest	254.5	69	175.6	Least Threatened
Licuati Sand Forests : Western Sand Forest	9.1	69	6.3	Least Threatened
Low Escarpment Moist Grassland	1,339.1	23	308.0	Least Threatened
Makatini Clay Thicket	323.0	19	61.4	Least Threatened

Maputaland Pallid Sandy Bushveld	613.5	25	153.4	Least Threatened
Maputaland Wooded Grassland	1,077.9	25	269.5	Least Threatened
Marine Saline Wetlands : Saline Grassland & Mud Flats	42.1	24	10.1	Least Threatened
Marine Saline Wetlands : Saline Reed & Sedge Beds	9.6	24	2.3	Least Threatened
Muzi Palm Veld and Wooded Grassland	528.6	25	132.2	Least Threatened
Northern Drakensberg Highland Grassland	708.2	27	191.2	Least Threatened
Northern Zululand Sourveld	4,702.5	19	893.5	Least Threatened
Pondoland Scarp Forests	48.9	61.61	30.1	Least Threatened
Southern Drakensberg Highland Grassland	898.6	27	242.6	Least Threatened
Southern Lebombo Bushveld	1,164.6	24	279.5	Least Threatened
Subtropical Coastal Lagoons : Estuary	400.4	24	96.1	Least Threatened
Subtropical Dune Thicket	12.6	20	2.5	Least Threatened
Subtropical Seashore Vegetation	6.9	20	1.4	Least Threatened
Swaziland Sour Bushveld	505.1	19	96.0	Least Threatened
Tembe Sandy Bushveld	1,105.5	19	210.0	Least Threatened
Thukela Thornveld	2,161.1	25	540.3	Least Threatened
Thukela Valley Bushveld	2,686.8	25	671.7	Least Threatened
uKhahlamba Basalt Grassland	1,203.5	27	324.9	Least Threatened
Wakkerstroom Montane Grassland	1,316.2	27	355.4	Least Threatened
Western Maputaland Sandy Bushveld	151.2	19	28.7	Least Threatened

3.3.4 Data analysis

To measure the extent to which each conservation feature is represented in the three types of conservation area, we used the CLUZ plugin (Smith 2019) for QGIS (QGIS 2016). Firstly, we took the EKZNW planning unit layer, which is used in their conservation planning system and includes data on the boundaries of the state PAs, and used the Union function in ArcGIS to combine it with the other conservation area boundaries to produce our final version of the planning unit layer. We then used the Tabulate Area tool to calculate the area of each elevation zone and vegetation type in a natural or natural/degraded condition in each planning unit, and imported the conservation feature data into CLUZ. We set the representation targets for each feature as those used by EKZNW, which range between 19% and 100% of the original extent of the vegetation types and are based on South African national legislation (Jewitt 2009) and the species-area curve relationship target-setting method presented in Desmet & Cowling (2004). Using CLUZ, we then calculated the amount of each conservation feature in the set of state PAs, non-state PAs and voluntary conservation areas. We also calculated the amount in state PAs + non-state PAs, and state PAs + non-state PAs + voluntary conservation areas to determine the extent to which these combined sets of conservation areas met the targets.

We calculated the protection equality scores using the ProtectEqual package (Chauvenet et al. 2015) in R 3.3.3 (R Core Team 2015), to assess representation in the different conservation area types, both in terms of proportion protected and proportion of the representation target met. We calculated three protection equality scores for the 'natural only' scenario: protection equality based on the absolute proportion protected of each feature; protection equality scores based on each feature's percentage target met, and protection equality scores based on each feature's percentage target met capped at 100%. The purpose of the cap is to reduce the influence of those features that are significantly over-represented. For example, if a conservation feature has a range of 1000 km^2 and a representation target of 30% (i.e. 300 km^2), but 600 km^2 is contained within conservation areas, then the proportion of its target met is 200%. This may lead to a low protection equality scores, even if all the targets have been met, and hence appear unnecessarily negative. We calculated the scores based on both uncapped and capped percentage target met because this target-based approach is new and we wanted to investigate the effect of using the raw data as well as the modified data.

3.4 Results

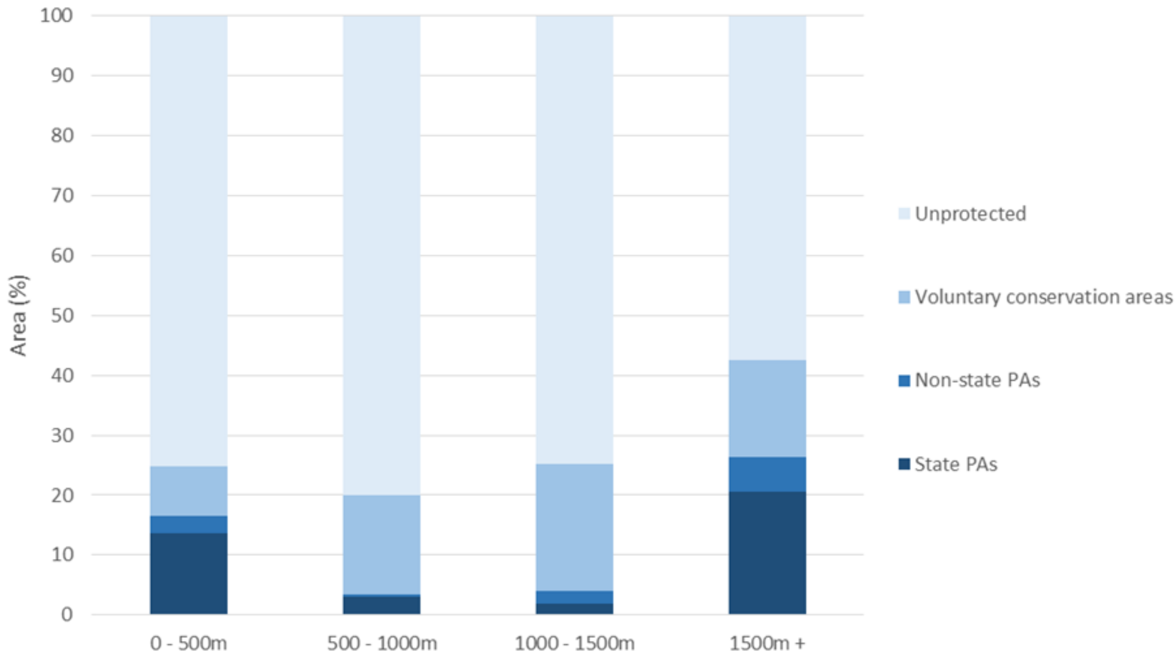
3.4.1 Total conservation area coverage

KwaZulu-Natal has 25,334 km^2 of land under some kind of conservation designation; this amounts to just over a quarter of the total land area of the province. State PAs account for 8%, non-state PAs 2.5%, and voluntary conservation areas 15.5%. The conservation areas vary greatly in size, from 0.02 km^2 to over 3,200 km^2. State PAs are largely confined to the north-east and the south-west around the uKhahlamba-Drakensberg Mountains on the border with Lesotho. The other conservation area types tend to be more evenly distributed. In particular, voluntary conservation areas cover a considerable area (nearly twice that of state PAs) and provide significant coverage across the centre and north-west of the province (Figure 3.1).

3.4.2 Conservation area coverage by elevation

Between 20 and 25% of the lower three elevation zones is protected, increasing to over 40% at 1500 m and above. Relatively little land is covered by state PAs in the 500-1000 m and 1000-1500 m zones, at 3.0% and 1.8% respectively. Meanwhile, 13.5% of the lowest elevation zone and 20.6% of the highest elevation zone is contained within state PAs. Non-state PAs show a similar pattern to the state PAs, with greatest coverage in the lowest and highest elevation zones, while voluntary conservation areas are more evenly spread across all elevation zones (Figure 3.2).

Figure 3.2. The percentage of each elevation zone protected/unprotected by the three types of conservation area.



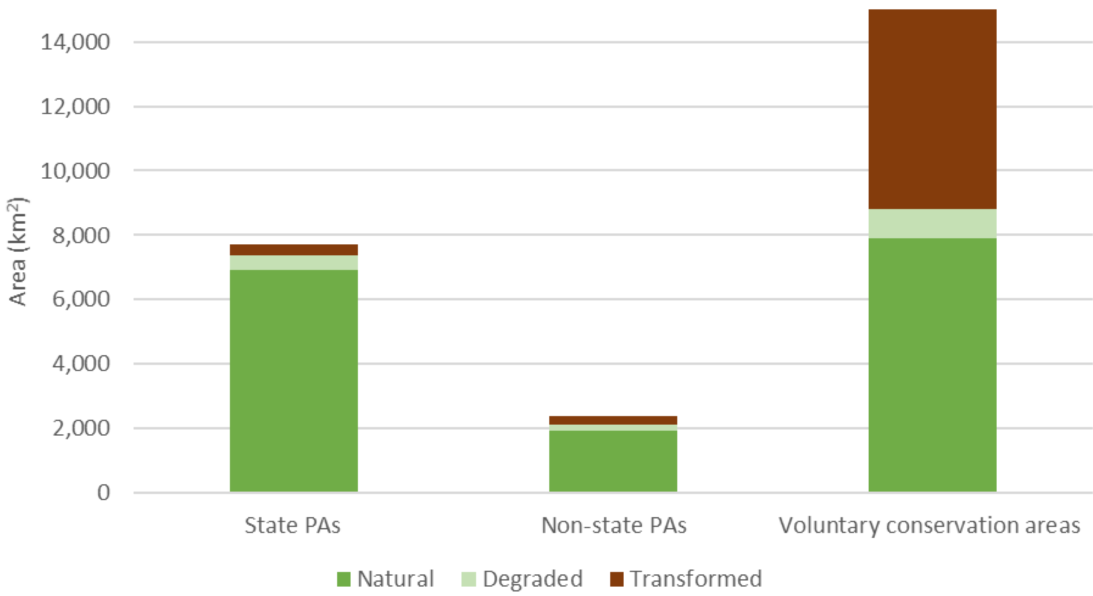
3.4.3 Habitat condition in conservation designations

Fifty-eight percent of all the land in KwaZulu-Natal is in a broadly natural state, while 9% is degraded and 33% transformed. The mean proportion of transformed land per vegetation type is 24.8% but varies greatly, with 'Freshwater Wetlands: Subtropical Freshwater Wetlands: Coastal Lakes & Pans: Lacustrine' being entirely transformed and others types, such as 'Freshwater Wetlands: Lesotho Mires', being entirely untransformed.

Across all conservation area types, 45% of the land is natural, 5% is degraded and half is transformed. However, the proportions in different conservation area types vary considerably. In state PAs, 89% of vegetation is natural. Non-state PAs also contain a high proportion of natural vegetation, at 80%. Voluntary conservation areas contain a considerably lower proportion of natural vegetation, at just 52%. They also have by far the highest proportion of transformed land, at 42%, compared to 4% and 12% for state and non-state PAs respectively.

Nonetheless, voluntary conservation areas cover more natural vegetation than any other conservation designation because of their much larger extent. Almost 8,000 km^2 of natural vegetation is contained within voluntary conservation areas, while state PAs cover just under 7,000 km^2 and non-state PAs nearly 2,000 km^2 (Figure 3.3).

Figure 3.3. The amount of land within the conservation areas that is natural, degraded or transformed.



3.4.4 *Representation of vegetation types in conservation designations*

In the 'natural only' scenario, state PAs alone met targets for 24 of the 101 vegetation types (Table 3.2). The median level of protection provided by state PAs is 37.4%, while 24 targets are met and 13 vegetation types are not represented at all. In state PAs + non-state PAs, the median is 43.6% and 26 targets are met while the number unmet falls to 11. In all conservation designations combined, the median rises to 69.5% and 38 targets are met, leaving 63 unmet. Seventeen features are significantly over-represented, their target having been met between two and four times over. Seven vegetation types are not found anywhere in the conservation area network.

Adding degraded areas to the analysis in the 'natural + degraded' scenario does not substantially improve the results. Across all the conservation area types, only one new target is met and no previously unrepresented features are now represented in the network. The average target shortfall is slightly reduced (Table 3.3).

Table 3.2: The number of targets that are entirely met, partially met, or entirely unmet, in conservation areas, when either only natural areas are considered, or when both natural and degraded areas are considered.

	State PAs		State PAs and non-state PAs		State PAs, non-state PAs and voluntary conservation areas	
	Natural only	Natural + degraded	Natural only	Natural + degraded	Natural only	Natural + degraded
Targets met	24	24	26	26	38	39
Targets 50-100% met	21	24	21	24	26	31
Features not represented at all	13	13	11	11	7	7

Table 3.3. The average percentage by which targets are missed in conservation areas, when either only natural areas are counted, or when both natural and degraded areas are counted.

	Natural only	Natural + degraded
State PAs	54.4	52.8
State PAs and non-state PAs	51.1	49.3
State PAs, non-state PAs and voluntary conservation areas	34.4	31.0

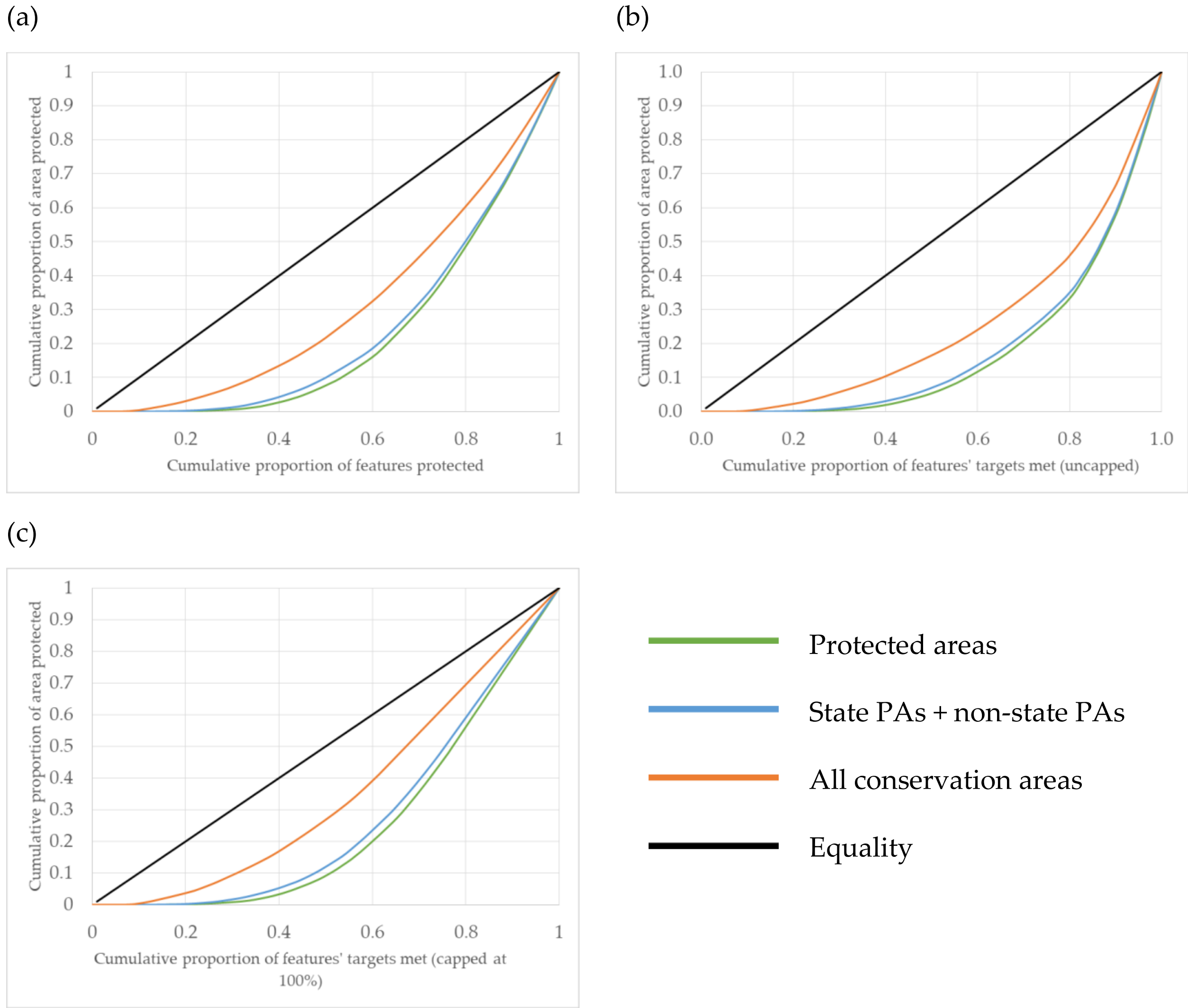
Table 3.4. Protection equality scores for the conservation areas, when no targets are used, when targets are used and the proportion of each feature's target that is met is not capped, and when targets are used and the proportion met is capped at 100%. Perfect equality would result in a score of 1.

	Non-target based	Target-based; no cap	Target-based; 100% cap
State PAs	0.44	0.35	0.50
State PAs and non-state PAs	0.47	0.37	0.53
State PAs, non-state PAs and voluntary conservation areas	0.62	0.50	0.70

3.4.5 Protection equality scores

The protection equality score (Table 3.4), calculated using the proportion of the features conserved, was 0.44 for state PAs alone and 0.62 for all conservation designations combined (Figure 3.4a). When we calculated the protection equality scores based on the proportion of the representation targets that had been met, equality was reduced. State PAs alone scored 0.35, while across the network the score was 0.50 (Figure 3.4b). After capping the proportion of target met at 100% to reduce the influence of over-represented features, the protection equality scores improved to 0.50 for state PAs alone and 0.70 for the overall conservation area network (Figure 3.4c).

Figure 3.4. Protection equality graphs for the conservation areas, when (a) no targets are used, (b) targets are used but without capping the proportion of each feature's target that is met, and (c) targets are used and the proportion met is capped at 100%.



3.5 Discussion

If we are to understand the potential role of non-state conservation areas for conserving global biodiversity then we need to measure the extent to which they complement the state PA network (Watson et al. 2016). Such studies are relatively rare because many countries lack data on the boundaries of these other types of conservation area, and have not translated their conservation goals into specific targets. This makes our analysis one of the first to answer this important question using data from KwaZulu-Natal in South Africa. In this section we begin by discussing how the non-state conservation areas add to the total extent and representativeness of the conservation network. We then discuss our protection equality analysis, and finish by discussing the policy implications of our work.

3.5.1 Area and representation

Across the world, non-state land plays an important role in extending the area of land protected for biodiversity. For example, 7% of Tanzania is covered by community wildlife management areas (Lee & Bond 2018), 3.1% of Guyana by community conservation areas (Bicknell et al. 2017), 2.0% of the USA by conservation easements (Adams & Moon 2013) and 1.2% of Australia by private PAs (Fitzsimmons 2015). The designations available for conservation on private or communal lands vary enormously by country, and these examples may be biased towards countries that have extensive and/or successful conservation area networks outside of state lands. Nonetheless, they suggest that there is willingness across the world among non-state actors to manage their own land for conservation.

South Africa is relatively unusual in having significant conservation initiatives operating on private and communal lands (Paterson 2010), and this is reflected in our results for KwaZulu-Natal. State PAs, non-state PAs and voluntary conservation areas collectively cover 26.3% of KwaZulu-Natal's land area. This figure is more than triple the 8% found in state PAs alone, but caution is needed when interpreting this result. This is because more than 40% of the voluntary conservation areas are covered by transformed land, which is mostly driven by the network of conservancies that contain large areas of farmland. Thus, a more relevant figure is the percentage of the province covered by conservation areas that contains natural and natural + degraded habitat. This is 17.0% and 18.6% respectively, which is still much greater than the percentages reported for other countries. This is partly because some of the conservation areas included in this analysis would probably not currently qualify as PAs or OECMs. Nonetheless, the area covered by non-state conservation areas is considerable and has real potential to make a major contribution to the province's conservation area network. In particular, despite containing a large area of degraded land, nonetheless voluntary conservation areas cover more land in a natural condition than all state PAs.

Another potential benefit of these non-state conservation areas is that they could conserve under-represented biodiversity elements. This is because state PAs are often biased towards higher elevation zones and other land with lower economic value (Joppa & Pfaff 2009; Hoekstra et al. 2005). This pattern is evident in KwaZulu-Natal, though not to the same degree found in some studies of networks in other countries (Schwartz et al. 2017; Oldfield et al. 2004). Coverage is highest in the lowest and highest of the four elevation zones, which can be explained by the presence of three large state PAs: Hluhluwe–iMfolozi Park and iSimangaliso Wetland Park on the east coast at low elevation, and the uKhahlamba Drakensberg Park in the Drakensberg Mountains at high elevation. This relatively high coverage by state PAs in the lowest elevation zone is unsurprising because the most valuable and productive farmland in KwaZulu-Natal is present in the middle elevation zones, rather than at the lowest levels on the eastern coast. Thus, our study suggests that economic value is a more important factor than high elevation when determining the location of state PAs (Joppa & Pfaff 2009). This also explains why the voluntary conservation areas play an important role in representing these middle elevation zones, increasing conservation area coverage across the two central zones from 7.5% to 37.8% compared to state PAs + non-state PAs, as many of these areas are conservancies that are often owned and managed by farmers. In addition to cultivated land, voluntary conservation areas cover many grassland habitat types, including threatened KwaZulu-Natal Mistbelt Grasslands IBA sites (BirdLife 2020).

Non-state PAs and voluntary conservation areas also improve network representativeness at a finer biodiversity scale. Results from our 'natural only' scenario showed that targets for 24 of 101 features are met in state PAs alone, increasing to 38 across all conservation areas. This is similar to a study on private conservation areas in the Little Karoo region of South Africa, which made a major contribution to biodiversity representation by tripling the number of targets met (Gallo et al. 2009). Likewise, the number of features that are not represented at all is 13 in state PAs but decreases to 7 across all conservation areas. These missing 7 mostly have quite restricted distributions; features with a smaller area are more likely to be either entirely protected or entirely unprotected. Their average proportion of area that has been transformed is also twice the overall average of all the features, at 49.8%. Thus their targets will be more difficult to meet.

Future studies could improve on our analysis by including species data to explicity assess how well the different governance types represent species, as vegetation types may not be effective surrogates for them (Rodrigues & Brooks 2007). However, we already know that at least some of the non-state and voluntary conservation areas are important for species. For example, Umvoti Vlei conservancy in central KwaZulu-Natal is one of several conservancies that cover designated IBAs. In winter, its wetland areas support large numbers of wattled crane (*Grus carunculatus*), classified by IUCN as vulnerable, and grey crowned crane (*Balearica regulorum*), classified as endangered. Its grassland areas also support a number of vulnerable and endangered species including the southern bald ibis (*Geronticus calvus*), blue crane (*Grus paradisea*) and black harrier (*Circus maurus*) (BirdLife 2020; IUCN 2020). Although more state PAs in KwaZulu-Natal have been designated as IBAs than the other governance types, this may be partly due to sampling bias in species surveys towards areas already seen as conservation priorities (Reddy and Dávalos 2003).

3.5.2 Protection equality

Setting quantitative targets is vital to the systematic conservation planning process because it provides transparency and allows outcomes to be measured objectively (Carwardine et al. 2009). Further, it allows the relative importance of different conservation features to be reflected in decision-making (Groves & Game 2016). This is also illustrated by our protection equality analyses. Studies thus far have analysed how evenly protection is distributed across the area of conservation features (Barr et al. 2011; Shwartz et al. 2017; Chauvenet et al. 2017). These analyses did not include conservation targets, thus implicitly assuming that all features are of equal importance and under the same level of threat. Yet we know this is not the case. Species and habitats vary enormously in their remaining extent, biodiversity value and vulnerability to future destruction (i.e. their irreplaceability; Margules & Pressey 2000). Therefore, conservationists' immediate priority should not be to achieve absolute equality of protection for all features, but levels of protection which take into account features' characteristics and context.

In this study, we conducted an analysis with the same assumption of equal value, but also two further analyses which incorporated each feature's representation target. In the first of these three analyses, we simply measured equality in the proportions of each conservation feature protected. In the second, we measured equality in the proportions of the targets met, i.e. we allowed for differences in the conservation value of the different features. In the final analysis, we again measured equality in the proportions of the targets met, but capped the proportion of the target met at 100%. This sought to reduce the influence of features whose targets have been met many times over (for example, some had a proportion met of nearly 500%). Over-representation of features is not ideal but may simply be due to the features having a restricted extent rather than solely to poor planning.

We found that the non-target based protection equality score for state PAs alone is 0.44, which is better than, for example, England's PA network which scores 0.32 for protection ecoregions (Schwartz et al. 2017). It is also comparable to protection equality scores calculated for ecoregion protection in six of the world's largest countries, which range between 0.14 and 0.46 (Chauvenet et al. 2017). When non-state PAs are added to the non-target based protection equality analysis, this increases to 0.47, while including voluntary conservation areas as well brings the score to 0.62. These are broadly comparable to protection equality scores found for England's broader conservation area network (Shwartz et al. 2017).

The target based, uncapped protection equality scores for states PAs alone was 0.35, which increases to 0.37 and 0.50 when non-state PAs and voluntary conservation areas are added, respectively. These scores appear worse than the non-target based score, as expected due to feature that are overrepresented in the network. Finally, the target based protection equality score when capped at 100% is 0.50 for state PAs alone, and increases to 0.53 when non-state PAs are added and to 0.70 when voluntary conservation areas are added as well. These scores are an improvement on both previous sets, which shows that taking in account the relative importance of conservation for different features indicates that KwaZulu-Natal's conservation area network is doing better than a simple non-target based protection equality analysis would suggest. It also provides further evidence that non-state PAs and voluntary conservation areas can make a substantial contribution to the representativeness of the overall network (Shwartz et al. 2017).

3.5.3 Policy implications

Our study demonstrates that non-state PAs and voluntary conservation areas have the potential to contribute hugely to KwaZulu-Natal's conservation area network in terms of both area covered and representativeness, provided that they are managed effectively to ensure positive outcomes for biodiversity

(Geldmann et al., 2019). This is not the case currently, because while non-state PAs are required to have management plans as part of their proclamation under the National Environmental Management Protected Areas Act, voluntary conservation areas are not required to do so and there is substantial variation in the quantity and quality of conservation activities carried out within them. Ensuring that voluntary conservation areas are providing the maximum possible benefit for conservation would require the compilation of detailed inventories of natural areas and natural assets that remain within them and the development of management plans, as well as plans for species re-introduction and rehabilitation where necessary (A. Armstrong; pers. comm.). It is also important to note that if this is achieved, they could qualify as other effective area-based conservation measures (OECMs), which IUCN define as: *"a geographically defined space, not recognised as a protected area, which is governed and managed over the long-term in ways that deliver the effective in-situ conservation of biodiversity, with associated ecosystem services and cultural and spiritual values"* (IUCN WCPA 2018) and therefore count towards South Africa's national contribution to Aichi Target 11.

Given the current consensus that much higher, more ambitious targets are needed for the expansion of the global PA estate post-2020, OECMs have a vital role to play in achieving them (Watson et al. 2016). Defining, recognising and supporting areas outside the formal network that nonetheless deliver tangible benefits for biodiversity will be a valuable tool in the effort to overcome the significant practical and political obstacles to PA expansion (Dudley et al. 2018). PAs are at times controversial because of economic, social and human rights concerns (Oldekop et al. 2016; Brockington & Wilkie 2015). Our study shows that OECMs could play an important role for conservation in South Africa, allowing the recognition of existing conservation efforts, as well as the expansion of effective conservation that potentially provokes less opposition from those who see PAs as exclusionary or socially damaging.

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Chapter 4. Developing a conceptual framework to understand the drivers of conservation area network expansion

4.1 Abstract

The global conservation area network has seen rapid expansion in recent years, but a further significant increase in the coming years will be necessary is we are to address the current extinction crisis. Public and scientific debate on how to drive the creation of new conservation areas is often centred on calls for increased funding and the need for further research. In addition, there are many studies that describe other direct and indirect factors that influence conservation area establishment. However, there is a need to understand how these different elements combine and interact, so here we present a conceptual framework based on a synthesis of the literature to understand what determines the relative extent of conservation area networks at the national level. We identify a range of factors linked to motivation (intrinsic value, rates of loss, human population density, socio-economic values and political ideology) and capacity (financial, legal, technical, and operational), discussing how these influence and interact to drive the creation of state-, privately- and communally-managed protected areas, as well as land under other effective conservation measures (OECMs). We also give suggestions on how these factors are likely to influence future growth, highlighting the conditions that should be encouraged to create an enabling environment for the growth of conservation areas.

4.2 Introduction

Protected areas are the cornerstone of efforts to stem the rapid loss of biodiversity now occurring across the world (Watson et al. 2014). Recent decades have seen a rapid and significant expansion of the global conservation area estate, such that 14.9% of the terrestrial realm is now protected by law (Lewis et al. 2019). However, under the Convention on Biological Diversity Aichi Target 11, at least 17% of global land area must be protected by 2020 (CBD 2010). Moreover, as the deadline for the completion of this target approaches, conservationists and policymakers are debating what our next steps should be. There is a strong consensus within the scientific community that 17% is far from sufficient to safeguard biodiversity into the future (Maron et al. 2018). There are pushes for a much greater, more ambitious coverage target to perhaps as much as 50% (Wilson 2016; Kopnina et al. 2018), as well as increased focus on improving the planning, placement and management of new and existing conservation areas (Coad et al. 2019; Visconti et al. 2019). In addition, there is a growing appreciation of the contribution that non-state conservation areas (i.e. those privately and communally owned) and other effective area-based conservation measures (OECMs) can make, both in terms of simply increasing area under conservation management (Garnett et al. 2018; Donald et al. 2019), and in representing species and habitats that may be under-represented or even absent altogether from state networks (Shwartz et al. 2017; Donald et al. 2019).

These discussions are vital to progress, but as we seek to expand and improve the global network of PAs and OECMs (referred to collectively hereafter as "conservation areas"), it is also important that we understand the fundamental drivers behind their establishment. In doing so we can help answer why it is that some countries' networks cover a greater percentage of their land area than others, and provide insights on where and how future increases could take place. To do this we present a conceptual framework describing the factors underpinning the growth of conservation area networks, with the aim of shedding light on the factors driving the patterns and extent of the national networks we see in the world today.

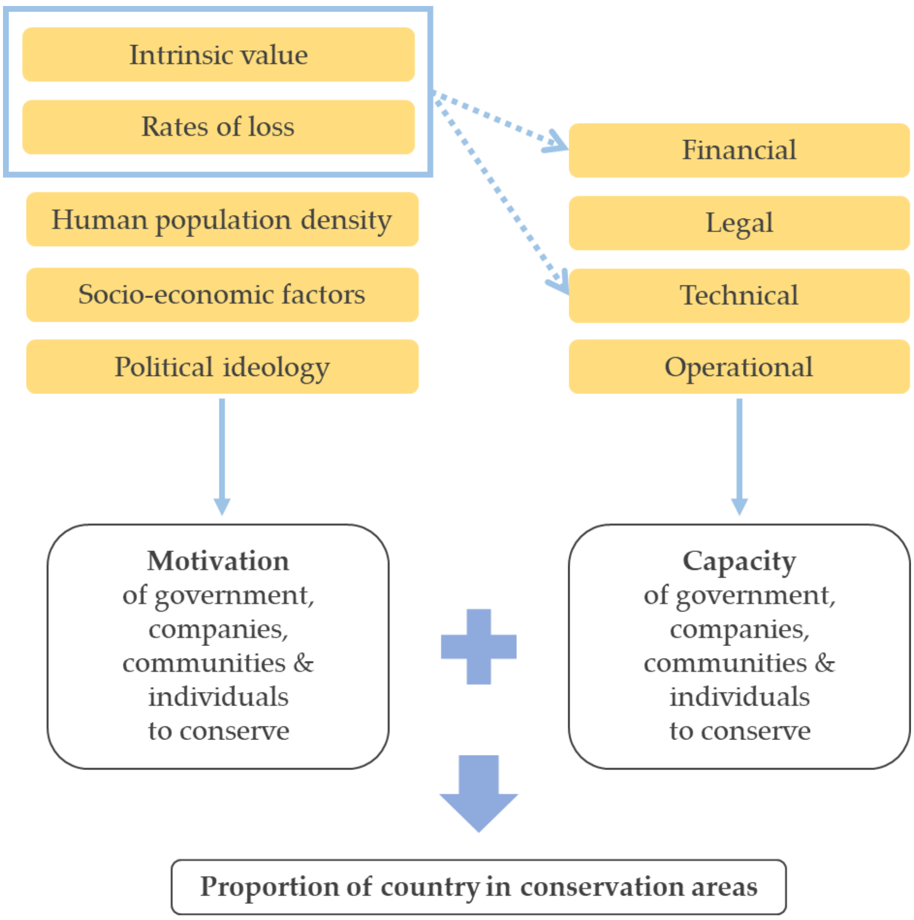
4.3 Introducing the conceptual framework

Conceptual frameworks are tools used to define and organise concepts, processes and the relationships between them. Often they are visualised in the form of a diagram setting out key ideas and linkages. They are especially useful and important when exploring research areas that are understudied, or for developing emergent theory (Rocco & Plakhotnik 2009). As such, they differ from a simple literature review by conceptualising a problem or research area, drawing out the essential factors and the network of relationships that connect them, and thus laying the foundations for developing new research questions in the area of interest.

Conceptual frameworks have been used widely in conservation to define and explore a variety of issues including: integrating human livelihoods with biodiversity (Salafsky & Wollenberg 2000), how to take effective conservation actions and management (Salafsky et al. 2002), understanding the public's relationship to urban biodiversity (Pett et al. 2016), and human-human conflicts over biodiversity and natural resources (White et al. 2009). A conceptual framework is ideally suited to exploring the question of why and how conservation areas come to be established. The literature on this topic is sparse, but defining the factors and relationships involved is vital to furthering our understanding of the enabling or disabling conditions that impact conservation area establishment.

While we lack accurate information on conservation area extent per country, current data in the WDPA show levels of protection vary from <1 % for 14 countries to 54% for Slovenia, and as much as 100% for some small island nations (IUCN 2019). This range suggests that the factors that have driven the creation of new conservation areas are likely to differ greatly between countries. However, most of the literature focuses on the factors that determine the spatial pattern of conservation area networks at national and global levels (Fearnside & Ferraz 1995; Ramesh et al. 1997; Joppa & Pfaff 2009) or global patterns in conservation area expansion or contraction (Jenkins & Joppa 2009; Mascia & Pailler 2011). Here we attempt to bring this literature together and produce a conceptual framework based around the factors that motivate, and the factors that impact capacity to act upon that motivation, for both national governments and individuals or communities (Figure 1). We use 'motivation' to mean the desire of decision makers to pursue a goal, for either intrinsic or instrumental reasons, and 'capacity' to mean the ability to achieve that goal because of a sufficiency in skills, knowledge and resources (OED 2019). Basing the framework on these two concepts makes the process clearer, as it shows that any decision to create new conservation areas depends on the decision maker being both willing and able to make the change. It also allows us to identify the different factors that underpin these two aspects, so that the role and interactions of these driving forces can be documented and understood.

Figure 4.1. A conceptual framework of factors that drive conservation area establishment.



4.4 Details of motivation

4.4.1 Intrinsic value

People value the natural world for many different reasons. While it provides physical resources and sustains human civilisation in a basic material sense, our aesthetic and spiritual appreciation of the beauty and intricacy of nature has been just as important to human life throughout history (Ulrich et al. 1993). While many people feel increasingly alienated from nature (Soga & Gaston 2016), these 'cultural ecosystem services' are no less significant today (MEA 2005). Humans have always sought to protect things that we value, so arguably conservation areas have existed for hundreds, if not thousands, of years in the form of sacred groves and hunting grounds, which conserved natural spaces and wild animals that were considered significant (Ormsby & Bhagwat 2010; Newton 2011). Early modern conservation areas such as Yellowstone and Yosemite in the US, which appeared in the 19th century, were intended to protect magnificent natural scenery for the enjoyment of the public (Leader-Williams et al. 1990; Watson et al. 2014). Today, many indigenous groups seek to conserve their ancestral lands in order to sustain a traditional way of life, as well as their shared identity and heritage, which is often intimately bound up with the natural world (Garnett et al. 2018). Meanwhile, private land owners, whether they are native or not, may turn their land into a conservation area because they value its wildlife and landscapes (Bingham et al. 2017).

However, today the extinction crisis has thrown a sharp focus on the need to protect biodiversity not only for its intrinsic beauty and uniqueness, but also for the role it plays in the fundamental functioning of natural systems which sustain life on Earth (Daily & Matson 2008). International agreements made in Bali, Caracas and Aichi over the last 40 years have urged nations to create more conservation areas that are bigger, better and more representative of all biodiversity (Le Prestre 2017). The Convention on Biological Diversity, created in 1992 and ratified by all but one nation on Earth, aims to ensure the worldwide conservation of biodiversity, sustainable use of biological resources, and equal sharing of benefits accrued (CBD 2018a). At the 10th Conference of the Parties in 2010, the Aichi Targets were agreed which contain specific goals due to be completed in 2020. Target 11 specifically addresses the expansion of conservation areas, under which signatories have committed to raising the proportion of the global land area protected to 17%, while each country has set its own national target taking into account its particular circumstances (CBD 2018b).

Biodiversity is not evenly distributed across the globe. In some regions there are high concentrations of variety, abundance and endemism, such as in the tropics, or where there is high geographic and climatic variation, or in places which have historically been very isolated (Myers et al. 2000). Thus the number of species and the diversity they represent in evolutionary terms is much higher in some countries than others (Brooks et al. 2006). The very fact of greater intrinsic biodiversity value can be a motivating factor by itself. The more diverse, abundant and globally recognised the biodiversity, the stronger impetus there can be to act to conserve it. This difference in richness is partially reflected in the differing national commitments to Aichi Target 11. While many countries also adopted a 17% target, they range between 3 and 50% overall (Butchart et al. 2015).

However, the relationship between biodiversity and land under conservation is not linear. Some countries, usually those that underwent industrial revolutions in the 19th and 20th centuries, and have dense, long established human populations, have lost much of their native biodiversity and habitats to agriculture and urbanisation (Kehoe et al. 2015). As a result, the few remaining species and fragments of native habitat may be highly prized. Even if they are not globally significant, on a national level there may still be a strong push to conserve and restore the natural environment simply because there is so little of it left (Geldmann et al. 2019).

4.4.2 *Rates of loss*

Globally, it is now widely recognised that the natural world is in a parlous state and biodiversity is being lost at an alarmingly and unacceptably high rate (Pimm et al. 2014). Under Aichi Targets 5 and 12, rates of habitat loss must be at least halved, and extinctions of vulnerable species must be prevented, respectively, by 2020 (CBD 2018b). Conservation areas are one of the key tools we have to help bring about a reversal of the downward spiral that the natural world is currently experiencing, as reflected in Target 11. The recent rate of expansion of the global conservation area network has been rapid, such that they now cover almost 15% of the global land area (IUCN 2018).

In individual countries, the motivating impact of high rates of loss of regional biodiversity is less clear. Focus on the decline of flagship species can prompt the establishment of new PAs to address their plight. For example, the steep decline of tiger populations across Asia prompted the establishment of tiger reserves across India, Nepal, Thailand, Indonesia and Russia from the 1970s onwards (Walston et al. 2010; Wei et al. 2015), while concern for giant pandas led the Chinese government to increase the number of reserves dedicated to their conservation from 4 to 67 (Wei et al. 2015), and recognition of the long decline of habitats across Europe led to the creation of the network of Natura 2000 sites under the EU Habitats Directive, which currently totals over 27,000 sites (Evans 2012). However, rates of biodiversity loss in some countries may be high precisely because there is little desire there for conservation, so threatened species and habitat remain vulnerable to destruction.

4.4.3 *Human population density*

Intuitively, there is less willingness to establish conservation areas in places with high human population density, because there is less natural habitat to protect and more resistance to restrictions on land use (Kehoe et al. 2015). While conservation areas can contain human settlements, and people often appreciate living among natural spaces, nonetheless there is still greater willingness to create new conservation areas in places with low population density (Fearnside & Ferraz 1995; Ramesh et al. 1997; Joppa & Pfaff 2009). This partly explains the persistent bias towards placement of conservation areas in regions that are characterised by difficult terrain, high elevation, an inhospitable climate, poor soils, inaccessibility and so on (Joppa & Pfaff 2009), because these factors tend to result in fewer people and less land use competition. Moreover, this issue is likely exacerbated by the introduction of global targets for conservation area coverage, leading many governments to create new PAs in remote areas (Barnes et al. 2018). Where conservation areas do exist among high human population density, they tend to be smaller and more fragmented networks (Oldfield et al. 2004).

However, two factors complicate this picture. Firstly, while conservation areas may be more likely in places with low human habitation and activity, equally where pressures on the natural world are low, there may be no real need to give formal protection to the land (Watson et al. 2018), although this is becoming less common because of international commitments for protected area coverage (Barnes et al. 2018). Secondly, human population density fluctuates over time and space. Places that have experienced rapid growth in recent decades may still have a large number of conservation areas that were created when densities were lower (Tyrrell et al. 2020). Meanwhile, urbanisation means that even in countries with a high national density, there may be areas with much lower density in more remote regions.

4.4.4 Socio-economic factors

Socio-economic factors are the primary driver of most land use decisions. This is illustrated by documented instances of PADDD (protected area downgrading, downsizing and degazettement), under which conservation areas have been reduced in size for the purposes of resource extraction and human settlement (Mascia & Pailler 2011). It is also partly reflected in the link between population density and conservation area establishment because more people generally leads to more economic activity (Global Change Data Lab 2018). However, broader opportunity costs are also relevant, so places with valuable land uses may not be conserved even when the human population density is low. In particular, state PAs are generally targeted away from lands which, if conserved, would introduce greater costs in terms of lost agricultural revenue and increased food insecurity (Venter et al. 2014) and thus towards less productive or valuable land (Joppa & Pfaff 2009). However, it is important to note that just because these areas are not seen as important by the politically well-connected, they may still support communities of subsistence farmers or hunters who have long made a living from the land (Homewood & Brockington 1999).

However, while setting aside land for conservation has a range of direct and indirect opportunity costs (Naidoo et al. 2006), the effective protection of ecosystem services provides enormous economic benefits (Costanza et al. 2014) that can drive the creation of conservation areas. Financial gain from recreational and spiritual services, in particular, is crucial to many private and communal conservation areas (Stolton et al. 2014). More broadly, in regions of particular natural beauty or with populations of charismatic and rare species, designating land for conservation and founding an ecotourism business can be a profitable venture (Balmford et al. 2015). In addition, the recent development of no net loss biodiversity policies have encouraged the creation of new conservation areas by industry through biodiversity offsetting, which can cover up to 10% of some countries (Bull & Strange 2018).

4.4.5 Political ideology

Political ideology, in both a broad and narrow sense, plays an important role in the development of conservation area policy (Büscher et al. 2017). Governments may create more conservation areas simply because environmental protection is a core part of their party values and manifesto. They may use the natural environment as a uniting force and source of national pride, shared identity and heritage, and in doing so promote nature conservation as a benefit to the nation and its people both materially and culturally (Duraiappah et al. 2005; MEA 2005). Additionally, they may focus on the need to maintain vital supporting, regulating and provisioning ecosystem services which underpin national prosperity, and push for conservation to ensure continued security and wealth long term (Harrison et al. 2016). Costa Rica is a prime example of this approach, with its focus on long term environmental sustainability, wildlife conservation and 'bioliteracy' in education (Pringle 2017). However, a direct appreciation of nature is not strictly necessary for a government to still be strongly motivated to create conservation areas. As well as the motivations that derive from the economic benefits described above, there is greater pressure on governments both domestically and from the global community to fulfil their international conservation commitments. Establishing new conservation areas can be seen as a way to gain kudos in the eyes the international community, which may in turn result in other benefits, such as greater influence on the world stage or new economic investment (Duffy 2006).

The particular political system of a country also has a significant influence on the implementation of policy, in the extent to which trade-offs are made with other competing priorities, and the speed with which action is taken. Authoritarian governments can, if nature conservation is a priority to them, create large new conservation areas relatively quickly and easily in a system that imposes little to no oversight or accountability on them (Dowie 2011). For example, many game reserves were established by European powers in countries they colonised, particularly during the 19th and early 20th centuries. These were largely intended for the enjoyment of the wealthy, while local communities were expelled or excluded (Adams & McShane 1996). However, many of these conservation areas have been maintained and expanded since independence (Pringle 2017). In addition, strict exclusionary conservation areas continue to be established in indigenous lands and overseas territories in controversial circumstances (De Santo et al. 2011).

By contrast, in a democratic society with a commitment to human rights, the creation of new conservation areas can be an extremely slow process. The administrative burden of the necessary consultation, planning, stakeholder engagement and so on that is required to establish a conservation area in a fair and equitable way, means that it may take years to complete. Furthermore, the fact that citizens and industry have a right to oppose or lobby against policy, means that the original plans may be significantly watered down and weakened by the time they are implemented (Lieberknecht & Jones 2016). Agriculture, fossil fuel and extractive industries are vastly wealthier than conservation and environmental advocacy groups, and are thus often able to leverage far more influence over decision making, even though lobbying activities are constrained and regulated by law (Smith & Walpole 2005). Their competing interests and influence over decision makers may result in fewer large conservation areas and less restrictive rules on resource use within those areas (Brailovskaya 1998).

However, in both cases the features of these political systems can also have the opposite effects. If authoritarian governments do not value nature then they can easily ignore the need for conservation or even degazette existing conservation areas with little opposition (Mascia & Pailler 2011), while the fact that democratic systems allow people a say means that the public can protest and campaign for greater conservation efforts. Furthermore, research suggests that popular support strengthens conservation efforts and underpins the long term success of conservation areas (Brockington & Schmidt-Soltau 2004). Thus, where people are involved in the political process, there may be a greater chance of achieving larger and stronger conservation area networks. However, irrespective of whether democratic or authoritarian, countries with high corruption levels are more susceptible to policy being influenced by bribe offering individuals or organisations, most of which have an anti-conservation agenda (Smith & Walpole 2005).

4.5 Details of capacity

4.5.1 Financial

Funding is a key limiting factor in the expansion of conservation area networks, because it can be very expensive to plan, implement and maintain them (Naidoo et al. 2006; Waldron et al. 2013, 2017; Venter et al. 2014). In theory, richer countries will be more able to shoulder the costs of establishing new conservation areas, and thus will have more land under conservation, while poorer countries may struggle in the face of other more immediately pressing issues (Balmford & Whitten 2003). However, sometimes it may simply be a case of priorities: where the motivation exists funds will be found, especially as conservation funding accounts for a tiny proportion of government spending (Bruner et al. 2004; McCarthy et al. 2012). In addition, it is worth noting that the world is increasingly getting richer: today, 86% of countries are classified as 'middle income' or 'high income' by the World Bank (World Bank 2019). Furthermore, while well-designed and effective conservation areas can be expensive, the process of creating a new conservation area is relatively cheap and so it is common for governments to add to their conservation area estate despite lacking the budget for future investment. Thus, while the relationship between national wealth and conservation area effectiveness is strong (Eklund & Cabeza-Jaimejuan 2017), the relationship between national wealth and conservation area coverage is likely to be much weaker (UNEP-WCMC & IUCN 2016).

There is, however, a general recognition that conservation areas need funding to achieve their goals (Waldron et al. 2013). This is why a number of intergovernmental agencies, governments and non-governmental organisations (NGOs) were established to help fund conservation area expansion and management in lower-income nations (Hickey & Pimm 2011), often with a particular focus on countries with high levels of biodiversity (Myers 2003; Miller 2014). This can take many forms, from supporting projects to identify and protect new areas (Bicknell et al. 2017) to core funding for conservation area running costs (Waldron et al. 2013). Such support can be particularly important for non-state conservation areas, as funding from government and NGOs to help with the bureaucratic process of creating and managing non-state conservation areas (Bingham et al. 2017; Wright et al. 2018). Other approaches are based on market mechanisms but these are more controversial (Kosoy & Corbera 2010; Muradian et al. 2013). For example debt-for-nature swaps, where countries have part of their foreign debt forgiven in exchange for conservation measures, including the establishment of conservation areas, taken within the debtor country (Hansen 1989). Similarly, payment for ecosystem services schemes can provide funding for new conservation areas that capture carbon or provide water (Wendland et al. 2010; Bicknell et al. 2017).

There are also situations where funding for conservation areas can come from sources other than government or external donors. Some conservation areas can generate large revenue surpluses through ecotourism, with those in less remote areas in richer countries being the most highly visited (Balmford et al. 2015). For state conservation areas this funding is often used to subsidise management of other, less popular areas (Bovarnick et al. 2010) but it could still create an incentive for governments to expand their conservation area coverage. Revenue generation is probably more of a direct incentive for privately managed conservation areas, as shown by the many profitable privately-owned game reserves in Africa (Langholz & Krug 2004). In addition, governments in higher-income countries can encourage new privately managed conservation areas by providing incentives based on tax rebates, agri-environment schemes or other financial instruments (Kamal et al. 2015).

4.5.2 Legal

Every country has legislation for establishing conservation areas and in a few cases, such as Brazil's Forest Code, this explicitly states how much land should be conserved under specific circumstances (Azevedo et al. 2017). This is rare though, so legal factors tend to have more indirect influences based on reflecting or tempering the motivational factors described above. When it comes to expanding conservation area networks, past increases in state protected areas were often underpinned by conservation legislation brought in by colonial powers and autocratic governments, over-riding existing tenure systems and taking away land without compensation (Homewood & Brockington 1999). This has largely changed; land for conservation is now usually either purchased or compensation provided to landowners to relocate or manage the land appropriately. In addition, many countries have brought in legislation to encourage the creation of privately and communally-managed conservation areas, although the success of this depends on the extent to which landowners trust the state to recognise and respect the conservation areas they develop (Bingham et al. 2017). Thus, limiting the powers of government to confiscate land can lead to lower conservation coverage, but also enables legislation to increase coverage through the creation of privately and communally-managed conservation areas.

There is a similar pattern when it comes to reducing conservation area networks. Such changes have been well documented as part of PADDD, where legislation allows governments to remove conservation area status to allow mineral extraction, agricultural expansion or other forms of natural habitat loss, even in the face of public opposition (Mascia & Pailler 2011; Golden Kroner et al. 2019). Frequently legislation provides higher protection for some types of conservation area types over others, but even the most prestigious conservation area types can be degazetted when governments consider it a priority (Qin et al. 2019). Uncertainty about respect for human rights and land tenure, either through changes in legislation or unstable government, can also lead to the loss of conservation areas, with landowners either selling their land or stripping it of natural resources (Robinson et al. 2018).

4.5.3 Technical

It is well known that technical capacity is important for good conservation area management effectiveness (Leverington et al. 2010), but there is less research on the extent to which capacity determines whether they are established in the first place. One might expect it to play a role, given that the process of identifying priority areas for conservation and then legally gazetting them is a technical one, requiring people with expertise in law, international policy, spatial prioritisation, conservation science and planning (Lausche & Burhenne, 2011). However, many existing conservation areas were created with little technical input, and today the minimum process can be run by consultants or other people without long-term roles in the conservation sector (Smith et al. 2009). Thus, lack of technical capacity at the government level is more likely to be a reflection of political priorities than a limiting factor. This is probably not the situation for privately and communally managed conservation areas, where landowners are unlikely to have all the required skills. In such cases training and support from government, NGOs and academia is likely to be very important when seeking legal recognition, especially when this involves producing conservation covenants and management plans (Bingham et al. 2017).

4.5.4 Operational

If governments, groups or individuals have an objective to create new conservation areas, and the legal, financial and technical capacity to do so, then in most situations this will lead to higher levels of conservation area coverage. However, achieving this depends on more than just the conservation sector, as a range of institutions are involved in planning, authorising and implementing action (Knight et al. 2006), so dealing with institutions with low bureaucratic effectiveness can slow progress (Barrett et al. 2001). Moreover, such institutions are more susceptible to corruption (McCreless et al. 2013), which can seriously hinder conservation efforts. Corruption is widespread across the world but lower-income countries tend to have higher corruption levels (Laurance 2004), depriving conservation of vital funds through embezzlement, and undermining political support and effective law enforcement through bribes for political influence or to overlook illegal activities (Smith & Walpole 2005; Irland 2008). The fact that poor countries tend to have a greater economic reliance on extractive and environmentally exploitative activities may mean that there is much greater opportunity for public officials to profit directly from destruction of the natural environment and, thus, from preventing the establishment of conservation areas.

Functional bureaucracies can also be severely undermined or destroyed by political upheaval, civil unrest and war (Hanson et al. 2009). Conflict can remove existing conservation areas and, by preventing the effective functioning of government, prevent the establishment of new ones (Kanyamibwa 1998; Baral & Heinen 2006). In addition, where social, economic and political conditions are highly volatile and unstable, individuals and communities are disincentivised from creating conservation areas because they have no guarantee of long-term use rights over the land, legal help or protection, or a reliable financial return, particularly if it depends on tourism. Instead, they may decide that it is preferable to strip the land of resources for short term gain, rather than attempt to manage it sustainably long term (Barbier & Tesfaw 2013; Robinson et al. 2018). However, times of great change, such as the fall of a colonial government, may also provide opportunities for new ideas and policies, which can result in the establishment of new conservation areas (Radeloff et al. 2013).

4.6 Implications for future conservation area coverage

As the 2020 CBD Convention of the Parties approaches, the latest data indicate that we will come close to the Aichi Target 11 goal of 17% coverage (IUCN 2019), but calls to increase the target further are already well established e.g. '30% by 2030' and 'half earth' (Wilson 2016; Dinerstein et al. 2017; Pimm et al. 2018), while others propose moving away from area-based targets to a more sophisticated, outcome-based target (Visconti et al. 2019). Thus, we must consider the social, political and economic conditions needed in order to achieve ever more ambitious targets.

Biodiversity loss and climate change continue apace, posing an existential threat to human civilisation. The global human population is still rising, increasing the spread of agriculture, resource demands and carbon emissions (Tittensor et al. 2014). However, there is also a trend towards urbanisation and rural land abandonment, which opens up further potential for rewilding and nature recovery (Sanderson et al. 2018). Moreover, growing social and political movements suggest increasing pressure on governments in the coming years to act with appropriate urgency to avert ecological collapse, particularly among younger people (Gardner et al. 2020). This suggests the motivations for creating new conservation areas will increase but in the face of increasing threats and pressures, so political ideology will play an enormous role in determining outcomes at the national and international level (Mace et al. 2018). It is also widely recognised that upscaling conservation area coverage cannot depend solely on creating new state PAs, so privately- and communally-managed PAs and OECMS will become increasingly important.

In terms of capacity, our review suggests financial and legal are the most important factors, both of which are strong reflections of public and government motivation. With increased political will, it would be easy to hugely increase funding and develop technical capacity at the agency and land-owner level. Political will is also the main constraint on creating legislation to devolve power and encourage non-government agencies to establish and maintain conservation areas. However, implementation also depends on a set of operational factors that are outside of the conservation sector and more difficult to manage and mitigate.

No single factor discussed is sufficient or explanatory by itself; some or all of these encouraging and enabling factors must exist and work in conjunction with one another to provide the opportunity and impetus to establish and maintain conservation areas. Debate around expansion of conservation areas often centres on fundraising for international NGOs and increasing knowledge through scientific research (McCarthy et al. 2012). Here we have hopefully illuminated some less frequently considered factors which, directly or indirectly, can be vital to the establishment of conservation areas. Chief among these is stable government, political commitment and strong human rights, all of which are necessary to produce conservation area networks and healthy societies that can co-exist peacefully in the long term. We hope that by bringing together and highlighting these factors, we will help inform discussions of what are the necessary conditions that underpin the establishment of large, healthy, functioning conservation area networks.

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Chapter 5. Developing a system to improve the accuracy of global estimates of conservation area coverage based on a sampled approach

5.1 Abstract

Monitoring progress towards global conservation targets is vital but often difficult, given the highly time- and resource-intensive nature of data collection. Measuring the global extent of conservation area coverage is a key component of several international conventions but currently depends on each national government providing relevant data, so the results depend on in-country capacity and often underestimate the area of non-state conservation areas. Here we present a new method to improve estimates of global conservation area coverage, based on identifying a representative sample of countries and regions for future data collection. This first involved identifying 10 biological and socio-economic factors that drive patterns of global biodiversity and conservation area extent, dividing them into 89 categories and mapping their distributions. We then used the spatial prioritisation software Marxan to select areas that contain at least 10% of the extent of each of the 89 categories, based on identifying (a) the minimum number of countries needed and (b) the minimum amount of land based on 100 km^2 grid squares within the selected countries. Marxan identified 25 countries, although there was some flexibility with the results meaning that some countries could be swapped with other nations without impacting the efficiency of the results. There was also flexibility when selecting the grid squares within the sample countries, especially in wilderness areas. Our sample should serve as the basis for focussed data collection, allowing quicker and more accurate estimations of conservation area coverage than is currently possible, and our approach could be adopted more broadly when developing global conservation metrics.

5.2 Introduction

Conservation areas are an essential component of efforts to prevent the loss of global biodiversity (Watson et al. 2014). To this end, the 196 signatories to the Convention on Biological Diversity have committed through Aichi Target 11 to conserve 17% of the global terrestrial area within protected areas (PAs) and land under other effective area-based conservation measures (OECMs) by 2020 (CBD 2010). Progress towards Aichi Target 11 and other international commitments is measured using the World Database of Protected Areas (WDPA), which is compiled and maintained by the UN Environment World Conservation Monitoring Centre (UNEP-WCMC), based on conservation area data provided by each national government (Bingham et al. 2019; Lewis et al. 2019). This makes the WDPA an extremely important source of information and a great deal of resources are spent maintaining its accuracy (Juffe-Bignoli et al. 2016). However, there are data limitations (Visconti et al. 2013), partly because some countries lack the capacity to provide up-to-date and accurate information, so that newer PAs are missing. More generally, non-state PAs and OECMs are generally underrepresented in the database (Stolton et al. 2014; Bingham et al. 2017; Garnett et al. 2018; Corrigan et al. 2018; Donald et al. 2019), partly because governments have only recently started collecting data on conservation areas not owned or managed by the state, and partly because some owners of non-state conservation areas are wary of providing information to the government on their land (Fitzsimons & Wescott 2007).

These limitations make it difficult to accurately measure progress towards international conservation targets, both in terms of percentage area conserved and representing biodiversity. They also make the process of setting new targets difficult, which is particularly important given that the international conservation community is pushing for more ambitious targets for conservation area extent and representativeness post-2020 (Dudley et al. 2018; Maron et al. 2018; Visconti et al. 2019). One way to address this is to invest in improving the quality of global conservation area datasets and there are a number of ongoing projects to increase the quality of data on state PAs and collect information on non-state PAs and OECMs. Such work is vital but expensive (Juffe-Bignoli et al. 2016), so complementary approaches are needed. One solution would be to collect these data from a representative sample of countries, as this would be quicker and cheaper, as well as less dependent on official government sources because the data would only be used to calculate global estimates, avoiding the need to report official national estimates. Here we present such an approach and identify a representative set of countries.

Identifying this set of countries can be framed as a conservation decision science problem, based on defining a broad objective, and then converting this into targets and costs (Groves & Game 2015). In this case, our objective is to identify a representative proportion of the terrestrial realm, so that conservation area data from this subset can be used to estimate the extent to which the global PA and OECM network meets area and biodiversity conservation targets. This involves considering two sets of factors. The first are drivers of conservation area establishment, which are likely influenced by a range of economic, political and social factors. For example, it is well known that conservation area coverage is higher on land that is less suitable for commercial agriculture or resource extraction (Pressey & Tully 1994; Joppa & Pfaff 2009). The second are drivers of biodiversity pattern, as species and ecosystems show strong variation across relatively few broad-scale patterns, such as latitude, elevation and other gradients (Gaston 2000). In this chapter, we use the term 'biodiversity' to mean 'the variety within and among living organisms, assemblages of living organisms, biotic communities, and biotic processes, whether naturally occurring or modified by human' (DeLong 1996). By mapping out the relevant factors that determine conservation area extent and biodiversity pattern, and setting targets for how much land belonging to each category of each factor should be selected, we can use spatial conservation prioritisation algorithms to identify a representative proportion of the terrestrial realm for future data collection on conservation area coverage (Groves & Game 2015).

The final component of the conservation decision science problem is choosing the cost metric, which must reflect the time and effort involved in collecting the conservation area information. PA and OECM data is generally collected and collated at the national level (CBD 2019), so each new country added to our sample would add an extra cost in terms of effort required to obtain their datasets. Thus, we define our cost metric as the number of countries in which our sample areas are found. Such a metric is a simplification, as the effort required will vary between countries based on their capacity and the number of conservation agencies that are responsible for national or regional data collection. We partially account for this in our study by dividing larger countries into their highest administrative units below the level of national government, such as states or provinces, which better matches the devolved nature of data collection in these countries.

In this study, we used the decision support tool Marxan (Ball et al. 2009) to identify a representative set of areas across the world that could be used in future to estimate global patterns in conservation area coverage. We did this by: (i) selecting and mapping the biogeographic and socio-economic factors which are representative of both global biodiversity patterns and drivers of conservation area extent; (ii) undertaking sensitivity analyses to understand the trade-off between increasing the percentage of each factor category in our sample and decreasing the cost of data collection by minimising the number and countries selected. We used these results to determine an appropriate target for the percentage of each category type that should be selected, and; (iii) running a two-stage selection process to identify our sample areas, first selecting broad nationally-defined areas, then selecting planning units within them.

5.3 Methods

5.3.1 Producing the feature data

We conducted a literature review to identify factors that determine patterns of global biodiversity and those that are likely to determine total conservation area network extent. We then ran a workshop with 12 experts on conservation area networks to identify which of these factors were most important (Supplementary Materials Table S5.1). Finally, we identified the available global datasets that mapped these important factors, producing a list of ten: biomes, elevation, government effectiveness, human population density, islands, landcover, latitude, per capita GDP, realms and subregions (Supplementary Materials Figure S5.1). Four of these factors were selected to represent global biodiversity patterns, three to represent both global biodiversity patterns and drivers of conservation area network extent and three to represent only drivers of conservation area network extent (Supplementary Materials Table S5.2).

We used two datasets for the four factors that only represent global biodiversity patterns. We chose not to use IUCN species polygons to represent biodiversity, because they cover major vertebrate groups but are patchy on others and so are not representative. Instead, we used the well-established WWF biogeographical realms and biomes datasets, which relate to broader biodiversity (He & Zhang 2009). We used WWF's global ecoregion shapefile to map the biomes and realms, where each of the 16 biomes is a broad ecosystem type and each of the 8 realms is a large biogeographic unit (Olson et al. 2001). We used the Global Administrative Areas shapefile (GADM 2018) as the basis of our islands layer, grouping them into five categories: < 1,000 km^2, ≥ 1,000 - 10,000 km^2, ≥ 10,000 - 100,000 km^2, ≥100,000 - 1,000,000 km^2 and continent. As part of this, we removed islands with an area < 1 km^2 because these are unlikely to contain important terrestrial biodiversity. In addition, we classified islands as having the continent category if they were both <10 km^2 and within 100 km of a continent or Greenland, as we argued that these would have similar species composition to their associated continents. We created the latitudinal zone layer in ArcMap 10 (ESRI 2011) by dividing the globe into 7 bands. Each band has a width of 20°, apart from at the poles where we used bands of 40° to avoid over-representing differences in these relatively small regions.

We used three datasets for the factors that represent global patterns of biodiversity and drivers of conservation area network extent. We used the Shuttle Radar Topography Mission's 1 km elevation data and divided these into five elevation categories (0 – 299 m, 300 – 799 m, 800 – 1399 m, 1400 – 1999 m and 2000+ m) based on existing studies of biodiversity and elevation gradients (Bruijnzeel & Veneklaas 1998; Linkie et al. 2010). We used the European Space Agency's GlobCover landcover map which divides the terrestrial realm into 12 broad landcover types: croplands, croplands mosaic, closed forest, open forest, mosaic grassland/shrubland, sparse vegetation, flooded forest/grassland, artifical surfaces, bare areas, water bodies, snow and ice, and unknown (ESA GlobCover Project 2009). We used the United Nations subregions classification, which assigns each country to one of 22 groups based on continental regions and homogeneity in sizes of population, demographic circumstances and accuracy of demographic statistics within groups: Australia and New Zealand, Caribbean, Central America, Central Asia, Eastern Africa, Eastern Asia, Eastern Europe, Melanesia, Micronesia, Middle Africa, Northern Africa, Northern America, Northern Europe, Polynesia, South America, South-eastern Asia, Southern Africa, Southern Asia, Southern Europe, Western Africa, Western Asia and Western Europe (United Nations Statistics Division 2019).

We used three datasets for the factors that only represent drivers of conservation area network extent. We used the World Bank's Worldwide Governance Indicators dataset to measure government effectiveness, grouping countries into four categories based on them having scores of 0 - 24.9, 25 - 49.9, 50 - 74.9 and 75 - 100 (World Bank 2019a). Similarly, we grouped countries into low, lower-middle, upper-middle and high income per capita GDP based on the classification used by the World Bank (World Bank 2019b). We used the UN's map of human population density per km^2 as the basis of our human population density layer (United Nations Population Division 2013). We wanted to ensure that our categories represented both areas with very low and very high densities, so grouped the values into 5 categories using a logarithmic scale: 0 - 0.9, 1 - 9.9, 10 - 99.9, 100 - 999.9 and > 1000 per km^2.

5.3.2 *Producing the spatial selection systems*

Our study aimed to identify a representative sample of the terrestrial realm for measuring conservation area coverage, whilst also minimising the set of countries selected to reduce the number of expert groups needed to provide data. To do this we adopted a systematic conservation planning approach, but rather than representing conservation features such as species or habitats as in a typical spatial conservation prioritisation analysis, we used the 10 factors at a global scale to select areas which are representative both biogeographically and socio-economically. This approach allowed us to identify sets of planning units that met targets for each of the categories from the ten factors (referred to as 'features' hereafter). This involved two stages: Stage 1 identified the set of countries; Stage 2 identified 100 km^2 grid squares within these countries, thus refining the sample from Stage 1 to avoid over-representing larger nations in the analysis. We used the Marxan spatial prioritisation software in both stages (Ball et al. 2009), which uses a simulated annealing approach to identify near-optimal portfolios of planning units that meet targets, whilst minimising planning unit and boundary costs. Each Marxan analysis involves running the software a number of times and producing a near-optimal portfolio each time. Marxan then produces two key outputs: the "best" output, which is the portfolio from the run with the lowest cost, and the "selection frequency" output, which counts the number of times each planning unit appears in each of the portfolios. Planning units with high selection scores are always needed to meet the targets, whereas lower scoring planning units could be swapped with similar, alternative planning units without affecting target attainment (Ball et al. 2009).

Running the spatial analysis involved creating a planning system for both Stage 1 and Stage 2 using the CLUZ plugin (Smith 2019) for QGIS 3 (QGIS 2009). This involved dividing the planning region into a series of planning units, giving each planning unit a cost for including it in a portfolio, and calculating the amount of each feature in each planning unit. For the Stage 1 planning system, our planning units consisted of countries for nations with an area < 1,000,000 km^2 and sub-national administrative level units for countries with an area ≥ 1,000,000 km^2 (Figure S5.1). We took this approach because larger nations tend to have sub-national conservation agencies and legislation, so we wanted to minimise the number of these sub-national administrative units selected to avoid having to collect data from a large number of sub-national expert groups. The threshold of 1,000,000 km^2 was decided with the workshop expert group as it provided a balance between dividing enough large countries with devolved systems while avoiding unnecessarily dividing up smaller countries. The boundaries of these planning units were derived from the GADM shapefile and we used the national (L0) level for the smaller countries and the highest sub-national administrative level (L1; such as states or provinces) for the larger countries. We followed established practice for reporting terrestrial coverage statistics by excluding Antarctica from our analyses (Butchart et al, 2014). We then converted each layer into a 1000 x 1000 m resolution raster in the Mollweide projection, calculated the area of each feature (i.e. each category type of each of the ten factors) in each planning unit using the Tabulate Area function in ArcMap, and imported these data into CLUZ.

We used a new approach in Marxan to ensure that our Stage 1 analysis identified a set of countries that represented all the features, while also minimising the number of countries selected. To do this we set the combined planning unit cost of each country as 1, so that selecting more countries was more costly. This involved accounting for the larger nations being split into several planning units, based on the L1 administrative units, so we set the planning unit costs of these L1 planning units as being the inverse of the number of L1 units in the country.

Thus, each of South Africa's nine provinces had a planning unit cost of 0.111. In addition, we needed to ensure that Marxan met targets by selecting the L1 planning units from the same countries whenever possible, rather than L1 planning units from different countries. To do this we manipulated the Marxan boundary cost file so it appeared that every L1 planning unit in the same country shared a boundary, so that if Marxan selected one L1 planning unit in a particular country then there would be less of a cost to selecting subsequent L1 planning units from the same country. To ensure that this cost would be the same per country, we set the length equal to the inverse of the number of different L1 boundary pairs in each country, so for example the nine provinces in South Africa produced 45 combinations of L1 pairs and so the boundary length was 0.0222. This manipulation of the boundary cost data has been used in previous studies to ensure that certain planning units are more likely to be selected together, even when they are not physically adjacent (Possingham et al. 2005; Hermoso et al. 2011; Makino et al. 2013).

The Stage 2 planning system was based on a set of 10 km x 10 km grid squares, which was created in QGIS 3 using the Create Grid tool. We then used the Union tool to combine this grid layer with the L0 and L1 planning units used in Stage 1 to produce the final planning unit layer, calculated the amount of each feature present in each of these smaller planning units using the Tabulate Areas function in ArcMap and imported the results into CLUZ. However, for this finer-scale analysis we used the planning unit area as the cost metric and did not create a boundary length data file. This was because in Stage 2 we were seeking to identify the smallest area of land needed to meet the targets and were not interested in selecting planning units that neighboured each other.

5.3.3 Spatial analysis

To ensure that the planning units selected in Stage 1 and 2 were representative of the terrestrial realm, we decided the analyses should use the same percentage target for every feature. Deciding on what that target should be was less straightforward, as it needed to balance the fact that selecting a larger number of planning units would make future conservation area extent estimates more accurate, but that collecting data for more countries and sub-national administrative units would be less feasible and more expensive. So, we used the conservation planning system developed for Stage 1 to run a number of preliminary Marxan analysis using targets of 1%, 2%, 5%, 10%, 20%, 30%, 40% and 50% of the total extent of each feature. Each run consisted of 100 runs of 10,000,000 iterations and we used a BLM of 1.5, a value that we determined through testing best ensured that Marxan chose enough sub-national administrative units from the same countries to meet the targets. We counted the number of whole countries and the number of planning units in the 'best' solution. This enabled us to investigate the trade-off between area selected and cost of data collection. We also produced 1,000 samples of randomly selected units using Python programming language (Python 3 2020) for target percentages between 1% and 50%.

We then ran Stage 1, which consisted of 1,000 runs of 10,000,000 iterations, using the target percentage for each feature that we identified in the sensitivity analysis and the BLM value of 1.5. Marxan identified a number of planning unit portfolios that had equally low costs, i.e. contained the same number of planning units. To choose between them we identified the portfolio with the most even spread of countries selected across the continents and containing planning units with the highest mean selection frequency score. This provided us with our final set of national and sub-national planning units that were the basis of Stage 2. Thus, our first step in Stage 2 was updating the planning system to specify in CLUZ that all of the 100 km^2 planning units found outside the national and sub-national regions selected in Stage 1 should be excluded from subsequent Marxan analyses because they had not been selected as part of the initial sample. The Stage 2 Marxan analysis also consisted of 1,000 runs of 10,000,000 iterations. However, we used a BLM of 0 so that Marxan did not account for connectivity and selected the smallest number of planning units needed to meet the targets.

To assess whether the samples represent global patterns, and whether the sample of grid squares is an improvement on the sample of administrative units in terms of representativeness, we undertook two comparative analyses. Firstly, using the Union tool in ArcMap we calculated how much of the global area and of the Stage 1 and Stage 2 samples is covered by protected areas (those with boundary data available) listed in the WDPA to see which of the latter is closest to the former. Secondly, we calculated what percentage of the total land area globally and in each of the samples is covered by each of the conservation features.

To assess whether the samples perform better than a set of units selected randomly, we ran further analyses using Python consisting of 1,000 runs in which a random number of planning units were selected until the area they covered was greater than or equal to the area selected in the best outputs identified in Stage 1 and 2.

5.4 Results

5.4.1 Sensitivity analyses

The number of L0 and L1 planning units selected by Marxan to meet the targets for the 89 features (Table 5.2) ranged from 23 for the 1% targets, for which at least 1% of the area of each feature must be included in the sample, to 206 for the 40% targets, for which at least 40% of the area of each feature must be included in the sample (Table 5.1). The number of planning units more than doubled when comparing results from using 10% and 20% targets, with a levelling off when the targets were ≥ 30%, although there was some variation due to the random component of the simulated annealing process. The number of countries selected had a narrower range, from 22 for the 1% targets to 32 for the 30%, 40% and 50% targets. Analysis of the randomly selected samples show that as the target percentage increases, so does the stability of the results (Figure 5.1). However, it is important to balance representativeness with feasibility. Therefore, we decided to base our decision on the number of L0 and L1 planning units selected in the sensitivity analyses and use 10% targets for the main analyses, balancing between including a sufficient proportion of the terrestrial realm and ensuring that data would need to be collected from a feasible number of administrative units.

Table 5.1: The number of planning units and countries selected to meet specific percentage targets for each of the 89 features. Planning units consisted of whole countries (L0) for nations with an area <1 million km² and highest level political administrative units (L1 = provinces, states, etc) for countries with an area ≥1 million km².

Conservation feature targets (%)	Number of L0 and L1 planning units selected	Number of countries (L0) selected
1	23	22
2	24	22
5	30	24
10	50	25
20	117	27
30	204	32
40	206	32
50	205	32

Figure 5.1: The standard deviation of the percentage of each of the 1,000 randomly selected samples, based on selecting between 1% and 50% of the planet, that is covered by sites in the WDPA.

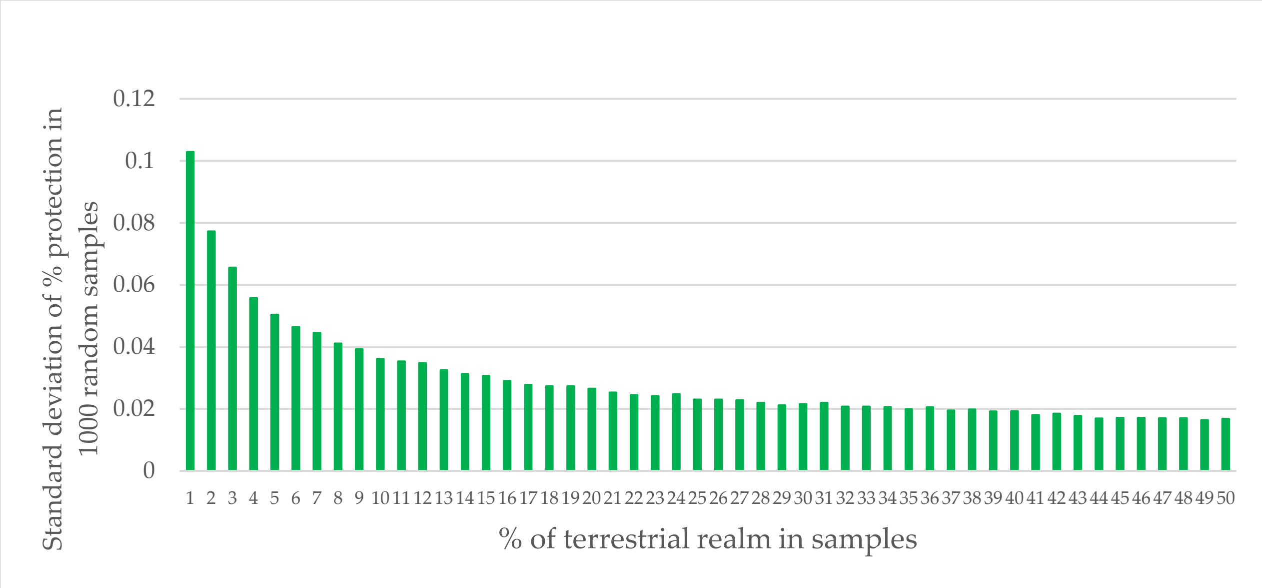


Table 5.2: The 89 'conservation features' included in the Marxan analyses. At least 10% of the global extent of each feature was included in the each sample.

Category	Feature
Biomes	Tropical & Subtropical Moist Broadleaf Forests
Biomes	Tropical & Subtropical Dry Broadleaf Forests
Biomes	Tropical & Subtropical Coniferous Forests
Biomes	Temperate Broadleaf & Mixed Forests
Biomes	Temperate Conifer Forests
Biomes	Boreal Forests/Taiga
Biomes	Tropical & Subtropical Grasslands, Savannas & Shrublands
Biomes	Temperate Grasslands, Savannas & Shrublands
Biomes	Flooded Grasslands & Savannas
Biomes	Montane Grasslands & Shrublands
Biomes	Tundra
Biomes	Mediterranean Forests, Woodlands & Scrub
Biomes	Deserts & Xeric Shrublands
Biomes	Mangroves
Biomes	Inland water
Biomes	Rock & ice

Realms	Australasia
Realms	Antarctic
Realms	Afrotropics
Realms	Indomalay
Realms	Nearctic
Realms	Neotropics
Realms	Oceania
Realms	Palearctic
Elevation	0 - 299m
Elevation	300 - 799m
Elevation	800 - 1399m
Elevation	1400 - 1999m
Elevation	2000+m
Islands	Under 1,000km^2
Islands	1,000 to 10,000km^2
Islands	10,000 to 100,000km^2
Islands	100,000 to 1,000,000km^2
Islands	Continents
Landcover	Croplands
Landcover	Croplands mosaic
Landcover	Closed forest
Landcover	Open forest
Landcover	Mosaic grassland / shrubland
Landcover	Sparse vegetation
Landcover	Flooded forest / grassland
Landcover	Artificial surfaces
Landcover	Bare areas
Landcover	Water bodies
Landcover	Snow & ice
Landcover	No data
Latitude	50N to 90N
Latitude	30N to 50N
Latitude	10N to 30N
Latitude	-10S to 10N
Latitude	-30S to -10S
Latitude	-50S to -30S
Latitude	-90S to -50S
Income classification	Low income
Income classification	Lower middle income

Income classification	Upper middle income
Income classification	High income
Population density	0 to 0.9
Population density	1 to 9.9
Population density	10 to 99.9
Population density	100 to 999.9
Population density	1000+
Government Effectiveness	0 - 24.9
Government Effectiveness	25 - 49.9
Government Effectiveness	50 - 74.9
Government Effectiveness	75 - 100
Realms	Water & ice
Subregions	Australia and New Zealand
Subregions	Caribbean
Subregions	Central America
Subregions	Central Asia
Subregions	Eastern Africa
Subregions	Eastern Asia
Subregions	Eastern Europe
Subregions	Melanesia
Subregions	Micronesia
Subregions	Middle Africa
Subregions	Northern Africa
Subregions	Northern America
Subregions	Northern Europe
Subregions	Polynesia
Subregions	South America
Subregions	South-eastern Asia
Subregions	Southern Africa
Subregions	Southern Asia
Subregions	Southern Europe
Subregions	Western Africa
Subregions	Western Asia
Subregions	Western Europe

5.4.2 *Stage 1 analysis*

Running Marxan 1,000 times identified that only 42 of the 900 planning units were needed to meet the 10% targets for all the features (Figure 5.2a). We found that 174 planning units were selected at least once across all the runs, with only 17 planning units selected in all 1,000 runs (Figure 5.2b). Planning units that were selected fewer times, and so could be swapped in any portfolio for planning units with similar characteristics, were mostly found in Brazil, central Africa, south-east Asia and the United States of America (Figure 5.2b). Using these results, we identified the Marxan output containing the smallest number of countries, most even spread across the continents and the highest mean selection frequency. This consists of 9 whole countries and territories and 33 administrative units within another 16 countries. These 25 countries and territories are: Argentina, Australia, Brazil, China, Democratic Republic of the Congo, Dominican Republic, France, French Polynesia, Greenland, Indonesia, India, Italy, Kazakhstan, Kiribati, Mexico, Mali, Papua New Guinea, Russia, Saudi Arabia, Sudan, South Georgia and the South Sandwich Islands, South Africa, Sweden, Tanzania and the United States of America.

5.4.3 Stage 2 analysis

The best portfolio identified by Marxan contained 4,581 grid squares from a set of 137,287, covering 10.9% of the terrestrial area (Figure 5.3a). Of the 42 planning units selected in Stage 1, only 7 had less than half their area selected in Stage 2. The amount of each Stage 1 planning unit also selected in Stage 2 ranged between 27.8% for French Polynesia and 100% for three US states, with a median of 92.5% (Figure 5.3a). The percentage of Stage 1 planning units selected in Stage 2 mirrors the selection frequency results, with low selection frequency scores for planning units where Marxan only needed to select a smaller proportion of the country or L1 administrative unit (Figure 5.3b).

Figure 5.2a. Sample of countries (L0) and administrative units (L1) that meet 10% targets, selected based on 1,000 Marxan runs and selecting the result with the smallest number of planning units, most even spread across the continents and with planning units with the highest mean selection frequency.

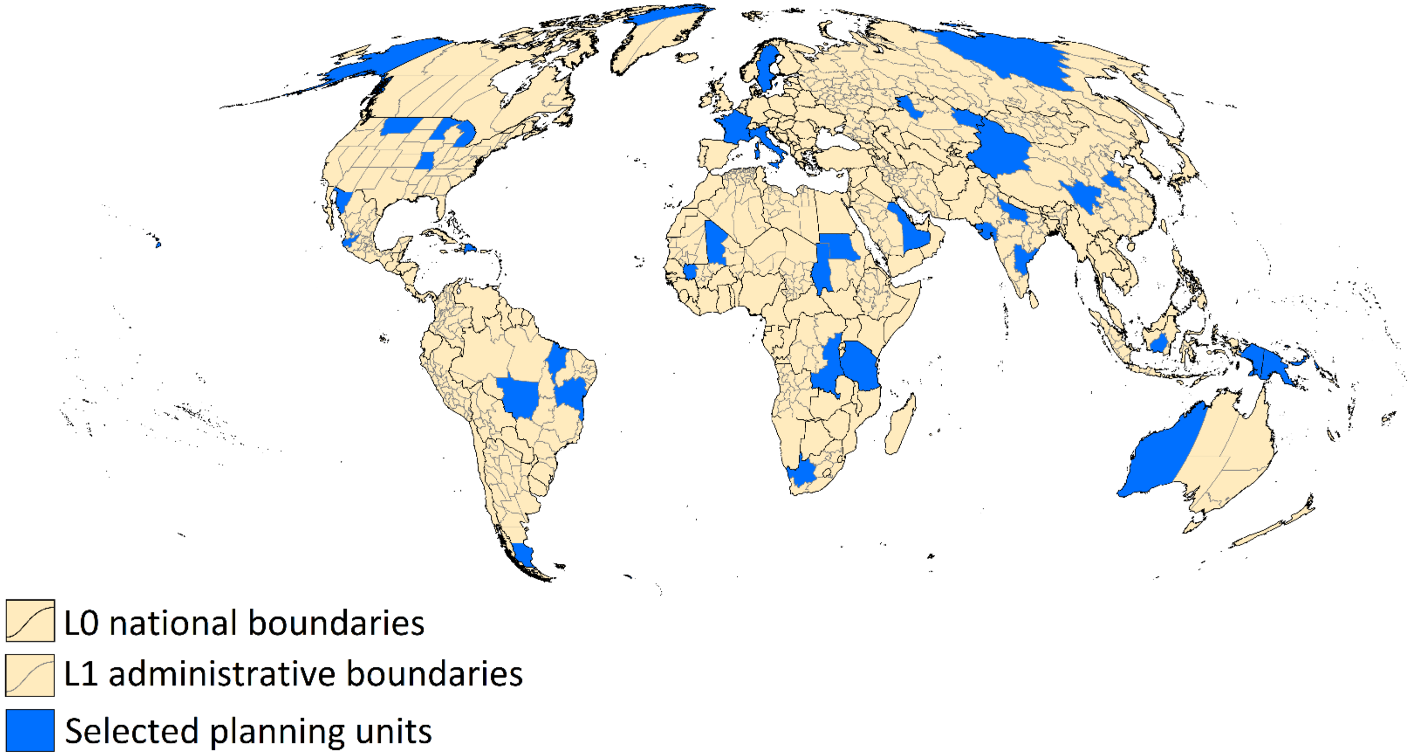


Figure 5.2b. Selection frequency scores from Marxan showing the number of times each planning unit was selected in the 1,000 runs used to identify the sample.

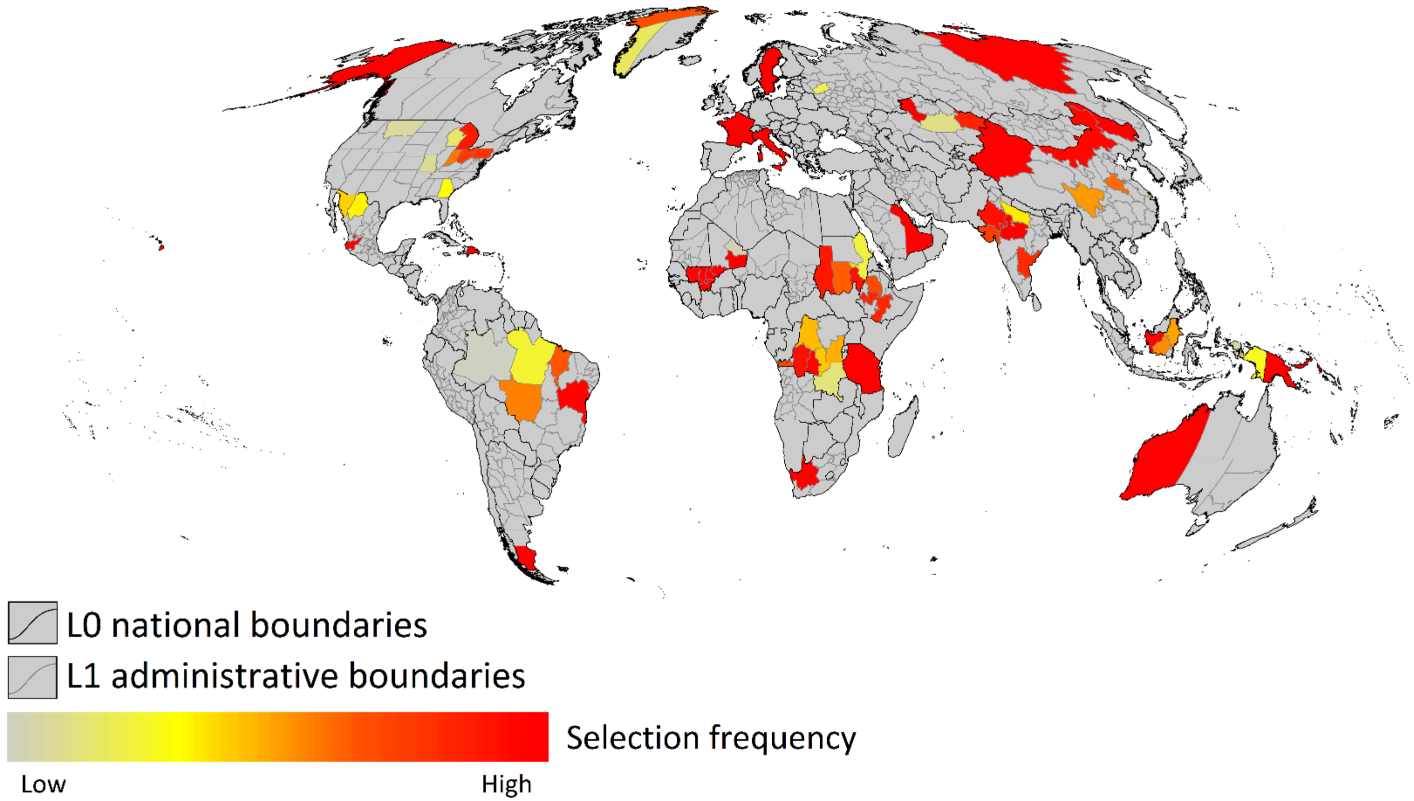


Figure 5.3a. Sample of 100 km² grid squares found in the focal countries (L0) and administrative units (L1) selected by Marxan that best meets 10% targets for biogeographic and conservation area extent factors while minimising sample area.

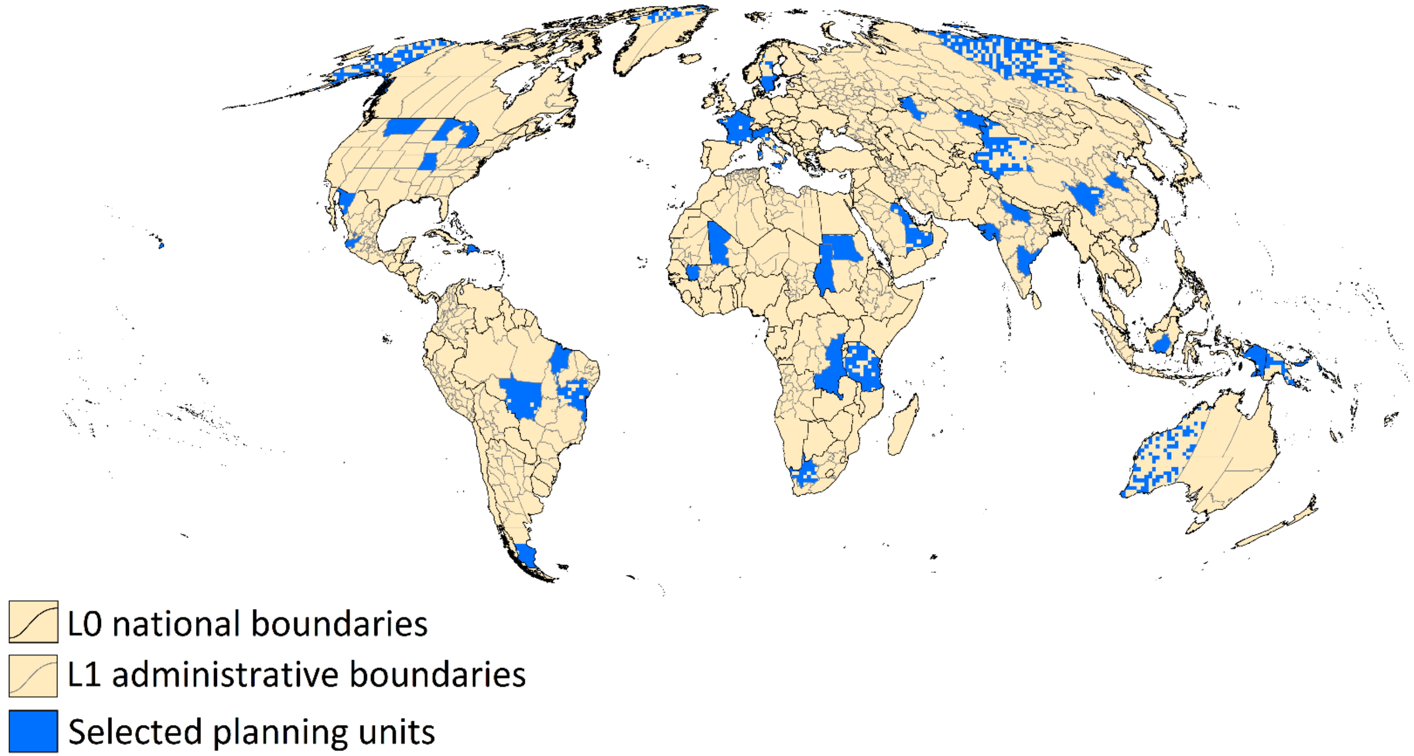
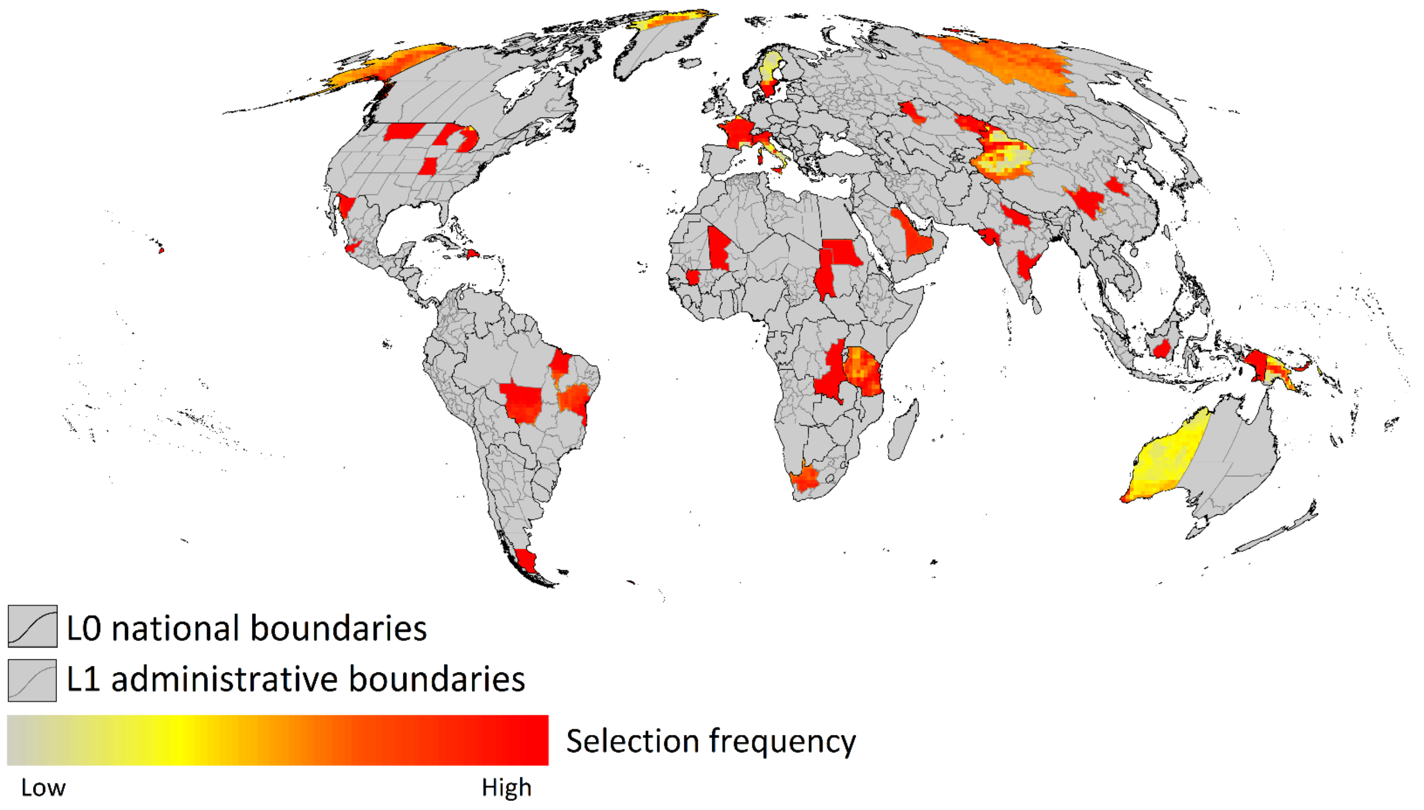


Figure 5.3b. Selection frequency scores from Marxan showing the number of times each planning unit was selected in the 1,000 runs used to identify the best sample.



5.4.4 *Sampling comparison*

The area of the terrestrial realm, excluding Antarctica, in our analysis is 135,306,346 km^2. The sample selected in Stage 1 is 15.5% of this study area and the sample selected in Stage 2 is 10.9%. The WDPA data shows that 14.41% of the terrestrial realm is under protection, compared to 16.9% of the area selected in the Stage 1 analysis, and 14.9% of the area selected in the Stage 2 analysis. The area of the conservation features varied widely, from the subregion of Micronesia that covers 0.003% of the terrestrial realm, to the upper-middle income classification which covers 44.91% of the terrestrial realm (Supplementary Materials Table S5.3).

5.4.5 *Random comparison*

The random selections of planning units never met all the 10% targets for every feature. For Stage 1, the number of targets missed by the random sample ranged from 15 and 49, while for Stage 2 the number of targets missed was 7 to 30.

5.5 Discussion

Well-defined, measurable conservation targets, and accurate on-the-ground data to compare against them, are vital for driving forward progress towards our goal of a sustainable and ecologically healthy future for the planet (Mace et al. 2018). However, obtaining accurate and up-to-date data for the entire globe can be costly and time-consuming (Juffe-Bignoli et al. 2016). An alternative is to make estimates on a sample of features, as pioneered by the Red List Index and other projects that monitor the status of a taxonomically representative set of species (Butchart et al. 2007). In this study we identify a sample of areas that are representative of global biogeographic and socio-economic factors, as a way of guiding the future collection of a subset of data to produce more accurate estimates of progress towards global targets. In this section we first discuss the features and targets we used in our analyses and how that influenced the areas selected, and finish by recommending how this approach could be used both to assess conservation area coverage and more broadly for monitoring global conservation trends.

5.5.1 *Choosing the features and targets*

When choosing features to include in our study, we sought to represent broad patterns of biogeographic diversity and relevant socio-economic conditions across the world. We included the former because we wished to ensure that the final sample was representative of global habitat and species diversity, both so that the sample could be used in future to measure the extent to which conservation areas meet species- and ecosystem-based targets, and because biodiversity patterns and geography may have an impact on whether and where conservation areas are established (Joppa & Pfaff 2009). We included the latter because these demographic, economic and governance factors drive the establishment of conservation areas, so differing socio-economic conditions will result in conservation area networks with differing extents and characteristics (Bohn & Deacon 2000).

The fact that there is an established literature on the factors that determine global biodiversity patterns means we can be confident that our final sample is representative at this global scale (Gaston & Spicer 2013). This is less the case for the socio-economic factors that drive conservation area coverage, although a number of previous studies have shown the importance of elevation, landcover and human population density (Joppa & Pfaff 2009). Other research has highlighted the link between government effectiveness and wealth in determining conservation outcomes (Waldron et al. 2017). Some factors that our expert group identified as potentially important have not been mapped at the global scale, so could not be included in our analysis. For example, political and public support for conservation may have an effect on conservation area establishment (Chapter 4), but data were not available. This may be resolved in future through polling data and citizen science initiatives (McKinley et al. 2017).

We would also have liked to include national land tenure systems as a factor, as this is likely to have a large impact on the extent of privately- and communally-managed PAs in each country (Bingham et al. 2017), but this was also unavailable on a global scale. We would argue that it is an important dataset to be collected in future. However, we did broadly account for differing land tenure types across the globe, as well as other potential factors, by using the geographic subregions dataset. This ensures representation of countries with shared legal, cultural and historical backgrounds. Another issue is that some of our datasets are a snapshot of the current situation but conservation area coverage reflects past and current circumstances, although governments do add or remove conservation areas in response to immediate conditions (Mascia & Pailler 2011; Radeloff et al. 2013).

As well as producing a sample of areas in the world that are broadly representative both biogeographically and socio-economically, the second key aim of our study was to ensure that this representative sample would also make a feasible basis for data collection and further study. This data collection is usually difficult and time-consuming (Juffe-Bignoli et al. 2016), so we clearly needed to make a trade-off between selecting a sample that was large enough to be sufficiently representative, while not so large that collecting data for all areas in that sample would be unfeasible. To produce a representative sample, we decided to use the same percentage target for every feature. We then set these targets as 10% because our sensitivity analyses showed that above 10%, the number of administrative units required to meet higher targets jumped considerably to a level we considered unfeasible for future data collection (Table 5.1). Our decision was based on the untested assumption that there is a relationship between how many countries or administrative units are included in the sample, and the effort data collection would require. More research is needed to assess this trade-off and the potential benefits of using a larger sample.

5.5.2 Defining the planning units and selecting the sample

In Stage 1 we identified countries and large country sub-regions to be included in our sample. This is because the nation state is the functional unit in conservation area data collection and reporting (Dallimer & Strange 2015) but large countries often have sub-national conservation agencies. Thus, we wanted to minimise the number of countries in our sample to minimise the number of agencies and organisations needed for data collection, based on the assumption that each extra country added to the sample would require a similar amount of time and resources to collect the relevant data. In addition, for the largest countries we also assumed that their conservation authorities would have a devolved structure, with national and sub-national agencies, hence our use of sub-national L0 boundaries as planning units for larger countries. To minimise the number of countries selected by Marxan, we used a planning unit cost metric where each country had the same cost. We also modified the Marxan input file detailing the shared boundary between planning units, so the software was much more likely to choose sub-national planning units from the same countries (Ball et al. 2009). Previous analyses have manipulated the Marxan boundary data to ensure the software preferentially selected non-neighbouring planning units that are ecologically connected (Possingham et al. 2005; Hermoso et al. 2011); our analysis did the same but for planning units connected by governance. We chose to use Marxan for the purpose of selecting this sample, because it is specifically designed to identify near-optimal samples of areas which are representative of features inputted and spatially efficient (Ball et al. 2009). Our analyses of randomly selected samples demonstrate that using Marxan produces a better sample because it ensures targets are met.

The best portfolio identified in Stage 1 comprised nine whole countries and 33 administrative units in a further 16 countries, meaning that 25 countries in total were represented. This result, in which 42 planning units were selected, was a slight improvement in efficiency compared to the result for 10% targets in our sensitivity analyses, which selected the same number of countries but 50 planning units overall. The reason for this improvement is because our Stage 1 analysis had 10 times the number of runs as the sensitivity analyses, which gave Marxan greater opportunity to identify more efficient solutions (Ball et al. 2009). The selection frequency scores, which are based on how many times each planning unit was selected in each of the runs, showed there is some flexibility in which countries could be included. Thus, while 17 planning units were selected every time, a further 68 were selected at least once during the analysis (Figure 5.2b), showing that many units are potentially interchangeable. Moreover, even more alternatives would be available if slightly more countries could be included. This is important because if obtaining data from a particular country was impossible for logistical or political reasons, the analysis could be run again to find suitable replacements. For our study, from the solutions with the smallest number of countries or sub-national regions, we selected the one with the most even spread of countries in each continent and the highest mean selection frequency score. We did this to further increase representation and ensure the best areas identified by Marxan were included.

Our Stage 2 analysis chose grid square planning units from within the countries and sub-national administrative units selected in Stage 1, removing areas that are not needed for meeting the targets and reducing the area of the terrestrial realm selected by nearly a third. Most Stage 1 planning units had a large proportion of their area included in Stage 2, apart from Western Australia, Alaska, parts of Siberia and northern China (Figure 5.2a). These remote regions were selected by Marxan to meet targets for low human population density areas and wilderness habitat types, but the large size of these planning units meant these features were then overrepresented in the Stage 1 result. This is reflected in the selection frequency map for Stage 2, which shows there is plenty of flexibility when choosing grid squares within these wilderness zones (Figure 5.2b).

These patterns are also illustrated by the total extent and PA coverage of the areas selected in Stage 1 and Stage 2. Stage 2 was more efficient, covering only

10.9% of the terrestrial realm compared to 15.5% for Stage 1. The Stage 2 result also shows the efficiency benefits of using a complementarity-based algorithm to select areas (Ball et al. 2009), as Marxan was able to meet the 10% targets for each feature in close to 10% of the planning region, even though features belonging to different factors have different spatial distributions. The proportion of land in the Stage 2 sample protected according to the WDPA was also closer to the global figure than the Stage 1 sample, as 14.3% of the terrestrial area outside Antarctica is contained within conservations areas in the WDPA, while for our Stage 1 and Stage 2 samples, the figure is 16% and 14.9% respectively. This was again likely due to the over-representation of wilderness areas in the Stage 1 sample, which generally have higher levels of protection (Pressey & Tully 1994; Joppa & Pfaff 2009). The fact that the percentage of the Stage 2 sample covered by conservation areas included in the WDPA is similar to the global figure is encouraging, suggesting that the sample is broadly representative of the factors driving conservation area coverage, and shows why the Stage 1 sample would not be suitable for making global estimates.

5.5.3 Implications and wider relevance

Ongoing monitoring of progress towards conservation targets is essential but the required data are often lacking (Brooks et al. 2015). Resolving this will need more resources and capacity building, especially at the level of the nation state where most action is carried out and thus where guidance is most needed (Smith et al. 2009). At the same time, we need global estimates of progress to inform international policy. This is currently limited by the quality of some of these national-level datasets, especially with privately- and communally-managed PAs and OECMs (Jonas et al. 2014; Bingham et al. 2017), but also sometimes with state-managed PAs (Visconti et al. 2013). Our proposed solution is to identify a representative sample of countries and collect better data just from them, taking advantage of the availability of accurate information that has not yet been officially approved. Importantly, such a study would not need to report the estimated conservation area coverage for each country, avoiding problems associated with reporting unofficial national datasets. In this study we have shown that it is possible to identify such a representative sample of areas across the globe within a small enough number of countries to make data collection possible. More research is needed on whether this holds true when marine conservation areas are included in the analysis, and the trade-off between the percentage of the terrestrial realm included in the sample and the number of countries and L1 administrative units required to provide data. Nonetheless, our study demonstrates proof of concept and has identified a sample of reasonable size that is also a realistic basis for data collection.

Thus, the first next step is to collect data on conservation areas within the sample we have identified from local experts, NGOs and other non-government sources. This can then be followed by a gap analysis using the new data, to calculate how well this improved measure of conservation coverage represents biodiversity and habitats (Butchart et al. 2015). This will be particularly important for OECMs, as national and regional scale data suggest these may cover different types of biodiversity as compared to state PAs (Chapter 3; Dudley et al. 2018). More broadly, we hope that this sampling approach could be used to produce global estimates of a range of other metrics that currently depend on collating patchy national data, such as management costs and management effectiveness (PAME). In doing so, this approach could help to more effectively monitor progress towards conservation targets and inform international conservation policy.

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Supplementary Materials

Table S5.1: Factors identified in a preliminary workshop as potential features to include in the analysis, due to their influence on biodiversity patterns and conservation area coverage. The following attendees were selected for their expertise and experience in conservation areas: Bob Smith, Zoe Davies, Matt Struebig, Neil Burgess, Naomi Kingston, Piero Visconti, Mike Hoffman, Lincoln Fishpool, Ben Collen and Diego Juffe-Bignoli.

Factor	**Included?**
Major habitat types	Yes – covered by biomes, realms and landcover
Ecoregions	Yes – broadly covered by biomes
Endemism	Yes – broadly covered by biomes and landcover
Continents	Yes
Freshwater	Yes – covered by landcover
Landcover trends	No – data unavailable on a global scale
Rates of habitat conversion	No – data unavailable on a global scale
Rates of forest loss	No – data unavailable on a global scale
Degraded and pristine areas	Yes – covered by landcover
Political stability	Yes – covered by governance indicators
Corruption	Yes – covered by governance indicators
PA management effectiveness	No – data unavailable on a global scale
PA management record	No – data unavailable on a global scale
Land tenure	No – data unavailable on a global scale
PA visitor numbers	No – data unavailable on a global scale
PA investment	Yes – broadly covered by GDP
Latitude	Yes
Islands	Yes
Political groupings, e.g. ex-Soviet, ex-colonial	Yes – broadly covered by UN subregions
Religious groupings	Yes – broadly covered by UN subregions
Sacred sites	No – data unavailable on a global scale
Size of country	Yes – PUs and cost allow small and large countries
Carbon payments	No – data unavailable on a global scale
African, Caribbean and Pacific Group of States (ACP)	Yes – covered by UN subregions
Completeness of WDPA country records	No – data unavailable on a global scale
Age of PA network	Yes – broadly covered by UN subregions
Language groups	Yes – broadly covered by UN subregions
Legal system type	Yes – broadly covered by UN subregions
Climate vulnerability indices	No – data unavailable on a global scale
Within country variability	No – data unavailable on a global scale

Table S5.2: All the factors used in our analysis to define the samples, divided by whether they drive global biodiversity patterns, conservation area extent or both, with our reasons for including each.

Factor	Global biodiversity patterns	Conservation area extent
Biomes	Biodiversity differs greatly between biomes, with ecosystem types sharing similar species compositions (Gaston 2000).	
Islands	Islands are often geographically and biologically distinct, with unique and highly threatened biodiversity (Sadler 1999).	
Latitude	Species composition shows strong latitudinal gradients (Willig et al. 2003).	
Realms	Biodiversity shows strong biogeographic patterns at the global scale (Gaston 2000).	
Elevation	Species composition varies across elevation gradients (Lomolino 2001).	Conservation area extent tends to increase at higher elevations (Joppa & Pfaff 2009).
Landcover	Species composition varies between vegetation types and land-uses (Gaston & Spicer 2013).	Conservation area extent differs between landcover types (Joppa & Pfaff 2009).
Subregions	Biodiversity shows strong biogeographic patterns at the sub-continental scale (Gaston & Spicer 2013).	Sub-sections of continents have relatively similar histories, economies and legislative frameworks (Siegfried et al. 1998).
Government effectiveness		Stable countries with higher bureaucratic quality have greater capacity to expand conservation area networks (Laurance 2004).
Human population density		Conservation area extent is lower in regions with high human population density (Joppa & Pfaff 2009)
Per capita GDP		Wealthier countries have more resources to fund the expansion of conservation area networks (Waldron et al. 2013).

Table S5.3: Details of all the features, their total extent, the proportion of the sample covered by each feature, and the proportion of each feature's total extent that is found in the samples identified in Stage 1 and Stage 2.

Category	Feature	Global extent (km²)	Terrestrial realm covered by feature (%)	Stage 1 sample covered by feature (%)	Stage 2 sample covered by feature (%)	Global extent found in Stage1 sample (%)	Global extent found in Stage 2 sample (%)
Biomes	Tropical & Subtropical Moist Broadleaf Forests	19,847,759	14.67	10.78	13.49	11.40	10.00
Biomes	Tropical & Subtropical Dry Broadleaf Forests	3,017,092	2.23	1.92	2.54	13.36	12.41
Biomes	Tropical & Subtropical Coniferous Forests	711,296	0.53	0.34	0.48	9.98	9.98
Biomes	Temperate Broadleaf & Mixed Forests	12,772,448	9.44	6.64	8.68	10.91	10.00
Biomes	Temperate Conifer Forests	4,075,868	3.01	1.97	2.77	10.16	10.00
Biomes	Boreal Forests/Taiga	15,046,636	11.12	14.11	10.99	19.68	10.75
Biomes	Tropical & Subtropical Grasslands, Savannas & Shrublands	20,285,917	14.99	14.04	15.73	14.53	11.41

Biomes	Temperate Grasslands, Savannas & Shrublands	10,098,291	7.46	4.92	6.86	10.22	10.00
Biomes	Flooded Grasslands & Savannas	1,094,839	0.81	0.63	0.83	12.03	11.13
Biomes	Montane Grasslands & Shrublands	5,203,199	3.85	3.28	3.58	13.23	10.13
Biomes	Tundra	8,206,496	6.07	9.22	7.90	23.59	14.16
Biomes	Mediterranean Forests, Woodlands & Scrub	3,210,402	2.37	3.59	2.34	23.50	10.70
Biomes	Deserts & Xeric Shrublands	27,969,796	20.67	25.21	21.03	18.92	11.06
Biomes	Mangroves	320,823	0.24	0.21	0.23	13.99	10.51
Biomes	Inland water	1,039,692	0.77	0.84	0.96	16.96	13.54
Biomes	Rock & ice	1,973,619	1.46	1.99	1.34	21.16	10.00
Realms	Australasia	9,232,561	6.82	15.77	8.78	35.85	13.99
Realms	Antarctic	11,159	0.01	0.02	0.01	33.00	10.88
Realms	Afrotropics	21,769,183	16.09	13.08	16.22	12.62	10.96
Realms	Indomalay	8,513,981	6.29	4.08	5.79	10.06	10.00
Realms	Nearctic	20,398,341	15.08	12.92	13.86	13.30	10.00
Realms	Neotropics	19,368,174	14.31	10.32	13.16	11.19	10.00
Realms	Oceania	43,247	0.03	0.09	0.10	45.13	32.70
Realms	Palearctic	52,705,510	38.95	40.89	39.84	16.28	11.12
Elevation	0 - 299m	55,813,693	41.25	37.82	39.26	14.23	10.35
Elevation	300 - 799m	43,299,328	32.00	35.59	33.97	17.26	11.54
Elevation	800 - 1399m	19,827,397	14.65	16.57	15.77	17.54	11.71

Elevation	1400 - 1999m	7,279,095	5.38	4.47	4.95	12.88	10.00
Elevation	2000+m	8,627,189	6.38	5.32	5.86	12.94	10.00
Islands	Under 1,000km^2	487,462	0.36	0.29	0.33	12.28	10.00
Islands	1,000 to 10,000km^2	660,808	0.49	0.60	0.50	18.95	11.07
Islands	10,000 to 100,000km^2	1,621,613	1.20	0.86	1.11	11.11	10.03
Islands	100,000 to 1,000,000km^2	5,009,245	3.70	4.04	3.95	16.92	11.60
Islands	Continents	127,492,420	94.23	94.20	94.10	15.51	10.86
Landcover	Croplands	10,044,523	7.42	7.92	9.62	16.54	14.08
Landcover	Croplands mosaic	17,948,478	13.27	10.76	12.81	12.59	10.50
Landcover	Closed forest	25,436,142	18.80	13.85	17.29	11.43	10.00
Landcover	Open forest	12,323,377	9.11	12.82	11.32	21.85	13.51
Landcover	Mosaic grassland / shrubland	26,265,135	19.41	20.05	18.32	16.02	10.26
Landcover	Sparse vegetation	13,551,920	10.02	12.60	9.22	19.52	10.01
Landcover	Flooded forest / grassland	1,902,386	1.41	1.53	1.45	16.92	11.19
Landcover	Artificial surfaces	317,365	0.23	0.20	0.22	12.94	10.04
Landcover	Bare areas	21,608,578	15.97	15.35	15.53	14.91	10.57
Landcover	Water bodies	2,980,599	2.20	2.08	2.18	14.68	10.75
Landcover	Snow & ice	2,913,595	2.15	2.82	2.02	20.29	10.21
Landcover	No data	14,186	0.01	0.02	0.02	29.48	18.71
Latitude	50N to 90N	31,826,862	23.52	27.39	22.89	18.06	10.58
Latitude	30N to 50N	32,126,360	23.74	20.68	21.83	13.51	10.00
Latitude	10N to 30N	26,501,375	19.59	15.65	21.21	12.40	11.78
Latitude	-10S to 10N	20,617,051	15.24	13.90	16.00	14.15	11.42

Latitude	-30S to -10S	18,842,279	13.93	18.04	14.11	20.10	11.02
Latitude	-50S to -30S	5,146,310	3.80	4.04	3.54	16.48	10.13
Latitude	-90S to -50S	214,058	0.16	0.30	0.41	29.83	28.50
Income classification	Low income	14,417,961	10.66	11.06	14.00	16.10	14.29
Income classification	Lower middle income	22,038,475	16.29	12.63	15.92	12.03	10.63
Income classification	Upper middle income	60,767,325	44.91	40.77	41.32	14.08	10.00
Income classification	High income	38,082,584	28.15	35.54	28.76	19.59	11.11
Population density	0 to 0.9	53,883,215	39.82	46.44	39.53	18.09	10.79
Population density	1 to 9.9	39,359,881	29.09	28.03	31.21	14.95	11.67
Population density	10 to 99.9	27,781,643	20.53	16.15	18.89	12.20	10.00
Population density	100 to 999.9	9,292,487	6.87	5.54	7.02	12.52	11.11
Population density	1000+	1,070,380	0.79	0.56	0.73	11.05	10.10
Government Effectiveness	0 - 24.9	23,463,373	17.34	11.34	15.95	10.15	10.00
Government Effectiveness	25 - 49.9	48,702,809	35.99	33.62	34.59	14.49	10.45
Government Effectiveness	50 - 74.9	28,037,538	20.72	23.55	24.89	17.64	13.06
Government Effectiveness	75 - 100	35,102,625	25.94	31.49	24.58	18.83	10.30
Realms	Water & ice	2,832,017	2.09	2.52	2.01	18.66	10.44
Subregions	Australia and New Zealand	7,985,635	5.90	12.08	5.43	31.75	10.00
Subregions	Caribbean	233,427	0.17	0.23	0.33	20.70	20.62

Subregions	Central America	2,481,651	1.83	1.25	1.70	10.57	10.05
Subregions	Central Asia	4,380,003	3.24	2.29	2.98	10.99	10.00
Subregions	Eastern Africa	7,049,679	5.21	4.52	4.79	13.45	10.00
Subregions	Eastern Asia	11,598,707	8.57	10.87	9.46	19.68	12.00
Subregions	Eastern Europe	18,604,967	13.75	14.62	12.64	16.49	10.00
Subregions	Melanesia	544,908	0.40	2.22	1.38	85.63	37.36
Subregions	Micronesia	3,576	0.00	0.00	0.00	27.68	15.27
Subregions	Middle Africa	6,608,246	4.88	3.55	5.01	11.27	11.15
Subregions	Northern Africa	7,647,985	5.65	4.80	6.74	13.18	12.97
Subregions	Northern America	21,581,549	15.95	14.61	14.67	14.21	10.00
Subregions	Northern Europe	1,803,994	1.33	2.13	1.23	24.83	10.00
Subregions	Polynesia	8,613	0.01	0.02	0.01	46.37	14.01
Subregions	South America	17,845,353	13.19	9.80	12.40	11.53	10.23
Subregions	South-eastern Asia	4,483,416	3.31	2.24	3.05	10.50	10.00
Subregions	Southern Africa	2,681,065	1.98	1.73	1.82	13.54	10.00
Subregions	Southern Asia	6,710,677	4.96	3.35	4.75	10.49	10.41
Subregions	Southern Europe	1,316,461	0.97	1.43	1.11	22.85	12.38
Subregions	Western Africa	6,082,789	4.50	2.99	4.20	10.33	10.15
Subregions	Western Asia	4,528,985	3.35	2.62	3.08	12.16	10.00
Subregions	Western Europe	1,102,673	0.81	2.62	3.24	49.81	43.21

Figure S5.1: Map of the planning units used in the Stage 1 analysis, distinguishing countries (L0) with an area $<1,000,000$ km^2 which were entered into the analysis as single planning units, and countries with an area $\geq 1,000,000$ km^2 which were divided into planning units defined by sub-national administrative units (L1).

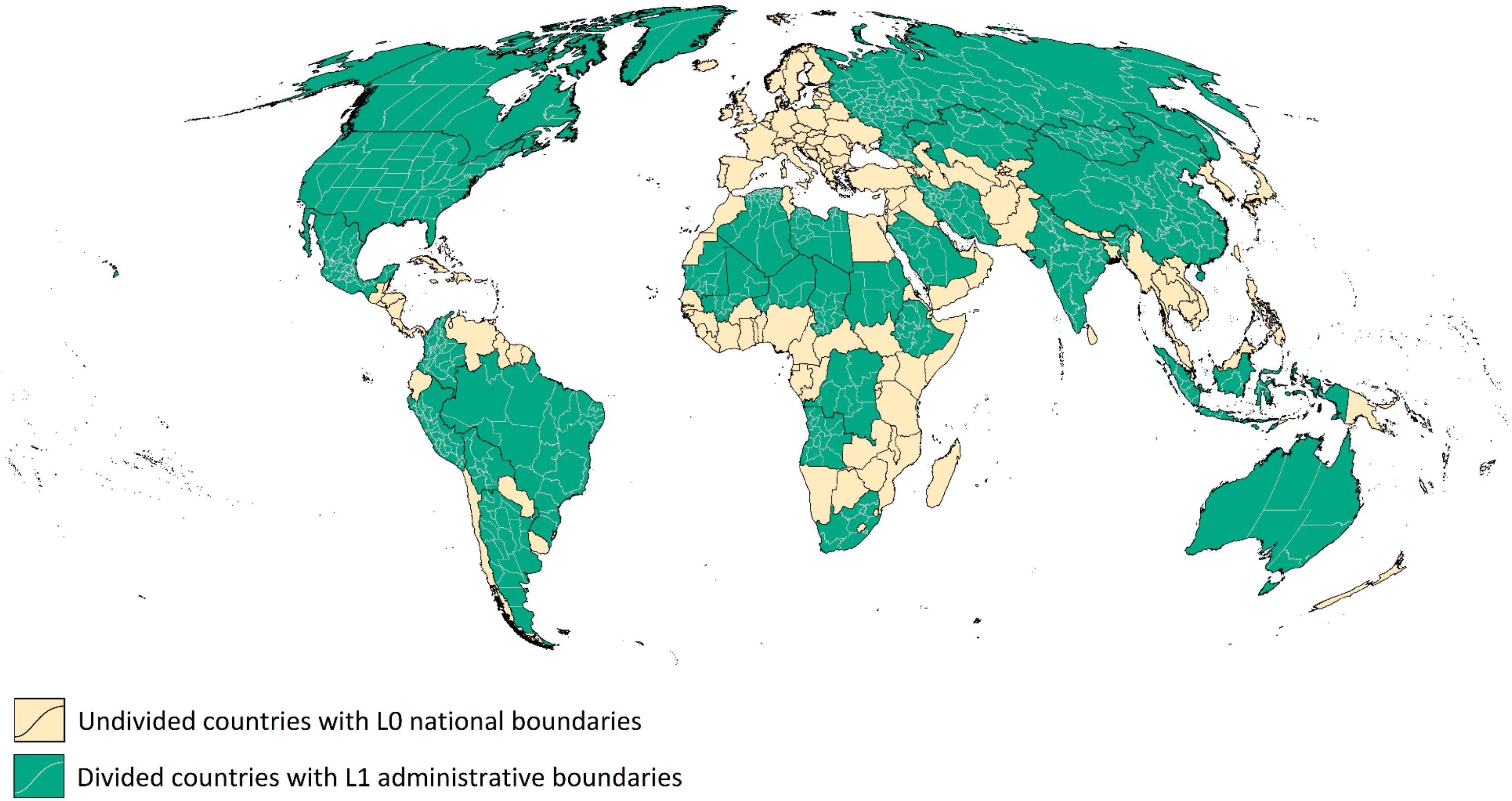
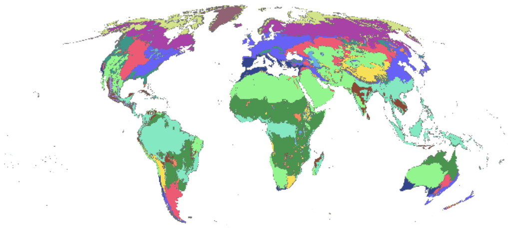
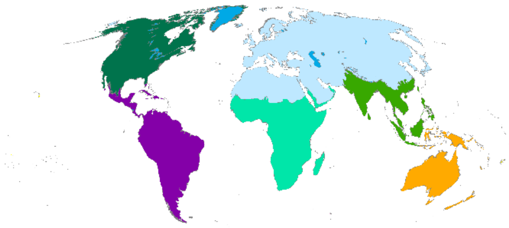


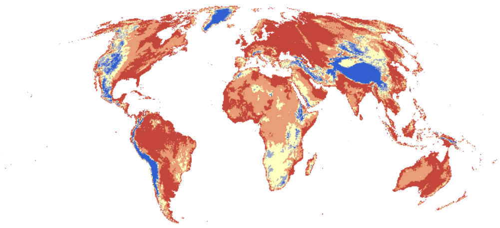
Figure S5.2: Maps of 9 of the 10 factors used to represent drivers of global biodiversity patterns and conservation area extent in the Stage 1 and Stage 2 analyses. The 'islands' factor is not displayed because of the small size of the features.



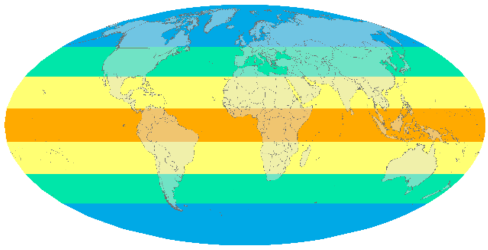
Biomes



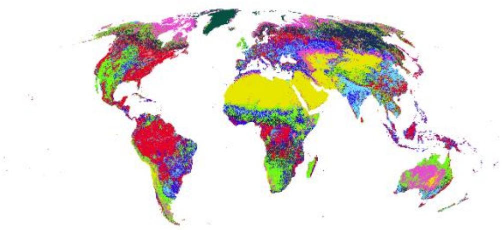
Realms



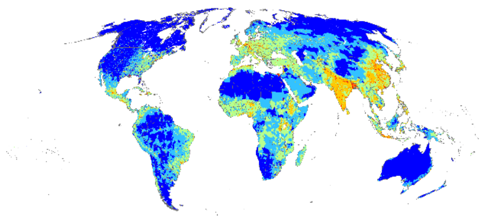
Elevation



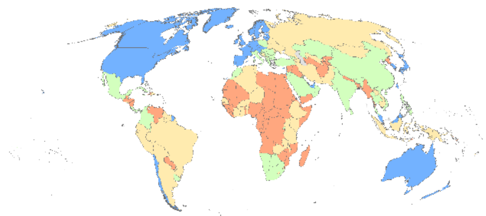
Latitude



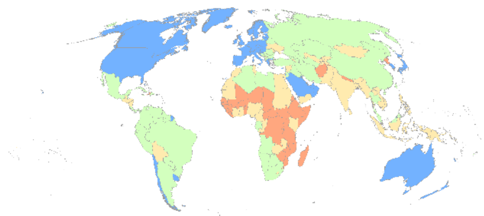
Landcover



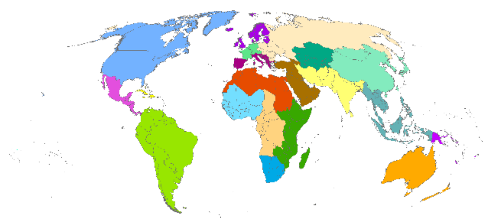
Population density



Government effectiveness



GDP



Subregions

Chapter 6. Discussion

6.1 Introduction

Protected areas have long been, and will remain, a staple of efforts to conserve the natural world (Watson et al. 2014). While their success is never guaranteed, they have been shown to be generally effective at preventing the loss of species and habitats, even in less than ideal conditions (Geldmann et al. 2013). Nonetheless, we urgently need more, and better-functioning, conservation areas across the globe to stem the rates of decline and extinctions. To do so, the planning of new areas must be improved, such that the resulting sites are representative of the regional biodiversity and sufficiently connected to neighbouring areas to create a functioning network of natural processes on a large scale. In addition, it is necessary to consider the wider context and the circumstances that really underpin conservation success and to provide robust metrics to monitor progress.

In this thesis I have covered a range of topics that address these key issues, using national-level case studies to understand and illustrate the important factors that need to be considered when understanding the effectiveness of conservation area networks, developing a conceptual framework that brings together the relevant literature on conservation area network expansion, and then proposing a new approach for measuring conservation area extent at the global level. In this final chapter I will explain how these different studies have contributed to the literature, discuss the research limitations and suggestions for further work, and then provide recommendations for practitioners.

6.2 Contributions to the literature

6.2.1 Chapter 2

Systematic conservation planning (SCP), an approach designed to produce effective and efficient conservation areas, is now widely used internationally in a great variety of landscapes and seascapes (Watson et al. 2011). It has been applied in an

English terrestrial context to investigate the value of biodiversity surrogates (Prendergast 1993; Hopkinson et al. 2000; Franco et al. 2009) or on a broad but coarse scale (Isaac et al. 2018). The Marine Conservation Zone Project (JNCC & Natural England 2010) produced guidelines for conservation in UK seas informed by SCP, but did not publish a national plan of priority areas. This chapter is part of a new effort to apply SCP to an English terrestrial context, and the first analysis designed specifically to inform the development of conservation areas in England. It follows a recent study conducted with Natural England to test the suitability of the approach for identifying priority areas for the expansion of the network of National Nature Reserves (Pett et al. *in press*), which found that a systematic planning approach was certainly applicable, but further refinements were necessary to adapt it to an English context of highly fragmented habitats.

Our study is thus the first to use a fine scale spatial conservation prioritisation approach for England, in which remaining habitat fragments are used to define the planning units, rather than solely a grid of large regular units. In addition, it is the first to use MinPatch (Smith et al. 2010) as part of a spatial conservation prioritisation analysis for England, in which the results of the initial prioritisation produced by Marxan (Ball et al. 2009) are then grouped into a network of larger and thus more practicable patches of land. We are also the first to be able to show, in our preliminary national plan, how many patches each conservation feature is represented in, which is vital to the overall resilience of the network. In using MinPatch, this is one of the first studies to begin putting to practice the principles defined in the Lawton review (Lawton et al. 2010), which stated the need for 'more, bigger, better, joined' conservation areas to enhance the resilience and coherence of England's ecological network. Our analyses also help to assess the potential costs of a significant expansion of the NNR network in terms of the area that would be required to meet more or less ambitious targets, and what the opportunity costs to agriculture could be.

6.2.2 Chapter 3

Systematic conservation planning approaches have long been used in South Africa to guide conservation actions (Balmford 2003; Knight et al. 2006; Smith et al. 2006; Rivers-Moore et al. 2011) and some studies have assessed the contribution of private conservation areas in particular to the overall network (Gallo et al. 2009; Maciejewski et al. 2016; Clements et al. 2019). This study is the first to bring together data on a variety of different non-state conservation areas in South Africa and analyse their contributions in concert, rather than focusing on only a single governance type. In doing so we deliberately highlight their role in increasing both absolute coverage and representation of habitats that are either under-represented in or entirely absent from the state conservation area network. Furthermore, this study is the first to use protection equality analysis in a new, target-based way to assess the representativeness of KwaZulu-Natal's conservation area network while taking into account the relative vulnerability of different priority habitats as reflected in the different protection targets assigned to each. Previous studies measuring protection equality have focused on absolute equality, under which the focus is on whether conservation features are represented equally, regardless of how much protection each feature actually needs (Barr et al. 2011; Shwartz et al. 2017; Chauvenet et al. 2017). Thus the method we demonstrate here is an improvement on these studies, because it provides a more realistic assessment of how well a network is representing biodiversity.

6.2.3 Chapter 4

This chapter is the first conceptual framework developed to examine the drivers of conservation area establishment globally, and thus to explain why conservation area networks differ so greatly between countries. Research to date has largely focused on a few particular factors that influence the success or otherwise of conservation area networks (Struhsaker et al. 2005; Nolte et al. 2013; Cetas & Yasué 2017), while others have identified some political (Radeloff et al. 2013) or ecological (Walston et al. 2010; Wei et al. 2015) triggers for major increases in conservation area networks in a number of countries. Our study is therefore among the first to attempt to bring this literature together and to highlight the conditions that should be encouraged to enable greater conservation area coverage in future.

6.2.4 Chapter 5

A key hindrance to the monitoring of progress towards global conservation targets is the difficulty in collecting sufficient accurate data in a timely and cost-effective manner (Juffe-Bignoli et al. 2016). This study is the first to develop a methodology aiming to overcome this hurdle, by producing a representative set of areas across the global which can be used as the basis for data collection. Building on our conceptual framework in the previous chapter, this study is one of the first to explicitly consider the factors that determine the extent and characteristics of conservation area networks; not only biological factors but also socio-economic and political. By allowing the collection of a manageable but representative subset of global data, we enable more accurate estimations of global patterns in conservation area coverage to be made. This approach is also applicable to data collection for other global conservation targets such as management effectiveness (Leverington et al. 2010) and conservation area funding levels (Bruner et al. 2004). Thus we are also first to consider the trade-off between data requirements and the resources needed to collect it, and to then produce a method specifically designed to tackle this trade-off in a satisfactory and efficient way.

6.3 Limitations and further research

6.3.1 Chapter 2

Our SCP analysis of England included only priority habitats as conservation features because the available species data were at too coarse a spatial scale to inform action on the ground. We also used uniform targets and differing levels of low, medium and high, without supporting research to suggest what level would be sufficient to ensure persistence. Any future analysis to identify priority areas for conservation would benefit from the inclusion of species data, as the priority habitats may not be satisfactory surrogates for threatened fauna and flora (Rodrigues & Brooks 2007; Grantham et al. 2010). In addition, more accurate targets should be developed, through a review of the literature and expert opinion, to better reflect the characteristics and vulnerability of each habitat and species (Pressey et al. 2003). Our cost metric, calculated using agricultural land quality data, was also fairly simplistic and could be improved with the inclusion of more detailed opportunity cost data encompassing a wider range of economic activities, or data on potential costs of land purchase and management (Pett et al, *in press*).

6.3.2 Chapter 3

Our gap analysis of KwaZulu-Natal also did not include species as conservation features, only vegetation types and elevation zones, because the data were unavailable at the time. However, we did have specific targets set for the vegetation types by Ezemvelo KZN Wildlife, the provincial conservation authority, and species-specific targets are available too. Thus any future gap analysis of the conservation area network could be improved through the inclusion of species data. In addition, it is not clear currently which conservancies in KZN are active and which are not. Thus it may be helpful to seek out further information on the status of each, to ensure that any no longer undertaking any conservation work are excluded from the study.

6.3.3 Chapter 4

There is an abundance of literature on all aspects of protected area planning, management and assessment, but the literature on establishment and the conditions that support it is much more limited. This is not surprising given the myriad factors that underlie the political and personal decisions needed to establish new conservation areas, which can be obscured or hard to pin down. Thus there are few studies which can draw a direct link between particular factors and the establishment of new conservation areas. As a result, our conceptual framework necessarily contains some assumptions about the potential impacts of certain factors which lack a substantial body of literature to support them. Nonetheless, our aim was not to produce an exact formula of supporting conditions that will apply in every circumstance, but to bring together and shed light on some lesser-considered factors that have a significant influence on the growth, or otherwise, of conservation area networks across the world.

6.3.4 Chapter 5

Our literature review and preliminary workshop with experts that we conducted prior to beginning the sampling project produced a list of 30 factors that could potentially be included in the selection of a representative sample. Of these, we used 10 in our study, while those excluded were either broadly accounted for by the other factors chosen, or were unavailable on a global scale. The sample thus could be refined if new global datasets on factors such as land tenure or support for conservation were to become available. Any further study could also look further at the trade-off between the size of the sample and the cost of data collection in terms of time, effort and funding, to see if a larger sample could be produced without an excessive increase in data collection effort.

6.4 Recommendations for practitioners

6.4.1 Chapter 2

The results of this study are not a finished template on which to base an expansion of the NNR network; further development and refinement will be required. Nonetheless, the SCP approach has numerous benefits (Smith et al. 2006). It enables conservationists to make best use of limited resources in the design of efficient and effective PA networks, and requires them to be explicit and transparent about their priorities and objectives and the trade-offs that will be required as part of the planning process. Furthermore, it is repeatable and adaptable to different contexts and goals. Therefore SCP should be taken up and fully incorporated into the process of planning new PAs and Nature Recovery Networks in England. In addition, fine scale distribution maps of conservation features – species, habitats and ecosystem services – should be produced, as well as specific targets for each based on their remaining extent and vulnerability.

6.4.2 Chapter 3

Our study of KwaZulu-Natal's conservation area network clearly demonstrates the potential of conservation areas outside the state network to make a unique contribution to the representation and conservation of biodiversity. We recommend that practitioners pay more attention to these alternative options for area-based conservation. There is clearly a desire and aptitude for conservation among private and communal landowners (Bingham et al. 2017; Corrigan et al. 2018), which conservationists must do more to engage with, in addition to collecting more data on where these conservation areas are and what biodiversity value they hold. If such landowners can be brought 'into the fold', and encouraged and aided in undertaking effective biodiversity conservation action on their land, they could be an enormous asset in global efforts to halt biodiversity loss (Corrigan et al. 2018). In addition, we recommend that target-based protection equality analysis should be used more broadly, to improve assessments of conservation area representativeness (Barr et al. 2011). This may be particularly important post-2020, as efforts continue to ensure that conservation areas are effective.

6.4.3 Chapter 4

In our conceptual framework we have shown the possible role of various non-biological factors in the growth of conservation areas, and thus highlight the conditions practitioners should (to the extent they are able) encourage to help create an enabling environment. This includes political engagement and consideration of the impacts of conservation on people, which can be negative (Oldekop et al. 2016). Practitioners must address these negative impacts and develop a more sophisticated understanding of what conservation means to those outside it, who do not necessarily consider growth in conservation areas to always be a positive development.

6.4.4 Chapter 5

While area-based targets are sometimes criticised for being crude and simplistic, they are likely to remain a core part of international efforts to improve biodiversity conservation for the foreseeable future (Woodley et al. 2019a, 2019b). Post-2020, calls for a 30% target to be achieved by 2030 are gathering significant support (IUCN 2019), and so the need for effective methods of data collection and monitoring of progress continues. Thus the next step following our development of a sampling methodology is to collect the data on conservation areas in our sample. Long term, ensuring adequate funding for capacity building and data collection is essential to help produce comprehensive and accurate national datasets on conservation areas.

6.5 Conclusion

The coming decades will see the fight to sustain the natural world intensify ever further as we attempt to tackle the enormous challenges of climate change, population growth and habitat loss. There is no single solution that can usher in a new, sustainable and harmonious world on its own, and working towards the creation of such a future is the responsibility of every sector of society. But for conservationists, the continued support for and improvement of conservation areas will be a vital part of our role in minimising the loss of precious species and habitats, and integrating healthy natural systems into human societies.

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