Conservation and Social Outcomes of Privately Protected Areas

Rachel Hannah Palfrey

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Confirmation of authorship

The candidate confirms that the work submitted is their own, except where work which has formed part of jointly authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below. The candidate confirms that appropriate credit has been given within the thesis where reference has been made to the work of others.

PhD publications

Parts of this thesis have been published in the following:

Chapter 2: Palfrey, R., Oldekop, J. & Holmes, G. (2020) Conservation and social outcomes of private protected areas. Conservation Biology. DOI: https://doi.org/10.1111/cobi.13668

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Chapter 4: Palfrey, R., Oldekop, J. & Holmes, G. (*in prep*) Effectiveness of privately protected areas to reduce deforestation in Colombia,

I am lead author on the above articles. They all originate from my PhD research meaning I designed the research questions, methodology; as well as collected and analysed the data. These articles were co-authored with my supervisors whose role was in the recommendation of revisions and edits to manuscripts.

Rationale for thesis by alternative format

This thesis explores the link between the governance of protected areas (PAs) and their environmental and social outcomes. In particular, this thesis focuses on the outcomes of protected areas which are governed by private actors. It uses three distinct methodological approaches to answer three distinct research questions:

1. A literature review to determine what we already know about the environmental and social outcomes of privately protected areas (PPAs)
2. Mapping methods to determine global spatial contributions of privately protected areas
3. Counterfactual analysis and matching methods to determine the effectiveness of privately protected areas in reducing deforestation and degradation within Colombia

Answering these research questions was therefore more suited to thesis by alternative format than a traditional thesis approach.

This thesis consists of an introductory chapter outlining the background and rationale for the research, a placing of the study within the wider literature, an identification of the research gaps being addressed and consequent contributions of this study and a summary of the research strategy, data collection and analysis procedures. This is followed by the three papers which are listed above. The first paper explores what we already know of the environmental and social outcomes of PPAs. The second and third papers focus on the environmental contributions of PPAs. The second examines to what extent PPAs contribute to increasing the overall area of the global PA estate, to enhancing ecosystem representativeness and to improving PA network connectivity. The third evaluates the effectiveness of PPAs to reduce deforestation in Colombia and how this compares with other PA governance types. The three results chapters are followed by a discussion and conclusion which brings together insights from the three papers and offers suggestions on how conservation actors and policy can assist in enhancing the contributions of PPAs to the global PA estate. The conclusion also reflects on the research approach, limitations of the thesis and possible future research directions.

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When I was 12 someone close to me told me that I would never to amount to anything, and I would be lucky to make it as a shelf stacker in Tesco.

Those few words have had a profound impact on my life and everything from that moment has led to the completion of this thesis and proving to them, but most of all to myself, that I can achieve whatever I set out to do.

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I would like to dedicate this thesis to anybody from the working classes who has ever been told that they will never achieve anything because of their background or where they come from. May it be proof that with a little bit of skill, a lot of determination and a few goodhearted people around you, you can achieve whatever it is that you wish to do. Finally, I would like to end on a quote:

“The worst fault of the working classes is telling their children they’re not going to succeed, saying: There is life, but it’s not for you.” ~ John Mortimer

Abstract

Global conservation governance is changing. A broad array of diverse actors are increasingly manging and implementing conservation interventions. In particular, the number of private actors implementing conservation initiatives is growing. Despite being an old conservation approach, privately protected areas (PPAs) have historically been understudied and many uncertainties remain regarding private actor involvement in biodiversity conservation. This thesis tackles some of these uncertainties by seeking to understand what the outcomes of PPAs are and how do these differ to those of other PA governance types. Using open-source data, this thesis takes a big data approach to increase the understanding on the outcomes of PPAs in terms of (i) what they protect, (ii) how effective they are at protecting it and (iii) what are the social impacts of these areas for landowners and local surrounding communities.

This thesis suggests that PPAs can play an important part in achieving conservation targets by increasing the coverage, complementarity and connectivity of PA networks. PPAs can protect areas where other PA governance types struggle to be implemented, in particular; unrepresented and threatened biomes and areas of high human pressure (see Chapter 4). Moreover, PPAs can be more effective than other governance types at mitigating threats (see Chapter 5). This thesis finds that PPAs in Colombia are effective at reducing deforestation and on average, are more effective than regional state PAs (see Chapter 5). PPAs can also have positive social impacts for landowners by improved social networks, increased property value, or a reduction in taxes. However, similarly to state PAs in some cases local communities surrounding PPAs may incur costs, including a reduction in social capital and loss of cultural identity (see Chapter 3).

Findings from this study have important implications for (i) the role of PPAs in meeting conservation biodiversity targets and (ii) general theories surrounding the role of private actors in conservation. Results suggests that PPAs deserve to be better integrated into regional, national and global biodiversity strategies as they can offer beneficial environmental biodiversity outcomes. Moreover, better integration and regulation of PPAs into biodiversity conservation strategies may increase PPA accountability and help to limit

negative social impacts such as inequalities in land ownership and the use of perverse economic incentives.

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List of acronyms and abbreviations

IUCN - International Union for Conservation of Nature

KBA – Key Biodiversity Area

NGO – Non-Governmental Organisations

OECMs – Other Effective Conservation Measures

PA – Protected Area

PAME – Protected Area Management Effectiveness

PPA – Privately Protected Area

PRISMA - Preferred Reporting Items for Systematic Reviews and Meta-Analyses

REDD+ - Reducing Emission for Deforestation and Degradation

RPPN - Reservas Particulares do Patrimônio Natural

UNEP – United Nations Environment Program

WCMC – World Conservation Monitoring Centre

WWF – World Wildlife Fund

WDPA - World Database on Protected Areas

Chapter 1: Introduction

1.1 Research Context

Biodiversity supports all life on the planet (Rockström *et al.* 2009; SCBD 2010; Millennium Ecosystem Assessment 2005). However, biodiversity loss is occurring at an alarming rate. Species extinctions are estimated to be 100 to 1,000 times greater than background levels (Diaz et al. 2019). Between 1970 and 2016, the population of vertebrate species fell by 68% on average worldwide (WWF Living Planet Report, 2020).Some scientists suggest that we are now entering a sixth mass extinction event (Barnosky et al. 2011). Halting biodiversity loss is therefore a global challenge.

The five main threats to global biodiversity have been identified as: changes in land and sea use, species overexploitation, invasive species and disease, pollution and climate change (The WWF's Living Planet Report, 2020). Changes in land and sea use account for the largest portion of biodiversity loss, making up 50% of recorded threats to biodiversity on average. Species overexploitation is the second biggest threat at 24% on average, whilst invasive species is third accounting for 13% of recorded threats to biodiversity.

In 2010, the Convention for Biological Diversity set out twenty "*Aichi Biodiversity Targets*" to support actions to conserve biodiversity and mitigate threats. In particular, Target 11 called for 17% of the world's surface to be conserved by protected areas (PAs) or other effective area-based conservation measures (OECMs). PAs are heralded as the cornerstone of global biodiversity conservation efforts and are a long-standing tool in the pursuit of nature conservation. Europe has had PAs for centuries in the form of royal hunting grounds and forest reserves (Hamin, 2002; Zupancic-Vicar ,1997), yet arguably the modern approach to PAs stems from the creation of Yellowstone national park in the USA in 1872 (Selman, 2009). PAs could play a key role in reducing the two largest threats to biodiversity by either enforcing strict protection allowing no changes to habitat within their borders or any hunting activities (thus preventing land and sea use change or species overexploitation) or by closely regulating activity sustainable use PAs reducing the overall impacts of land and sea use change and reducing any hunting or extractive activities to sustainable levels. PAs can also have management plans which may include the removal of invasive species and remediation of pollution within their borders and PAs may help reduce climate change by protecting environments such as rainforests, peatlands and sea grass which act as carbon sinks.

Protected Areas (PAs) numbers have been increasingly rapidly, particularly over the past decade (Maxwell et al., 2020) and 257,889 PAs are currently reported in the World Database of Protected Areas (WDPA). PAs cover approximately 15.4% of the terrestrial and 3.4% of the marine biosphere (UNEP-WCMC, IUCN & NGS, 2021). The IUCN defines a PA 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*".

Despite the rapid proliferation of PAs, we know relatively little about their outcomes. Conservationists need to evaluate the outcomes of PAs because (i) there are growing pressures on biodiversity, (ii) increasingly PA funders require evidence of PA success and (iii) resources for conservation are inadequate and therefore it is vital to ensure those that are available are used as efficiently as possible (James et al., 1999). PA evaluation can be defined as the collection of information about the characteristics, activities and outcomes of PAs for the purpose of making judgements about PA effectiveness, improving PA performance, and/or informing decisions about future PA development. Historically, conservationists have failed to evaluate the outcomes of PAs and reasons for PA failures. Of the 257,889 PAs reported to the WDPA only 28,283 (11%) PAs have a protected area management effectiveness (PAME) evaluation reported. This is equivalent to about 11% of all PAs in the WDPA (UNEP-WCMC, IUCN & NGS, 2021). A lack of evaluation is not limited to just PAs. Across all conservation interventions Catalano et al., (2019) found that only 59 peer-reviewed articles have been published which analyse the failures of conservation programs. This failure to evaluate PA effectiveness and reasons for failures may help to explain why although PA coverage has increased significantly over the past 10 years, and PA coverage is approaching percentage targets in many parts of the terrestrial and marine realms (Butchart et al., 2015), global biodiversity is still declining (WWF, 2016). Alarmingly, declines are also being seen within PA boundaries (Laurance et al., 2012). The discrepancy between increasing PA coverage and negative biodiversity trends has resulted in a surge of interest into how to evaluate PAs and their outcomes (Kapos et al., 2008).

The current IUCN definition of a PA and most studies into PA outcomes treat PAs as apolitical spaces (Dudley, 2008). However, PAs are more than just geographical spaces. Within this thesis I consider PAs as institutions. That is to say, I think of PAs as political entities; each with a unique set of rules, regulations and regulators that condition or compel people to behave in certain ways within their boundaries. For example, some PAs may have rules that prevent all public access and extraction of resources (e.g., a strict nature reserve), some may allow restricted access for certain purposes (e.g., a private ecotourism reserve), and some may have open access and allow multiple uses and extractions of resources (e.g., a UK national park) (Stolton et al., 2014). The rules and regulations of a PA can be formal and legally enforced or informal and enforced via social pressure and norms. These rules can be drafted and imposed by a diverse array of actors including federal and local government, indigenous communities and private organisations or individuals. The actors who make and impose the rules of a PA are known as the PAs governors. This main question I explore in this thesis is *"how does <u>who</u> governs a PA impact its outcomes?"*

I focus on PAs governed by private actors. These areas are referred to as privately protected areas (PPAs). There are numerous definitions for PPAs (Holmes 2013), but Stolton et al. (2014) provide a comprehensive and widely accepted definition that I use within this thesis: "*a protected area, as defined by IUCN, under private governance (i.e. individuals and groups of individuals; non-governmental organizations; corporations – both existing commercial companies and sometimes corporations set up by groups of private owners to manage groups of PPAs; for-profit owners; research entities (e.g. universities, field stations) or religious entities)."* I focus on PPAs for several reasons; (i) PPAs may be increasing in number due to rising trends in neoliberal conservation approaches that facilitate a role for private actors (Büscher and Whande, 2007), (ii) because there is a pressing need for conservation on private land to help achieve global conservation goals (Kamal et al., 2015), (iii) because they are the least well understood form of PA (Holmes, 2013), and (iv) because PPAs may have different outcomes than those of other PA types (see Section 1.4 – Private Governance).

Although there are many different theories on how the outcomes of PPAs may vary compared to those of other types of PAs, very little empirical evidence exists. This may be because in some countries PPAs are poorly recorded and are not legally recognised (Bingham et al., 2017) (see Section 2.4.1). The lack of evidence in understanding how PPAs work has led to numerous calls for rigorous evaluations of PPA performance (e.g., Schleicher et al., 2018). My thesis aims to provide a piece of the puzzle by undertaking a comprehensive literature review to pool together what we currently know about PPAs outcomes and by conducting empirical research to determine how PPAs contribute to the global PA conservation estate and how effective they are reducing deforestation within Colombia (a global biodiversity hotspot). This knowledge can help us determine: (i) what are the potential outcomes of PPAs (chapter 3), (ii) where PPAs make the greatest contributions to the global conservation estate (chapter 4), and (iii) whether PPAs are effective (chapter 5). It is critical we gain this information to help facilitate the integration of private actors into conservation to improve the diversity, quality and vitality of PAs, and maximize the potential of PPAs to successfully conserve biodiversity for future generations.

The remainder of this section is structured as follows. Firstly, I give a general overview of the environmental and social outcomes of PAs. Secondly, I discuss the three main factors that affect PA outcomes and lastly, I explore the possible benefits and criticisms of private governance, and state the research justifications, aims and objectives.

1.2. Protected Area Outcomes

The following section provides a general overview of the impacts of all PA governance types. I provide this overview because: (i) most studies fail to disaggregate PA outcomes by different PA types; (ii) the majority of issues and outcomes discussed within these following sections are applicable to all PAs (although some issues may be more applicable to some PA types than others); and (iii) to act as a reference with which to compare the unique outcomes and contributions of PPAs explored within the following parts of this thesis. Although PAs can have multiple related outcomes, for the purpose of brevity and clarity I divide PA outcomes into two main categories: i) environmental outcomes and ii) social outcomes.

1.2.1. Environmental outcomes

Most studies into the environmental outcomes of PA use quantitative spatial methods and report the environmental outcomes of PAs in two main ways: (i) what PAs protect, and (ii) how effective they are at protecting it (Rodrigues & Cazalis, 2020). With regards to the former, outcomes are reported as to what extent PAs represent or fail to represent particular biodiversity features (e.g., threatened species, ecoregions) within their boundaries. With regards to the latter, outcomes on PA effectiveness are reported in a multitude of ways. Firstly, how well are PAs connected to allow species movement, secondly, to what extent do PAs reduce land cover change (e.g., deforestation and degradation), thirdly, to what extent do they restore degraded ecosystems and lastly, how do species populations (both richness and abundance) either change after PA establishment or compare to areas outside PA boundaries. Here, I explore these approaches in turn to demonstrate what we currently know about the impacts of PAs in general. **Error! Reference source not found.** summarises what and how PA impacts are measured.

1.2.1.2 Ecological Representativeness

PAs are unevenly distributed and are biased towards areas that are remote and have lower opportunity costs, higher elevations and steeper slopes (Joppa & Pfaff, 2009). This is because historically, there has been a lack of planning and PAs have been established in an opportunistic, ad hoc manner (Joppa & Pfaff, 2009), based on aesthetics (e.g., Yosemite National Park) or created for political reasons and for militarisation and control of land (e.g., Argentine NPs in Patagonia (Martin and Chehebar, 2001)). As such, we do not yet fulfil global conservation targets to design and produce an ecologically representative PA network (Joppa & Pfaff, 2009). Certain biomes are under and overrepresented within the global PA estate. Gap analysis is used to assess the representativeness of a given PA network and to determine if a particular species, biome or ecosystem function is under-represented (Jennings, 2000). Different metrics have been used to assess gaps in existing PA networks. Studies show protection levels for global biomes are highly variable. Protection ranges from 4% for temperate grasslands, savannas and shrub lands to 25% for temperate conifer forests (Hoekstra et al., 2004). Six of the 14 biomes have less than 10% of their total area protected (Jenkins and Joppa, 2009). Key Biodiversity Areas (KBAs) are sites contributing significantly to the global persistence of biodiversity. 19% of KBAs are estimated to be completely covered by PAs, while 39% have no protection (UNEP-WCMC, IUCN & IGS, 2021). The IUCN Red List determines which species are at threat of extinction. Only 15% of threatened vertebrates (617) are effectively represented in PAs (adequate overlapping of species range with a PA to a level consistent with their likely persistence - Rodrigues et al. 2004; Venter et al. 2014), and 17% of threatened vertebrates (700) are not found in a single PA (Venter et al., 2014). There are currently no global datasets or analysis that provides a measure of how well PAs cover areas of particular importance for ecosystem services (UNEP-WCMC, IUCN & IGS, 2021).

1.2.1.3 Connectivity

Ecological connectivity is defined as the ability of an organism to move between two distinct units. For example, a rodent that will only disperse up to 1km away from a grassland patch will regard patches 2km apart as being disconnected, whereas a raptor that files tens of kilometres in a day may regard patches of grassland 10km apart as being connected. Ecological connectivity is important for the movement of species to reduce genetic bottlenecking, assist populations in the evasion of natural disasters, and for species with large roaming distances and migration routes. Using the ProtConn indicator, which quantifies how well PA systems are designed to support connectivity, Saura et al., (2019) found that globally 7.7% of protected land is well connected (at a dispersal range of 10km). Connectivity is greatest in the Americas (11% of protected land is well connected) and lowest in Europe and Asia (5% of protected land is well connected).

1.2.1.4 Deforestation and degradation

Remote sensing and subsequent analysis (e.g., calculating of the normalized difference vegetation index (NDVI)) has shown that on average PAs reduce deforestation and degradation. Examples across continents show that tropical deforestation rates are routinely lower inside PAs than comparable sites outside PAs (e.g., Andam et al., 2008; Canavire-Bacarreza & Hanauer, 2013, Sims & Alix-Garcia,2017). Effect sizes of PA impacts vary per country, per PA and within PAs. For example, Shah & Baylis (2015) found that the Sebangau National Park in Kalimantan, Indonesia, performed significantly better in reducing deforestation than the Kerinci Seblat National Park in Sumatra. Within the Kerinci Seblat

National Park, reducing deforestation was more successful in the South than in the North part of the park. On average effect sizes are modest. In Costa Rica, Andam et al. (2008) found that PAs avoided 10% of deforestation. PAs may also have positive ("blocking") or negative ("leakage") impacts outside their boundaries, but the evidence remains inconclusive (Herrera et al. 2019). By "blocking" I mean that a PAs presence reduces deforestation in surrounding areas outside of its boundaries. "Leakage" refers to deforestation that may have occurred within a PAs boundary which is displaced (or "spilt over") into an area outside of the PA. Fewer studies provide evidence on PA success outside of tropical forests. Geldmann et al., (2019) conducted a global study of the effectiveness of PAs and found whilst many PAs show positive outcomes, compared with matched unprotected areas, PAs on average did not reduce a compound index of pressure change (Temporal Human Pressure Index – THPI) between the years 2004 – 2019. Despite the relative success of PAs in slowing down deforestation in comparison to unprotected land, 3% of the global protected forest was lost from 2000 to 2012 and 10% of the total forest loss occurred within PA boundaries (Heino et al. 2015).

1.2.1.5 Smaller anthropogenic disturbances

Smaller scale anthropogenic disturbances such as fire and logging have received limited attention. Selective logging is difficult to determine because traditional remote sensing techniques are unable to detect it due to pixel sizes being too big to capture small-scale habitat changes. Using techniques to analyse forest degradation at subpixel levels one study has shown that PAs in the Brazilian Amazon show limited signs of selective logging (Asner et al. 2005). A global study of forest fires (an indicator of land clearing in forests) compared inside PAs vs. outside PAs and found fewer fire occurrences in PAs (Nelson & Chomitz, 2011).

1.2.1.6 Development Prevention

Evidence on the ability of PAs to prevent development is mixed. A global study of PAs found that between 1995 – 2010 increases in human population density and night lights have been smaller inside PAs compared to matched areas outside (Geldmann et al., 2019). Both are potentially significant indicators of environmental degradation so evidence that PAs are effective at slowing their growth is encouraging. However, the same study finds that in most of the world, cropland increased more inside PAs between 1995 – 2010 than in comparable areas outside PAs. Land conversion to agriculture is particularly pronounced in the Afrotropics where the area of cropland inside PAs increased at almost double the rate seen in comparable unmatched lands. Duran et al, (2013) found that large-scale mining activities are occurring in at least 6 % of PAs globally. Reports suggest that 44 World Heritage Sites are, or potentially will be, impacted by large-scale mining operations (Koziell and Omosa, 2003). PAs in some countries perform better than others. Spatial analysis of Canadian PAs shows that land development was far less extensive inside than outside PAs. However, several PAs had substantial development inside their boundaries and nearly half of all PAs had roads (Leroux & Kerr, 2013). It is important that development near PAs is reduced because studies show that forest loss in PAs increases with increased proximity to roads and high human population densities (Padma, 2018).

1.2.1.7 Ecosystem Restoration

"Ecological restoration" is defined as *"intentional activity that initiates or accelerates the recovery of an ecosystem with respect to its health, integrity and sustainability"* (Society for Ecological International Science & Policy Working Group, 2004).. Restoration activities may include reforestation, reducing pollution or the removal of the invasive species. Studies on restoration projects are uncommon – perhaps in part because no baseline data was collected before restoration activities commenced or because the effectiveness of restoration projects is difficult to measure. For example, evidence on the ability of PAs conduct successful reforestation projects is difficult because reforestation is hard to detect via remote sensing methods (Demina et al., 2018). What studies exist show that PAs can have important impacts for land and ecosystem restoration. Using remotely sensed data, a quasi-experimental study in Costa Rica showed that over a 40-year period, 15,000 Has of degraded land was reforested within PAs (Andam et al., 2012).

1.2.1.8 Species protection

Evidence of the effectiveness of PAs to conserve species is mixed. PA performance differs across species and locations. Data collected by more than 1,000 camera traps in 15 PAs in the tropics shows that of 250 ground-dwelling mammal and bird species; 17% of populations increased, 22% remained constant and 22% decreased (Bergen, 2016). European PAs appear to perform better than African PAs (Barnes et al., 2016) and Southern African PAs are maintaining populations whereas West African PAs are suffering the most severe declines (Craigie et al., 2010). However, it is difficult to evaluate the true success of these PAs as these studies lack an appropriate counterfactual to compare PA impacts against what would have happened had the PA have not been in place. The most comprehensive study using a counterfactual compared species richness and abundance of 13,669 vertebrate species inside 359 PAs against areas with matched unprotected land (Gray et al. 2016). They found higher species richness and abundance inside PAs than outside. However, wide confidence intervals highlighted that evidence of PA impacts on species could be improved. They also found that differences in species were mostly due to differences in habitat instead of wildlife exploitation (e.g., hunting and trapping). Evidence on how successful PAs are at limiting wildlife exploitation is therefore very limited. Hunting within tropical PAs is common (Castilho et al., 2017). However, overall hunting pressure is generally lower in PAs than outside (Laurance et al. 2012; Benítez-López et al. 2017). Studies suggest that PAs might fail to prevent the extinction of commercially valuable species (Symes et al. 2018).

Table 1.1 Summary of what and how PA impacts are measured

What PAs protect	How impact is measured
Ecological representativeness	• Biomes • Ecoregions • Geodiversity
Species	• Richness / abundance • Threatened species • Keystone species • 'EDGE' Species (Evolutionarily Distinct Globally Endangered)
Ecosystem services	• Regulating services (e.g., carbon sequestration, pollinator services) • Provisioning services (e.g., productive fisheries, non-timber forest products)
Important areas for conservation	• Key biodiversity areas (KBAs) • Important bird and biodiversity areas (IBAs) • Area of low human disturbance (e.g., 'low impact areas')
How effective are PAs?	
Connectivity	• Adjacency • Ecological connectivity
Comparison of mitigation of threats inside / outside of PA boundaries	• Land cover change • Deforestation and degradation • Forest fires • Small anthropogenic threats (e.g., selective logging) • Prevention of development (e.g., road building)

Restoration	• Increases in species richness / abundance • Increase in forest cover • Reduction of pollution

1.2.1.9 Environmental outcomes of PAs – Knowledge Gaps

Current evidence on PA outcomes is mostly limited to case studies of state-owned PAs (Oldekop et al., 2016). Moreover, most studies fail to include a counterfactual to compare the outcomes of PAs to what would have happened had the PA have not existed (Pullin et al., 2013). Most larger scale assessments of PA outcomes fail to account for the diversity of PAs and therefore do not distinguish PAs outcomes by PA type (e.g., Nolte et al., 2013; Gallo et al., 2009). Therefore, we do not know how PAs under different governance types perform relative to one another. It remains unclear if (i) certain PA types are more likely to represent certain types of biodiversity than others, (ii) how different kinds of PAs might complement one another to enhance connectivity and (iii) if certain types of PAs are more effective at producing positive environmental outcomes (e.g., reducing deforestation) than others. What factors led to PAs producing positive environmental outcomes is unclear because barely any studies specifically test the casual pathways of PA outcomes (e.g., Andam et al., 2010). It is also unclear if the factors that lead to PAs having positive environmental outcomes are the same across all PA types. Gaining this knowledge can lead to the better planning and design of PA networks because different types of PAs can be established in areas in which they are best placed to overcome particular environmental challenges or address PA network shortfalls.

1.2.2 Social outcomes of protected areas

PAs can have positive and negative socio-economic impacts for people. These are important not just in themselves, but because socio-economic outcomes influence biodiversity outcomes and vice versa. For example, social outcomes of PAs can determine PA legitimacy and the level of support they receive from local communities. This affects their ability to achieve their biodiversity conservation goals. In a global review of PA impacts, Oldekop et al. (2016) found that PAs with positive socio-economic outcomes were also more likely to report positive biodiversity conservation outcomes.

PAs can impact people’s lives in a multitude of ways. Within this thesis, I categorise social outcomes into the five livelihoods assets (financial, social, human, physical and natural capital) with the sustainable livelihoods’ framework (DFID, 2000). I adopted the sustainable livelihoods framework because it takes a holistic view of livelihoods, incorporates governance processes and has had some use in assessing conservation impacts (Ward et al., 2017, Bennett et al., 2010). Capital is defined as assets that all humans require to make a living. Social capital is defined as social resources, including networks for cooperation, mutual trust, and support, human capital is defined as the amount and quality of knowledge, skills and labour available in a household as well as psychological benefits obtained from the creation of a PA and natural capital is defined as supporting, provisioning and regulating services that humans gain from the natural environment. It is important to not only measure the impacts of PAs averaged across a population but also to understand how different members of that population are impacted (e.g., different genders, ethnicities etc.). With my thesis I categorise to whom the impact accrued using the categories: PPA owner, local community surrounding the PPA, general public or local government. Within the wider literature most studies use qualitative methods such as questionnaires and interviews however some quantitative studies have also been conducted using mostly financial analysis or counter-factual methods are just starting to be used. These quantitative methods are particularly used to assess the impacts of PAs on income.

1.2.2.1 Financial outcomes of protected areas

Income and assets remain the dominant indicators used to assess the social impacts of PAs (de Lange et al., 2016). PAs can have positive financial outcomes for communities by creating employment opportunities, particularly through ecotourism. In changing land from farming to private ecotourism reserves in South Africa, employment figures increased by a factor of 3.5 (Sims-Castley et al., 2005). Ecotourism can have a profound impact on national economies (Coria & Calfucura, 2012). In Costa Rica, earnings from tourism amount to more than $1.7 billion US dollars per year and 80% of all visitors to the country come to do eco-tourism related activities (Embassy of Costa Rica in Washington D.C.,. n.d.). Ecotourism is also linked to higher wages. In South Africa, individuals employed by private nature reserves had an average annual salary 4x greater than that of individuals employed in local farming (Sims-Castley et al., 2005). A study of >60,000 households across 34 developing countries found that households near PAs with tourism had 17% higher wealth levels and a lower likelihood of poverty than similar households living far from PAs (Naidoo et al., 2019). Whilst ecotourism can lead to improved income and employment it can also cause an increase in inequalities (West et al., 2006). Poorer households, those less able to capitalize on tourism opportunities, or those living farther from reserve boundaries and park entrances benefitted less than others from PA establishment (Serenari et al. 2017; Hora 2017). Studies show higher levels of inequality lead to more environmental degradation, whereas greater equality leads to better environmental protection (Ceddia, 2019).

1.2.2.2 Outcomes of protected areas on social capital

Anthropological research, key informant interviews, focus groups and household surveys can be used to determine the more intangible impacts of PAs on social capital. During interviews, local communities have stated that PA creation has undermined their freedom of choice and action (Abunge et al., 2013). This is not surprising because consultation and participation of local communities to guide PA management is often lacking. For example, only 8 of 34 PAs surveyed in the Congo Basin involved local communities in management decisions (Pyhälä et al. 2016). Accusations of human rights violations of local people by park rangers cause further concern about the power dynamics associated with PAs, and their lack of freedom of choice and action (Matsuura, 2017). Case studies in Chile suggest that PAs have led to a loss of cultural identity through the erosion of traditional practices and related social interactions (Serani et al., 2017). PAs can reduce community cohesion and introduce social tensions surrounding the reporting of illegal activities and create conflicts around the distributions of any development project activities which a PA may undertake (West et al., 2006). However, studies in Indonesia and Kenya show that where PAs improve natural resource governance and local communities have meaningful influence over decisions, they can increase feelings on empowerment (Gurney et al., 2014) and a sense of pride and ownership (Mahajan & Daw, 2016).

1.2.2.3 Outcomes of protected areas on physical capital

The loss of land rights, eviction and displacement of local people is one of the most controversial and contested impacts of PAs (Agrawal & Redford, 2009; West et al., 2006). Evidence of eviction and displacement is not well recorded but could have affected millions of people (Agrawal & Redford, 2009). Eviction is the forced removal of residents from an area. Displacement is the “putting out of place” of local residents. This could be though physical displacement where local residents are relocated by the PA or through economic displacement where the rules and regulations enforced by a PA make it untenable for the residents to remain in-situ. A review of 250 PA reports covering ~200 PAs found that 50% of all documents mentioned displacement but offered no details or who or how many people were displaced and if they were afforded any compensation (West et al., 2006). Studies in environmental history show that many national parks across the US were created through the containment of Native Americans onto reservations and by keeping areas free of human presence. Displacement for biodiversity conservation under apartheid was also commonplace in South Africa (West et al., 2006). A study of 36 PAs in the Congo Basin shows that the creation of 26 of them resulted in the partial or complete displacement of local people and in no case was compensation provided (Pyhälä et al., 2016). It is unclear how many people may be displaced to create PAs in the future, but the half-earth vision may encourage human displacement (Schleicher et al., 2019). Local people may be economically displaced through restrictions on the use and access to resources (Cernea & Schmidt-Soltau 2006). A review of empirical data and evaluation analyses undertaken by the World Bank and African Development Bank concluded that people living in PAs are made materially worse off and impoverished by the introduction of restriction of access to natural resources, enforced by PAs (Weber et al., 2011). These restrictions can disproportionately affect poor segments of the population who rely on natural resources for their livelihoods (Holmes, 2007). It is important note that not all PAs act in the same way. Examples from community-based conservation projects have shown how PAs can sustain livelihoods and safeguard ecosystem services (Weber et al. 2011). One study from India shows how local communities were successfully resettled to make room for the Bhadra Wildlife Reserve (Karanth, 2005).

1.2.2.4 Outcomes of protected areas on Human Capital

Results of studies on the impacts of PAs on health and well-being are mixed. A global assessment of the impact of PAs found no negative effects of PAs on human health and living standards (Naidoo et al. 2019). They also found that PAs with documented tourism were associated with better children's health and more assets. These results suggest that additional income earned from ecotourism can be spent to pay for food, medicine or medical clinic visits that improve children's health. However, a systematic literature review by Pullen et al., (2013), found evidence to the contrary. Forest evictions created by PAs exposed evictees to new diseases when they integrated with other groups; in particular, children were seriously affected by malaria (First Peoples Worldwide, 2006). It also prevented evictees' access to traditional medicinal plants they had previously used to stay healthy.

There is weak evidence for improvement in education provision following PA establishment (Pullen et al., 2013). One study of Ramonafana National Park in Madagascar found no difference in the percentage of girls in primary schools inside or outside of the park (Korhonen et al., 2004). However individual case studies do exist where PAs have invested heavily into the education and capacity building of the local community (e.g., Chumbe Island Coral Park in Zanzibar, Tanzanaia) (Dodds, 2012).

1.2.2.5 Outcomes of protected areas on natural capital

PAs can safeguard vital ecosystem services, including water provision, food security and carbon storage (Clements et al., 2014). For example, a third of the world's 100 largest cities rely on PAs as a significant source of drinking water (Dudley & Stolton, 2003). Globally, PAs play an important role in climate change mitigation efforts, such as REDD+ (reducing emissions from deforestation and forest degradation) (Scharlemann et al., 2010). However, the distribution of ecosystem service benefits from PAs may not be received equally. Villamagna et al., (2017) found that PAs offer ecosystem benefits for all, but the benefits disproportionately accrue to households with greater income. PAs can offer cultural services such as aesthetic inspiration and the spiritual experience of nature. Yet some PAs may only value and preserve the culture of certain groups. For example, National Parks in the USA have been criticised for valuing the wilderness experience of white urbanites over the cultural values of Native Americans (Fortwrangler, 2007).

1.2.2.6 Social outcomes of PAs – Knowledge Gaps

In summary, PAs have wide ranging social impacts, but they are rarely quantified. Counterfactual studies, to determine what would have happened had a PA have not been established, are also rare due to difficulties in accessing the data necessary to conduct these studies (Pullin et al., 2013). Like environmental outcomes, it is uncertain if (i) PAs of certain management or governance types more likely to produce certain social outcomes (e.g., increases in employment, education, or health), (ii) what factors led to PAs producing positive social outcomes and (iii) if the factors that led PAs having positive social outcomes are the same across all PA types. This knowledge can ensure that PAs have positive social impacts increasing their legitimacy, the level of support they receive from local communities and ultimately their effectiveness in achieving their biodiversity conservation goals.

1.3 Determinants of protected area outcomes

Many factors may influence the outcomes of PAs. Here I focus upon three key factors: (i) management, (ii) location and (iii) governance.

1.3.1 Protected Area Management

PA management refers to the means and actions that are taken to achieve the PAs objectives (Borrini-Feyerabend et al., 2013; Table 1.3). PAs are categorised into seven management categories based on primary management objectives (Borrini-Feyerabend et al., 2013; Table 1.1). The objectives of a PA influence its outcomes. For example, the establishment of a strict nature reserve with a primary objective to minimise human disturbance may have a ‘fines and fences’ management approach which forces the displacement of people living in that area and prevents the collection of resources such as firewood or medicinal plants. In comparison, the establishment of a sustainable use zone with a primary objective to protect nature and encourage the use of natural resources sustainably may take a co-management approach and permit access to the PA and the extraction of resources.

Table 1.2 IUCN PA management categories and associated definitions (adapted from Borrini-Feyerabend et al., 2013)

Management category	Definition	Primary objective
Ia **Strict nature reserve**	Protected areas that are strictly set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values. Such protected areas can serve as indispensable reference areas for scientific research and monitoring.	To conserve regionally, nationally or globally outstanding ecosystems, species (occurrences or aggregations) and/or geodiversity features: these attributes will have been formed mostly or entirely by non-human forces and will be degraded or destroyed when subjected to all but very light human impact.
Ib **Wilderness area**	Usually large unmodified or slightly modified areas, retaining their natural character and influence, without permanent or significant human habitation, protected and managed to preserve their natural condition	To protect the long-term ecological integrity of natural areas that are undisturbed by significant human activity, free of modern infrastructure and where natural forces and processes predominate, so that current and future generations have the opportunity to experience such areas.
II **National park**	Large natural or near-natural areas protecting large-scale ecological processes with characteristic species and ecosystems, which also have environmentally and culturally compatible spiritual, scientific, educational, recreational and visitor opportunities	To protect natural biodiversity along with its underlying ecological structure and supporting environmental processes, and to promote education and recreation.
III **Natural Monument or feature**	Areas set aside to protected a specific natural monument, which can be a landform, sea mount marine cavern geological feature such as	To protect specific outstanding natural features and their associated biodiversity and habitats.

	cave, or a living feature such as an ancient grove	
IV **Habitat/species management area**	Areas to protect particular species or habitat, where management reflects this priority. Many will need regular, active interventions to meet the needs of particular species or habitats, but this is not a requirement of the category	To maintain, conserve and restore species and habitats.
V **Protected landscape or seascape**	Where the interaction of people and nature over time has produced a distinct character with significant ecological, biological, cultural and scenic value: and where safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values	To protect and sustain important landscapes/seascapes and the associated nature conservation and other values created by interactions with humans through traditional management practices.
VI **Protected areas with sustainable use of natural resources**	Areas which conserve ecosystems, together with associated cultural values and traditional natural resource management systems. Generally large, mainly in a natural condition, with a proportion under sustainable natural resource management and where low-level non-industrial natural resource use compatible with nature conservation is seen as one of the main aims	To protect natural ecosystems and use natural resources sustainably, when conservation and sustainable use can be mutually beneficial.

Results on whether strict nature reserves or sustainable use zones are more effective are mixed. In the Brazilian Amazon, Nolte et al., (2013) found that strict PAs have consistently avoided more deforestation than sustainable use areas. However, a global study by Nelson & Chomitz (2011) found sustainable use PAs were more effective than strict PAs at reducing forest fires. An assessment of global PA outcomes by Oldekop et al., (2016) found positive conservation and socioeconomic outcomes were more likely to occur when PAs adopted co-management regimes which empowered local people, reduced economic inequalities, and maintained cultural and livelihood benefits. Stern (2008) found that the perceived trustworthiness of PA managers and positive personal interactions between PA managers and the public made people more likely to comply with PA rules and regulations.

PA management can falter due to funding shortfalls (Waldron et al., 2017) and understaffing (Coad et al., 2019). An assessment of 2,167 terrestrial PAs found that only 25% have adequate funding and staffing (Coad et al., 2019). Insufficient resources impact boundary demarcation, natural resource management and can also lead to reduced communication with stakeholders, potentially creating conflicts (Watson et al., 2014). A lack of ability to enforce restrictions and demarcate boundaries has led to criticisms that some PAs only exist on paper and offer no meaningful environmental protection (Oates, 1999). Corruption and bureaucracy are also important factors that influence PA management and performance. For example, between 1985 to 2001, 56% of lowland forest in PAs was destroyed in Indonesia largely due to corruption (Laurence, 2004). Upgrading the management of current PAs could result in significantly improved conservation outcomes (Pringle, 2017). For example, due to civil war the Parque Nacional de Gorongosa in Mozambique suffered great ecological damage. In 1992 the park had lost more than 90% of its large mammals but populations recovered to almost 80% of pre-conflict levels following an upgrade of PA management (Pringle 2017).

1.3.2 Location of protected areas

The location a PA influence its potential, perceived effectiveness and ability to mitigate negative outcomes. Many PAs have limited potential to mitigate environmental change because they are in areas where pressures on nature are expected to have remained low even without formal protection (Joppa & Pfaff, 2009). Therefore, they only passively protect land through the absence of human pressure, and thus have no meaningful environmental outcomes. These remote PAs may also have limited social outcomes as they do not directly impact anyone living in the immediate vicinity. A PAs perceived effectiveness to mitigate environmental change can be influenced by the immediate socio-economic context in which it is located (Geldmann et al., 2019). A PA may be in an area of high pressure (e.g., an area with greater proximity to roads, urban centres and navigable waterways (Barber et al., 2014)) and prevent 100Ha being deforested but be deemed ineffective as it still permits deforestation in its border. However, a PA in an area of low pressure may only reduce 10Ha being deforested but be considered effective because it prevents all deforestation within its boundaries. The regional context in which a PA is located can affect its ability to prevent environmental change (Oldekop et al., 2016). Regional differences in the representation and empowerment of rural people in national politics (Galvin & Haller, 2008), differences in economic stability, and the robustness and transparency of national governance can drive divergence of PA outcomes in different regions (Nelson & Agrawal, 2008).

1.3.3 Protected Area Governance

There is no universally accepted definition of governance, but in the simplest terms it can be described as the political processes that exist in and between different institutions. In reference to PAs, Graham et al., (2003, p.2) define governance as "*the interactions among structures, processes and traditions that determine how power and responsibilities are exercised, how decisions are taken and how stakeholders have their say*". Essentially, governance refers to who decides what the objectives of a PA are, how decisions are made, who holds the power, authority and responsibility and how they are held accountable. PAs are divided in four main governance categories and into two to three further governance subtypes (Borrini-Feyerabend et al., 2012; Table 1.1).

Table 1.3 A summary of governance types and critiques (Borrini-Feyerabend et al., 2012; Macura et al., 2013)

Governance Type	Definition	Critiques
1. Government	Government protected areas are owned and managed by a centralised governmental agency (ministry or park agency reporting directly to the government) that enforces decisions, has authority, responsibility and accountability for management	Unequal distribution of rights, power and benefits, therefore creating social conflicts (Coad and Campbell, 2008)
2. Shared	Co-managed or multi-stakeholder protected areas exist where a governmental agency and other stakeholders, such as local/indigenous communities that depend on the area culturally or for their livelihoods share power and responsibility to make and enforce decisions.	Lack of biodiversity promotion and protection (Terborgh, 2004) In some cases the communities are not as involved in governance processes as claimed (Virah-Sawmy et al., 2014)
3. Private	Private protected areas exist where private landowners, individuals, NGOs and other organisations make and enforce decisions, have control and/or ownership over resources	There are questions about the long-term security of privately owned areas (Adams and Hutton, 2014)
4. Indigenous peoples and local communities	Protected areas where the management authority and responsibility rest with indigenous peoples and/or local communities through various forms of customary or legal, formal or informal, institutions or rules	Some concerns about a lack of biodiversity protection (Eklund and Cabeza, 2017) In some cases local communities or indigenous populations do not have the funds or capacity to manage these areas (Macura et al., 2015)

PA governance is now recognised as “central to the conservation of PAs throughout the world” (WCPA, 2004: 257). Governance is the variable with greatest potential to affect conservation coverage (Borrini-Feyerabend et al., 2012). Governance is also the main factor in determining the effectiveness and efficiency of PA management. This is because it is a key detriment of the appropriateness and equity of management decisions (Borrini-Feyerabend et al., 2012).

Governance, which is legitimate, appropriate, equitable and effective is known as good governance (Johnston, 2002). Good governance ensures that PAs are well embedded within society and in supportive environments where they have the potential to succeed. It also means that processes and institutions produce results that meet the needs of stakeholders while making the best use of resources at their disposal. An overview of the characteristics of good governance can be seen in Figure 1.1.

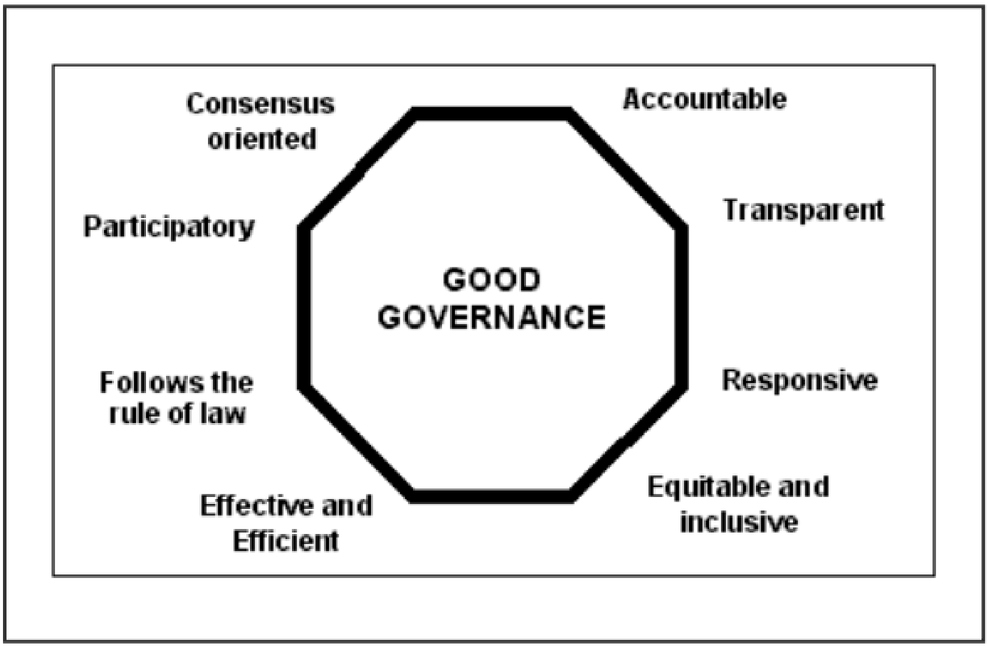

Figure 1.1 Characteristics of good governance (taken from UNESCAP, 2008)

Good governance can help (i) create ecological links to the wider landscape / seascape (ii) create support with local communities living in and around PAs and (iii) make sure that PAs are taken into account in broader decision-making by the institutions which both affect and are affected by PAs. Good governance maximises the ecological, social and cultural benefits of PAs without incurring unnecessary costs. Good governance practices means that PA governersshould be aware of and aim to avoid potential negative effects of PAs to local communities. Where negative effects (e.g., displacement, restriction of access to resources) is unavoidable, suitable compensation and mitigation should be put in place.

A survey of 110 national PA agencies in 2005 found that 90% of respondents felt that PA governance had improved over the previous decade and 67% felt that this has led to improved PA management effectiveness and better conservation and social outcomes of PAs (Dearden et al., 2005).

In summary, when assessing how PA governance affects PA outcomes governance impacts can be categorised in two main ways: firstly, *who* is doing the governing (i.e., the 4 IUCN governance types) and secondly how are they governing. Within this thesis I am primarily interested in assessing how does *who* governs PAs affect their outcomes and indirectly assess how (effectively) PAs are being governed and if they can be deemed as being under "good governance".

1.3.3.1 Knowledge Gaps of PA governance affecting PA Outcomes

Knowledge on how governance may influence a PA's conservation and social outcomes is lacking (Macura et al., 2015), in part because evaluations of PA performance to date have focused on PA management. This may be because there is a diversity of methodologies to evaluate management elements of planning, inputs, processes, outputs and outcomes such as the Rapid Assessment and Prioritization of Protected Area Management (RAPPAM), the Management Effectiveness Tracking Tool (METT) and UNESCO's Enhancing our Heritage (EoH)) (Stoll-Kleemann, 2010). Over 55,000 PA management evaluations have been completed and are available through the Global Database on Protected Area Management Effectiveness. No such dedicated tools exist to evaluate the effectiveness and outcomes of PA governance although it may be possible that existing tools used to assess PA management (e.g., the RAPPAM) could be repurposed. Studies which evaluate the outcomes of PA governance mostly focus upon state or community PAs with fewer studies including multiple governance types in the same study or examining private or co-governed PAs (Munoz Brenes, 2018). In a global review of reported PA outcomes Oldekop et al.,

(2016) show that research is significantly biased towards state owned national parks in Africa and Asia. They find PPAs are grossly under-represented and studies on the outcomes of PAs in Europe and Oceania are also lacking. Most analysis comparing the distribution of PA governance types are conducted at the regional or national level and focus on biomes or plant species (e.g., Gallo et al., 2009: Shanee et al., 2017). No global level studies have been conducted to assess if PPAs protect different biomes than other PA governance types. Schulze et al., (2017), examine the threats that PAs face, but they fail to differentiate threat levels between different PA governance types. Gallo et al., (2009) find that PPAs are in areas of lower elevations and higher populations than state PAs but no study explicitly compares the difference in PA governance types in protecting areas experiencing different levels of human pressure. One Australian study investigates how PPAs contribute to protecting KBAs and finds they may protect 39.7% of all KBAs, represent 10 KBAs that would otherwise not be protected, and enable 5 KBAs to reaching their target for adequate protection (Ivanova & Cook, 2020). Studies into how PAs of different governance types contribute to protecting threatened species or EDGE species are absent. Few studies (e.g., De Vos and Cumming, 2019; Graves et al., 2019) examine how PAs of different governance types contribute to PA network connectivity. (See section 1.2.1).

Studies using quasi-experimental designs to infer a strong causal link between governance and ecological and social outcomes are uncommon (Curzon & Kontoleon, 2016). Evidence from the Peruvian Amazon and South Africa suggest that PA governance does have an impact on their effectiveness to reduce deforestation and degradation (Shumba et al., 2020; Schleicher et al., 2017). However, it is not known if this is the case elsewhere. PA governance can also influence the social impacts of PAs. In a global analysis on the impacts of PAs, Oldekop et al., (2016) found that PA governance is to some extent associated with empowerment, monetary impacts, livelihood impacts and the unequal distribution of impacts from PAs. This thesis looks to address a key knowledge gap in the literature by exploring the link between PA governance and their environmental and social outcomes. This thesis focuses on the outcomes of PAs which are governed by private actors.

1.4 Private Governance

This thesis focuses on PAs under private governance. Private governance refers to private landowners, individuals, NGOs and other not-for profit and for-profit organisations that make and enforce decisions and have control and/or ownership over resources in PAs (Macura et al., 2015). PPAs are an old conservation approach, with some countries (e.g., the UK) having established PPAs decades before state governed PAs (Hodge and Adams 2012). However recently they are increasing in number (UNEP-WCMC, IUCN and NGS (2021). A global survey to assess changes in governance of PA systems across 41 countries reported an increase of involvement of the private sector between the years 1992 and 2002 (Dearden et al., 2005). Parcels of land owned and formally protected by individuals, NGOs and for-profit organisations are referred to as private protected areas (PPAs) (see Section 1 for definition). PPAs are highly diverse in their form, ownership, size and location. 13,103 PPAs are currently reported to the WDPA. Yet, this may be a substantial underestimate because only a small proportion of countries report PPAs to the WDPA and these may also report only a subset of existing PPAs (Fitzsimons 2015; Bingham et al. 2017) (see chapter 2 section 2.5.1. - Challenges to evaluating PA network capacity).

Opinions on the role that private actors can play in conservation are divided. Gooden & Sas-Rolfes (2020) catalogue criticisms of private land conservation into three main groups; (i) implementation effectiveness, (ii) value conflict and (iii) economic inefficiency, across 25 themes. Issues of implementation effectiveness centre upon; (i) the permanence of PPAs, (ii) how effective they are in terms of their size and the skills, financial sustainability and resources of PPAs owners, (iii) a lack of information on PPAs which reduces the ability of systematic conservation planning and (iv) the PPA accountability. Proponents of PPAs refute these criticisms. They state that PPAs can be highly effective because (i) although most are small in size they can greatly increase connectivity of the PA network by being located adjacent to other PAs or by forming wildlife corridors (ii) in many cases, PPA staff have better training, equipment, and funding than their national park counterparts and they are in a position to conduct superior community outreach and development and (iii) they can be permanent due to formal long-term agreements such as easements and because some are owned by NGOs who by their very nature are dedicated to long term nature conservation.

Moreover, issues of permanence are not related to PPAs. A study of protected area downgrading, downsizing, and degazettement (PADDD) identified 543 instances of PADDD in 57 countries, affecting more than 503,591 km^2 of protected lands and waters across all PA governance types (Mascia et al., 2014). The highest number of PADD events (>30) were reported in Russia, Malaysia, Uganda and Zambia where no PPAs are reported in the WDPA (PADDD itself does not keep a record of PA governance type). This means that PAs in these countries undergoing events may be either state, community and co-governed PAs. Issues of value conflict centre upon (i) cultural conflicts between PPA owners and local communities and subsequent disruption of local social systems (ii) elitism, accumulation of land control, green-grabbing and neo-colonialism, (iii) neoliberalism and commodification of nature, (iv) exclusion, and physical and economic displacement and (v) the legitimacy of private conservation actors. In response, proponents of PPAs state that not all PPA owners are wealthy or foreigners to the area in which a PPA is established. For example, many PPAs in Colombia are owned by local farmers designating a part of their property to biodiversity conservation (RUNAP, 2021). They also state that when PPAs are governed by local individuals or communities they can increase local empowerment and devolve power to regional or local institutions away from the national state. Issues of PPAs regarding economic inefficiency include moral hazard and hubris of the present. Issues of moral hazard relate to situations in which the risks of a private activity are involuntarily shared by the wider public, thereby creating perverse incentives for private actors. Issues of hubris of the present relate to issues of over intergenerational equity. It is argued that the landowners of today should not “lock in” land use decisions which may not be optimal for future generations. In relation to these claims, proponents state that not all PPA owners are motivated by perverse monetary incentives and evidence suggests many PPA owners are motivated by conservation values, place attachment and social learning (Selinske et al., 2015). Whilst many hypotheses exist regarding the potential impacts of PPAs, there is little empirical evidence with which to evaluate these claims.

1.5 Research justification

It is widely acknowledged that government action alone will be insufficient to reach global PA targets (Butchart et al., 2015; Watson et al., 2014). Therefore, there is a need to better integrate and include non-state actors into PA management and governance. Yet, we currently lack the evidence to do this effectively. This is because we do not know how impacts between PA governance types differ, how these impacts are influenced by the social, political and economic contexts in which they are located and how different types of PAs operate as institutions. A structured programme of PA governance assessments can help facilitate the integration of non-state actors into conservation and improve the diversity, quality and vitality of PAs. To assess PA governance, we must (i) systematically assess and evaluate the outcomes of different forms of PA governance in a range of contexts and (ii) determine the circumstances in which different PA governance types yield the best outcomes, both ecologically and socially. This thesis aims to explore the role of governance in determining the environmental and social outcomes of PAs via a focus on private forms of governance. I focus on private governance because it has been under-studied in the literature, PPAs numbers are increasing in number and extent and because private governance may have unique and significant contributions to make to achieving global biodiversity targets.

1.6 Aims and objectives

The overall aim of the thesis is to determine the environmental and social impacts of privately protected areas. Each individual objective builds upon the findings of the previous to answer this over-arching research aim.

Objective one: To determine what is currently known about the environmental and social outcomes of privately protected areas

Objective two: To analyse what attributes privately protected areas protect within their boundaries and how this compares to PAs of other governance types and to random placement

Objective three: To compare how effective private protected areas are at reducing deforestation compared to PAs of other governance types

1.7 Thesis Outline and Structure

Chapter 2 outlines the methodological approach underpinning the research. The following three chapters are the academic journal papers produced from the research conducted for this thesis. Chapter 3 is a literature review which collates out the documented outcomes of PPAs. It determines what PPA outcomes are reported, where and at what scale have studies on PPA outcomes been conducted, and to whom the outcomes are accrued. Chapter 4 focuses on one of the main findings to emerge in chapter 3 which was that PPAs can increase PA network complementary. Chapter 4 assesses the global spatial contributions of PPAs to the PA conservation estate and how this compares with other PA governance types and that of random distribution. Chapter 3 also found that the few studies in existence show that PPAs can be more effective than PAs of other governance types. Chapter 5 looks to contribute to the PPA effectiveness evidence base by assessing how effective PPAs are at reducing deforestation in Colombia and compares this of other PA governance types. The three results chapters are followed by a discussion and conclusion (chapter 6) that brings together insights from the three papers and highlights the unique role that PPAs can play in reaching global biodiversity targets. The final chapter also reflects on the research approach, and possible future research directions.

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Chapter 2: Methods and approach

This chapter outlines the methodological approach underpinning the research. Firstly, it justifies the methodological approach for the thesis. Secondly, it describes what approaches have been used before to determine PA outcomes. Lastly, this chapter summarises the data collection and analysis procedures used within each paper. Full details on the methods used within each paper are included within chapters 3, 4 and 5.

2.1 Rationale for a quantitative research approach

Within this thesis I use quantitative methods for three key reasons. Firstly, this thesis is concerned with identifying the global outcomes of PPAs and making generalizable statements regarding the outcomes of PPAs that can be applied over a large number of sites and in a broad range of settings. Quantitative research methods are best suited for this task because they can handle large numbers of cases to produce broadly applicable information (Cresswell, 2008). This is opposed to qualitative research methods that are more suited to in depth studies on a small number of cases within a context specific setting (Yin, 2014). These methods are better for detailed explorations of the casual pathways of the outcomes that I am looking to identity within my thesis. Secondly, this thesis uses a deductive research approach. This approach is better suited to quantitative methods because it is more aimed towards testing hypotheses and theories (Morgan, 2014). I use a deductive approach because I already had a broad array of theories that support reasons why the outcomes of PPAs may be different to those of other PA governance types (see chapter 1) and I test the outcomes of different types of PAs through observations (see chapters 4 & 5). This is opposed to an inductive research approach where first data is collected, and observations / theories arise from that collected data (Morgan, 2014). Moreover, in chapter 5 I use an experimental research design to evaluate the outcomes of PPAs. Overall, experimental interventions offer a set of strengths that are well matched to purposes and procedures of quantitative research (Morgan, 2014). It is important to note that even though I use quantitative research method, the theories tested within this thesis are informed by a body of qualitative research.

2.2 Rationale for a big data approach

I take a big data approach to evaluate the effectiveness of PPAs and explore how private governance affects the outcomes of a PA. I define a big data approach as a research method which uses large datasets that are produced in a digital form and can be analysed through computational tools (Malpas, 2012). Big data approaches are increasingly being used within conservation science (Farley et al., 2018). This is due to significant advances in the way that we can generate, analyse and store large volumes of ecological data (Farley et al., 2018). Areas of large and growing ecological data streams include; remote sensors on earth observing systems, aggregation of individual observations into larger community data resources, investment in long-term ecological monitoring networks, deployment of automated sensors networks (e.g. camera traps and temperature loggers) and citizen-science initiatives. Big data approaches have previously been used to provide an over-arching picture of where PAs are located and what they protect (e.g., Butchart et al., 2015; Pouzols et al., 2014; Venter et al., 2014), to determine how successful PAs have been at resisting anthropogenic pressures (e.g., Geldmann et al., 2019) and to find 'bright spots' where conservation initiatives have been successful and the reasons for their success. For example, while Hansen et al., (2013) revealed dramatic declines in forest extent across the globe, forest loss in Brazil was decreasing by $1318km^2$ y−1 between 2000 and 2012, primarily due to progressive legal framework covering forests during the study period. Similarly, recent analyses of satellite data by Chen et al., (2019) showed that direct human-land management has led to greening over large expanses in China and India. Big data approaches create new opportunities to study ecological systems at broad scales to better understand underlying ecological processes and to improve ecological forecasting (Dietze, 2017). They are also highly efficient, inexpensive and effective at determining the overall picture of what is happening.

I use a big data approach for several reasons. Firstly, because my research is conducted at both global (see chapter 4) and national (see chapter 5) scales. I choose these scales because the main aim of my thesis is to obtain a generalized statement regarding the outcomes of PPAs across a broad range of contexts. It is important to study conservation interventions on a global scale because (i) global biodiversity is unevenly dispersed

(Somveille et al., 2018), environmental problems are rarely local in scale (e.g., climate change) (Clayton, 1991) and countries ratify international level policies to protect nature (e.g., the Kyoto Protocol) and the success of these policies needs to be monitored. It is important to study PPAs across national borders because different countries and areas have different PPA systems and motivations for PPA establishment. It is useful to be able to compare PPA outcomes across national borders to see to what extent different systems and motivations produce different outcomes. Secondly, I also use a big data approach in order to create a counterfactual to compare my obtained results on PA outcomes to what have happened had a PA have not been established (see Section 2.5). A big data approach allows for the analysis of counterfactuals. Thirdly, I also use a big data approach because very few studies have researched the outcomes of PPAs at global and national scales and majority of studies are located at a local or sub-national scale (see chapter 3). An analysis at the international level allows for more holistic conclusions about the global outcomes of PPAs and more nuanced discussions which provide greater insights into the outcomes of PPAs in different regions.

Big data approaches have certain limitations. The key limitations of big data approaches are that they cannot (i) provide an accurate picture of what is happening in any given location and (ii) deduce causal mechanisms for why something is happening. This is because PAs are much smaller than the sample units typically used in global scale conservation analysis (Runting et al., 2020). Moreover, the scale at which big data studies are conducted is far too big to understand the unique socio-political or cultural setting in which a PA is located which may give rise to certain outcomes being observed (Runting, 2020). It can therefore be hard to draw concrete conclusions of the impacts of small or individual PAs based on coarse scale data (Kullberg et al., 2019). A solution to this problem is to better integrate the findings of both big data approaches with local case studies. Local case studies can provide quality and assurance checks against the findings of big data studies and can draw out finer details and nuances that big scale data approaches miss (Yin, 2014). Moreover, case study approaches can provide guidance for what to include in big data modelling approaches (Morgan, 2014). Although this thesis does not use a case study approach, I use these studies to inform the inputs and to help explain and validate the results of larger scale big data approaches. The design of chapters 4 & 5 and results interpretation was heavily guided using the case study literature reviewed within chapter 3. I discuss the challenges of the specific big data approaches used in this thesis (spatial mapping and quasi-experimental design) in sections 2.4.1 – Challenges to evaluating PA network and 2.5.1 – Challenges of counterfactual analysis, respectively.

2.3 What to evaluate?

The effectiveness of PAs and PA networks arises from a set of intertwined factors (Figure 2.1), including both decisions taken at the time of PA establishment (e.g., design, location, connectivity to other PAs, representativeness), as well as subsequent management decisions to mitigate threats (e.g., how well rules enforced, relationship with surrounding community). These factors translate into conservation outcomes through two mechanisms: resilience enhancement though location and design factors determining the capacity of PAs to conserve biodiversity over the long team and threat abatement through effective management. Therefore, PA outcomes needed to be evaluated in two ways. Firstly, what is the capacity of a PA or PA network to conserve biodiversity over the long term (resilience enhancement)? And secondly, how effective are PAs at mitigating threats (threat abatement)?

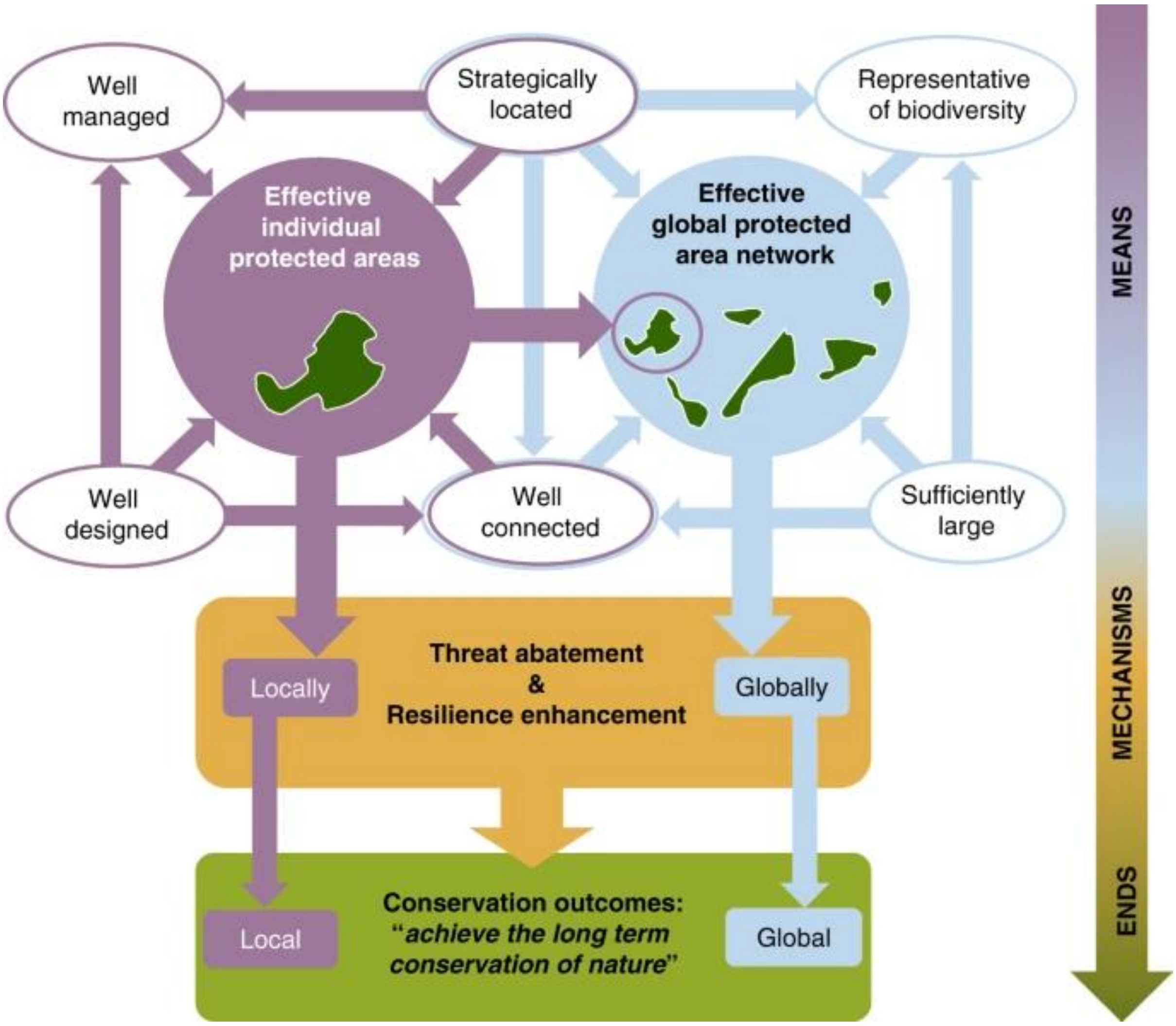

Figure 2.1 The many facets of protected area effectiveness, from the means to the mechanisms, to the ends. (Source: Rodrigues & Cazalis, 2020)

2.4 Evaluating PA network capacity

The most primitive way to evaluate a PA networks capacity to protect biodiversity is to measure its extent. This is important to deduce how much area is protected. After this, it is important to determine what is and is not represented within those areas that receive protection. The technique used to assess the representativeness of PA networks is called gap analysis. There are multiple biodiversity features (e.g., threatened species, biomes, ecoregions, KBAs etc) that can be studied within a gap analysis. In a review of the sensitivity of studies which assess PA representativeness, Vimal et al., (2011) found that different features provide different results on the assessment of the completeness of an existing reserve network to be representative of biodiversity and can affect the strategy for expanding PA networks. Bonn et al., (2005) state that focusing on any single biodiversity feature alone is insufficient to protect other features. Therefore, gap analyses should use multiple representation targets. Representation targets can include biomes, ecoregions, individual species, areas of threat or areas of biological importance (e.g., KBAs). Within this thesis, chapter 4 uses multiple biodiversity features (biomes, areas of threat and KBAs) to assess the contribution of PPAs to the global conservation estate to protect biodiversity.

When evaluating the capacity of a PA network to protect diversity, it is also important to consider its connectivity. The connectivity of PA networks can be defined and measured in two ways. Firstly, structural connectivity is how many PAS are physically connected together to form one contiguous land mass and secondly dispersal connectivity, which is how easily can species move between PAs which are not necessarily connected (Saura et al., 2017). Connectivity is important to allow the movement of species to reduce genetic bottlenecks, assist populations in the evasion of natural disasters and to maintain the ecological function of species with large roaming distances and migration routes (Cantu-Salazar & Gaston, 2010). Within this thesis, chapter 4 uses both connectivity measures (structural connectivity and dispersal connectivity) to assess the contribution of PPAs to the global conservation estate to increase PA network connectivity and therefore to protect biodiversity.

2.4.1 Challenges to evaluating PA network capacity

2.4.1.1 Problems measuring PAs in general

Gap and connectivity analyses require spatial boundaries of PAs. The World Database of Protected Areas (WDPA) is the international baseline database for tracking PAs. Data is submitted to the WDPA as one of two feature classes; a polygon boundary representing the area of land under protection, or if the boundaries of the PA are not available, a point location representing the centremost point. Submitted alongside spatial data is a table of 29 descriptors, referred to as data attributes, which describe each PA record in the WDPA (UNEP-WCMC, 2016) (Appendix A). Attributes are categorised as either “minimum” or “complete”. There are 12 minimum attributes which represent information that is mandatory and nine complete attributes which represent data that is not mandatory for a PA record to be integrated into the WDPA. Although not mandatory, complete attributes contain priority information needed to be able to perform analysis of PA effectiveness and equity (Milam et al., 2016). Less than 10% of all PA records have all 29 descriptors completed (Milam et al., 2016).

Spatial analysis is susceptible to variations in data quality. Milam et al., (2016) warn that the quality of data submitted to the WDPA is highly variable depending on who is the original data source. Until recently data quality on the WDPA has been measured and reported rather than controlled (Milam et al., 2016). The WDPA have a caveat in place which states that due to providers having different capacities and resources to collate PA information and digitize PA boundaries, issues with the accuracy of WDPA data should be expected (UNEP-WCMC, 2016). Issues with the quality of data reported have negative impacts on spatial PA studies. Firstly, there may be issues which the total area of a PA reported. Quality indicators used by the WDPA show that on average there is 33% chance that the area of a PA calculated from the boundaries provided will have a difference of more than 5% compared with the area reported. This can result in the total extent of PAs being over or under-estimated. Some PAs are given as point locations which do not convey the actual shape of the PA, and the area covered on the ground (Bingham et al., 2017) hampering connectivity analysis as the actual location of the PPA is uncertain. Lastly, it is rarely reported if a PA is fenced or not (Jakes et al., 2018). This is important as it impacts the connectivity and effective contiguity of PAs. A workaround for this is by doing local detective work for example reading legal PA documents which may describe the features of PAs or by looking on GIS systems (e.g., Google Earth) to determine the presence of fencing.

2.4.1.2. Problems measuring PPAs

Spatial boundaries of PPAs and can be freely downloaded online from the WDPA. Data is free, quick and simple to access enabling studies to be easily conducted and replicated at national or global scales. The WDPA is updated monthly enabling the latest data on PPAs to be accessed. However, there is currently a lack of PPA reporting therefore the current number of reported PPAs are a grave underestimation of the total number of PPAs in existence (Bingham et al., 2017; Stolton et al., 2014). This is because there are many challenges to obtaining data regarding PPAs, which vary from country to country. Issues include concerns around privacy (e.g., Australia (Fitzsimons, 2015)), failure to legally recognise PPAs (e.g., Canada (Wilkinson, 2014) and Spain (Rafa, 2014)) and technical issues such as low data management capacity and poor communication between PPA owners and national government bodies who report on PPAs. In a review of 17 countries believed to be the most advanced regarding the recognition and support of PPAS, only 3 have established national databases for PPAs and 9 counties have databases in development (Stolton et al., 2014). Another further confounding factor is that it is not mandatory to report a PAs ownership or governance type when submitting PA boundaries to the WDPA. A review of the WDPA shows that currently 191,414 PA records (82%) do not have their ownership type reported and 24,337 PA records (10%) do not have their governance type reported. Some of these could be PPAs but it is not possible to tell with great certainty. Missing data on PPA boundaries makes gaining a comprehensive picture of the total number and areal coverage of PPAs extremely difficult (Stolton et al., 2014). It means that the data within the WDPA on where PPAs are located is biased towards regions that have adequate PPA reporting in place and the data we do have is biased towards other PA governance types. The best workaround is to get national level data through local governments or organisations (e.g., Chile's *Asociación de Iniciativas de Conservación en Areas Privadas y de Pueblos Originarios* (http://asiconservachile.cl/acch/) to supplement the WDPA as some PPA boundaries are reported at the national but not international level.

2.4.1.3 Problems using ancillary datasets

Gap analysis of PPAs requires reported PPA boundaries to be overlaid on ancillary data sets, such as those derived from remote sensing products and from human population censuses, to determine what is and is not excluded within PPA boundaries. As gap analysis relies on PPA boundaries, it is vulnerable to the same issues as spatial analysis in that PPAs boundaries are not reported or they may be reported in the wrong location. Issues of missing or incorrectly reported PPA boundaries means that gap analysis may not correctly calculate the total area of PPAs within different areas of interest (e.g., threatened biomes, areas of high human pressure, or KBAs) and therefore their contributions to protecting these areas may be over or underestimated.

These are limitations to using ancillary data sets derived from remote sensing products and from human population censuses. Whilst they are very effective in enabling large scale studies, they do not identify targets for fine-scale conservation action (Eken, 2004). For example, whilst the IUCN red list data can show that a PPA overlaps with a target species range, it does not show specifically where a target population is living day to day and therefore give an accurate picture of species richness and abundance within a PPA of interest or where best to place a PPA to protect a target population.

2.5 Evaluating the effectiveness of PAs to mitigate threats (biodiversity outcomes)

Analyses of PA outcomes are generally referred to as impact evaluations. Impact evaluations are concerned with determining the impact of one variable on another. In the case of PAs, impact evaluation looks to determine the impact of a PA (often called a treatment or intervention) on a given variable (e.g., deforestation or financial income) (Gertler et al., 2016). Early methods to determine PA effectiveness used inside - outside PA comparisons. However, these methods overestimated the effectiveness of PAs (Joppa& Pfaff, 2010). This is because PA distribution is not random and PAs are biased towards areas of little economic interest (i.e., greater remoteness, higher altitudes and lower agricultural potential) (Joppa & Pfaff, 2009), which are less likely to have suffered from human pressure both before and after protection. Before and after PA establishment deforestation rates are also unsuitable to test the effectiveness of PAs because deforestation pressure is not consistent over time. Demand for timber can rise and fall and this can affect deforestation rates and could influence results but not be considered as a confounding factor within the analysis (Joppa & Pfaff, 2010. A study in Costa Rica using both inside-outside and before and after methods over-estimated avoided deforestation by 65% (Andam et al., 2008).

To address these shortcomings more complex counterfactual analyses are now beginning to be used. These methods were inspired by randomised control trials in the field of medicine. These studies ask the question "*what would have happened had a unit received no treatment?*" The central question of chapter 5 of this thesis is "*how much deforestation would have occurred had an area not received any protection?*". In order to answer this question matching methods are used to pair PAs with unprotected lands that have similar characteristics (termed confounding factors) that may affect the dependent variable (e.g., land cover change or deforestation rate). The matched site is termed a counter-factual and acts a proxy to show what would have happened to the area inside the PA had it not received any protection. Confounding factors are often related to either (i) physical attributes; slope, elevation, rainfall (ii) accessibility; distance to nearest road, distance to nearest market or (iii) human related, (e.g., distance to forest edge, population density). Chapter 5 uses matching methods to determine the impacts of PPAs on deforestation in Colombia compared to other PA governance types.

2.5.1. Challenges of counterfactual analysis

Counterfactual analyses overcome issues of the non-random location of PAs, issues of before-after studies and issues of spatial spill overs (because one can specify that matches are not drawn for a PAs buffer zone). However, they do have certain weaknesses. Many of these studies use datasets of coarse scales which are subject to many threats of validity, including noncompliance, attrition and randomization biases (Adams et al., 2019). They are also criticized for their frequent lack of external validity and inability to elucidate mechanisms (Adams et al., 2019). For example, Galiatsatos et al., (2020) tested the suitability of the global forest change dataset (Hansen et al., 2013) and found that whilst it offers a good first approximation of forest loss it should not be relied upon to provide a precise annual loss/gain or rate of change estimate for audit purposes and they suggest that data is checked against independent high-quality reference data. Many also criticise the global forest change dataset because it is unable to distinguish between natural forests and plantations when assessing forest cover and forest change. A potential way to validate datasets such as the Global Land Cover change dataset is to ground truth what the dataset shows compared with onsite observations.

2.5.1.2. Review of methods to determine social impacts of PAs

Here I provide a brief overview of research methods to determine the social impacts of PAs. This is intentionally brief as my thesis is predominately focused on the environmental impacts of PAs determined by large scale quantitative analysis. Evaluations of the social impacts of PAs aim to determine what impact a PA has had on human wellbeing. Wellbeing is a broad term with multiple meanings (Leisher et al. 2013), but there is increasing agreement that it encompasses objective material components, relational aspects, and subjective experiences (Stiglitz, Sen & Fitoussi, 2009). Empirical research has shown that there are broadly five aspects which are held in common; material assets, health, social relations, security, and freedom of choice and action (Narayan et al. 2000; Millennium Ecosystem Assessment, 2005). A review of approaches used to evaluate the social impacts of PAs by de Lange et al., (2015) found that 99% of studies (n = 95) analysed assessed material aspects of wellbeing (e.g., income) whilst 51% of studies assessed nonmaterial aspects such as health, social relations, security and freedom. Only one study examined the full breadth of aspects (Silva, 2006). 53% of studies attributed impacts of PA through the perceptions of the people being studied, 36% attributed impacts through inference of the researcher, and 23% used a comparison with control. However only one study employed a before-after control intervention. As with ecological impacts, in the past few years (post 2015) quasi-experimental designs using counterfactuals are increasingly being used to determine social impacts of PAs (e.g., Naidoo et al., 2019; Sims et al., 2019). These studies have the benefit of assessing what has happened with what might have happened had the PA not been established. However, the trade-off is that these studies have a more limited idea of wellbeing in order to allow for quasi-experimental design. For example, they focus on community rather than household income or on income versus other aspects of wellbeing such as psychological wellbeing or food security. At present, the most common method of data collection is through semi-structured interviews (de Lange et al., 2015). Other common tools include key informant interviews, focus group discussions, ethnographic approaches, self-complete questionnaires and meta-analyses and literature reviews (e.g. Oldekop et al., 2016). Few studies use secondary data sources such as census data, but this may increase as counter-factual research designs are more widely used because these methods require this form of data. However, gaining access to this data at suitable levels for analysis (e.g., at a household level) can be difficult due to privacy issues. Additionally, census data is not able to give you all the necessary components in order to determine someone's wellbeing.

2.6 Data Collection

All data used within this thesis is freely available online. Reviewed literature was obtained from Web of Science, SCOPUS and Google Scholar. Spatial data was obtained from a variety of sources (See Table 2.1 for details).

Table 2.1 Data used with this thesis

Data	Format	Source(s)
Literature on PPAs	.pdf	Web of Science, SCOPUS, Google Scholar
PA boundaries	.shp and .kml files	World Database of Protected Areas Multiple national level databases of PPA boundaries (see supp info.)
Terrestrial ecoregions of the world	.shp file	https://www.worldwildlife.org/publications/terrestrial-ecoregions-of-the-world
Key Biodiversity Areas	.shp file	Received upon email request to http://www.keybiodiversityareas.org/kba-data/request
Global Human Footprint	.shp file	https://sedac.ciesin.columbia.edu/data/set/wildareas-v2-human-footprint-geographic
Global Forest Change	.tif	https://earthenginepartners.appspot.com/science-2013-global-forest/download_v1.7.html
Precipitation Data	.tif	https://www.worldclim.org/data/index.html
Estimated travel time to the nearest city of 50,000 or more people in the year 2000	.tif	https://forobs.jrc.ec.europa.eu/products/gam/
Elevation data	.tif	https://dwtkns.com/srtm30m/
Colombia waterways	.shp	https://mapcruzin.com/free-colombia-arcgis-maps-shapefiles.htm
Colombia Roads	.shp	https://mapcruzin.com/free-colombia-arcgis-maps-shapefiles.htm
Population Data – Colombia	.tif	https://www.worldpop.org/geodata/listing?id=69

Administration regions – Colombia	.shp	https://data.humdata.org/dataset/colombia-administrative-boundaries-levels-0-3

Table 2.2. Summary of the research objectives, methods and data analysis used within the thesis

Objective	Research Questions	Data needed	Method	Data Analysis
1. To determine what is currently known about the environmental and social outcomes of privately protected areas	1. What is distribution of peer reviewed literature? 2. What are PPA environmental and social outcomes and how these have been measured? 3. Are outcomes are positive or negative and for whom? 4. What are the challenges of measuring PPA outcomes and what are future research needs?	Peer reviewed literature	Comprehensive literature review	Inductive Coding. Codes include: Environmental Outcomes *(e.g., Connectivity, Ecosystem Restoration)* Social Outcomes following the sustainable livelihoods frameworks (REF). Codes include: • Financial Capital • Social Capital • Human Capital • Physical Capital • Natural Capital Landowner type *(e.g., Individual, NGO, Corporate)* Governance entity *(e.g., Covenant, Individual,)*

				Protection Mechanism *(e.g., Conservation Easement, NGO Freehold*)
2. To globally analyse what attributes PPAs protect within their boundaries and how this compares to PAs of other governance types and to random placement	1. What is the overall coverage of PPAs? 2. To what extent do PPAs protect threatened or underrepresented biomes? 3. To what extent do PPAs protect areas of high human disturbance? 4. To what extent do PPAs protect KBAs? 5. To what extent to PPAs contribute to PA network connectivity? 6. How does PPA distribution compare with random placement?	PPA Boundaries WWF ecoregions Global Human Footprint Key Biodiversity Areas	Use of open-source data	Overlaying of spatial boundaries' in ArcMap10.4 Connectivity analysis using Conefor2.6 PPA distribution modelling in R

3. Compare how effective PPAs are at reducing deforestation compared with other PA governance types	1. How effective are PPAs reducing deforestation? 2. How do PPAs perform relative to other PA governance types?	Global Forest Change PA Boundaries Precipitation Data (WorldCLIM) Elevation / slope (STRM30) Access to market (Global Environment Monitoring Unit) Colombian Road Network Colombian waterways Population density (Worldpop)	Quasi-experimental design	Counter factual matching Generalised linear regressions

2.7 Research ethics

There were no particular sensitivities or serious ethical considerations in the completion of this thesis as the methods used reanalysed existing datasets rather than gathering original data. Moreover, no controversial topics or issued were studied. There are three main ethical considerations to this thesis. Firstly, it could be possible to identify landowners of PPAs studied in Chapter 5 due to existing datasets and therefore the decision was made to anonymise individual PPAs via a numbering system. Secondly, data ownership issues were minimal as all datasets used were open-source and free to use providing that a citation to the original data source was provided. Thirdly, some land designated as PPAs may have contested ownership and I have acknowledged this and been sensitive to this issue within my three empirical research chapters.

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Chapter 3: Conservation and Social Outcomes of Privately Protected Areas

3.1 Abstract

Government administered protected areas (PAs) have dominated conservation strategies, discourse, and research, yet private actors are increasingly managing land for conservation. Little is known about the social and environmental outcomes of these privately protected areas (PPAs). We searched the global literature in English on PPAs and their environmental and social outcomes and identified 412 articles suitable for inclusion. Research on PPAs was geographically skewed; more studies occurred in the United States. Environmental outcomes of PPAs were mostly positive (89%), but social outcomes of PPAs were reported less (12% of all studies), and these outcomes were more mixed (65% positive). PPAs increased the number or extent of ecosystems, ecoregions, or species covered by PAs (representativeness) and PA network connectivity and effectively reduced deforestation and restored degraded lands. Few PPA owners reported negative social outcomes, experienced improved social capital, increased property value, or a reduction in taxes. Local communities benefited from increased employment, training, and community-wide development (e.g., building of schools), but they reported reduced social capital and no significant difference to household income. The causal mechanisms through which PPAs influence social and environmental outcomes remain unclear, as does how political, economic, and social contexts shape these mechanisms. Future research should widen the geographical scope and diversify the types of PPAs studied and focus on determining the casual mechanisms

through which PPA outcomes occur in different contexts. We propose an assessment framework that could be adopted to facilitate this process.

3.2 Introduction

Biodiversity is in crisis, with extinction rates 1,000 times higher than expected background rates (Diaz et al. 2019). In response the international community has explicitly included biodiversity protection and the expansion of protected areas (PAs) in multiple international agendas, including the Aichi Biodiversity Targets and Sustainable Development Goals. Government administered PAs have dominated conservation strategies, discourses, and research for decades (Adams, 2004; Watson et al., 2014). However, a variety of private actors, including individuals, nongovernmental organizations (NGOs), and businesses are increasingly purchasing and managing significant tracts of land for conservation. These areas are collectively known as privately protected areas (PPAs) and are highly diverse in their form, ownership, size, and location. There are numerous definitions for PPAs (Holmes, 2013), but Stolton et al. (2014) provide a comprehensive and widely accepted definition that we use in this review: *"a protected area, as defined by IUCN, under private governance (i.e. individuals and groups of individuals; non-governmental organizations; corporations – both existing commercial companies and sometimes corporations set up by groups of private owners to manage groups of PPAs; for-profit owners; research entities (e.g. universities, field stations) or religious entities)."*

In contrast to other forms of PAs, PPAs have received relatively little scholarly attention (Capano et al., 2019). This despite their being an old conservation approach; some countries (e.g., United Kingdom) established PPAs decades before state-governed PAs

(Hodge and Adams, 2012). PPAs deserve greater attention because they may be increasing in number due to rising trends in neoliberal conservation approaches that facilitate a role for private actors (Büscher and Whande, 2007) and because there is a pressing need for conservation on private land to help achieve global conservation goals (Kamal et al., 2015). The World Database on Protected Areas (WDPA) reports 13,103 privately governed PAs (UNEP-WCMC et al. 2020). Yet, this may be a substantial underestimate because only a small proportion of countries report PPAs to the WDPA and these may also report only a subset of existing PPAs (Fitzsimons, 2015; Bingham et al. 2017).

PPAs can potentially make significant contributions to conservation in some countries (Holmes, 2013) and may operate differently from other forms of PAs due to different owner motivations and incentives, access to financial resources, and levels of accountability (Langholz and Lassoie, 2001). Existing global reviews of PPAs have focused on PPA typologies (Langholz and Lassoie, 2001; Carter et al., 2008; Kamal et al., 2015), their differences relative to other effective conservation measures (Mitchell et al., 2018), their geographical distribution (Stolton et al., 2014; Bingham et al., 2017), and PPA reporting (Clements et al., 2019) and management guidelines (Pasquini et al., 2011; Mitchell et al., 2018). Recent studies focusing specifically on outcomes have been region specific and explore the outcomes of private land acquisitions for forest conservation in the United States (Nolte, 2018), contributions of PPAs to the regional persistence of large- and medium-sized mammals in South Africa (Clements et al., 2019) and Brazil (Laurindo et al., 2017), how PPAs contribute to ecosystem representativeness in Victoria, Australia (Fitzsimons & Wescott, 2001), and the outcomes of conservation concessions in South America (Schleicher, 2018).

However, a global understanding of PPA outcomes for people and nature is lacking. We address this gap by synthesizing the published literature on PPAs to describe the geographic distribution of peer-reviewed PPA literature, summarize PPA environmental and social outcomes and how these have been measured, whether outcomes are positive or negative and for whom or what, and examine the challenges of measuring PPA outcomes and future research needs. We assessed ecological outcomes to see to what extent PPAs contribute to global biodiversity conservation goals. We assessed social outcomes of PPAs because it is now accepted that PA governers should be aware of and aim to avoid potential negative effects of PAs to local communities. Where negative effects (e.g., displacement, restriction of access to resources) is unavoidable, suitable compensation and mitigation should be put in place. . Social outcomes of PAs can determine their legitimacy and the level of support they receive from local communities and therefore their long-term persistence and effectiveness in achieving the biodiversity conservation goals they were meant to achieve. Social outcomes for owners are also important for the longevity and number of PPAs.

3.3 Compiling the Literature

We used the PRISMA method - which allows for the search of literature using key terms in conjunction with snowballing (Moher et al., 2009) to conduct extensive literature searches in Web of Science, SCOPUS, and the first 500 papers from Google Scholar in October 2019. We focused on PPAs in peer-reviewed journals in English. We assessed the gray literature on PPAs but decided to exclude it because of its limited scope. We assessed the gray literature through searches on Google Scholar, snowballing, and searching NGO and land trust websites (e.g., The Nature Conservancy, World Land Trust). Much of this literature in English focuses on defining PPAs (e.g., Stolton et al., 2014), how they should be managed (e.g., Mitchell et al., 2018), and where they can be found (e.g., American Bird Conservancy, 2013). Few reports focus on environmental outcomes (*n* = 2), and social outcomes centre on changes in land value following the establishment of conservation easements (*n* = 7). The gray literature was also difficult to systematically collate and posed challenges related to research quality and potential duplication of information (Oldekop et al., 2016; Hajjar et al., 2016). Although we excluded gray literature from our review, we believe our results nonetheless reflect important PPA trends and gaps and the way key issues are currently covered in the peer-reviewed literature. PPAs take many different forms (e.g., conservation easements or private game reserves). Using the comprehensive International Union for Conservation of Nature (IUCN) report The Future of Privately Protected Areas (Stolton et al., 2014), we compiled search terms to cover the diversity of forms of PPAs, which are widely reported and accepted. Our search strings utilised truncation or wild card symbols, as appropriate, to search for alternative spellings and endings or complete titles of known PPA forms. The complete search string is given below:

TOPIC = "private protected area*" OR "private nature reserve*" OR "private natural Heritage Reserve*" OR PNHR* or RPPN* OR "conservation concession*" OR "conservation easement*" OR "Conservation Conservan*" OR "conservation Covenant*" OR "Private Forest Reserve*" OR "Natural Reserve* of Civil Society" OR "private wildlife reserve*" OR "private wildlife refuge program" OR "private conservation area*" OR "private game reserve*" OR "informal community group".*

We screened all results in a three-stage process based on title, abstract, and full text, according to our study inclusion criteria. To be included, studies needed to first meet our definition of PPA. Confusion still exists as to what exactly classifies as a PPA, and the boundaries between what constitutes a PPA versus PAs under other forms of governance or other effective conservation measures can be ambiguous. We based our definition on that of the IUCN (Stolton et al 2014) and define PPAs as areas under private forms of governance; primarily used for biodiversity conservation; (iii) designated based on long-term intent; and (iv) that afford legal or other effective means of biodiversity protection. Although it can be argued that South African conservancies do not meet Stolton's definition we include them in our study because they are governed by a private entities (namely individuals), the main driving force of their establishment is conservation values (Selinske et al., 2014) and a survey of South African conservancy owners found that 92% of them undertook actions to conserve or protect biodiversity (e.g., eradicating invasive species from their properties) (Downsborough et al., 2011). Like Capano et al., (2019), we discarded articles reporting ecological surveys inside PPAs that did not relate the results to PPA management or governance (n = 15). We coded PPAs by landowner type, governance entity, and protection mechanism. We coded environmental and social outcomes according to the primary research question asked in the literature. We further categorised social outcomes based on the five livelihoods assets (financial, social, human, physical and natural capital) in the sustainable livelihoods' framework (DFID, 2000). We define capital as assets that all humans require in order to make a living. We define social capital as social resources, including networks for cooperation, mutual trust, and support, human capital as the amount and quality of knowledge, skills and labour available in a household as well as psychological benefits obtained from the creation of a PPA and natural capital as supporting, provisioning and regulating services that humans gain from the natural environment. We adopted the sustainable livelihoods' framework as it takes a holistic view of livelihoods, incorporates governance processes and has had some use in assessing conservation impacts (Ward et al., 2017, Bennett, 2010). We coded outcomes as positive (+), negative (-), or no discernible effect (~), and to whom or what the outcomes accrued too.

Our initial search returned 1,325 articles, which we reduced to 373 following title and abstract screening. We selected a further 39 papers from references lists, resulting in a final sample of 412 articles.

3.4 Results

We found an increasing trend in the number of peer-reviewed articles in English focusing on PPAs, but the overall number of articles remained small (*n* = 412, Figure 3.1) relative to the current number of PPAs (*n* = 13,103). The environmental and social outcomes of PPAs only recently received scholarly attention (Figure 3.1). The literature was substantially skewed in geographic focus (perhaps due to a sampling bias of conducting the literature search only in English) (Figure 3.2); the types of PPAs studied (Table 3.1); the types of questions asked about PPAs (Table 3.2); and the spatial scale at which research was conducted. Most studies were conducted at a subnational (*n* = 261) or national scale (*n* = 78). In contrast, landscape-level studies were uncommon (*n* = 21). Most studies were conducted in only five countries (United States *n* = 226, Brazil *n* = 31, Australia *n* = 31, South Africa *n* = 30, Chile *n* = 19), and studies on conservation easements in the United States dominated the literature (Figure

3.2, Table 3.1). There was marked overlap between country and PPA type studied (e.g., conservation easements and the United States). Marine PPAs, were largely absent (*n* = 6).

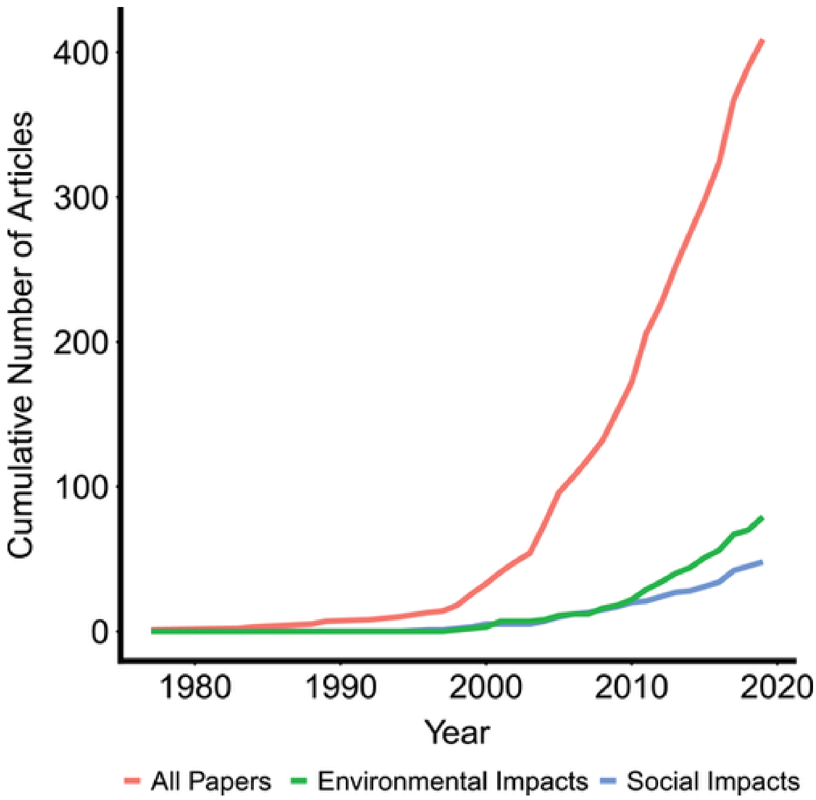

Figure 3.1 Cumulative number of peer reviewed articles on privately protected areas

Figure 3.2 Geographical distribution of articles on privately protected areas

Table 3.1 Types of privately protected areas (PPAS) included in a synthesis of the published literature

PPA Characteristics	No. of articles (n = 412)
Landowner type	
Individual	254
Multiple undefined*	89
Nongovernmental Organisation (NGO)	38
Unspecified**	18
Corporate	8
Informal Community Group	5
Governance Entity	
Convenant (unspecified**)	130
Multiple undefined	93
Individual-NGO Partnership (e.g., Landowner and The Nature Conservancy)	54
Individual	52
NGO	29
Individual-State Partnership	28
Unspecified	17
Corporate	8
Informal Community Group	2
Protection mechanism	

Conservation easement or convenant	250
Multiple undefined	56
Landholder agreement in perpetuity (e.g., RPPN in Brazil)	44
Unspecified	29
NGO freehold	26
Long-term landholder agreement	8

* Studies in which PPAs were reviewed or generalizations were made across PPAs but certain specific characteristics were not given

** Studies in which a case study was undertaken on a certain subset of PPAs in a specific region (e.g., conservation easements in Wyoming), but specific details were not provided

Studies largely focused on what drives PPA establishment, their geographical locations, and PPA definitions (Table 3.2). In contrast, relatively less attention was given to environmental and social outcomes of PPAs (Table 3.3, Figure 3.1).

Table 3.2 Focus of papers on private protected areas (PPAs)

Focus	**No. of articles (*n* = 412)**
Ownership characteristics, incentives, or motivations	84
Coverage (e.g., spatial distribution, representativeness, connectivity)	70
Opportunities, challenges, and constraints	5
Defining PPAs (e.g., typologies, classifications, history)	42
Management actions	38
Ecological effectiveness and impacts (e.g., reduce deforestation or prevent development)	31
Permanence	26
Financial analysis (e.g., how establishment affect land prices)	25
Governance (e.g., participation of local communities, collaborative governance)	17
Social impacts	16
Political economy (e.g., neoliberalism, land grabbing, resource nationalism)	11

Table 3.3 Assessments of the environmental outcomes of privately protected areas (PPAs)

Study Focus	**No. of Articles**	**Method**	**Increase (+), decrease (-), or no discernible effect (~)**	**Study**
Species conservation	37			
Species abundance	8	Biodiversity survey	+	Burgi et al. 2011; Higgins et al. 1999; Tapp et al. 2015; Benson et al. 2018
		Spatial analysis	+	Herzog & Vaughan (1998), Pegas & Castley, (2016), Child et al., (2013)
		Spatial analysis	-	Olmstead et al., (2013)

Projected estimates of PPAs to conserve species in future scenarios	19	Analysis of secondary data or modelling	+	Cox & Engstrom, (2001), Stralberg et al., (2011), Copeland et al., (2013) Smith et al., (2016), Lewis et al., (2019)
		Biodiversity survey	+	Cabral et al., (2017) Dos Santos & Da Costa, (2008), Falcão et al., (2012), Gatti et al. (2017), Laurindo et al., (2017), Porfirio et al., (2014), Posso et al., (2013), Ruiz-Esparza et al., (2016), Sánchez-Lalinde et al., (2019), Talamoni et al., (2014), Zortéa et al., (2008), Jones & Jiménez-Saa, (2017), Clements et al., (2019)
		Spatial analysis	~	Sandker et al., (2011)
Compliment species protection in other PAs	9	Biodiversity survey	+	Rambadli et al., (2005), Colletta et al., (2016), Shanee et al., (2017), Negroes et al., (2011), Lovett-Doust & Kuntz (2001)

		Spatial analysis	+	Pegas & Castley, (2016), Munks et al., (2004), Alarcón & Cavieres, (2015), Maslo et al., (2015)
Protect species of conservation concern	1	Biodiversity survey	+	Ortiz-Lozada et al., (2017)
Ecosystem representativeness	20			
Ecosystem representativeness	18	Analysis of ecoregions, plant species diversity, or ecosystems in PPA boundaries compared with other PA types	+	Squeo et al., (2011), Martinez-Tilleria et al., (2017), Pliscoff & Fuentes-Castillo (2011), Lemeanger et al., (2014), Baldwin et al., (2015), Gallo et al., (2009), Von Hase et al., (2010), Shanee et al., (2017), Graves et al., (2019), De Vos & Cumming (2019), Yuan-Farrell et al., (2005), Fitzsimons & Wescott, (2001)

		Analysis of ecoregions, plant species diversity, or ecosystems in PPA boundaries compared with other PA types	~	Jackson & Gaston (2008), Larrea-Alcazar et al., (2010), Schutz (2017), Yuan et al., (2015), Lacher et al., (2019)
		Analysis of ecosystems protected in future PPA scenarios	+	Chomitz et al., (2006)
Protect or restore conservation priorities and human values	2	Spatial analysis of overlap of ecosystems protected and desirable human values	+	Fisher et al., (2012), Cronan et al., (2010)
Connectivity and adjacency	15	Spatial analysis of PPA locations assessing contiguity and connectivity of PAs	+	Crouzeilles et al., (2013), Chomitz et al., (2006), Gatti et al. (2017), Langholz and Lassoie (2001), Rissman & Merenlender (2008), Meyer et al., (2015), Graves et al., (2019), Tack et al., (2019), Lawley et al., (2015), De Vos and Cumming (2019),

				Lovett-Doust & Kuntz (2001), Pegas & Castley., (2016)
		As above	~	Rissman (2013), Cronan et al., (2010), Lacher et al., (2019)
Land restoration	8	Field surveys to determine reduction in pollutants, increases in ecosystem function	+	Benson et al., (2018), Burgi et al., (2011), Forshay et al., (2005), Bunnell-Young et al., (2017), Sonnier et al., (2018), Tang et al., (2016), Tapp et al., (2018)
		Spatial analysis of reforested area	+	Zambrano et al. (2008)
Deforestation and Degradation	5	Biodiversity surveys	+	Turyahabwe & Tweheyo (2010)
		Matched similar areas under different PA governance types to determine deforestation rates	+	Scheicher et al., (2017), Vuohelainen et al., (2012), Song et al., (2014)

		As above	~	Noone et al., (2012)
Land-cover change (nonforests)	3	Matched similar areas under different PA governance types to determine land cover change	+	Braza (2017), Wu (2000)
		Spatial analysis	~	Gonzalez-Roglich et al., (2012)
Development prevention	4	Modelled projection of development with or without PPAs	~	Byrd et al., (2009),
		As above	+	Smith et al., (2016)
		Analysis of degree of naturalness of protected land under different governance types	~	Fouch et al., (2019)

Inside PPA and outside PPA comparison of road densities	+	Pocewicz (2011)

Environmental outcomes were considered in 79 studies and focused mainly on species coverage (*n* = 37) and ecosystem representativeness (*n* = 20). Social outcomes were discussed in 48 studies, the majority of which focused on financial outcomes (*n* = 36).

Results of studies on ecological outcomes of PPAs showed many positive outcomes, particularly for species conservation (increase *n* = 35, decrease or no effect, *n* = 2) (Table 3.3). Social outcomes of PPAs were far less reported and more mixed. Studies that researched social outcomes of PPAs showed local communities benefitted from skills training (*n* = 6), infrastructure development within the local area (n=4), improvements to the regional economy (*n* = 5), and increased employment opportunities (+, *n* = 9, -/~, *n* = 3). However, there was little improvement in household income (+, *n* = 2, -/~, *n* = 5). Some local communities reported feeling a loss of power and cultural identity (*n* = 9). Due to the bias in papers focusing on individual landowners (Table 3.1), results also showed that the general public lost tax revenue (*n* = 4) and access to open space (*n* = 4) and that landownership inequality increased (*n* = 3). In contrast, landowners benefitted from increased land value (*n* = 8), reduction in tax payments (*n* = 4), and strengthened community involvement, relations, and networking (*n* = 6). Within our study, few PPA owners reported any negative impacts (*n* = 6).

3.5 Discussion

3.5.1 Characterising the literature on PPAs

Research on PPAs is geographically and ecologically limited, reflecting global skews in conservation research (Oldekop et al., 2016; Capano et al., 2019). We found a marked overlap between the country and PPA type studied, perhaps because certain types of PPA management may be unique to, or more dominant in, specific countries (e.g., Private Natural Heritage Reserves [Reservas Particulares do Patrimônio Natural] [RPPNs] in Brazil). The bulk of the PPA literature focused on conservation easements in the United States (*n* = 216, 52% of all studies), perhaps due to their prominence and large numbers (Nolte, 2018; UNEP-WCMC et al., 2020). Conservation easements and covenants are contractually binding agreements between landowners and a third party (e.g., land trusts or governments) that dictate how properties should be managed alongside conservation goals (Merenlender et al., 2004).

Mexico, Canada, Colombia, Namibia, Spain, and Finland have growing PPA networks (Stolton et al., 2014) and receive limited scholarly attention. Studies commissioned by NGOs in these countries were not published in the peer review literature. Countries with a large number of PPAs reported to the WDPA received greater scholarly attention than countries with few reported PPAs. The United Kingdom was an exception. It has a large number of PPAs managed by NGOs (Stolton et al., 2014) reported to the WDPA (*n* = 690), but they were little discussed in the peer-reviewed literature (*n* = 2).

Limited questions have been asked about PPAs; 38% of articles (*n* = 155) investigated the location of PPAs or ownership characteristics, incentives, and motivations for PPA establishment (Table 3.2). These research questions reflect an exploratory research agenda and demonstrate a trend of research heavily dominated by factors shaping PPA establishment and aims (inputs), rather than results (outputs) (Tables 3.3 & 3.4).

Table 3.4 Assessment of the social outcomes of privately protected areas (PPAs)

Study Focus	No. of Articles	Method	Increase (+), Decrease (-), and No discernible effect (~)	Assessment recipient	Study
Financial	36				
Employment opportunities	12	Questionnaires and interviews	+	Local community	Hora (2018), Hora (2017), Zambrano et al., (2010), Sims-Castley et al., (2005), Barany et al., (2010)**, Langholz (1996)
			+ / -		Serenari et al., (2017)
			~		Serenari et al., (2016)
			-		Louder & Bosak (2019)
		Case study	+	Local community	Dodds (2012)

			-		Buergin (2016)
		Quasi-experimental design	+	Local community	Sims et al., (2019)
Household income	10	Questionnaires and interviews	+	Local community	Hora (2017), Sims-Castley et al., (2005)
			~	Local community	Hora (2018), Spenceley & Goodwin (2007), Zafra-Calvo & Moreno-Penaranda, (2017)
		Case study	+	PPA owners	Rissman & Sayre (2011), Maynard et al., (1998)
		Financial analysis	+	PPA Owners	Ulisses Saraiva Farinha et al., (2019)
		Modelling	-	Local community	Sandker et al., (2011)

		Quasi-experimental design	~	Local community	Sims et al., (2019)
Land or property value	12	Questionnaires and interviews	+	Local community	Hora (2018)
		Mmodeling using secondary data	+	Landowners of PPAs	Ulisses Saraiva Farinha et al., (2019)
		Financial analysis	+	PPA owners	Schilling et al., (2013)
			+	Landowners surrounding PPAs	Zhang et al., (2018), Reeves et al., (2018), Yoo & Ready, (2016), Chamblee et al., (2012), Armsworth et al., (2006), Farja (2017)
			-	PPA owners	Lawley et al., (2014), Anderson & Weinhold, (2008)

			-	Nonland owners (renters)	Farja (2017)
Tax payments	4	Financial analysis	+	PPA owners	Sandre-Drake (1999), Crompton (2009), Jurinski & Goveia, (2000), Forshay et al., (2005)
Tax revenue	4	Financial analysis	~	Local government	King & Anderson (2004)
			-		Vercammen (2017), Crompton (2009), Anderson & King (2004)
Regional economy	5	Interviews & questionnaires	+	Local community	Zambrano et al., (2010), Child et al., (2013), Sims-Castley et al (2005), Barany et al., (2010)**
			+ / -	Local community	Serenari et al., (2017)
Ability to access grants or funding	1	Interviews	+	PPA owners	Horton et al., (2016)

Physical capital	5				
development in the area (e.g., road improvements, building schools)		Interviews and questionnaires	+	local community	Hora (2017), Serenari et al., (2017), Zambrano et al., (2010)
			~		Hora (2018)
		Case study	+	local community	Buergin (2016)
Social capital	12				
Strength community involvement, relations and networking	7	Interviews and questionnaires	~	local community	Hora (2018),
		Case Study	+	PPA owners	Rissman & Sayre (2011), Horton et al., (2017)

		Questionnaires	+ / -		Maciejewski et al., (2016), Selinske et al., (2015), Pasquini et al., (2010)
		Interviews	+ / ~		Harrington et al., (2006)
Strengthen or maintain cultural identify	3	Interviews and questionnaires	+	local community	Hora (2018)
			-		Louder & Bosak (2019)
			+	PPA owners	Maynard et al., (1998)
Strengthen power relations or ability to make decisions	3	Interviews	-	local community	Louder & Bosak (2019), Serenari et al., (2017)
			+ / -	PPA owners	Horton et al., (2017)
Land-ownership equality	3	Interviews	-	local community	Langholz et al., (2000)***, Serenari et al., (2017)
		Case study	-	local community	Quintana & Morse (2005)***

Human capital	5				
Improve environmental education	4	Questionnaires and interviews	~	local community	Hora (2018)
		Interviews	+	local community	Serenari et al., (2016), Serenari et al., (2017)
		Case study	+	local community	Dodds (2012)
New skills (e.g., diving, tour guiding, baking, cooking)	3	Case study	+	local community	Dodds (2012)
		Interviews	+	local community	Hora, (2017), Serenari et al., (2017)**
Natural capital	9				
Access to open space, cultural heritage, or	7	Interviews and questionnaires	-	general public	Crompton (2009), Owley (2015), Rissman & Merenlender (2008), Lieberknecht (2009)

recreation (cultural services)					
			+	PPA visitors	Clements & Cumming (2017), Langholz (1996)
		Modelling	+	PPA visitors	Nahuelhual et al., (2013)
Regulating services (e.g., erosion control, surface water regulation)	1	Modelling	+	everyone (but PPA owners benefit more)	Villamagna et al., (2017),
Access to forest resources (provisioning services)	1	Interviews	-	local community	Serenari et al., (2017)

*We define local community as a group of individuals who live in the area immediately surrounding a PPA

**Outcomes especially for women

*** Outcomes felt most by non-wealthy community members

3.5.2 Environmental outcomes of PPAs

We found that PPAs made unique and significant spatial contributions to achieving some global conservation targets and overwhelmingly had positive ecological outcomes (89%, *n* = 70). Globally, state PAs account for 82% of total PA coverage, whereas PPAs account for ~7% (UNEP-WCMC et al. 2020). PPAs added little to the total protected land area (additionality), and extent of PPA coverage was much smaller than that of state PAs. However, 72% of papers (*n* = 13) discussing ecosystem representativeness suggest PPAs add complementarity to the PA matrix by existing in ecoregions not represented or underrepresented by state PAs or by existing in less remote areas that are more suitable for agricultural or urban development (Pegas & Castley, 2016; De Vos & Cumming, 2019). PPAs have been reported to protect species not recorded in state PAs (Shanee et al., 2017). Eighty percent of papers (*n* = 12) discussing connectivity showed PPAs increase the contiguity and connectivity of PAs by being adjacent to other PAs (Rissman & Merenlender, 2008) or by forming parts of wildlife corridors increasing connectivity between PAs of other governance types (De Vos and Cumming, 2019). The remaining 20% (*n* = 3) exclusively studied conservation easements in the United States and showed they add little to PA network connectivity because they are often small and do not border other PAs (Graves et al., 2019).

Overall, different countries had unique spatial configurations of PPAs that lead to varied conservation outcomes, potentially because within each country, PPAs establishment is shaped by different factors (Nolte, 2018).

Few studies monitored or evaluated the ecological effectiveness of PPAs. Those that did defined effectiveness as the degree to which a PPA achieves a successful outcome for biodiversity conservation as defined by their own unique study criteria. Eighty percent (*n* = 5) of papers in which deforestation rates were analysed showed that PPAs are more effective at reducing deforestation and degradation than PAs under other governance types (Schleicher et al., 2017; Nolte et al., 2019). Sixty six percent of studies examining landcover change (*n* = 2) showed PPAs are effective at reducing landcover change in non-forest areas. All studies assessing ecological restoration (*n* = 8) showed PPAs have positive outcomes for restoring degraded lands. Most of these studies focused on wetlands in the United States and showed PPAs can increase wetland functionality, reduce pollution, increase flora and fauna diversity, and contribute to recovery of species in greatest conservation need (Benson et al., 2018). Half the studies (*n* = 2) that empirically assessed the impacts of PPAs on development prevention reported reductions in development and the other half reported no discernible changes.

Ninety five percent (*n* = 34) of papers examining species conservation showed PPAs achieve positive outcomes. Empirical exploration of PPAs' ability to protect or increase specific species' populations showed PPAs can significantly increase numbers of wetland bird species compared with unprotected sites (Tapp et al., 2018) and that they may play a substantial role in the long-term conservation of large- and medium-sized mammals (Laurindo et al., 2017; Clements et al., 2019). Model-based studies to predict future PPA impacts suggested they may contribute to the conservation of key species (Copeland et al., 2013). Only 1 study explored the spillage effects of PPAs (Wu, 2000).

3.5.3 Social outcomes of PPAs

Social outcomes of PAs take different forms, including economic, livelihood, and cultural outcomes (Oldekop et al., 2016). We found PPA outcomes echoed the common outcomes of other types of PAs; however, private entities may have different levels of accountability than non-private equivalents. Moreover, accountability across different PPA types (e.g., private landowner and NGO) may also vary widely.

We found studies on the social outcomes of PPAs focused predominantly on financial outcomes (n = 35, 73%). Eighty two percent of studies (n = 9) discussing employment reported PPAs increase employment opportunities for local communities, and Sims et al., (2019) suggest PPAs may have greater positive impacts for employment than state PAs. However, only 29% of studies (n = 2) commenting on household income reported that PPAs increase the household income of local communities, and Sims et al., (2019) found no difference in median household income between state and private PA governance types. Moreover, some studies reported PPAs could increase inequalities within communities because poorer households, those less able to capitalize on tourism opportunities, or people living farther from reserve boundaries benefitted less than others from PPA establishment (Serenari et al., 2016; Hora, 2017)

Eight studies (80%) quantifying changes in land value showed landowners benefit from increased land value after designating a PPA. However, Farja (2017) reported this can have detrimental effects for nonlandowners by facilitating a concentration of land ownership and exacerbating inequalities. Last, where PPAs were used in tourism, studies in Costa Rica, Nicaragua, Chile, and South Africa showed PPAs can have a positive impact for regional economies. However, in the United States (where easements are not used in tourism and more likely to be family ranches), studies showed PPAs reduce regional tax revenue (Crompton, 2009).

The broader social costs and consequences of livelihood shifts linked to PPAs have not been systematically studied (Spierenburg & Brookes, 2014). Trade-offs may exist between financial gains and social and cultural costs. Two studies reporting on cultural identity (66%) showed that local communities sense a loss of cultural identity and values and community cohesion. This may be because non-locals move into the area and introduce new cultures and ideas, and as opportunities for greater financial income increase, it can generate competition within communities (Serenari et al., 2017; Büscher et al., 2018).

PPAs can redistribute political resources, particularly control over land. They have sometimes been perceived as land grabs, illegitimate and harmful land acquisitions by foreign and local elites with negative outcomes for local communities (e.g., Langholz et al., 2000; Serenari et al., 2017; Büscher et al., 2018). All the studies in our review commenting on landownership inequality (n = 3) reported an increase in land ownership inequality and negative outcomes for non-wealthy community members in areas where PPAs were established. In contrast, 6 studies (80%) showed that individuals who own, create, and govern PPAs (e.g., through conservation easements) may obtain greater social benefits (e.g., building social networks) and political empowerment (e.g., being able to have greater influence over development decisions [Rissman & Sayre, 2011]) and are able to maintain their cultural identity (Maynard et al., 1998).

Nine studies discussed PPA outcomes on natural capital. Villamagna et al. (2017) reviewed the distribution of ecosystem service benefits from PAs. They found that PAs offer benefits for all, but the benefits disproportionately benefit households with greater income and beneficiaries of ecosystem services from PPAs in particular have a significantly greater household income than all other beneficiaries of ecosystem services from other PA governance types. Crompton (2009) found public benefits of conservation easements emerge serendipitously to the public and that most benefits accrue to landowners. These findings are important because enhancing the equity of benefit delivery from PPAs will build public and private support for them as a long-term conservation strategy and increase conservation efficacy. We found no empirical studies on the magnitude of impacts that PPAs have on sequestering carbon or improving water quality, although Kreuter et al., (2010) found private nature reserves exhibit some of the critical conditions for the sustainability of common-pool resources. These studies are needed because PPA creation may be driven by REDD+ incentives that claim to provide ecosystem services, such as carbon sequestration (Schleicher, 2018). Due to a bias in articles focusing on individual landowners Table 3.1), 100% of studies investigating cultural services showed PPAs have negative impacts for local communities (*n* = 5) (e.g., access to open space and forest resources) (e.g., Serenari et al., 2017), but have positive impacts for paying PPA visitors (Clements & Cumming, 2017). It is unclear the extent to which people had access to land before its establishment as a PPA because the land may have been privately owned with limited public access.

A small number of articles (*n* = 7) briefly mentioned PPA outcomes on physical and human capital. Some PPAs may encourage infrastructure developments for local communities (e.g., roads and building of schools) (Barany et al., 2001; Serenari et al., 2016), and PPAs involved in tourism may offer training or facilitate access to education for local staff (Hora, 2017).

3.5.4 Current approaches to determine PPA outcomes

Research approaches varied in the scale and rigor of analysis (Table 3.3 & Table 3.4). Quasi-experimental designs to measure PPA effects on deforestation and forest degradation reflect broader trends in the use of such methods to assess outcomes of natural resource management and conservation interventions (Ferraro & Hanauer, 2014). We are aware of only one study that applied these methods to assess PPA outcomes for land restoration (Sims et al., 2019). In assessments of PPA outcomes for species of conservation interest, researchers either modelled projected future outcomes (e.g., Copeland et al., 2013) or focused on individual case studies based on primary data (e.g., Negroes et al., 2011). Methods to assess the social outcomes of PPAs almost exclusively focused on semi-structured interviews and mailed questionnaires. In most studies, a variety of stakeholders (e.g., government officials, PPA owners, local communities) were interviewed and the number of respondents was large relative to the total population size. Only three (Langholz et al., 2000; Hora et al., 2017; Serenari et al., 2017) out of 36 studies combined methods and data sources to triangulate results, raising questions about the strength of many conclusions regarding the social outcomes of PPAs. Only 1 study (Sims et al., 2019) used quasi-experimental techniques to assess the social outcomes of PPAs.

3.5.5 Challenges to assessing PPA outcomes

The global number of reported PPAs is believed to be a significant underestimation of total number in existence (Stolton et al., 2014; Bingham et al., 2017). While we acknowledge some countries have good national-level spatial data for PPAs (e.g., South Africa) (De Vos &

Cumming, 2019), others do not (e.g., Canada (Stolton et al., 2014) or data may not be publicly available (e.g., Australia (Stolton et al., 2014)). Moreover, the quality of spatial point and polygon data on the location of PPAs is highly variable, depending on the original data source (Milam et al., 2016). For example, there may be mismatches in the reported area and actual area of the PPA or PPA locations may be given as points (with a written area attached) that do not convey the actual shape of the PPA and the area it covers on the ground (Bingham et al., 2017). There is rarely data that would allow a detailed assessment on the contribution of PPAs to landscape-scale conservation, beyond presence or absence of a PPA, making any assessment of their additionality, complementarity, or connectivity a best guess. For example, in some areas, such as South Africa, PPAs are often fenced and thus impermeable to animal movements, limiting their effective contiguity, yet such data are rarely reported (Jakes et al., 2018). Quasi-experimental approaches are increasingly being used to address limitations of before and after and inside versus outside reserve comparison methods to determine PPA environmental and social outcomes (Schleicher, 2018). Yet these studies rely on good quality spatial data, which may be scarce for PPAs in some regions.

3.6 Limitations to the literature

Only peer-reviewed literature written in English was reviewed in this study possible leading to a regional bias in the studies reviewed and a bias towards individual owners. Moreover, our findings may be biased due to a risk of cherry picking in the literature. For example, within our sample, studies in Chile almost exclusively discuss the negative social costs of PPAs. PPAs in Chile has been surrounded in controversy due to the purchase of large swathes of land by Douglas Tompkins. The authors of these papers may have chosen to study PPAs in Chile (as opposed to any other country) as PPAs are known to have negative connotations in this area and so studies here will confirm and reinforce their own existing beliefs and opinions regarding PPAs. There may also be a bias in the papers found in this study due to how PPAs are described and defined in different countries. For example, there are over 6,000 PPAs recorded in the WDPA in the UK but our study only found two UK based studies. This may be because UK reserves are talked about in terms of "private land conservation" rather than as PPAs due to the complicated governance arrangements as it may be private actors who take on the daily management on PA (e.g., the Royal Society for the Protection of Birds (RSPB)) but the land is owned by the state.

Our study uses vote counting to calculate the number of studies which report the positive or negative impacts of PPAs. There can be issues with using vote count. Firstly, whether a study is determined to express a positive, negative or no impact is subjective based on the person conducting the literature review. Secondly, vote counting does not account for the magnitude of the effects in the studies. For example, you may have three studies which state that PPAs have very minor positive effects for reducing deforestation and one studies which show that PPAs have large negative effects and induce extensive deforestation – when conducting simple vote counting this nuance would be lost. Despite these limitations, I used vote counting in this study because standard meta-analytical methods could not be applied because papers did not have a consistent outcome measure

3.7 Future Research needs

Our study offers a comprehensive review of PPA peer-reviewed literature, but this could be expanded by including non-English and NGO literature, which would help address regional biases and bias toward individual owners within our results. We found there is a need to measure and report the diverse outcomes of PPAs, as well as examine the underlying factors that make PPAs effective, which is currently absent within the literature. These insights could help maximise potential PPA benefits and minimise negative outcomes. We propose an assessment framework that could be adopted to facilitate this process. The framework should include determining the extent to which PPAs achieve their desired environmental and social outcomes (e.g., extent to which landscape restored or poverty alleviated); how PPAs operate as institutions (e.g. who are PPAs stakeholders, what are the distributions of power and agency between different stakeholders, and to whom are the stakeholders accountable); and how the positive outcomes of PPAs (if any) are shared among stakeholders and the local communities surroundings PPAs. We envisage this framework could be used by PPA owners to self-report and by academics and government bodies to objectively assess PPA outcomes. This will require strengthening data collection efforts on the distribution of PPAs and their environmental and social impacts (e.g., deforestation rates with PPAs boundaries or changes in multi-dimensional poverty surrounding PPAs) to accompany the rise in quasi-experimental approaches, as well as qualitative research initiatives to assess more intangible social impacts of PPA interventions.

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Chapter 4: Private protected areas contribute to global protected area coverage and increase PA network connectivity

4.1 Abstract

Privately protected areas (PPAs) are increasing in number and extent. Yet, we know little about their contribution to conservation, and how this compares to other forms of protected area (PA). We address this gap by assessing the contribution of 17,561 PPAs to the coverage, complementarity and connectivity of existing PA networks in 15 countries across 5 continents. We find that PPAs (i) are three times more likely to be in biomes with <10% of their area protected than other PA governance types and twice as likely to be in areas with the greatest human disturbance; (ii) that they protect a further 1.2% of Key Biodiversity Areas; (iii) that they account for 3.4% of land under protection; and (iv) that they increase PA network connectivity by 7.05%. Our results demonstrate the unique and significant contributions that PPAs can make to the conservation estate and that PPAs deserve more attention, recognition and resources for better design and implementation.

4.2 Introduction

Terrestrial protected areas (PAs) cover approximately 16% of the world's land mass UNEP-WCMC and IUCN (2021). However, PAs are disproportionately established in higher and steeper areas that have lower agricultural and economic potential (Venter et al., 2018). Therefore, the global PA network underrepresents key species and ecosystems, lacks connectivity, and does not adequately protect areas of high conservation importance. The global PA network thus fails its own goal to comprehensively conserve biodiversity (Ward et al., 2020; Venter et al., 2018). State governed PAs dominate conservation strategies in most countries (Adams, 2004), but government action alone will be insufficient to reach global PA targets (Butchart et al., 2015; Watson et al., 2014). Co-managed, community governed and privately protected areas (PPAs) are increasingly being used as tools to increase PA coverage and connectivity and complement existing state PA networks.

PPAs are defined as areas that (i) are governed by private actors; (ii) are primarily engaged in biodiversity conservation activities and have long-term intent to remain in place; and (iii) have legal or other effective means of protection (Stolton et al., 2014). PPAs vary in landowner types and governance authorities (e.g., individuals, non-governmental organisations (NGOs) or corporate businesses) and protection mechanisms (e.g., conservation easements, NGO freeholds or perpetual landholder agreements). As of November 2018, the World Database on Protected Areas (WDPA) reported 13,250 PPAs representing 5.7% (324,851 km^2) of the total number of all PAs (UNEP-WCMC and IUCN, 2018), although this is likely to be a significant underestimation because less than 20 countries legally recognise or report PPAs (Bingham et al., 2017). Despite apparent global increases in PPA establishment, recognition and reporting efforts, very little is known about their contribution to the global conservation estate. Moreover, most countries fail to plan or co-ordinate PPA establishment to maximize their conservation benefits (Schitz, 2018; Gallo et al., 2009). It is thus critical to assess the distribution of PPAs to better understand their contributions to the global conservation estate and identify their potential to help achieve global biodiversity targets.

Previous studies suggest that PPAs make different contributions to the conservation estate, compared to PAs under other forms of governance. PPAs tend to be located at lower elevations (Gallo et al., 2009), closer to human settlements (Gallo et al., 2009), in underrepresented ecoregions (Schutz, 2018), and in areas of high conservation priority (Ielyzaveta & Cook, 2020). PPAs have also been found to increase overall PA network connectivity (Graves et al., 2019; De Vos & Cumming, 2019). Yet, these studies have been conducted at national or sub-national levels: to date no international-level analysis has been conducted. Such international-level analyses are needed to provide a more nuanced picture of the current contributions of PPAs to global conservation efforts. This information is critical for better informed global conservation planning, including ecoregion-based conservation strategies that support transnational ecological processes and biodiversity. Furthermore, most studies compare PPAs to state PAs and exclude co-managed or community governed PAs (Schutz et al., 2018; Gallo et al., 2009). As conservation approaches continue to diversify, comparisons of different approaches will become more important to determine where, when and why different PA governance types deliver positive biodiversity outcomes.

Here, we conduct the largest study to date on the contributions of PPAs to the global conservation estate. We analyse the contributions of 17,561 PPAs to terrestrial PA networks in 15 countries (Australia, Belize, Brazil, Canada, Chile, Colombia, Finland, Guatemala, Honduras, Kenya, Mexico, Namibia, Peru, South Africa and the USA). Collectively, our case countries represent a wide variety of global ecoregions (377) (Olson et al., 2001) and biodiversity hotspots (13) (Myers et al., 2000). Our study seeks to understand the contributions of PPAs to conservation on an international scale and to assess how these contributions differ to state, co-managed and community governed PAs.

State PAs are governed by federal or national ministries, sub-national ministries or agencies or are areas that have sub-delegated management (e.g., to an NGO) (Borrini-Feyerabend et al., 2013). Co-managed PAs have collaborative management arrangements across different organisations or groups (e.g., La Reserva Nacional Pampa Galeras Bárbara D'Achille, which has a collaborative governance arrangement between the Peruvian government and resident indigenous communities); or transnational boundaries (e.g., Roosevelt Campobello International Park, which is owned and governed by both the American and Canadian government) (Borrini-Feyerabend et al., 2013). Community governed PAs are defined as indigenous peoples' conserved areas and territories, or community conserved areas that are declared and run by local communities (Borrini-Feyerabend et al., 2013).

Specifically, we focus on PPA contributions to: (i) PA network coverage; (ii) coverage of threatened or under protected biomes (<10% of biome under protection in case countries); (iii) coverage of Key Biodiversity Areas (KBAs); (iv) coverage of areas of high human disturbance; and (v) PA network connectivity. To generate a better estimate of the relative contribution of PPAs to areas of conservation importance and connectivity, we compare the performance of existing PPAs to a counterfactual dataset with random PPA placement. We limit our analysis to countries with a minimum of 10 PPAs reported to the WDPA to ensure our results offer a more balanced interpretation of the contributions of PPAs. Moreover, to ensure adequate reporting of PPAs, and to justify inclusion within our study, countries in our sample have at least one of the following: (i) legal recognition of PPAs; (ii) national PPA legislation; or (iii) a national PPA database (Gallo et l., 2009). Although our dataset does not represent a full census of PPAs, it compiles the best currently available data to determine the contributions of PPAs to the global PA estate and provides an important insight into the potential contributions of PPAs in the future (see 4.5 Methods).

4.3 Results & Discussion

4.3.1 Coverage

We find that across our 15 case countries, PPAs cover 246,586 km^2 (an area equivalent to the size of the United Kingdom), accounting for 3.4% of total PA network coverage in these countries. By comparison, state PAs, co-managed, and community governed PAs account for 4,620,065 km^2 (63%), 572,278 km^2 (7.8%) and 1,852,381 km^2 (25%) of total land area under protection, respectively (see Appendix C– Table A.3). Across the entire WDPA, PPAs account for 1% of the total area of PAs with a reported governance type, with state, co-managed and community governed PAs accounting for 70.5%, 28% and 0.5%, respectively (UNEP-WCMC and IUCN, 2018).

We find substantial variation in the contribution of PPAs to PA networks in individual countries. South Africa has the highest PPA coverage (accounting for 25% of total protected area within the country) and Canada the lowest (PPAs account for 0.02% of the total protected area in the country) (see Appendix C– Table A.3). This variation is likely the result of historical, environmental, demographic, and economic idiosyncrasies. Across our case countries, differences in PPA distribution could arise from: (i) the difference between the common law system, a legacy of British Colonial Settlement that facilitates private land ownership, and civil law systems used by other European colonial powers, which make private land ownership harder to obtain (Lee & Schultz, 2012; Acemoglu et al., 2001); (ii) presence of established non-governmental PPA networks (e.g. RESNATUR in Colombia and ICMbio in Brazil) that encourage the creation and facilitation of PPAs from a grassroots level; and (iii) presence of and differences in economic incentives. PPAs in South Africa are, at least in part, the result of provincial ordinances. These ordinances have allowed game management and ownership of private land (De Vos et al., 2019), providing an incentive to establish PPAs in grasslands and next to national parks to take advantage of nature-based tourism activities. In the USA, the six of the largest conservation incentive programs (The Conservation Reserve Program (CRP), The Agricultural Conservation Easement Program (ACEP), The Environmental Quality Incentives Program (EQIP), The Conservation Stewardship Program (CSP), The Regional Conservation Partnership Program (RCPP) & Conservation Techincal Assistance (CTA)) target agricultural land (mostly in grassland biomes) and encourage farmers and ranchers to take land out of intensive agricultural production (United States Department of Agriculture, 2019). Financial incentives for grassland conservation also exist in Australia (e.g., plainstender) (Zimmer et al., 2010).

Gaining a better understanding of how different incentive mechanisms (both within and between countries) shape the establishment of PPAs will be essential for the creation of more effective management and monitoring systems.

4.3.2 Representation of Biomes, Key Biodiversity Areas and Human Disturbance

In line with Aichi Target 11, we assess the extent to which PPAs contribute to a conservation estate that is ecologically representative (i.e., a conservation estate that contains adequate samples of the full range of existing ecosystems and ecological processes, including at least 10% of each ecoregion within each country) and protects areas important for biological conservation. Biomes represent biodiversity at a broad level and are the most suited biodiversity metric for assessing ecosystem representativeness at an international scale (Olson et al., 2015). KBAs highlight sites of global importance for biodiversity that should be prioritised for conservation interventions (IUCN, 2016). We assessed the contribution of PPAs to overall representativeness of PA networks by calculating the area of each biome protected by PPAs and the contribution of PPAs to protecting areas important for biological conservation by calculating the area of KBAs protected by PPAs. We also assess to what extent PPAs protect areas of high human disturbance by calculating the Human Footprint (HF) both within and outside of PAs (Venter et al., 2018). We choose the HF because it shows to what extent PPAs are situated in threatened areas and whether they conserve areas of potential conservation concern. PPAs in these areas may protect the last best habitat in a matrix of otherwise degraded lands or be situated in already degraded lands that PPA owners may potentially aim to restore.

4.3.2.1 Biomes

Within our sample, we find PPAs are three times as likely to be in biomes that do not have 10% of their total area under protection, compared to other PA governance types. We find that 12% of total area of PPAs is in biomes with <10% of their total area protected, compared with 3.9%, 2.3% and 0.5% of state, co-managed and community governed PAs, respectively (Figure 4.1). We find that 3.2% of randomly placed PPAs are present in biomes with <10% of their total area under protection (Table 4.1). PPAs are most prevalent in Mediterranean forests and woodlands and account for 12% of their total protected area of this biome (Figure 4.2). This biome experiences the fourth highest conversion rate from natural vegetation to other land uses (41% of biome area converted globally) and is protected by a skeletal network of PAs (5% of biome protected globally) (Hoekstra et al., 2005). Our results suggest PPAs can play a key role in increasing the ecological representativeness of the global PA network and that they are present in biomes that are threatened and underrepresented (<10% of total biome protected).

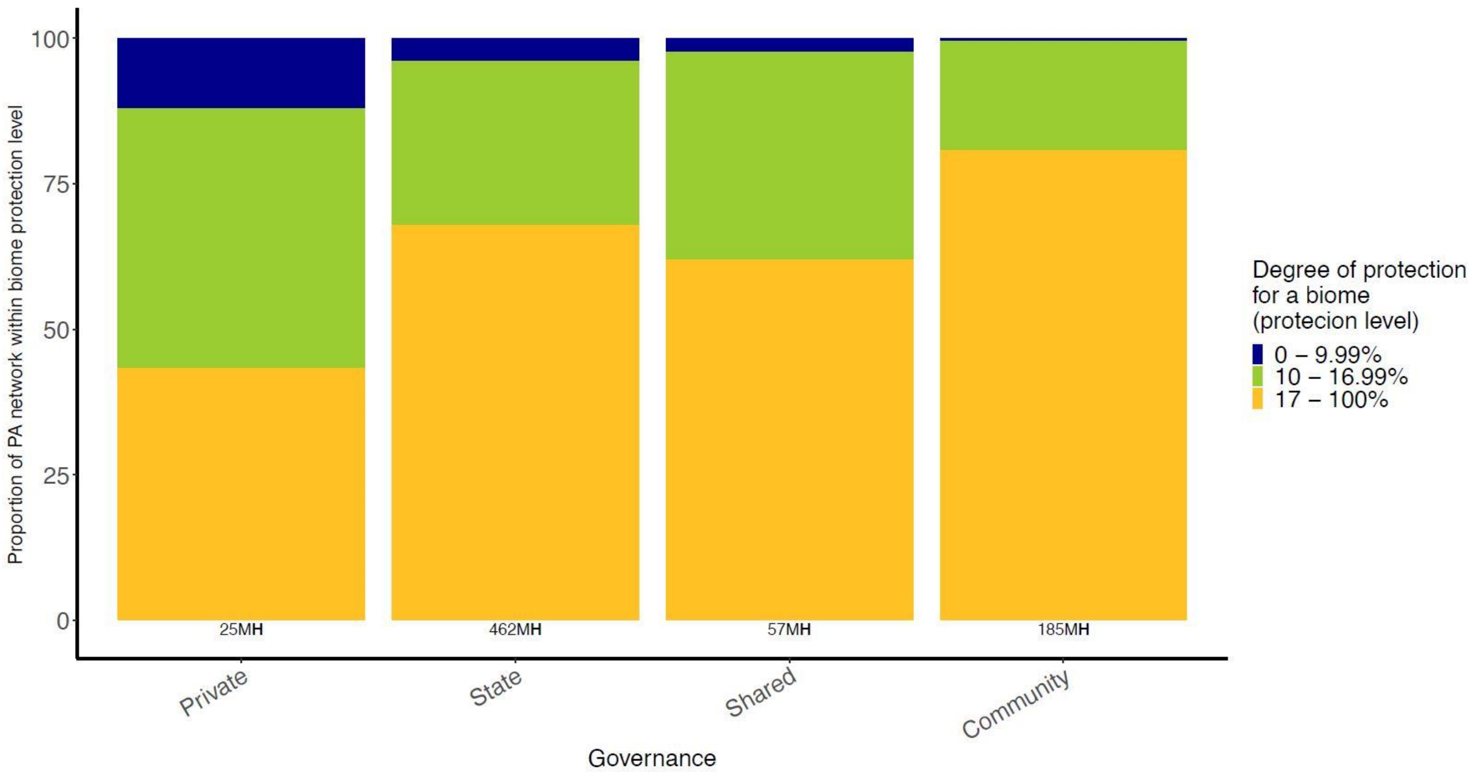

Figure 4.1 Proportion of each biome protection level protected by PA governance types

Proportion of areas of protection level; 0 – 9.99% of biome protected, 10 – 16.99% of biome protected and 17 – 100% of biome protected by state, co-managed, community and privately governed PAs. MH = Million Hectares.

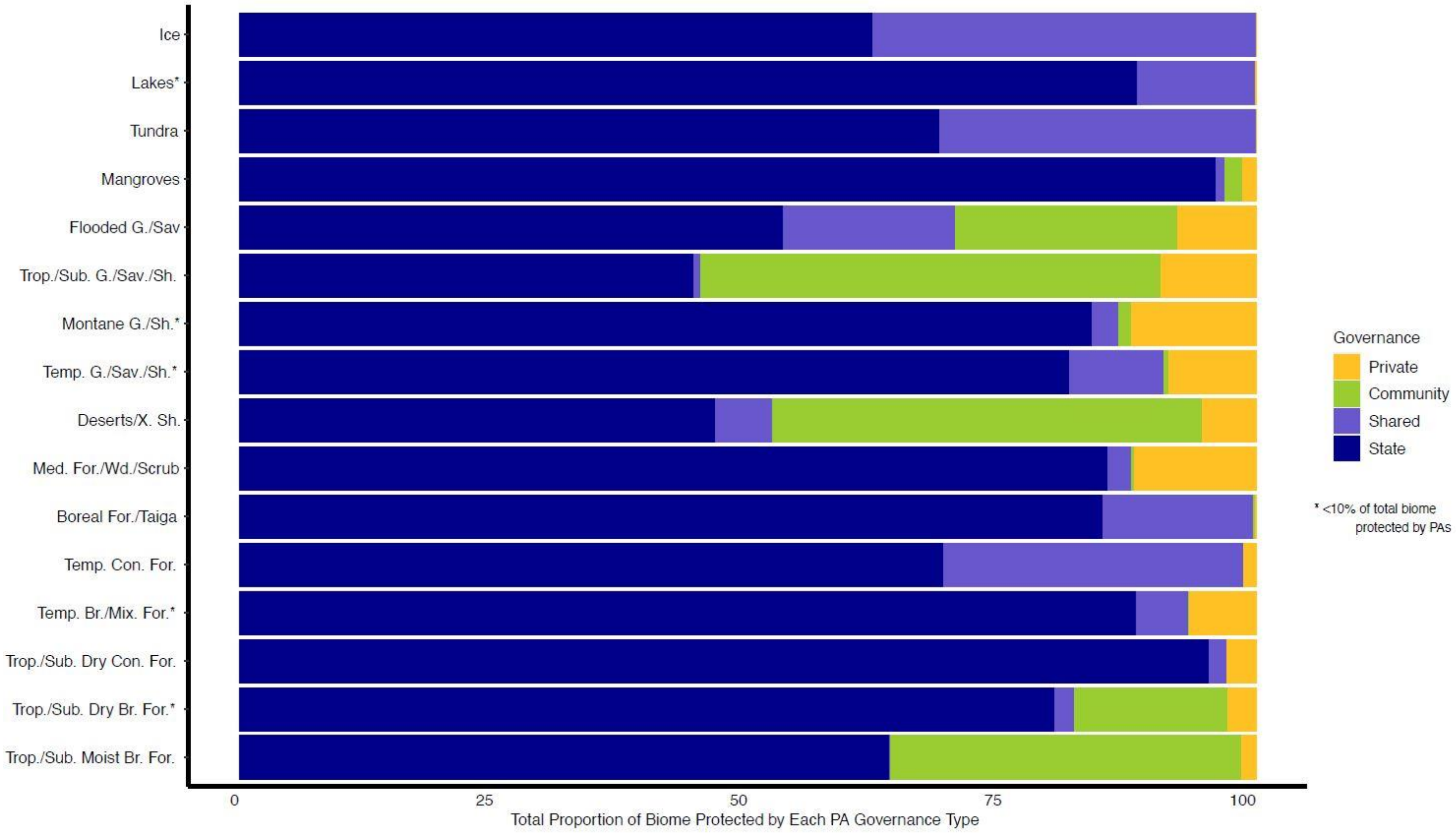

Figure 4.2 Proportion of each terrestrial biome protected by protected areas

Proportion of total area of biome protected across our 15 case countries covered by each governance type (state, co-managed, community, private), ordered by proportion of private governance. A star (*) indicates biomes where <10% of their total area is protected by any form of PA. Biome abbreviations: Flooded g./sav. = Flooded grasslands and savannas; Trop./sub. g./sav./sh. = Tropical and subtropical grasslands, savannas, and shrublands; Montane g./sh. = Montane grasslands and shrublands; Temp. g./sav./sh. = Temperate grasslands, savannas, and shrublands; Deserts/x. sh. = Deserts and xeric shrublands; Med. for./wd./scrub = Mediterranean forests, woodlands, and scrub; Boreal for./taiga = Boreal forests/taiga; Temp. Con. For. = Temperate conifer forests; Temp. br./ mix. for. = Temperate broadleaf and mixed forests; Trop./sub. con. for. =Tropical and subtropical coniferous *forests; Trop./sub. dry br. for. = Tropical and subtropical dry broadleaf forests; Trop./sub. moist br. for. = Tropical and subtropical moist broadleaf forests.*

We use a complementarity metric (see 4.5 Methods) to assess whether PPAs protect more or less of a particular biome than would be expected, given the total area of PPAs and that of state, co-managed and community managed PAs. We conduct this analysis to determine if PPAs complement other forms of PAs or if they are generally conserving the same elements of biodiversity. We find that PPAs have greater than expected complementarity for all grassland biomes and for at least seven biomes in total for all other PA governance types (Figure 4.3). These results show that PPAs are better at representing grasslands than any other PA governance type. This result is critical because grassland biomes are the most significantly degraded biomes globally (Newbold et al., 2016; Hoekstra et al., 2005), because habitat conversion in grasslands is exceeding habitat protection by a ratio of 8:1 (Hoekstra et al., 2005), and because grasslands offer a multitude of important ecosystem services (Bengtsson et al., 2019).

Within our case countries, there is a positive relationship between biomes and ecoregions with large proportions of their area under private ownership (e.g., grasslands biomes and the Atlantic Forest) and the total area protected by PPAs. In Australia, private ownership of grasslands averages 10% across the country but can be as high as 60% in certain states (e.g., Victoria (Zimmer al., 2010)). In the USA, 70% of the Northern Great Plains are privately owned (WWF, 2021). Within Brazil the majority of PPAs are located within the Atlantic Forest biome, which has 80% of its range under private land ownership (Henderson et al., 2016). PPAs may be more present in grasslands due to financial incentives for owners of private grasslands (Kamal et al., 2015; Zimmer al., 2010). It may also be due to a large number of institutions (e.g., The Nature Conservancy or the Land Trust Alliance) that can support landowners wanting to dedicate their land to private conservation. In addition, the amount of rural development grants and number of NGOs (Non-Governmental Organisations) positively influence the number of conservation easements along the Pacific coast of the USA (Williamson et al., 2018).

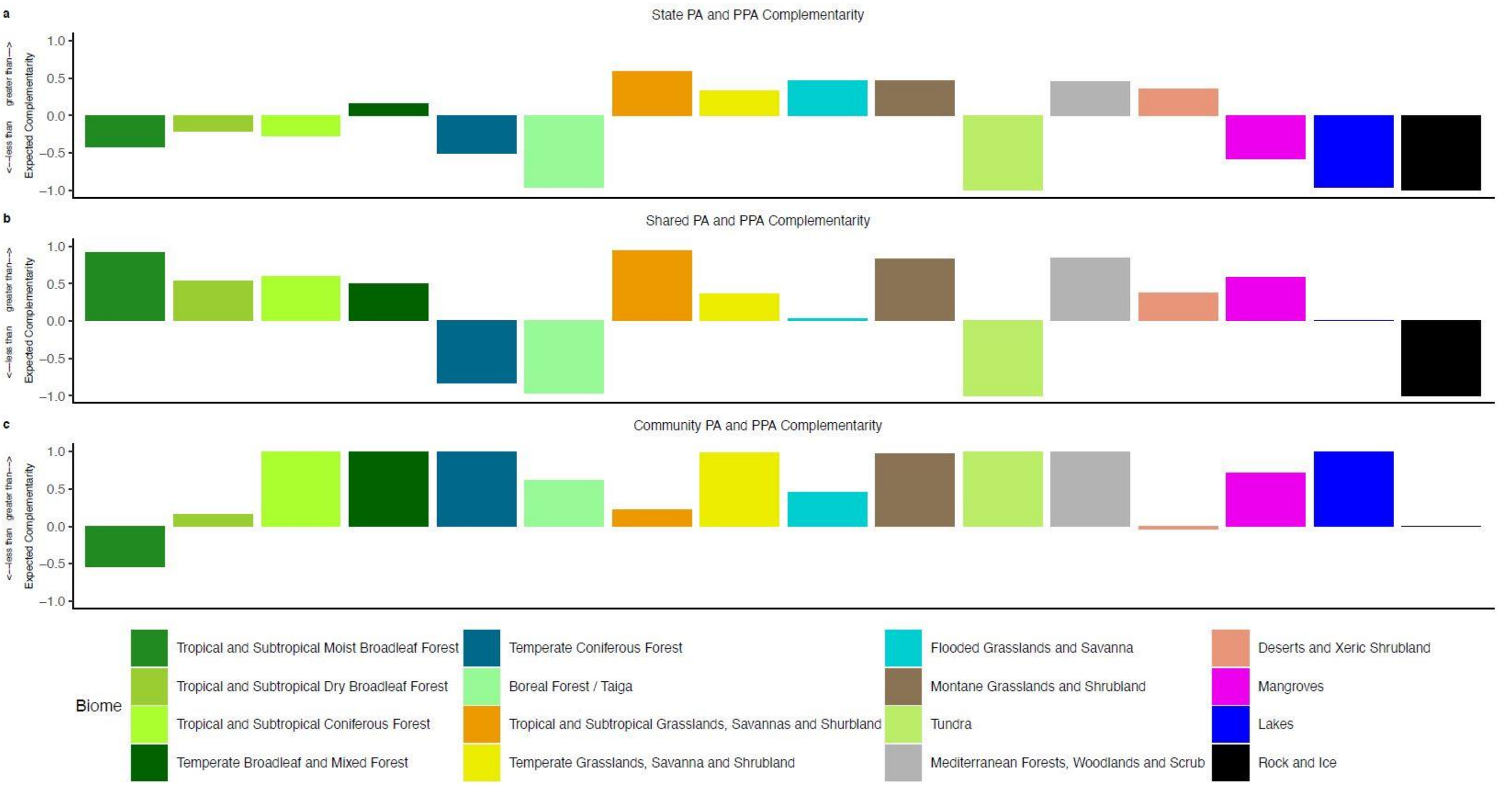

Figure 4.3 Complementarity of PPAs to other governance types in protecting terrestrial biomes

a – c, *Relative proportions of biomes protected by* ***(a)*** *state,* ***(b)*** *co-governed and* ***(c)*** *community governed PAs compared with and privately governed PAs. Positive (+) values (0 – 1) = greater than expected complementarity of a given biome between PPAs and either state/co-managed/community PAs accounting for the difference in the total protected area* *of PPAs and state, co-managed or community PAs. Negative (-) values (0 – -1) = less than expected complementarity of a given biome between PPAs and either state/co-managed/community PAs accounting for the difference in the total protected area of PPAs and state, co-managed or community PAs. A value of 0.2 indicates twice as much of a biome represented by a PPA than would be expected given the total ratio of PPAs to a PA of a state/co-managed/community PA.*

4.3.2.2 Key biodiversity areas and areas of high human disturbance

Within our sample, we find that PPAs protect 1.2% of the total area of KBAs, compared with state (32%), co-managed (1.1%) and community governed PAs (2.6%) – (see Appendix C– Table A.4), and randomly placed PAs (0.68% -Table 4.1). Twenty percent of the total area of PPAs within our case countries are located within KBAs compared with state (28%), co-managed (1.1%) and community governed PAs (5.8%) (see Appendix C– Table A.4) and randomly placed PAs (11% - Table 4.1). Our results show PPAs make a small but nevertheless important contribution to protecting KBAs.

Within our case countries, we found that a greater proportion of the area of PPAs is situated within higher HF areas, compared to other PA types: 47% of the total area of PPAs is in areas with HF ≥3 compared with state (23%), co-managed (11%) and community governed PAs (12% - Figure 4.4) and randomly placed PAs (43% - Table 4.1), respectively. We use a human disturbance score of 3 as a threshold, which represents when land can be considered as “human-dominated” (Watson et al., 2016). Previous analyses show that this threshold is where species are far more likely to be threatened by habitat loss (Di Marco et al., 2015). To further test that PPAs are situated within areas of higher HFs, rather than being identified as being under high human pressure themselves (as is the case with some

PAs) (Jones et al., 2018), we determine the HF scores of the immediate areas surrounding PPAs (1 km, 5 km and 10 km). We find that 53% of PPAs have a HF score of <3 compared with 34%, 29% and 37% of land within 1 km, 5 km, and 10 km of PPAs respectively (see Appendix C – Table A.5). We also find that 62%, 60% and 58% of individual PPAs have the same or lower HFs than 1 km, 5 km and 10 km buffers surrounding them, respectively. These results show that PPAs have lower human footprint scores than their immediate surroundings. Our findings suggest that PPAs have a key role in conserving areas facing greater pressure from urban and agricultural expansion and other external threats. Furthermore, areas with greater human pressure are also more likely to be substantially degraded (Sanderson et al., 2002) and PPAs could thus play a key role in the restoration of degraded lands. PPAs may be more likely to be present in areas of higher HFs due to historic biases in the distribution of private- and state-owned land across high and low productivity landscapes, respectively (Clements et al., 2018).

We also find that at least twice as much of the total area of PPAs is in areas with the highest HF scores (between 12 - 50) than any other PA governance type: 4% of total area of PPAs compared with 2%, 0.66% and 0.47% for total area of state, co-managed and community governed PAs respectively (Figure 4.4). We find that PPAs with HF scores between 12 - 50 were situated in large conurbations (e.g., suburbs of São Paulo, Brazil). Urban PAs are distinctively important for two reasons. First, urban PAs can offer key ecological benefits, such as water regulation to reduce flooding, improving air quality and helping to reduce the urban heat island effect (Song et al., 2020). Second, urban PAs can offer experiences in nature to large numbers of people living close to them. Visitors to these areas may be more socially and economically diverse than visitors to more remote PAs

(Trzyna, 2014). PPAs in urban areas could thus help to broaden and diversify access to nature, promote human health and well-being in under-privileged groups, and help build greater political support for nature conservation within urban populations. However, the proportion of PPAs which allow public access, and the extent to which these potential benefits are realised is unknown. As urban areas and urban populations continue to grow, understanding and protecting biodiversity in cities is of global conservation importance (Li et al., 2019).

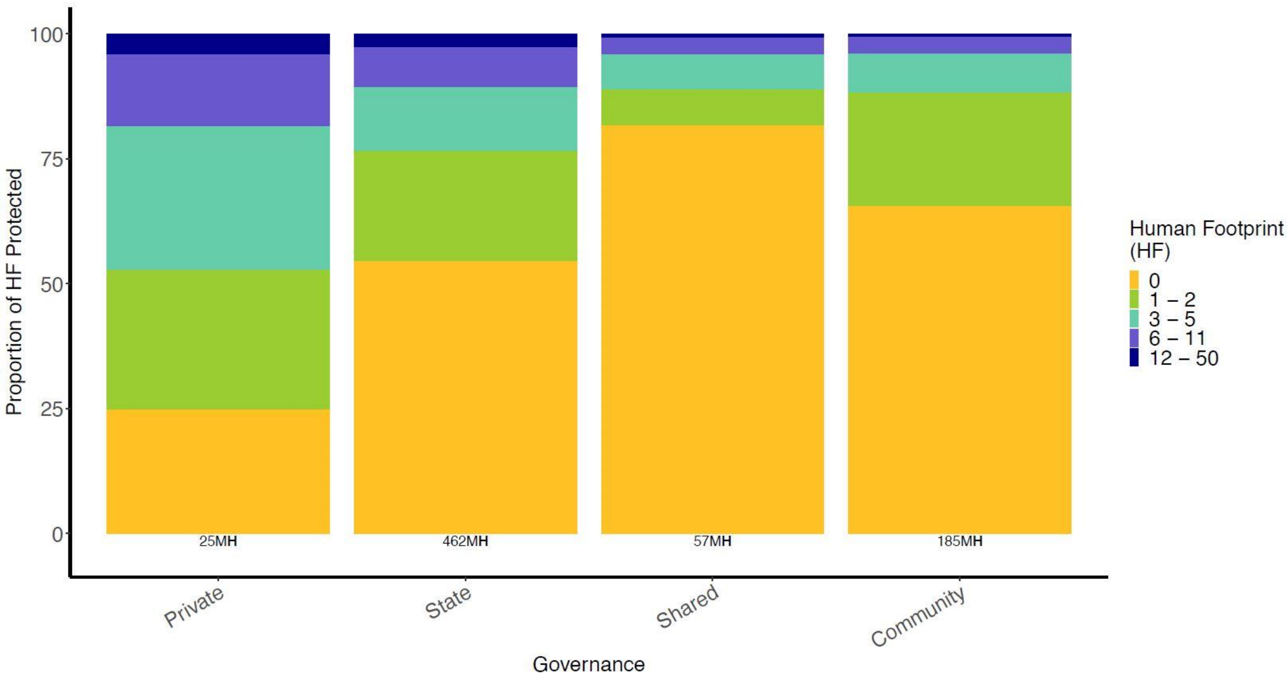

Figure 4.4 Proportion of each HF category protected by PA governance types

Proportion of areas of human disturbance (ranked between 0 and 50) protected by state, co-managed, community and privately governed PAs. 0 = no human pressure; 1 – 2 = low human pressure; 3 – 5 = moderate human pressure, 6 – 11 = high human pressure; 12 - 50 = very high human disturbance.

Table 4.10Comparison of the current PPA distribution with random placement

Percentage of PPAs within biomes receiving 0 – 9.99%, 10 – 16.99% and 17% + overall protection		
	Current PPA distribution	Random Placement
0 – 9.99% of biome protected	12	3.2
10 – 16.99% of biome protected	45	42
17 – 100% of biome protected	43	54

Percentage of PPAs within each Human Footprint grouping		
	Current PPA distribution	Random Placement
0	25	26
1 – 2	28	33
3 - 5	28	25
6 – 11	14	12
12 – 50	5	4

Percentage of PPAs within Key Biodiversity Areas		
	Current PPA distribution	Random Placement
% of KBA protected by a PPA	1.2	0.7
% of PPA area within KBAs	20	11

4.3.3 Adjacency

Many species need large areas for roaming distances to reduce genetic bottlenecks, to assist populations in the evasion of natural disasters, and for migration routes (Venter et al., 2016). Animal movement can be difficult if PAs are disconnected from one another. We find that on average 11%, 13%, 14% and 18% of PPAs are located within 0 m, 30 m, 100 m and 500 m of a PA of another governance type, respectively. For random placement, 0%, 0.5%, 1.7% and 6% of PPAs were located within 0 m, 30 m, 100 m and 500 m of a PA of another governance type respectively.

We find substantial variation in the percentage of PPAs adjacent to PAs of other governance types across our case countries (Mean = 168, SE = 65). Belize, Kenya, and Namibia have PPAs with the highest levels of adjacency with a PA of another governance type; 60%, 46% and 44% of PPAs at 30 m of state, co-managed and community governed PAs, respectively. However, in Canada, Colombia, Honduras, and Mexico <5% of PPAs are adjacent to PAs of another PA governance type.

Adjacency may be highest in Belize because so much of the country is under some form of protection (37%). Adjacency may also be greater in Kenya and Namibia than other countries due to ecotourism reserves siting along national park boundaries. Adjacency in South Africa is lower than expected perhaps due to the removal of UNESCO biosphere reserves in our analysis. Removal of UNESCO biosphere reserves has a bigger impact in South Africa than other countries because the total area of UNESCO sites in South Africa (109,705 km^2) accounts for 25% of the total area of UNESCO sites across our 15 case countries (429,347 km^2). In comparison, UNESCO sites account for <10% of PAs in all other study countries. Adjacency may be low in Canada, Colombia, Honduras and Mexico because PPAs make up <2% of the total protected area within these countries and therefore there is a reduced probability that they will be located next to PAs of other governance types. Our findings differ to regional or local scale studies showing that PPAs have high adjacency with other PA governance types (Graves et al., 2019; De Vos & Cumming, 2019). This discrepancy may be because these studies have focused on areas of limited size where PPAs are known to be present (De Vos & Cumming, 2019; Rissman & Merenlender, 2008). Increasing the distance from 0 m to 500 m had no substantial effect on our calculations (See Appendix C– Table A.6).

4.3.4 Connectivity

Global biodiversity targets call for PAs to be well connected to one another (CBD, 2011). To determine the contribution of PPAs to total connected protected land in each country, we performed a with and without PPAs scenario analysis using four dispersal distances scenarios of 1 km, 10 km, 30 km, and 100 km (Birdlife International, 2020). We define dispersal distance as the distance a terrestrial vertebrate species is able to travel between existing populations. We preferentially show results for a dispersal distance of 10 km because that is the median dispersal distance for a terrestrial vertebrate (Saura et al., 2017). The exclusion of PPAs decreased contiguous protected land by an average of 7.05% across our case countries (see Appendix C– Table A.7). This compares to 5.6% for random placement. In our study, the inclusion of PPAs in Kenya made the greatest contributions to PA connectivity increasing the total protected connected land by 29%. This increase may be because PPAs are mainly clustered in one area and are located on the border of national parks. PPAs subsequently connect several national parks together creating one large contiguous patch of connected land. Increases in connectivity are low where there are few PPAs, where PPAs are small are in size, and where they have limited adjacency with other forms of PA. We found that changing dispersal distances (1 km – 100 km) had a limited effect on percentage change of total protected land including/excluding PPAs (see Appendix C– Table A.8).

4.4 Conclusion

Our analysis reveals three important insights. Firstly, PPAs can protect areas that are under-represented by PAs under other forms of governance and contribute to protecting KBAs. Across our case countries, a greater proportion of PPAs are found in biomes that have <10% of their total area protected and areas of higher human pressure. Secondly, PPAs' contribution to PA coverage is significant in some countries but negligible in others. For the five countries in our sample with the greatest contribution to national PA coverage (South Africa, Guatemala, Belize, Namibia, and Peru), PPAs account for 15% of the total area protected in those countries. Thirdly, PPAs make a modest contribution to the connectivity of national PA networks. We found that 38% of PPAs are adjacent to a PA of another form of governance and that the inclusion of PPAs increase protected connected land by 19% for the five countries with highest PPA adjacency and connectivity within our sample. It is important to note that due to underreporting to the WDPA and national-level platforms, our findings represent a "bare minimum" of the contribution of PPAs. Improvements in PPA reporting would likely reflect a greater contribution from PPAs to the global conservation estate.

We suggest that greater legislative, technical and financial support for PPAs and a more co-ordinated approach to their establishment could help maximise their benefits. These forms of support could incentivise and facilitate the establishment of PPAs and help PPA owners implement better land management and restoration practices. Greater co-ordination of PPA establishment could be achieved by: (i) creating frameworks for the inclusion of PPAs into national conservation strategies; (ii) the creation of PPA support networks (such as RESNATUR in Colombia); and (iii) supporting countries with the recording and reporting of PPA boundaries (with the consent of relevant authorities and organisations) to national authorities and the WDPA.

Reporting of PPA boundaries requires time, resources and institutional infrastructure, which some governments may lack. In some countries, political situations may mean PPA land holders and governance authorities and/or governments may be less willing to gather and report data on PPA boundaries (Bingham et al., 2017). Civil society organisations, land trusts and PPA networks working in these countries can play a key role in facilitating the reporting of PPAs to the WDPA and other authorities with appropriate consent. Additionally, indigenous and local communities may have competing claims to the land contained within some PPAs, often based on customary tenure, which may or may not be recognised by governments. PPA owners and networks therefore have a responsibility and moral duty to ensure that these claims are adequately addressed and resolved in an appropriate and ethical manner, recognising the power disparities that often exist between conservation organisations and indigenous and local communities. Lastly, we encourage future research to assess the spatial contributions of PPAs in other regions to examine the underlying factors and governance structures that lead to specific spatial configurations of PPAs. Such analyses should include efforts to better understand the role of different stakeholders (e.g., private landowners, NGOs and land trusts) and their motivations for the establishment of PPAs, as well as assessments of national policies and incentives that support PPAs.

4.5 Methods

Our study uses PA boundaries and global spatial datasets of biome distribution (Olson et al., 2001), key biodiversity areas (KBAs) (Birdlife International, 2020) and human disturbance (Venter et al., 2018) to determine the contributions of PPAs to global conservation. We used ArcMap 10.4, Conefor2.6 (Saura & Torne, 2009) and R (R Core Team, 2014) for all our analyses.

4.5.1 Spatial layers and processing

We downloaded the November 2018 version of the WDPA from http://www.protectedplanet.net/ as a primary source for PA boundaries (UNEP-WCMC & IUCN, 2018). As in other PA assessments (Saura et al., 2017), we excluded from subsequent analysis 233 PAs with a “proposed” and 439 with a “not reported” status, 29 PAs reported as points without an associated area and 75 UNESCO Man and Biosphere Reserves. When point data was included, we created a circular buffer around the point in ArcMap to account for the total size of the reported PA area. We buffered 16 points. These circular buffers are unlikely to represent the real shape of the PA or their exact location because location points provided by the WDPA can be either in the centre of the PA or on an outer edge. This discrepancy could impact our connectivity analysis by affecting the distance to the nearest PA by up to the half the actual width of the PA (if the point is located on an outer edge of the PA). However, we feel that these discrepancies are likely to have a limited impact on our study because buffered PAs were few and small (mean size = 5 km^2). Remaining PA boundaries were classified into five reported governance types (state, co-managed, community, private and non-reported) using the GOV_TYPE field in the WDPA. We filtered for PPAs using the following GOV_TYPE values; For-profit organisations, Non-profit organisations and Individual landowners. All PA management types (Ia to VI) are included within our study. As of November 2018, the WDPA reported 13,250 designated PPA boundaries. We identified a further 11,074 PPAs that had been incorrectly reported through the DESIG field, which details the designation of a PA at the national level (i.e., Private Natural Heritage Reserves, *Reservas Particulares do Patrimônio Natural*, in Brazil are mistakenly reported as being under government management when they are, in fact, privately protected (Bingham et al., 2017) - see Appendix C -Table A.9. We cross-checked these potential PPAs by consulting with national PPA experts and conducting document analyses of open access materials (e.g., in Honduras, we contacted employees working for the National Institute for Conservation and Forest Development, Protected Areas, and Wildlife (ICF)). We based our definitions of PA governance types on that of the IUCN (IUCN, 2016) (see 4.1 Introduction). We excluded 1,346 PAs with a non-reported governance type. Excluding PAs with non-reported governance type accounted for 6% of the total area of PAs in our 15 case countries. The minimum and maximum size of a PA that we excluded from the study due to no governance being reported was 1 km^2 and was 30,893 km^2, respectively, with a mean size of 306 km^2.

We classified remaining PA boundaries by ISO3 country code. We excluded countries with <10 PPAs reported to the WDPA from subsequent analysis resulting in a global subset of 15 countries as any potential PPA effects would be negligible. We also excluded the UK due to difficulties in establishing the governance structure of potential PPAs because areas under habitats directives and other such initiatives are all reported as government PAs, even when managed by private entities. We obtained additional PPA data for our case countries from multiple sources outside of the WDPA (e.g., Chile's Asociación de Iniciativas de Conservación en Areas Privadas y de Pueblos Originarios (http://asiconservachile.cl/acch/) (see Appendix C– Table A.9). This resulted in an extra 1,038 PPA boundaries (70,240 km^2) that had not been reported to the WDPA (see Appendix C– Table A.9). We ensured all additional PPA boundaries met our standard definition of a PPA through consultation with PPA experts in their regions of expertise and document analysis of open access material.

We dissolved PA boundaries with the same governance type to remove overlaps and erased overlaps between PAs of different governance types to avoid double counting (Schutz, 2018). To determine which governance classification to retain, we created a governance hierarchy: state governance, co-management, community governance, and private governance (Schutz, 2018). This hierarchy is based on the strength of legal recognition and environmental protection security that each governance type offers (Schutz, 2018). We designated state PAs as the highest tier because they have the strongest legal standing across all countries and can provide strict environmental protection. We designated PPAs as last in our hierarchy because in some countries (e.g., Chile) PAS have no legal recognition, regulation and no guaranteed permanence. Hereafter these layers are referred to as 'PA governance layers'. While establishing this hierarchy was necessary for the analysis, we recognise that the assumptions made will not reflect reality in all cases, since the level of recognition, strictness of protection and quality of conservation outcomes will all vary within and between governance types. We removed 6% (105,441 km^2), 5%

(664,824 km^2) and 3% (439,589 km^2) of private, co-managed and community governed PAs respectively, due to boundaries overlapping with a governance type prioritised by our hierarchy.

The number of reported PPAs is believed to be a significant underestimation of their total number (Bingham et al., 2017). The quality of available data is highly variable depending on the original data source (Milam et al., 2016). Here, we have used the best available data, collected from multiple sources (see Appendix C – Table A.9), to provide initial insights into the spatial outcomes of PPAs. After the removal of overlaps, our final dataset included 13,206 PPA boundaries originally reported to WDPA, 3,317 PPA boundaries from within the WDPA that had been incorrectly reported as having another PA governance type and 1,038 PPA boundaries from additional sources (see Appendix C– Table A.9). This resulted in 17,561 PPA boundaries in total.

We used the World Wildlife Fund's (WWF) terrestrial ecoregions layer to assess biome complementarity between PPAs and PAs under other governance types (Olson et al., 2015). We used the 14 biomes (i.e., the natural vegetation that would be expected in that area assuming minimal human disturbance) identified by WWF as our unit of analysis because we could not be confident enough in the accuracy of the PA boundaries or ecoregions to make comparisons at the ecoregion level. Hereafter this layer is referred to as the 'biome layer'.

We used the Key Biodiversity Area (KBAs) dataset (Birdlife International, 2020) to assess what degree PAs under difference governance types protect KBAs. KBAs are sites contributing significantly to the global persistence of biodiversity. We used the 2018 released Global Footprint Dataset (V3) (Venter et al., 2018), which compiles the cumulative human environmental pressure in 2009, to assess to what degree PAs under difference governance types protect areas of greater human disturbance. We used the Global Human Footprint dataset as it the most complete and highest-resolution globally consistent terrestrial dataset on cumulative human pressures on the environment (Venter et al., 2016). All data were projected in Mollweide (World) as this is an equal area protection to calculate the total area of PAs within different biomes or degrees of human disturbance.

4.5.2. Analysis

We conducted spatial analyses in ArcMap 10.04 and Conefor 2.6 (Saura & Torne, 2009) to determine the total area of PAs within different biomes, degrees of human disturbance and their overlap with KBAs and the contribution of PPAs to national PA network connectivity. We determined the total area of PAs within each PA governance layer using the calculate geometry tools. As per previous studies that determine what PAs protect (Schutz, 2018), we clipped each of the biome, HPF and KBA layers with the different PA governance layers to determine the overlap between each.

To determine the complementarity of PPAs to other governance types for what biomes they protect we used an adapted a complementarity metric (Gallo et al., 2009). We define complementarity in this context to mean to what extent PPAs supplement the biome coverage of PAs of other governance types and increase overall biome representation within the PA network of our 15 case countries:

$$Complementarity\ metric\ of\ a\ biome = Mc = \frac{Pp *\ R - Op}{Pp * R + Op}$$

Where Pp = the percentage of a particular biome conserved by PPAs; R = (Area of State or Co-managed or Community governed PA) / PPA Area; Op = the percentage of a particular biome conserved by either state, co-governed or community governed PA. This metric is on a scale of -1 to +1, where negative values indicate less than expected complementarity and positive values indicate greater than expected complementarity between PPAs and PAs under other governance regimes. Expected complementarity is determined by the ratio of the area of PPAs to PAs under other governance types.

We then generated a network of random reserves, equal in area to the current PPA network within each country, to evaluate the coverage of existing PPAs relative to random counterfactuals. We generated this network by randomly selecting cells from a grid until reaching the area of the current reserve network within each country was reached. As per previous studies, the grid size was equal to the average size of each PA governance type within each country (Mason et al., 2018). This process was repeated 1,000 times using R (R Core Team, 2014) to provide an average of the total area of each biome, HF and KBA present within our model. We summed totals for each country to give a general overview for our case countries.

4.5.2.1 Connectivity

To conduct our connectivity analysis, we used undissolved polygons. We assessed connectivity using two metrics: adjacency and connected protected land.

4.5.2.2 Adjacency

We measured the adjacency of PPAs to PAs under other governance regimes using the select by location tool. Due to small misalignments in polygon boundaries, PPA adjacency may be inflated because only a small portion (i.e., 1 – 2% of the total area of a PPA) does not overlap with a PA of another governance type. This was the case for 5,102 PPAs (20% of the total number and mostly from Finland) and they were removed from this part of the analysis. To further account for small inaccuracies in the location of PA polygon boundaries we considered four within distance measurements of 0 m, 30 m, 100 m and 500 m to see what difference changing the distance of the buffer made to our results (see Appendix C – Table A.6). As with previous studies (e.g., Rissman & Merenlender, 2008), we preferentially show a 30 m buffer as we believe that it accounts for most minor inaccuracies in the location of PA boundaries. To test if the placement of PPAs around other forms of PAs occurs by chance or there if there are underlying factors, we generated 20 randomised maps in which the same PPA polygons were moved and rotated at random to new locations within each study country. We used the '*sp*' package in R (R Core Team, 2014) to generate a new random centroid for each PPA around which the polygon shape was then redrawn. If there were any overlaps between polygons, the script would rerun until a map of non-overlapping PA could be drawn. We re-ran the selection by location tool in ArcMap10.4 for each randomized map and averaged the results and compared that to those for the existing protected area network (De Vos & Cumming, 2019).

4.5.2.3 Connected protected land. To determine the contribution of PPAs to connected PA networks, we used Conefor2.6 (Saura & Torne, 2009) in command line (http://www.conefor.org). We performed a with and without PPAs scenario analysis using four dispersal distances of 1 km, 10 km, 30 km, and 100 km (as per previous studies (Saura et al., 2017)), to determine the equivalent connected area (ECA) of PA networks in each country. The ECA equates to the size of a single patch (PA) that would provide the same value of the probability of connectivity than the actual PA network in a country or continent. In effect, it summarizes the amount of reachable area in the PA network (Saura et al., 2017). From the ECA we computed the normalized Equivalent Connected Area (ECA_{norm}) (Saura et al., 2017), a connectivity metric that summarizes the percentage of reachable area in a PA network compared to the total country area, generally referred to as protected connected land. The protected connected land indicator assumes that PAs are effectively managed for connectivity (i.e., there are no important barriers for species movements and other ecological flows within PAs) (Saura et al., 2017). We preferentially use a dispersal distance of 10 km as per previous studies (Saura et al., 2017).

To test if the current PPA network distribution performs better or worse than random at increasing connectivity we also ran scenarios incorporating the 20 randomised PPA maps created for each country (previously created for the adjacency analysis). We calculated the average mean of the 20 randomised scenarios and compared these to those for the existing protected area network (De Vos & Cumming, 2019).

4.5.3 Limitations

We identify three potential limitations to our analysis. First, this analysis predominately relies of PA boundaries reported to the WDPA. The quality of the data reported to the WDPA can be highly variable depending on the original data source (Milam et al., 2016). Until recently, data quality on the WDPA has been measured and reported rather than controlled (Milam et al., 2016). Data quality issues may include incorrect or missing attributes (e.g., GOV_TYPE (Bingham et al., 2017; Milam et al., 2016), differences in the reported PA area and the submitted polygon boundaries (Milam et al., 2016), and presence of PA boundaries that may be degazzetted (Lewis et al., 2017)). These issues can cause both under and over-estimations of the coverage of PAs. Most pertinent to our study is the underreporting of PPAs to the WDPA, which is widely discussed in the literature (Bingham et al., 2017). An underreporting of PPAs means that results regarding how much PPAs contribute to total PA network coverage, protecting of KBAs and connectivity within our case countries are an under-estimate and should be regarded as a bare minimum. We are also aware that PPAs can be underreported in a biased way. For example, in some countries (e.g., Australia and Canada) certain states or provinces do not report or legally recognise PPAs (Stolton et al., 2014). We have attempted to mitigate this by contacting local experts who may have access to data currently unpublished at international and national levels. The failure of some states / provinces to report PPAs may lead to a bias of our results regarding the representativeness of PPAs, however the impacts of this may be limited. This is because biomes are mapped at such a large scale that each biome in each country covers multiple states. Therefore, if one or two states fail to report PPAs it is likely that those biomes will still be represented by PPAs within other states. Additionally, the omission of a small subset of states within the country will have a limited impact of the general trend of where PPAs are located. The impact upon HF is harder to determine. However, most states/regions share similar characteristics of having more remote and less remote areas. Therefore, it is not implausible that the characteristics of PPAs in states that fail to report PPAs may be similar to those of PPAs in states or regions that do (see Appendix C – Figure A.2). It should also be noted that we have analysed countries with good PPA networks and/or reporting. Therefore, our results cannot be more broadly applied to other countries that have not been included within our study. However, our study shows what may be possible if PPA creation is supported and encouraged by a wider number of countries.

Second, there is a temporal mismatch between HF dataset (2009) and the PPA dataset (2018). With human pressures continuing to rise this could mean that our calculated HFs within and surrounding PAs are an underestimate of the true values in 2018. However, the mean global HF only rose by 9% between 1993 and 2009 (16 years) despite an increase of 23% in global population and 153% in the world economy. Therefore, the effects of the temporal mismatch in our data (8 years) are likely to be small (Venter et al., 2016). Moreover, for the 60% (10,537) of PPA boundaries for which we have the designation year, 70% (7,376) were established before 2009. Third, we do not assess the extent to which PPAs protect threatened species, beyond their coverage of KBAs (despite some PPAs being set up for specific species) due to a lack of high-quality information on the presence/absence of species in individual PPAs (particularly difficult due to their small size and infrequent use of comprehensive species lists), population densities, minimum viable population sizes of threatened species.

Lastly, although a PPA may be reported in a given location, this does not mean that it is successfully conserving biodiversity, or that it will remain in place in perpetuity (Lewis et al., 2017). Assessing the effectiveness of PPAs is beyond the remit of this study and future studies should assess the effectiveness of PPAs in different countries. The few studies that have assessed the performance of PPAs to mitigate deforestation and degradation (Schleicher et al., 2017) and land cover change (Shumba et al., 2020) found that PPAs are more effective than other forms of PA. Studies of the permanence of PPAs showed that only

6.2% of PPAs were degazetted in a 92 year period (compared with 2.2% for state PAs) (Acemoglu et al., 2001).

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Chapter 5: Effectiveness of privately protected areas to reduce deforestation in Colombia

5.1 Abstract

Privately protected areas (PPAs) are increasingly recognised as important conservation initiatives, and national governments are now acknowledging, supporting and documenting them alongside protected areas (PAs) under other governance types. However, few studies have assessed the performance of privately protected areas (PPAs) and how it compares relative to other forms of PA governance. We address this shortfall by conducting an impact evaluation using propensity score matching to assess the effectiveness of PPAs to curb deforestation in Colombia and compare it with regionally governed and national PAs. We find that on average all PA governance types curb deforestation compared to unprotected areas with similar characteristics, but no governance type guarantees complete protection. We find that different measures of PA outcomes (e.g., absolute effects vs. relative effects vs. percentage of PAs which completely avoid deforestation) offer varying results on which PA governance type is most or least effective We find that national PAs have both the greatest absolute (1% reduction in deforestation rate per PA on average) and relative effects (83.92%). PPAs have the lowest absolute effect (0.43% reduction in deforestation rate per PA on average) and regionally governed PAs the lowest relative effect (61.10%). We find that 76% of PPAs completely avoid deforestation – compared with 40% and 26% of national and regionally governed PAs respectively. We find that PPAs have the most heterogeneous results (range of relative effects in individual PPAs: -500% - 100%). This is potentially due to PPAs being in areas of higher deforestation pressure than other PA governance types and the varying efforts and motivations of multiple individual landowners. This study provides both the first assessment of PPA effectiveness for forest protection in Colombia and the first national-scale evidence that PPAs can be an effective mechanism for reducing deforestation.

5.2 Introduction

Tropical forests cover ~1,172 million hectares globally (FAO & UNEP, 2020), directly affect the lives of over 1.35 billion people (FAO, 2014), store ~247 Gigatons of carbon (Saatchi et al., 2011) and harbour as much as half of the world's biodiversity (Gibson et al., 2011; Myers et al., 2000). Yet despite their global importance, 1.1 million km^2 of tropical forests were lost globally between 2000 – 2012 (Hansen et al., 2013), and few contiguous forest blocks remain (Brandon, 2014). Protected areas (PAs) and other effective area-based conservation measures (OECMs) have long been the dominant tool in the fight against deforestation, especially for stemming tropical forest loss (Cazalis et al., 2020). PAs cover approximately 16% of the terrestrial biosphere (UNEP-WCMC, IUCN & NGS, 2021) and a proposed target for 30% of the world's surface to be protected by 2030 has recently been announced (CBD, 2020). Even with the rapid proliferation of PAs (Maxwell et al., 2020), and billions spent on their creation and management (Gaston & Balmford, 2001), the conversion of tropical forests for agricultural and infrastructure expansion and timber extraction continues largely unabated (Potapov et al., 2017; Achard et al., 2002). This has led many to question the effectiveness of PAs and ask to what extent PAs actually limit deforestation.

Numerous studies suggests that on average PAs moderately reduce tropical deforestation (e.g., Andam et al., 2008; Canavire-Bacarreza & Hanauer, 2013; Sims & Alix-Garcia, 2017), however effect sizes vary by PA and by country (e.g., Shah & Baylis, 2015). What we currently lack information on is why this variability exists (Rodrigues & Cazalis, 2020; Ferraro & Pattanayak, 2006). Consequently, calls have mounted for studies to determine what factors make PAs effective at reducing deforestation (Baylis et al., 2015; Ferraro & Pattanayak, 2006). So far, PA management categories (as defined by the IUCN (Dudley, 2008)) have attracted the most attention and evidence suggests that whether a strictly protected or sustainable use management approach is more effective depends on a PAs location (e.g., Leberger et al., 2020; Bonilla-Mejia & Higuera-Mendieta, 2019; Nolte et al., 2013; Nelson et al., 2011; Joppa & Pfaff, 2009). Other factors have also been shown to affect PA effectiveness, such as PA age and size (Maiorano et al., 2008), proximity to human settlements and municipal characteristics (Bonilla-Mejia & Higuera-Mendieta, 2019) and differences in economic stability, robustness and transparency of national governance

(Nelson & Agrawal, 2008). Increasingly, studies are now looking to understand how PA governance affects PA impacts (e.g., Vergara-Asenjo & Potvin 2014; Pfaff et al., 2013; Nolte et al., 2013). This is because (i) PA governance is now being recognised as *"central to the conservation of PAs throughout the world"* (WCPA, 2004: 257), (ii) the number and extent of non-state governed PAs, such as community governed and privately protected areas (PPAs) are increasing (Aswani et al., 2017, Bingham et al., 2017), and (iii) the increased incorporation of non-state PAs into national PA frameworks. As biodiversity loss continues, comparing the performance of different PA governance types will be crucial to determine why, where and when different approaches deliver positive biodiversity outcomes.

The IUCN categorises PA governance into four types; state governance, community governance, co-governance (shared between the state and other actors), and private governance (Borrini-Feyerabend et al., 2013). This study places a particular emphasis on private governance because to date PPAs have rarely been included within effectiveness studies, or where included their outcomes have not been independently examined (Palfrey et al., 2021). A review of the impact of governance type on conservation effectiveness found that only 6 of 66 studies included PPAs (Macura et al., 2015), the least of any PA governance type.

PPAs are defined as areas that are governed by private actors, are primarily engaged in biodiversity conservation activities, have long term intent to remain gazetted, and have legal or other effective means of protection (Stolton et al., 2014). PPAs vary in landowner types and governance authorities (e.g., individuals, NGOs or corporate businesses) and protection mechanisms (e.g., conservation easements, NGO freeholds or perpetual landholder agreements) (Palfrey et al., 2022). PPAs have potentially been excluded from previous studies due to a lack of reporting on PPA boundaries (Bingham et al., 2017), because many PPAs have only recently been gazetted or legally recognised and due to their small size. Of the 17,961 PPAs currently reported to the WDPA, 13,378 have a reported area of <1KM^2. Most studies conduct analysis at 1km^2 resolution and exclude PAs under 1km^2 due to underlying data used within their analysis, and thus exclude a large majority of PPAs (e.g., Bonilla-Mejia & Higuera-Mendieta et al., 2019; Nolte et al., 2013).

Here we assess and compare the impacts of Colombian PPAs, regionally governed PAs (*Parques Naturales Regionales*) and national PAs on rates of deforestation (defined as forest cover loss to <30% of pixel) at a scale of $30m^2$ (Lwin et al., 2019), using the Hansen Global Forest Change dataset (Hansen et al., 2013) (see 5.3 Data and Methods for a description of different PA governance types in Colombia). We use a quasi-experimental design utilising matching methods to pair PAs with unprotected lands that have similar characteristics (termed confounding factors) that may affect PA location and the likelihood of an area being protected or undergoing deforestation. The matched site is termed a counter-factual and acts a proxy to show what would have happened to the area inside the PA had it not received any protection. We use this design because PA distribution is not random and is biased towards certain areas, such as those of little economic interest (i.e., greater remoteness, higher altitudes and lower agricultural potential) (Venter et al., 2018), and because deforestation pressure is not consistent over time (Austin et al., 2017). Earlier inside-outside or before-after comparisons to determine PA impacts have overestimated their effectiveness (Joppa & Pfaff, 2010). Quasi-experimental designs and 'matching' approaches in conservation science enable researchers to more accurately determine PA impacts and eliminate competing explanations (Schleicher, 2018).

We conduct our study across the tropical forest biomes of Colombia and in States where both PPAs and regionally governed PAs are present (Figure 5.1). Colombia is an excellent case study because it is a mega-diverse country, its environmental policy relies heavily on PAs (to date national parks over >13.5% of Colombia's continental area (Colombia Natural National Parks, 2019) (this figure can increase to up to 50% when regional and private PAs and OECMs are included) and because despite extensive PA coverage, deforestation rates are particularly high. Government statistics indicate that Colombia lost 8.3% of its baseline forest coverage between 1990 and 2016 (IDEAM, 2017) and deforestation has accelerated in PAs in recent years after Colombia's peace agreement with the Revolutionary Armed Forces of Colombia (FARC) in 2016 (Clerici et al., 2020). Private land conservation within Colombia is important to study due to the numerous external pressures on private land, e.g., gold mining, cattle ranching and the growth of illicit crops (International Crisis Group, 2021) and because in some areas (e.g., the Bajo Cauca region), most private land is used for activities that, to one extent or another, contribute to forest loss (International Crisis Group, 2021). We exclude Indigenous and Afro-Colombian territories from our study because they may operate in significantly different ways to other PA governance types, and they are rarely located in similar ecosystems to PPAs. Afro-Colombian territories are communal land titles granted to distinct ethnic groups and are broadly analogous to indigenous territories. Previous studies in the Colombian Pacific and across the Amazon show that indigenous lands do reduce deforestation (Velez et al., 2020; Blackman & Veit, 2018). We use a rolling 5-year baseline between the years 2001 and 2019 to assess PPA outcomes. Our first years of study are between 2001 and 2006 and final years of study are between 2014 and 2019. We choose this time period due to the temporal distribution of PPA establishment and because it covers the pre, post and transitional phase of the signing of the Colombian Peace Treaty with FARC.

5.3 Data and Methods

5.3.1 Study Area

For this study we considered the entire tropical forest biomes of Colombia as defined by Olson et al., (2001) and States where both PPAs and regionally governed PAs are present (Figure 5.1). For those States we also include national PAs in our analysis. Our study area covers 38% of the total land area of Colombia. For each rolling baseline year, we divided the study area into areas under private, regionally governed and national protection or areas under no formal protection. We excluded areas under any other form of protection (e.g., indigenous and afro-Colombian communal titles) or areas where PAs were established within the 5-year study period. In Colombia, national PAs are administered by the Ministry of Environment and regionally governed PAs are administered by the Regional Environmental Agencies (*Corporaciones Autonomas Regionales* – CARs). As of March 2022, there are 280 regionally governed PAs reported to the WDPA and 109 national PAs (UNEP-WCMC and IUCN, 2022). Regionally governed PAs are smaller in size (mean area of 109km^2) compared with national PAs (mean area of 1,654km^2). Regionally governed PAs are clustered within the central Andean region and north of the country whereas national PAs are more evenly distributed, and the largest PAs are found in Amazon region.

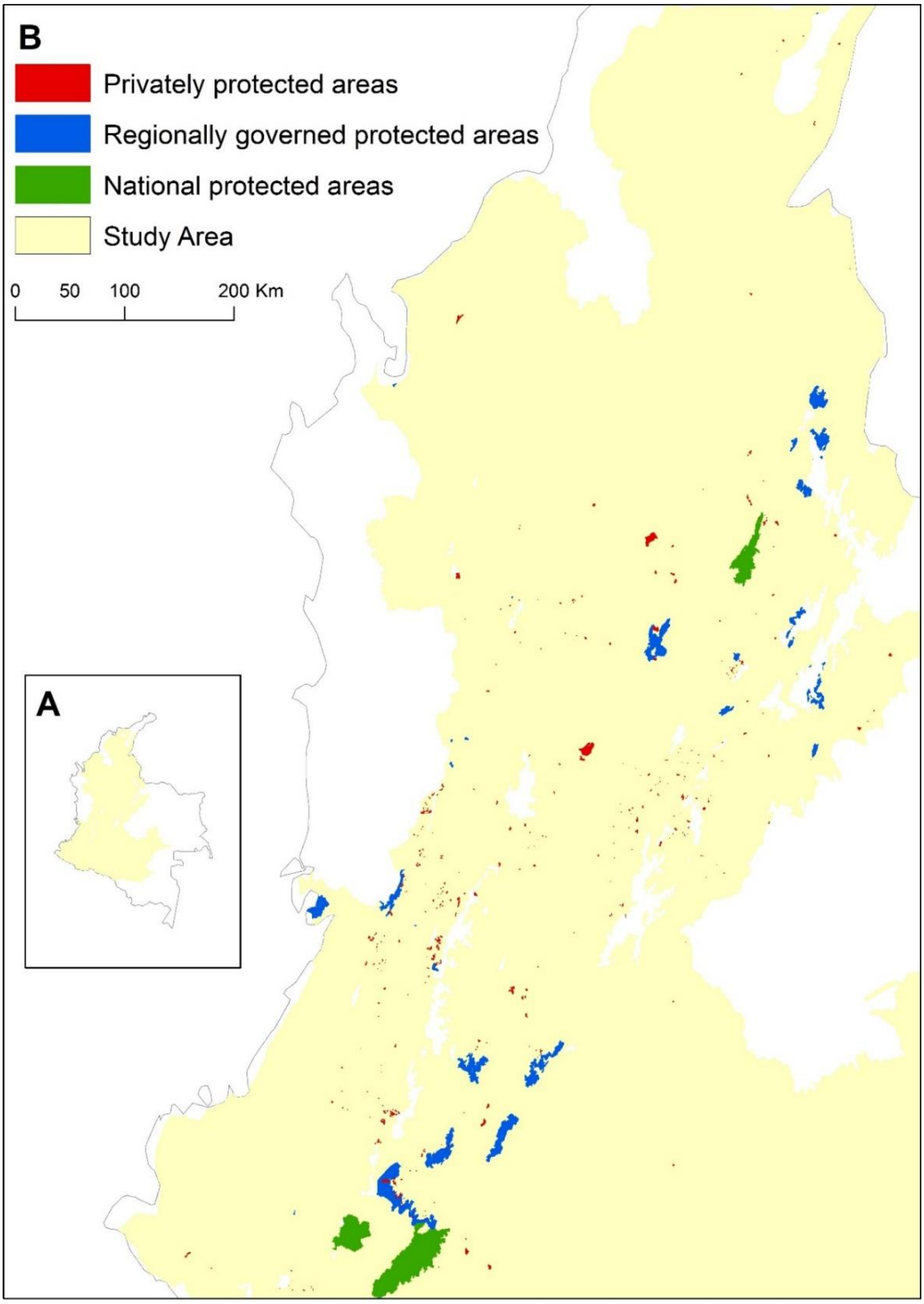

Figure 5.1 Map of (A) Colombia and (B) the study area showing the national protected areas (n=5), regionally governed protected areas (n=34) and privately protected areas (n=271) included in the study

The map was produced in ArcMap 10.4 based on shapefiles collated for this study.

5.3.2 Data

We obtained PA boundaries from the World Database of Protected Areas (WDPA) and El Registro Único Nacional de Áreas Protegidas (RUNAP) in Colombia. We removed any overlap of PA boundaries. We obtained boundaries for 329 PPAs, 36 regionally governed PAs and 5 national PAs. We downloaded indigenous reserves and Afro-Colombian communal territories from the Colombian Geographic Information System for Planning (*Sistema de informacion geografica para la planifacion y el ordenamiento territorial;* SIGOT) for excluding these areas from our study. We obtained baseline forest cover in 2000 from Hansen et al., (2013). Like Vieilledent et al., (2018) we combined the forest cover map of the year 2000 with annual tree cover loss maps to create annual forest cover maps from 2001 to 2019 at 30m resolution. Hansen et al.'s (2013) forest-gain data was not used in our analysis given that the relatively short duration of our study could not render significant forest regeneration in each pixel (Bowker et al., 2016). We computed distance to roads and navigable waterways in ArcMap 10.4. Mean precipitation was calculated from WorldClim. Slope and elevation were calculated from STRM30. Population density was obtained from World Pop and access to market obtained from Global Environment Monitoring Unit. We projected all datasets into MAGNA_Colombia_Bogota and resampled them to 30m^2 resolution (See Supplementary Information – Appendix D - Sampling Design).

5.3.3 Sampling design

In contrast with the PA-network level assessment that samples separately within PAs and control areas (e.g., Joppa & Pfaff, 2011), in this study we created an independent control area for each PA (Zhao et al., 2019). We used this approach to better account for spatial autocorrelation and to help diminish the heterogeneity of unobservable or omitted factors because closely neighbouring control areas are, to a certain degree, homogeneous with their corresponding PAs (Negret et al., 2020; Zhao et al., 2019; Blackman et al., 2015). Previous studies have found that there are regional differences in the biophysical characteristics that help determine deforestation patterns in Colombia (Etter et al., 2006). We set a 1km buffer outside the boundary of each PPA and a 5km buffer outside each state PA, where no control pixels were sampled, to avoid local leakage (Alves-Pinto et al., 2022; Zhao et al., 2019; Schleicher et al., 2017). Next, we generated a buffer of 100km as the control area outside each of the leakage buffers (Zhao et al., 2019). If a PA's control area was overlaid by neighbouring PAs or their leakage buffer areas, we erased the overlapped areas from the control area polygon to ensure each PA had an independent control area. We removed pixels with <30% forest cover at the baseline year and pixels where a PA was established within the 5-year study period. This left PPAs, regionally governed PAs, national PAs and control area pixels that were not contained within another private, regionally governed or national PA, were located outside of the leakage buffer, had never been protected up to the end of the 5-year study period and that were forested at the baseline of the study year (See– Appendix D - Sampling Design). We removed any PAs that had <10 pixels (*n* = 58), as we deemed these too small for use in our analysis. This resulted in a total of 271 PPAs (covering 237km^2), 34 regionally governed PAs (covering 3,667km^2) and 5 national PAs covering (586km^2) for use in our study. The number of pixels sampled within each control area was dependent on the size of the PA. For each PA, the control pixels were selected from the area from the leakage buffer to the outer buffer (e.g., 1km ->100km), where data was available for all the different variables, and there was forest cover in the base line year. To make the matching analysis computationally manageable, we sampled up to a maximum of 1,000 pixels per PA. This resulted in 62,162 PPA pixels, 31,477 regionally governed PA pixels and 5,000 national PA pixels being sampled. We established our sampling design following a rigorous testing processing investigating the impacts of using a leakage buffer or not, buffer sizes of 25km or 100km and whether or not to include a 50m sampling distance restriction on covariate balance, spatial autocorrelation (Morans' I) and the overall deforestation result (See Appendix D - Matching Testing).

5.3.4 Matching technique

We performed 1:1 Propensity-score matching individually for each PA using the *MatchIt* package in R Version 3.6.3. We conducted individual matching due to the uneven distribution of large and small PPAs and to reduce issues of spatial autocorrelation. Each PA was matched with six out of nine possible covariates. For each possible combination of the covariates, the Std. Mean Diff. for each covariate and the mean of the absolute Std. Mean Diff. for the combination of covariates were used to work out which combination of covariates produced the best matches. Values <0.25 were deemed acceptable (See

Appendix D - Covariate Balance). Possible covariates for selection were elevation, slope, precipitation, population density, access to market and distance to all roads, main roads, navigable rivers and forest edge. These covariates were chosen because these are well known drivers of deforestation (e.g., Schleicher, et al., 2017; Nolte et al., 2013; Joppa & Pfaff, 2010).

5.3.5 Statistical analysis

We assessed individual PA effectiveness by comparing forest loss inside PAs with matched unprotected control sites outside of PAs. As per previous studies (Alves-Pinto et al., 2022; Zhao et al., 2019), we calculated deforestation rates within each PA and its control by dividing the number of pixels deforested in a PA (DR_{pa}) or control (DR_{ca}) by the total number of pixels sampled. The absolute effect (AE), sometimes referred to as an impact estimate, of each PA was calculated as the difference between deforestation rates within each PA (DR_{pa}) and its corresponding control area (DR_{ca}), expressed as $AE = DR_{ca} – DR_{pa}$. Accordingly, a positive AE indicated that a PA suffered less deforestation than its control area. The relative effect (RE) of each PA was defined as AE/DR_{ca} x 100%, which measured how far the baseline deforestation had been altered by the PA (Carranza et al., 2014). Since the DR_{ca} of 116 PPAs and 1 national PA equalled zero, only the REs for 155 PPAs and 4 national state PAs were calculated. The mean AE and mean RE of all three PA governance types, as well as their standard errors (SEs) were calculated. The 95% confidence intervals (CIs) of AE mean and RE mean were estimated by the mean value ± 2 X SE (Cumming et al., 2007). We used a paired Wilcoxon test to assess whether there were significant differences in percentage rates of change between PAs and the matched controls. We used the Kruskal-Wallis test to assess whether there were significant differences in the AEs and REs of the three PA governance types.

5.3.6 Robustness check

To test our findings, we also ran post matching logistic regressions for each individual PA (general linear model with binary function) to produce bias adjusted match estimators for each PA. Protection was a binary treatment (0, outside PA; 1, inside PA and forest cover was a binary response (0, forest retention; 1, forest loss). These regressions evaluated the probability of PA experiencing forest cover loss. If the probability of forest loss inside PAs was significantly more than that in unprotected control areas, PAs were deemed ineffective. If the probability of forest loss inside PAs was significantly less than that in unprotected control samples, PAs were deemed effective (See Appendix D – Robustness Check).

5.4 Results

5.4.1 Overall Effects of PAs in Colombia

We found that forest loss was lower inside all analysed PA governance types than their mean matched controls (Wilcoxon signed ranked test, V = 4928, p<0.001). We calculated that PPAs, regionally governed PAs and national PAs avoided a total of 2.40Km^2 (1.01%), 24.49Km^2 (0.69%) and 30.04km^2 (1.03%) of forest cover loss during the study period and they reduced deforestation rates by 0.43%, 0.91% and 1%, respectively, compared to their matched controls (See Appendix D - Deforestation Rates for the results of individual PAs). Kruskal-Wallis tests demonstrated that all three governance types were similar with regard to their AEs (H(2) = 3.9, p = 0.14), however their RE's were significantly different (H(2), = 16.2, p = <0.001). Overall, national PAs exhibited the greatest mean absolute and relative effects and the least heterogeneity of results. PPAs exhibited the lowest mean absolute effect and greatest heterogeneity of results. Regionally governed PAs exhibited the lowest mean relative effect (Figure 5.2) (See Appendix D - Absolute and Relative Effects for the results of individual PAs).

5.4.2 Effects of PPAs

Deforestation rates inside PPAs ranged from 0% to 29.4% with a mean of 0.89%, of which 206 (76%) PPAs experienced no evident forest loss. In contrast, deforestation of control areas varied from 0% to 11.25% with an mean of 1.32% (Figure 5.3; Figure 5.4). Deforestation was not detected in 116 (42.8%) control areas. Deforestation rates within 135 PPAs (50%) were lower than their corresponding controls, whereas 34 PPAs (12%) suffered more deforestation within their boundaries than their controls. Deforestation rates in 102 PPAs (38%) were equivalent to those in their corresponding controls. The AEs of PPAs varied from -29.41% to 11.25% and the REs of PPAs varied from -500% to 100% (Figure 5.5). The mean AE of PPAs was 0.43% (95%CI: -0.02%, 0.88%) whereas the mean relative effect of PPAs was 69.57% (95% CI: 57.25% - 81.88%) (Figure 5.2).

5.4.3 Effects of regionally governed PAs

Deforestation rates inside regionally governed PAs ranged from 0% to 3.1% with an mean of 0.52%, of which 10 (25.64%) PAs experienced no evident forest loss. In contrast, deforestation of the controls varied from 0% to 7% with a mean of 1.43% (Figure 5.3; Figure 5.4). Deforestation was not detected in one control. Deforestation rates within 30 regionally governed PAs (88.24%) were lower than their corresponding controls, whereas 1 regionally governed PA (2.94%) suffered more deforestation within their boundaries than their control. Deforestation rates in 3 regionally governed PAs (8.82%) were equivalent to those in their corresponding controls. The AEs of regionally governed PAs varied from -0.90% to 4.39% and the REs of PPAs varied from -100% to 100% (Figure 5.5). The mean AE of regionally governed PAs was 0.91% (95%CI: 0.54%, 1.28%) whereas the mean relative effect of regionally governed PAs was 61.10% (95% CI: 45.62%, 76.58%) (Figure 5.2).

5.4.4 Effects of national PAs

Deforestation rates inside national PAs ranged from 0% to 0.5% with a mean of 0.22%, of which 2 (40%) experienced no evident forest loss. In contrast, deforestation of the controls varied from 0% to 2.7% with a mean of 1.22% (Figure 5.3; Figure 5.4). Deforestation was not detected in 1 control. Deforestation rates within 4 national PAs (80%) were lower than their corresponding controls. 0 national PAs suffered more deforestation within their boundaries than their controls. The deforestation rate in 1 national PA (20%) was equivalent to that in its corresponding control. The AEs of national PAs varied from 0.00% to 2.20% and the REs of PPAs varied from 70% to 100% (Figure 5.5). The mean AE of national PAs was 1.0% (95%CI: 0.21%, 1.79%) whereas the mean relative effect of national PAs was 83.92% (95% CI: 71.56%, 96.28%) (Figure 5.2).

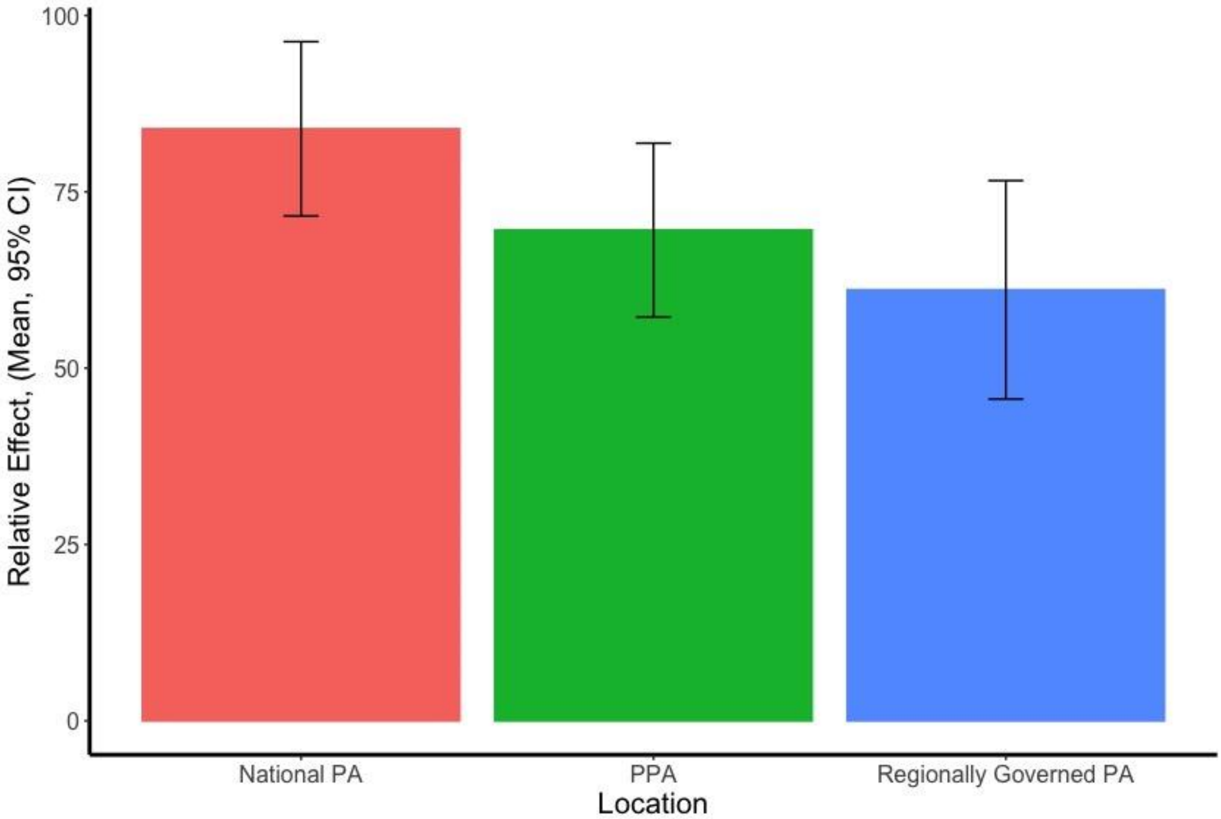

Figure 105.2 Mean relative effect of avoided forest cover loss for PPAs, Regionally Governed and National PAs.

The relative effect of a PA is defined as the absolute effect (difference between deforestation rates with the PA and it corresponding control) / deforestation rate in the control X 100%. The relative effect measures how far the baseline deforestation rate has been altered by the PA. Error bars represent the standard errors of means and only one arm is plotted for each SE bar for clarity.

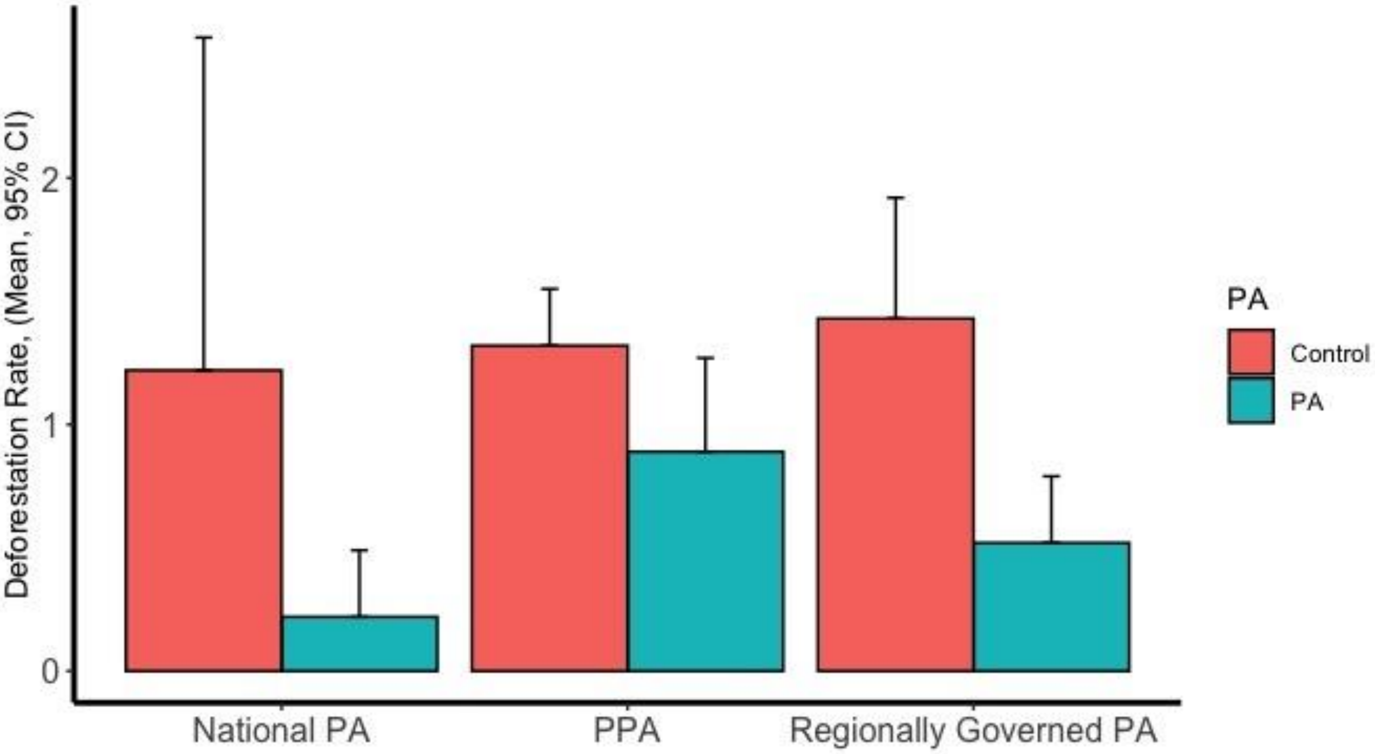

Figure 5.3 Mean deforestation rates in PAs vs. Controls

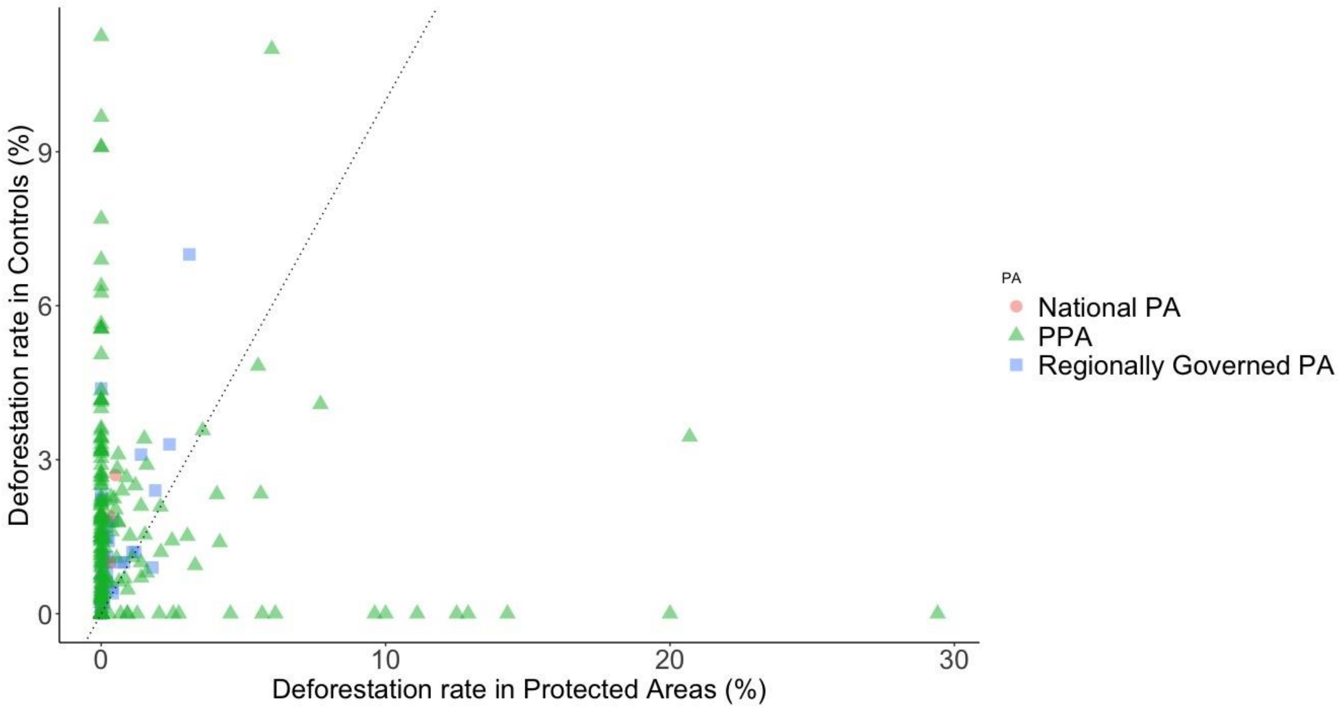

Figure 5.4 The deforestation rates of individual PPAs, Regional Governed PAs and National PAs and their corresponding control areas

PA symbols above the dashed line indicate deforestation rate in the PA < Control Area, and symbols below the dashed line indicate deforestation rate in PA > Control Area.

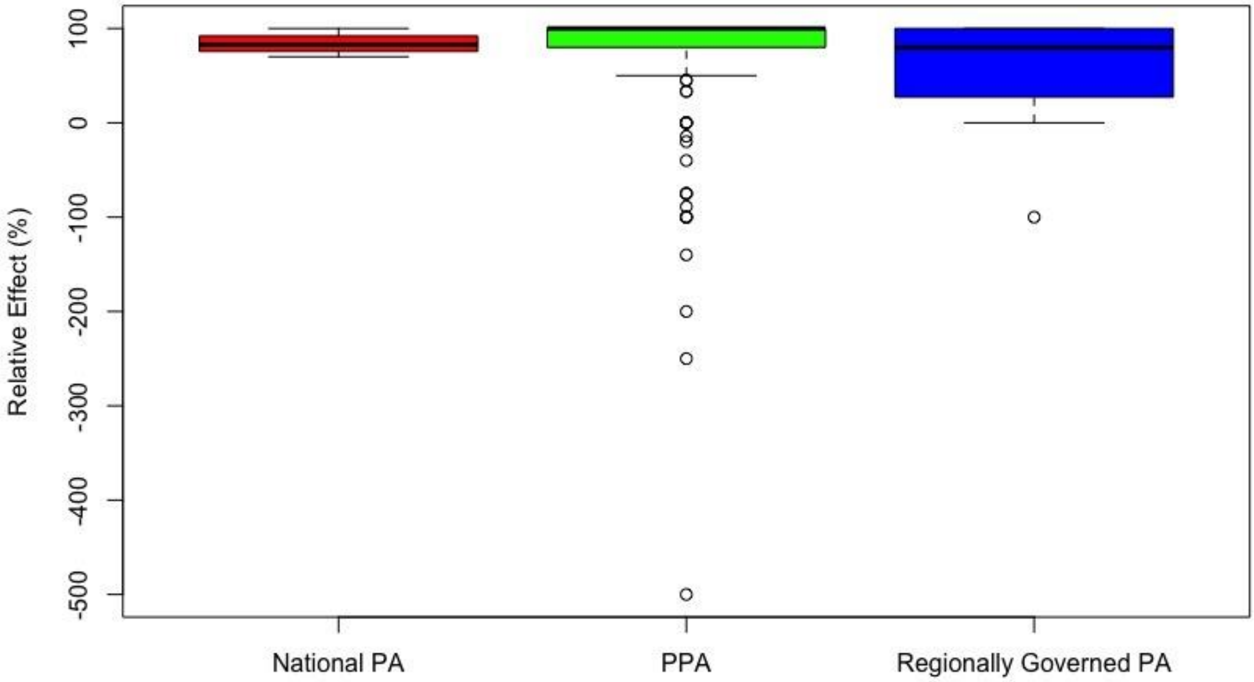

Figure 5.5 Effectiveness of PPAs, Regionally Governed PAs and National PAs in the study area

The relative effect of a PA is defined as the absolute effect (difference between deforestation rates with the PA and it corresponding control) / deforestation rate in the control X 100%. The relative effect measures how far the baseline deforestation rate has been altered by the PA. The length of each whisker extends no more than 1.5 times the interquartile range. Values which fall out of the range of two whiskers in a boxplot are shown as open circles.

5.5 Discussion

5.5.1 The conservation performance of PPAs, regionally governed and national PAs

Our study adds to the growing body of evidence that PPAs can be effective at reducing deforestation (Shumba et a., 2020; Schleicher et al., 2017). Our finding that PAs reduce deforestation by 0.91% (during the study period –5 years) is similar to studies in other countries within South America (Koskimaki et al., 2021; Schleicher et al., 2017). Our result that PPAs reduce deforestation by 0.89% is similar to Schleicher et al., (2017), who found conservation concessions in Peru reduce deforestation by 1.53%. Our analysis confirms that all three PA governance types have contributed to avoiding deforestation, compared to control areas with similar characteristics. This finding is consistent with previous counterfactual matching studies both within and outside of Colombia which find that PAs reduce deforestation (Bonilla-Mejia & Higuera-Mendieta et al., 2019; Negret et al., 2019; Zhao et al., 2019; Shah & Bayliss, 2015).

5.5.2 The conservation performance of PPAs relative to other PA governance types

A previous study into the outcomes of PPAs shows that they can be more effective than state PAs (Schliecher et al., 2017), however this study does not separate state PAs into those that nationally and regional governed. Our study builds on this previous study and shows that whilst PAs may be more effective than regionally governed PAs, they may not perform as well as PAs under national governance. Similarly to other studies, we find that national PAs were more effective than regionally governed PAs, (Herrera et al., 2019; Blackman et al., 2015; Carranza et al., 2014). National PAs may outperform regionally governed PAs and PPAs in our study firstly because national PAs have better access to political, technical and scientific resources and funding. There is little up to date or reliable information on PA finance globally or for Colombia (Emerton et al., 2006), however Zhang et al., (2017) found that in China, national PAs were generally better funded than regionally governed PAs. In neighbouring Peru, Lam (2017) found that high visitor numbers and larger areas were predictors of overall funding allocation within the national PA system and that that 81% of PA funding came from international donors. National PAs in Colombia possess the characteristics which are more likely to attract funding (Lam, 2017), as they account for the top five most visited PAs in Colombia and are much larger than regionally governed or private PAs. Several studies also suggest that larger parks are perceived to have a higher probability of long-term success and may generate more NGO involvement than smaller parks, due to phenomena such as edge effects (Blackmann et al., 2015; Joppa et al., 2008; DeFries et al., 2005). It is also probable that larger, national government run PAs have higher media profiles and are better known to international donors and therefore receive more funds. The majority of PPA owners in Colombia are local small landowners who may only have access to limited funds, political resources and technical expertise. Secondly, national PAs in Colombia are more likely to be strictly protected than regionally governed state PAs and PPAs which are more likely to be sustainable use areas (See Appendix D - PA Management Category by PA Governance Type). Within our study, 80% of national PAs (n = 4) are assigned management category of II and the remaining 20% (n = 1) is assigned a management category of IV. All regionally governed PAs are assigned a management category of II and all PPAs are assigned a management category of VI. Category II PAs (National Parks) are managed mainly for ecosystems protection and recreation, category IV PAs (Habitat / Species Management Areas) are managed for conservation through management intervention and category VI PAs (Managed Resource Protected Areas) are managed mainly for the sustainable use of natural ecosystems. Previous studies in Colombia have found that differentiation in use restriction legislation limits the conservation potential of some PA categories (Aldana & Mitchley, 2013). Our study also corroborates findings from other countries which found that strictly protected areas are more effective at reducing forest cover loss than sustainable use areas (Jones et al., 2018; Pfaff et al., 2017; Nolte et al., 2013). However, other studies have found the opposite - that sustainable use areas are more effective (Pfaff et al., 2015; Pfaff et al., 2013). Thirdly, enforcement of protection may be greater in national state PAs than regionally governed PAs. The protection of national PAs in Colombia is enforced by the Ministry of Environment and Sustainable Development, while regionally governed PAs are under the jurisdiction of decentralized environmental authorities, which are more prone to capture by special interests (Bonilla-MeJia & Higuera-Mendieta, 2019). There are different general theories regarding the ability of private actors to enforce sanctions and rules on their land. Kramer et al., (2002) hypothesise that whilst private landowners may be relatively effective in establishing and demarcating PA boundaries and detecting encroachment, they may have difficulties enforcing sanctions because of their lack of political status and influence with local governments. Whereas Cooke et al., (2011) argue that local people may see private conservation as more legitimate than state conservation as it does not challenge private property rights in the same way as many state conservation initiatives, exposing them to loss opposition and making them more effective.

PA size may also be a factor causing differences in PA effectiveness. Previous studies have shown that generally smaller PAs are less effective than larger parks (Bowker et al., 2016; Tranquilli et al., 2014; Pfeifer et al., 2012; Maiorano et al., 2008). This is because they have a large boundary-to-area ratio resulting in proportionally more boundary violations and they are more likely to follow the dominant land-use change patterns into which they are embedded. It could be that because regionally governed PAs and PPAs are much smaller than national PAs in Colombia they more prone to boundary violations and influenced by land-cover change occurring outside of their boundaries. However, others argue that smaller PAs may be more effective at reducing deforestation as guards have less ground to cover making intrusions more detectable and therefore, they are easier to protect. There is some evidence from the Brazilian Cerrado that smaller PAs do perform better at protecting natural habitat than larger areas (Paiva et al., 2015). However, the Brazilian Cerrado is a grassland and so the results found there may not translate to forest ecosystems. A meta-analysis of the effects of 49 tropical PAs reported that the size of a PA does not appear to correlate with deforestation outcomes (Naughton-Treves et al., 2005).

5.5.3 Heterogeneity of PPA effectiveness

Like other studies, we show that no PA governance type guarantees complete protection (Schleicher et al., 2017; Nolte et al., 2013). Despite the consistency of average patterns, like other studies, we observed individual cases with high and low relative effects for all protection types (Koskimaki et al., 2021; Zhao et al., 2019; Butsic et al., 2017; Shah & Bayliss, 2015). We found outcomes of PPAs were considerably more variable than regionally governed and national PAs (-500% - 100% relative effect). Conservation concessions in Peru also exhibit greater variation in terms of avoided deforestation than state PAs (Schleicher et al., 2017). PPAs may have a greater variation in their effectiveness as they are more likely to be in closer proximity to external threats than other PA governance types (Palfrey et al., 2022). Multiple studies have shown that differences in the level of threat and external pressures can affect a PAs effectiveness to reduce deforestation (e.g., Nolte et al., (2013); Pfaff et al., (2008)). Murillo Sandoval et al., (2020) show that some PAs in Colombia occur in areas of high armed conflict and intense illegal activities which are both factors that increase deforestation. Using a deforestation probability map of Colombia produced by Negret et al., (2019), we ran a primarily analysis to determine to what extent the three PA governance types were in areas of high deforestation probability. We found that PPAs were much more likely to be in areas of greater deforestation probability (as predicted by the model) compared with national and regionally governed PAs. 95% of the total area of PPAs is in areas with a deforestation probability of 90 – 100% compared with 79% of the area of regionally governed PAs and 24% of the area of national state PAs (See Appendix D – Proportion of each PA governance type within each probability of deforestation level). The fact that PPAs are a greater risk of deforestation, due to their proximity to cities and their suitability for agriculture, may be another contributing factor to their highly variable effectiveness, compared to other PA governance types.

The high heterogeneity of PPAs effectiveness in Colombia may also be explained in part due to PPA legislation (Law 99) and the multiple motivations for PPA creation from various PPA owners. Legally, for a PPA to be established in Colombia, part of the land needs to be dedicated for conservation purposes and left to its natural state. The rest of the land can be used for agriculture, ecotourism, education and research or sustainable extraction, but the area must be used in line with sustainable practices (Myron et al., 2019; Lopez & Arbelaez, 2015). PPAs are created by a diverse range of actors including; individuals, local community groups, NGOs, research organisations and corporations. Some actors create PPAs with the sole intention of that area being dedicated to conservation, whereas some owners have a proportion of the PPA dedicated to conservation but use other areas for agriculture, ecotourism or sustainable timber extraction. In some departments, PPA creators receive financial incentives such as property tax exemptions and this may drive PPA establishment, rather than solely intrinsic conservation motives, and this might impact on their effectiveness. However, a survey of PPA owners in Valle de Cauca by Lopez & Arbelaez (2015) found that the strongest drivers for PPA establishment were for conserving nature for utility and owner well-being and personal satisfaction. Within our study 17% (*n* = 46) of PPAs are solely dedicated for conservation purposes. We found that on average; 49.65% of a PPA's area is dedicated to conservation, 49.34% is dedicated to agricultural systems (e.g., agro-forestry or pastures), 8.81% is used as buffer zone for sustainable timber and firewood extraction and 3.53% have intensive uses and infrastructure (e.g., houses or stables). Overall, within our study 63.42% of the total area of PPAs is dedicated to conservation, 24.66% to agricultural systems, 10.10% for buffer zones for timber / firewood extraction and 1.82% for intensive use and infrastructure (see Appendix D – Uses of Private Protected Areas). We find that in some cases where PPAs have extremely high deforestation rates they have dedicated buffer zones (*Zona de Amortiguación y Manejo Especial*) for firewood extraction (see https://runap.parquesnacionales.gov.co/categoria/SINAP/20 for further information on the demarcation of activities within individual PPAs in Colombia).

5.5.4 Strengths and limitations

We present one of the first studies to individually assess the effects of PAs with different governance categories. Mort studies evaluate the effectiveness of PAs at the entire network level by separately sampling within a PA system and the wider unprotected area (Andam et al., 2008; Joppa & Pfaff, 2011). However, this approach leads to larger PAs being given more weight than smaller ones (Carranza et al., 2014). By sampling and setting up an independent control area for each PA, the effectiveness of each individual PA could be quantified, and the median relative effect calculated by weighing each PA equally. This was important for this study as the majority of PPAs were <1km^2 (86.7%, *n* =235) and a very few >10km^2 (1.8% > *n* = 5). This study shows that considerable heterogeneity on the effectiveness of PAs exists within different PA governance categories. This suggests that broad generalisations may be misleading for conservation decisions at the local scales. Our results indicate the distinct need to complement broad generalised findings of the effectiveness of different PA groups with the impacts of individual PAs. This is particularly relevant for PPAs, given the huge potential for heterogeneity that exists among individual PAs within this category. Conducting an analysis at the individual level can help determine what makes a PA successful (Koskimaki et al., 2021; Ament & Cumming, 2016). Although not investigated within our study, previous studies which conduct individual PA assessments assess the impacts of governance and management quality (Zhao et al., 2019; Eklund et al., 2019). Our results emphasise that the degree to which different forest governance regimes effectively preclude deforestation depends on which measures of impacts are being assessed. For example, when comparing absolute vs. relative effects using one or the other means than either PPAs or regionally governed PAs are deemed more effective within our study. This is also the same for median avoided deforestation rate vs. percentage of PAs which completely avoid deforestation. Our study shows that PPAs have the highest deforestation rates of all three PA governance types. However, our study also shows that PPAs most consistently have no deforestation with their borders (76%) compared with 40% and 25% of national and regionally governed PAs respectively. We also show that how groups of PAs are compiled can affect results. For example, Schleicher et al., (2017) show that PPAs can outperform state PAs, however they do not break state PAs down into their subcategories (e.g., federal or national ministry or agency PAs, sub-national ministry or agency PAs, government-delegated management PAs). In our study we show that when national and regionally governed PAs are analysed separately, national PAs outperform PPAs, but regionally governed PAs do not.

We acknowledge several limitations within our study. Firstly, our study relies upon the Hansen Global Forest Change dataset (Hansen et al., 2013). This dataset has been critiqued in the literature since it poorly differentiates forests from other similar high biomass vegetation in Colombia (Fergusson et al., 2020; Tropek et al., 2014). We are aware that Colombia has a national level forest cover dataset (IDEAM), however we did not use this as it is only released every five years and so did not fit in with the timeline of our study. We used the Hansen dataset as it represents the most comprehensive globally and regionally available data set, it allows forest loss to be calculated year on year and is useful at a local scale (Bowker et al., 2016). Secondly, our study does not make the distinction between illegal deforestation which all protection types seek to reduce, and subsistence deforestation driven by the livelihood needs of local people which is legally sanctioned in regionally governed PAs and PPAs. Moreover, we are unable to determine if any of the deforestation which took place within PAs was part of restoration work to remove non-native or invasive species. Therefore, our study is not able to determine how much illegal deforestation has been prevented or how much non-native or invasive species have been removed, rather it looks at total forest cover loss. It is estimated that 10% of deforestation in Colombia is attributable to illegal logging (Clerici et al., 2020). Thirdly, PAs can be effective along multiple dimensions not just reducing deforestation. While the lack or presence of deforestation is a clear representation of PA impact and it is informative of the condition of the environment and its threats, it should not be considered as the sole determinant of PA effectiveness. We do not assess forest degradation, declines in species diversity, ecosystem services or functioning or the presence of invasive species that can all be detrimental to ecosystem health (Lewis, Edwards and Galbraith, 2015). Lastly, in this study we compare the performance of different PA governance types but it important to remember different PA groups may be able to be established where others are not. Palfrey et al., (2022) show that PPAs are more likely to be in grasslands than other PA types because so much of global grasslands are under private ownership. It may also be the case that different PA types are more likely to be in native or secondary forests although we have not tested this.

5.6 Conclusion

In conclusion, we found that all PA governance types experienced less forest loss than their matched unprotected controls. Moreover, our findings add to the growing body of literature which finds that non-state actors can be effective at implementing conservation initiatives in areas of high deforestation probability. Although superficially our study points to PAs as effective tools for forest conservation, the complete picture is much more complex, as demonstrated by the high variability of PA effectiveness at the individual PA level. Our results emphasise the importance of comparative individual-level impact estimates for PAs to help guide the conservation of tropical forests and we hope that our study encourages more of these studies to determine why, where and when different PA governance types deliver positive biodiversity outcomes and what makes PAs successful.

5.7 References

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Chapter 6. Discussion and Conclusion

6.1 Overview

Biodiversity loss is a worldwide problem requiring global and local level solutions (Bishop 2012; Reade et al. 2015; Steffen et al. 2015). To combat this issue, as part of the Post-2020 Global Biodiversity Framework, the United Nations has called for 30% of the Earth's land and sea to be under some form of protection by 2030. PAs are heralded as the cornerstone of global biodiversity conservation efforts and are a long-standing tool in the pursuit of nature conservation. There is a wealth of research into the outcomes of PAs but a relative paucity of work focussed on what factors affect these outcomes (Palfrey et al., 2021). Of the work that exists the majority has focused upon how PA outcomes are affected by different management strategies (e.g., strict protection vs. sustainable use). This is despite the increasing acknowledgement that governance (i.e., who has power, who is involved in decision making and how are benefits shared) is a key determinant of PA outcomes (WCPA, 2004: 257). Within this thesis, I have primarily assessed how does *who* governs PAs affect their outcomes and indirectly assessed how effectively PAs are being governed and if they can be deemed as being under "good governance". Within the parameters of good governance discussed in the introduction (see section 1.3.3. – Protected Area Governance), this thesis has assessed to what extent PPAs are "effective and efficient" at contributing to achieving global biodiversity targets such as Aichi Target 11. Of course, all aspects are "good governance" are important to study and section 6.7.6 suggests ways in which other aspects of good governance not studied in this thesis could be investigated.

This thesis reviewed both environmental and social outcomes of PPAs. However, due to the size of the task and quality of work input into determining the environmental outcomes of PPAs it provided difficult to also include a detailed analysis of social outcomes within this thesis. Empirical analysis predominately focussed upon the environmental outcomes of privately protected areas (PPAs) and assessed both what do PPAs protect? And how effective are they at protecting it? These outcomes have been studied at a range of scales using a variety of techniques moving from a broad scale literature review in chapter 3 to a narrow, detailed, and forensic counterfactual modelling approach in chapter 5. Through the three results chapters, the thesis has made a range of contributions to the current knowledge of how PPAs contribute to the conservation estate and achieving global biodiversity targets and how this compares with other PA governance types.

This chapter begins by summarising the findings from the three results chapters, how they fulfil the three research questions, and outlines how the three empirical chapters join together to form one cohesive piece of research. It then considers the implications of this research in relation to policy and practice and concludes by outlining the priorities and opportunities for further research.

6.2 Summary of findings

Each of the result chapters focussed on one of the three research questions. This section briefly summarises the findings from each chapter in relation to the three research questions. The main findings of these chapters, how they advance the knowledge in this area and policy recommendations is summarised in Table 6.1 and how these interlink is considered in section 6.3.

6.2.1. What is currently known about the environmental and social outcomes of privately protected areas?

Chapter 3 assessed the existing knowledge of PPA outcomes through a comprehensive literature review. The findings show that environmental outcomes of PPAs were mostly positive but social outcomes were more mixed. PPAs increased the number or extent of ecosystems, ecoregions or species covered by PAs (representativeness) and PA network connectivity and effectively reduced deforestation and restored degraded lands. Few PPA owners reported negative social outcomes, experienced improved social capital, increased property value, or a reduction in taxes. Whilst local communities benefited from increased employment, training, and community-wide development (e.g., building of schools), they also reported reduced social capital and no significant difference to household income after the establishment of a PPA. However, the evidence base from which these findings was drawn was limited. The overall number of articles regarding PPAs is small (*n* = 412) relative to the current number of PPAs (*n* = 13,103). Most studies were conducted at national or sub-national scales in only 5 countries (United States *n* = 226, Brazil *n* = 31, Australia *n* = 31, South Africa *n* = 30 and Chile *n* = 19). Environmental outcomes were considered in 79 studies and focused mainly on species coverage (*n* = 37) and ecosystem representativeness (*n* = 20). Social outcomes were discussed in 48 studies, the majority of which focused on financial outcomes (*n* = 36). The findings demonstrate that greater research is needed into PPAs outcomes and why they occur, across a more diverse number of countries and contexts.

6.2.2. What is the spatial distribution of PPAs and how does this compare with other PA governance types and random placement?

Building upon one of the main findings of the literature reviewed in Chapter 3, Chapter 4 used spatial mapping techniques to determine the coverage and representativeness of PPAs and how they contribute to PA network connectivity compared with other PA governance types and random placement. The study focused on 15 countries across 5 continents that have >10 PPAs reported to the WDPA. The findings show that PPAs (i) are three times more likely to be in biomes with <10% of their area protected compared with other PA governance types, (ii) are twice as likely to be in areas with the greatest human disturbance compared with other PA governance types, (iii) protect a further 1.2% of KBAs (Key Biodiversity Areas) and (iv) make a greater contribution to conservation than if they were randomly placed. The findings show that PPAs contribute significantly to PA coverage in some countries, but their contributions are negligible in others. Overall, PPAs account for 3.4% of land under protection in our chosen study countries. The findings also show that their ability to increase PA network connectivity is moderate. An average of 38% of PPAs are adjacent to a PA of another form of governance and they increase protected connected land by 7.04%. The findings demonstrate that PPAs have unique and important contributions to offer to the global conservation estate and in particular protecting highly threatened areas and unrepresented biomes. Highlighting the potential of PPAs to contribute to the global conservation estate is particularly timely, given the recent discussions taking place with regards to post 2020 global biodiversity targets.

6.2.3 To compare how effective private protected areas at reducing deforestation and degradation compared to PAs of other governance types

Chapter 5 evaluated how effective PPAs are in Colombia at reducing deforestation as opposed to regional state and national PAs. I conducted an impact evaluation using propensity score matching to assess the individual performance of privately protected areas (PPAs), regionally governed PAs and national state PAs to curb deforestation. The findings show that on average PPAs, regionally governed PAs and national state PAs are all effective at curbing deforestation compared to unprotected areas with similar characteristics, but no PA governance type guarantees protection. On average, national state PAs are the most effective at curbing deforestation (average relative effect of 83.9%) and regionally governed PAs the least effective (average relative effect of 61.10%). PPAs have the most heterogeneous results (range of deforestation rates in individual PAs: 0% - 29.41%). The results show that PPAs can be effective at reducing deforestation and emphasises the need for more PA impact studies that compare multiple PAs at the individual level.

Table 6.11Summary of research objectives, key findings and policy recommendations

Paper	Objective	Justification for chapter	Method	Key findings	Importance of findings	Policy Implications
1	Determine the current state of knowledge on PPA outcomes	PPAs have received relatively little scholarly attention and existing studies have never been pooled to create a synthesis of knowledge on PPA outcomes.	Comprehensive literature review	1. Research into PPAs is limited and dominated by conservation easements in the USA (n = 216, 52% of all studies). 2. PPAs have overwhelmingly positive ecological outcomes (89%, n = 70) yet social outcomes are more mixed (65%, n = 48). 3. Few PPA owners reported negative outcomes however whilst surrounding residents benefited from increased	This paper offered the first comprehensive review of the outcomes of PPAs. It showed that whilst PPAs can have positive ecological outcomes they may have similar drawbacks as other forms of PAs in relation to the social outcomes for surrounding local communities.	1. Better reporting of PPA outcomes is required. 2. A deeper understanding of how PPAs operate and what outcomes these leads to is required. 3. PPAs may have similar negatives social impacts as other forms of PA governance and these need to be mitigated against.

				employment, training and community wide developed they reported a loss of social capital and no change to household income.		
2	Determine the coverage, connectivity, and representativeness of PPAs	Studies of the coverage and connectivity of PPAs have been limited to sub-national regions. No multi-country study has ever been conducted to assess to what extent PPAs contribute to the global PA estate and how this compares to other PA	Spatial mapping	1. PPAs account for 3.4% of land under protection (in study countries chosen) 2. PPAs increase PA network connectivity by 7.04% 3. PPAs have a greater proportion of their total area in underrepresented and threated biomes, areas of high human pressure and	These findings suggest that PPAs have unique and significant contributions to offer to the conservation estate and they may be present in areas where it is unlikely that PA of other forms of governance can/ will be established.	1. More resources into PPA research, especially in finding the strategies and instruments to strengthen the persistence, quality, and extent of PPAs is needed. 2. Increased institutional support from governments,

		governance types.		protect a further 1.2% of KBAs (Key Biodiversity Areas). 4. PPAs perform better than random placement.		conservation organizations, and funding agencies for actions that strengthen PPAs is needed. 3. PPAs should be better integrated into national biodiversity conservation strategies
3	Determine how effective PAs are at reducing deforestation and what effect governance type has on PA effectiveness	Few studies have been conducted to assess the individual impacts of PAs on reducing deforestation and what impact PA governance	Quasi-experimental design	1. All PAs types in Colombia are effective at reducing deforestation compared to unprotected areas with similar characteristics. 2. National PAs are the most effective at reducing	This study offered the first individual-level assessment of PA effectiveness for forest protection and to assess the impacts of PPAs, in Colombia. It showed that whilst on	1. PPAs can be effective at reducing deforestation but do not guarantee protection 2. PPAs establishment efforts may be best placed into areas of highest human threat

		type has on PA outcomes.		deforestation and regional state PAs the least effective. 3. PPAs have highest variation of results.	average PPAs can be effective at reducing deforestation there is substantial variation among them.	3. Different measures of PA outcomes may generate different result on which is most effective – it is best to use a variety of results

6.3 Linking the empirical chapters together / wider implications of the thesis

This thesis has followed a sequential structure with each empirical chapter (chapters 3 – 5) building upon and answering questions posed by the findings of the previous. In chapter 3, I reviewed the existing published literature on PPAs in order to determine and categorise *“What are the possible outcomes of privately protected areas?”* Outcomes of PPAs were divided into two main types: environmental and social. Environmental outcomes were further sub-divided into two main types; (1) what are the spatial attributes of PPAs which provide positive environmental outcomes? And (2) how effective are they? Social outcomes were subdivided into impacts on different types of capital as defined by the sustainable livelihoods’ framework (DFID, 2000).

Whilst chapter 3 determined the possible outcomes of PPAs it did not measure to what extent these outcomes are realised. In chapter 4, I built upon the findings of chapter 3 by measuring to what extent the spatial distribution PPAs produces positive benefits for the existing conservation estate. I chose to focus on the spatial contributions of PPAs as this was one of the most reported positive outcomes emerging from PPAs and spatial data easily lends itself to interrogation via big data methods.

Although chapter 4 determined the location of PPAs and that PPAs exist as institutions, it left questions of *“how effective they are as institutions within these locations?”* Findings from chapter 3 also stated that PPAs are more effective than other governance types. It is important to determine the effectiveness of PPAs because many PAs are reported as being paper parks (e.g., Borg, 2019: Jones et al., 2018). That is to say that legally PAs exist as entities, but they have little impact in reality. In chapter 5 I determined the effectiveness of PPAs by assessing to what extent they reduce deforestation. I choose this metric because this is one of the main ways in that PA effectiveness is currently determined and so my study can be compared with findings from other countries and secondly because remotely sensed data of deforestation is readily available. Figure 6.1 depicts the connections of these three empirical chapters.

Chapter 3	
Categories of PPA outcomes	
Environmental Outcomes	**Social Outcomes**
• Spatial • Effectiveness	• Financial Capital • Social Capital • Human Capital • Physical Capital • Natural Capital
Unanswered question: To what extent to do PPAs produce these outcomes?	

Chapter 4
Spatial outcomes of PPAs
• ↑ Redundancy • ↑ Complementarity • ↑ Connectivity
Unanswered question: How effective are PPAs in the locations in which they are found?

Chapter 5
Effectiveness of PPAs to mitigate deforestation
On average, PPAs are effective at reducing deforestation relative to what would have happened had the area have not received treatment. However, there is substantial variation in among them and some are shown to induce rather than avoid deforestation.

Figure 6.1 Linking the empirical chapters together

6.4 Contributions to theory

As outlined in Chapter 1 Section 1.4 – Private Governance, there has been much scepticism and critiques regarding the outcomes of PPAs. Within this section I outline how my research has contributed to proving or disproving these critiques and how my research findings align with other studies on PA outcomes.

Critiques of PPAs have stated that they are a less than optimally effective conservation mechanism (Clements and Cumming, 2017). One main reason for this is because with a few exceptions, PPAs tend to be small and therefore it is argued that they cannot conserve megafauna effectively or increase make meaningful contributions to increasing PA network connectivity (Langholz and Lassoie, 2001). My research findings in Chapter 4 showed that whilst in some countries PPAs offer minimal benefit, in others, where PPAs are more numerous and benefit from legal recognition and integration into national conservation policies (e.g., South Africa) they can substantially increase total PA network coverage and connectivity. My findings support the research of De Vos & Cumming (2019) and Rissman and Merenlender, (2008) who both found that PPAs moderately increase the connectivity of existing PA networks. A second criticism of PPAs is that their establishment is driven by individualistic and opportunistic parcel-level protection efforts rather than by a systematic landscape-scale process (Ryan et al., 2014). As such, important habitats may not be represented and the benefits derived from PPAs is limited (Ladle et al. 2014; Ryan et al. 2014). However, my research findings show that despite less than optimal conservation planning, PPAs perform better than random across a range of metrics. This result contrasts with findings by Clancy et al., (2020), Kuempel et al., (2019) and Mason et al., (2018) who all found that PA networks (composed of a mixture of all governance types but predominately state PAs) perform worse than random placement. Moreover, my research findings have highlighted that PPAs have unique contributions to offer to the global conservation estate by (i) representing ecoregions and biomes which are under-represented by other forms of PA (ii) protecting areas which are surrounded by high human pressure and (iii) are the governance type which is most likely to be located within a KBA. My research adds to the existing evidence provided by Schutz (2017), Shanee et al., (2017) and Gallo et al., (2009) that PPAs are located in areas where other forms or PA are not and as such, they increase the complementarity of PA networks.

Questions are also raised regarding the effectiveness of PPAs to provide positive environmental benefits. Some sceptics are concerned that PPA owners may lack expertise to successfully manage PPAs or that they have insufficient institutional strength to enforce PPA boundaries (Holmes, 2015; Pasquini et al., 2011; Langholz & Lassoie, 2001; Barany et al. 2001). Others are concerned that PPA owners may lack resources and funding which may render PPAs ineffective, particularly in the long term (Fitzsimons 2015; Ryan et al., 2014). Some PPAs are funded through eco-tourism, and this may introduce a further layer of complications reducing the effectiveness of PPAs. Baum et al., (2017) warns that PPAs engaged in ecotourism may risk degrading the resource they were set up to conserve. This is because PPA owners may have to make decisions based on tourist satisfaction rather than what is environmental beneficially (de Santo, 2012). Others raise concerns about how the outcomes of PPAs are monitored. For example, Rissman et al., (2007) found in a study of easement in the US, only 20% had quantitative monitoring programs. Similarly, in Australia's covenanting programs, organizations measured outcomes inconsistently, making it difficult for researchers to identify positive biodiversity outcomes across the covenanting programs in a systematic way (Fitzsimons and Carr, 2014). Contrary to these criticisms and concerns, in Chapter 5, my research shows that PPAs can be effective at reducing deforestation. My findings support research by Schleicher et al., (2017) conducted in Peru who found that PPAs are more effective than others form of PA at reducing deforestation and Shumba (2020) in South Africa who found that PPAs are more effective than other forms of PA at reducing land cover change.

6.5 Contributions to methods

Historically within the conservation literature, there has been a divergence in scale at which studies are conducted. Whilst some have perused large-scale quantitative studies to assess conservation intervention outcomes (e.g., Andam & Ferraro, 2010; Clements & Milner-Gulland, 2014), others have championed a localised case study-based approach utilising qualitative methods (e.g. Yin, 2014). However increasingly there have been calls for integrating natural and social sciences' perspectives and conducting interdisciplinary research which uses different types of data and methodologies to better understand environmental phenomena and how the environment and people interact (e.g., Mascia et al., 2003: Agrawal & Ostrom, 2006). Calls are also being made for more cross level and cross scale research (Soranno et al., 2014). Cross-level research refers to research which looks to explore interactions among levels within a scale (e.g., interactions between local, regional, and national levels within the spatial scale, which can influence response variables) (Cash et al., 2006). Cross-scale research refers to research which considers interactions across different scales, for example, between spatial domains and jurisdictions (Cash et al., 2006). Cross level and cross scale research are important because many environmental problems are complex and cannot be adequately addressed by being viewed through a single lens. Understanding cross-scale and cross-level interactions are necessary to help better predict likely outcomes of alternative management strategies. Ignoring cross-scale and cross-level interactions may lead to errors of extrapolation from one region to another.

Despite the importance of cross-level and cross scale research it is rarely done. Cash et al., (2006) note three key barriers to implementing cross scale and cross level research, these are: "ignorance", "mismatch" and "plurality". Issues of ignorance relate to the fact that the dynamics of the human-environment system at even just one level or scale may be so complex that attempting to understand or influence cross-level and cross-scale interactions may be interactions may be extraordinarily difficult. Issues of mismatch occur because there are differences in the scale of what we know about the world and the scale at which decisions are made and taken (Kates et al. 2001). For example, large-scale scientific knowledge (e.g., satellite imagery) can be at a resolution that can have little relevance to local decision making. Similarly, due to issues of confidentiality, census data may be aggregated as too large of a scale to reliably determine social impacts of interventions on an individual or household level. Additionally, local, tacit or indigenous knowledge that is not seen as credible may be disregarded (Gadgil et al., 2003; Berkes, 2002) and research using this data is less likely to be funded because there is a prevailing discourse that big data is 'better'. This narrative implies that qualitative research has only a limited, or no role, in advancing the analytical practices that big data offers (Davidson et al., 2018). The issues of plurality arise out of the incorrect assumption that there is a single, correct, or best characterisation of the scale and level challenge that applies to the system as a whole or for all actors. The drive to frame issues at a single level comes from the need to both simplify and control. For example, governments may frame environmental issues as solely "national issues" so that they become tractable within their jurisdictions (Lebel et al., 2006)

In Chapter 4 I "levelled up" existing studies on the spatial distributions of PPAs and in Chapter 5 I "levelled down" studies into the effectiveness of PPAs to combat deforestation. In Chapter 4 I also used a cross-scale research design looking at the spatial distribution of PPAs across different jurisdictions (i.e., different countries). Moreover, whilst in this thesis I have exclusively used a big data approach, I have drawn upon smaller case studies to help guide and design the research aims, questions and methods and to inform the interpretation of the results. Within chapter 3 I mostly reviewed studies which were conducted on a sub-national scale and used the findings of these studies to shape and influence the design of chapters 4 and 5. In chapter 4, I "levelled up" smaller scale studies which investigated the spatial distribution of PPAs at sub-regional, regional and national scales (e.g. De Vos & Cumming, 2019; Shanee et al., 2017; Gallo et al., 2009) to determine the contributions of PPAs at an international level. My international level analysis compliments smaller scale studies by determining if the trends seen in specific localised conditions are applicable across broader scales. Within this chapter I also I conducted cross-scale research by investigating the spatial distribution of PPAs across different jurisdictions (i.e., different countries). The benefit of this research is that it can explore if there is something unique about the political, social, and economic context of certain locations which produce different results to other areas of interest. My research found that across multiple levels (e.g., regional, national, international) the contributions of PPAs to complement the representativeness of PA networks holds true. This is despite different motivators and drivers of PPAs at different levels and in different countries. However, contributions towards connectivity declines as spatial scales (regional to national) increase.

In chapter 5 I assessed the effectiveness of PAs to reduce deforestation using a counter-factual matching approach and remote sensing data. Most studies using these methods have assessed the difference in effectiveness of different PA groups (e.g., different governance types: Schleicher et al 2017, different management regimes: Notle et al., 2013). Fewer studies (e.g., Koskimaki et al., 2021; Zhao et al., 2019) individually assess the outcomes of PAs. Here I "levelled down" existing studies into the effectiveness of PAs in Colombia by conducting an individual assessment of the effectiveness of PPAs, regional state and national state PAs in the country. An individual assessment of PAs is important because as my study shows, there is high heterogeneity on the effectiveness of PAs within PA groups. My research demonstrates that remote sensing research can be used to identify potential study subjects for on-the-ground research at the local level to clarify what the ingredients of a successful PA are.

6.6 Recommendations for policy and practice

6.6.1 PPAs deserve better integration into national biodiversity conservation planning

Despite the potential benefits of PPAs, as highlighted within my thesis, in many countries PPAs are not yet included within national biodiversity conservation strategies (Bingham et al., 2017; Stolton et al., 2014). My findings promote the beneficial opportunities afforded by PPAs and that they deserve to be better integrated in national and global strategies to achieve post 2020 global biodiversity targets. Better integration into national policies could even help PPAs achieve even better social outcomes. For example, binding reporting requirements to government funding on private land, such as with the Farm Bill in the USA, the accountability of PPAs owners could be increased (Rissman et al., 2007). Better integration into national and global conservation strategies could be achieved by (i) creating frameworks for the inclusion of PPAs into national conservation strategies, (ii) the creation of more PPA support networks (such as RESTNAUR in Colombia) and (iii) countries being supported and encouraged to record and report PPA boundaries at the national level and to the WDPA.

6.6.2 Better recognition of the diversity of PPAs

As outlined in Chapter 1 Section 1.4 – Private Governance, there has been much scepticism and critiques regarding the social impacts of PPAs and PPAs are associated with neo-colonialism, elitism, and land grabbing (Ramutsindela et al., 2011; Gooden & Sas-Rolfes, 2019). However current literature and debate treats PPAs as a homogenous group, but this is far from accurate. Chapter 3 shows that PPAs are highly diverse in terms of their form, ownership, management, and purpose and can have a variety of different social impacts (both positive and/or negative) for different stakeholders. In particular, Chapter 3 shows that PPAs can have positive impacts for landowners. In some cases, these landowners may be affluent individuals who are not native to the area in which they are establishing a PPA (upon which most of the criticism around PPAs is centred) however in other cases land owners are local small holder farmers. Moreover, just because a PPA is governed by an affluent international, is does not automatically mean that it has negative social impacts for the local communities. Take for example Chumbe Island Coral Park Ltd, a privately established and managed marine park in Zanzibar, Tanzania. This PPA has received multiple awards not only for environmental effectiveness by also for being equitably governed and providing positives benefits for the local surrounding community (Dodds, 2012). My research highlights the need for a more nuanced discussion surrounding what are the outcomes of PPAs taking in account the different purposes for PPAs establishment and who are the main stakeholders. More broadly, my research suggests that debates regarding PA outcomes need to shift in focus from “who” is governing the PA to “how” it is being governed.

It may prove difficult to have these discussions at present as we currently lack data on who governs PPAs, for what purpose and how benefits (if any) are shared. Findings from this thesis support the need for better recording of (i) how PPAs operate as institutions (e.g., who are PPA stakeholders, what are the distributions of power and agency between different stakeholders, and to whom are the stakeholders accountable); and (ii) how the outcomes of a PPAs shared among stakeholders and the local communities’ surroundings PPAs.

6.6.3 Better data on PPAs

There are many reasons why PPA boundaries are not reported, and these vary across different countries. These can include concerns over privacy (e.g., Australia) (Fitzwilliams, 2015), failure of nations or regionally authorities to legally recognise PPAs (e.g., Canada and

Spain) (Wilkinson, 2014; Rafa, 2014), and technical issues such as low data management capacity and poor co-ordination between various groups at the national level. In many countries the compilation of national reporting is still based on methods of manual documentation rather than through semi-automated data management systems (Joppa et al., 2016). Lack of access to information about PPA can hinder conservation planning, the measuring of PPA outcomes and research (Gooden & Sas-Rolfes, 2019). Systematic conservation planning requires knowledge on which land are already protected, and the characteristics of those lands, to make the most effective decisions about future purchases (Rissman et al., 2007). A lack of information on the location of PPAs hampers the ability of conservation planners to design the most effective PA network possible with limited resources. It is also desirable to have complete datasets for outcomes measurement to identify outcomes with certainty (Fitzsimons and Carr, 2014). Within this thesis I have attempted to create the best dataset of PPA boundaries available by using multiple PPA boundary sources, however the dataset is incomplete. This is because only a small proportion of countries report PPAs to the WDPA and these may also report only a subset of existing PPAs (Fitzsimons, 2015; Bingham et al. 2017). Therefore, the findings of this research act as a best guess for the contributions and effectiveness of PPAs. A lack of information on PPAs also reduces the transparency of PPAs and the extent to which they can be held to account. Holding PPA to account is important because they may have negative consequences for the environment or local communities and in some countries (e.g. the USA) PPAs are incentivised through the government via tax reductions and therefore there is a need to determine if these public funded interventions are offering value for money (Mitchell et al. 2017). The reporting of PPA boundaries could be improved by several measures including (i) better legal recognition of PPAs, (ii) more equitable and transparent process for data handling and (iii) bottom-up support for PPA networks who can share knowledge and expertise on spatial mapping techniques. For improved reporting of PPA outcomes, in countries where PPA establishment is in part funded by the tax payer, PPAs could be better held to account by binding reporting requirements to government funding for conservation of private land, as is the case with the Farm Bill in the US (Rissman et al., 2017). For ecotourism reserves, reporting and measuring of PPA outcomes could offer benefits of recognised awards or certifications.

6.7 Future research directions

Future research can build on the methodological and empirical insights from this thesis, and throughout this discussion chapter there have been several research questions and gaps highlighted by the results from this thesis.

6.7.1 Spatial contributions of PPAs

Whilst this study used several metrics to assess the representativeness of PPAs to the global conservation estate it was not exhaustive, and it would be of value to determine the representativeness of PPAs to other metrics of interest. This may include to what extent PPAs protect threatened species, various ecosystem services (e.g., forest carbon stocks, non-timber forest products or freshwater ecosystem services or productive fisheries (Neugarten et al., 2020; Maxwell et al., 2020), or geodiversity. As the conservation community develops protected area targets post 2020 these studies could guide and inform the role and contribution that PPAs can play in achieving these new and ambitious targets.

6.7.2 Effectiveness of PPAs

Like previous studies (e.g., Shumba et al., 2020; Schleicher et al., 2017) this study has shown that PPAs can be effective at reducing deforestation. Whilst this is an important finding and this study is one of only a few studies into the effectiveness of PPAs, like Schleicher et al., (2017) it is also located in Latin America. It would be interesting to assess the effectiveness of PPAs in other regions of the world that may have different legal statuses of PPAs, drivers for their creation and ideas around the legitimacy of private land ownership.

Studies into the impacts of governance of PA outcomes mostly focus on only one, specifically ecological, type of outcome; land cover change studies that focus on deforestation rate only (and sometimes forest degradation) (Macura et al., 2015). While the lack or presence of deforestation is a clear representation of PA impact and it is informative of the condition of the environment and its threats, it should not be considered as the sole determinant of PA effectiveness. There are several other are measures that could also be used to determine the effectiveness of PPAs. This may include to what extent PPAs reduce smaller anthropogenic disturbances such as forest fires and selective logging? Prevent development? And to what extent they increase ecological restoration? Studies into the effectiveness of PPAs to protect and conserve carbon stocks and biodiversity are especially timely given the international spread of initiatives aimed at reducing carbon emissions from deforestation and degradation (REDD) and more recently plans for biodiversity off setting.

6.7.3 Social impacts of PPAs

Increasingly attention is being paid not only to the environmental outcomes of PAs but also their social outcomes. It is important to assess social outcomes because it is now accepted that PAs should be aware of and aim to avoid potential negative effects of PAs to local communities. Where negative effects (e.g., displacement, restriction of access to resources) is unavoidable, suitable compensation and mitigation should be put in place. Social outcomes of PPAs can determine their legitimacy and the level of support they receive from local communities and therefore their long-term persistence and effectiveness in achieving the biodiversity conservation goals they were meant to achieve. Social outcomes for owners are also important for the longevity and number of PPAs. Whilst chapter 3 found that PPAs have mixed social outcomes, due to time constraints, the empirical research chapters (4 & 5) of this thesis focused solely on the environmental outcomes of PAs. Future research should be conducted to investigate the impacts of PPAs on all components of multi-dimensional poverty and how this compares with other PA governance types. Below I review the ways that previous social impact research has been conducted and how I would suggest this type of study should be done.

Multiple qualitative and quantitative studies have been implemented to assess PA impacts across the world (Pullin et al., 2013). However, many studies faced severe limitations. Most studies have only considered a single site at a single point in time and generally did not record pre-intervention levels, did not include a control group to compare the impact of the intervention against, and did not randomise allocation of the intervention (Baylis et al., 2015). Approaches such as these put emphasis on the contextual and place-based impacts, but these impacts may not be representative of impacts in other areas or conditions. As a result, their findings are mostly unsuitable for generalisation and are unable to serve as a basis for incorporation into broad policy interventions (Miteva et al., 2012;

Agrawal & Redford, 2006). Such approaches can also exaggerate PA impacts. Study designs such as before-after-control-impact or randomised control trials that incorporate pre-intervention sampling, counterfactuals and/or random allocation result in more accurate effect estimations (Christie et al., 2019). However, these approaches can be expensive, infeasible, or even unsuitable for many research questions in conservation and development (Deaton, 2009), and more specifically for PPA assessments.

One way to overcome these challenges, which is increasingly being used within conservation and development, is to use a quasi-experimental econometric design (like in chapter 5 of this thesis). Just like in studies that explore the environmental impacts of PAs (e.g., deforestation), factors that may affect the social outcomes of people living nearby to a PA are controlled for. These can include biophysical characteristics, measures of accessibility (e.g., distance to markets or roads) and socioeconomic features (e.g., baseline level of poverty before intervention, population density). The response variable studied (like forest cover loss in Chapter 5) could be changes in household health, education, living standards or household income. Recently, studies have combined a variety of different factors to create multi-dimensional poverty indices (den Braber et al., 2018). Due to confidentiality, it can be very difficult or impossible to get data at the household level and so a common unit of analysis in these studies is the census tract. Example studies into social impacts of PAs using quasi-experimental econometric designs include Naidoo, et al., (2019); den Braber et al., (2018), Andam et al., (2010). As I as I am aware, no study has been conducted to individually assess the impacts of PAs on social outcomes. Lessons learned through chapter 5 of this thesis could be used as a basis for such a study.

6.7.4 Mechanisms

To develop effective PAs, researchers and practitioners must better understand the mechanisms through which PA affect environmental and social outcomes. A mechanism is an outcome that, once affected by the treatment, affects the final outcome of interest. With strong evidence about the casual mechanisms of PA impacts, the key elements of success can be strengthened, and the elements of failure can be addressed (Ferraro & Hanauer, 2015). Unfortunately, empirical evidence about the effect of casual mechanisms on PA impacts is limited. I am aware of only one study which investigates the casual mechanism through which PA deliver impacts. This study investigated the casual what effect the impacts of tourism and recreational services, changes in roadless volume induced by protection area establishment and changes in forest cover caused by protection had on poverty in Costa Rica (Ferraro & Hanauer, 2014). This study asked a more elaborate question than that of studies like chapter 5 which just look to assess "what would have happened had the protection not have occurred?". Instead, it asked "What would have happened had the study area been exposed to protection, but protection had not affected the mechanisms?". Whilst chapters 4 & 5 in thesis have postulated reasons for why PPA have the outcomes shown within this chapters, further research could be undertaken to empirically evaluate these and other possible theories. This could be done using the study design of Ferraro & Hanauer, (2014). Future studies also need to be conducted which investigate under which conditions impacts are moderated by exogenous variables (e.g., Bonilla-Mejia & Higuera-Mendieta, 2019; Ferraro et al., 2018; Hanauer & Canavire-Bacarreza, 2015).

6.7.5 Other Area Based Conservation Measures

Methods used in this thesis can also be applied to determine the contributions of other forms of other area-based conservation measures (e.g., Integrated Community Conserved Areas or Other Effective Conservations measures).

6.7.6 Can PPAs be classified as being under "good governance"?

This thesis has assessed how does *who* governs PAs affect their outcomes and indirectly assess how (effectively) PAs are being governed and if they can be deemed as being under "good governance". More research should be conducted to directly assess how PPAs are governed (i.e., who has power, who is involved in decision making, how are benefits / costs shared), if these factors differ to those of other PA types, and how this is associated with PA outcomes. PAs could be assessed using the newly developed Site-level assessment of governance and equity (SAGE) tool developed by the International Institute for Environment and Development (IIED) and results compared across PA governance types.

6.8 Conclusion

This thesis sought to advance our understanding of the outcomes of PPAs. In doing so, it adds to the growing body of literature examining how actors outside of the state can contribute to achieving conservation biodiversity targets. This knowledge gap on the outcomes of conservation initiatives outside of the state is one that urgently needs to be addressed, as we see policy targets balloon and the types of designations envisioned to help meet these targets diversify. This thesis considered what PPAs protect, how effective they are protecting it and the social outcomes associated with PPA establishment. A big data approach enabled this thesis to assess PPA outcomes at larger scales than have been assessed previously and in doing so can provide generalised statements regarding the outcomes of PPAs across a broad range of contexts. It also demonstrates how remote sensing research can be used to identify potential study subjects for on-the-ground research at the local level to clarify what are the ingredients or a successful PA.

Results from this thesis provided new empirical evidence into the contributions of PPAs to the global conservation estate and has helped answer a gap in our understanding of how PA governance can affect PA outcomes. This thesis demonstrated that PPAs can make unique and significant contributions to achieving international biodiversity coverage targets. This finding is timely as we see discussions in the wider conservation literature about 30 x 30 (i.e., the recent 30 x 30 target to how 30% of all land and sea under some form of protection by 2030), and the half-earth whole earth debate (which strives to have 50% of the world protected). My research suggests that PPAs can protect areas where other PA governance types struggle to be implemented, in particular unrepresented and threatened biomes and areas of high human pressure. They also protect 1.2% of Key Biodiversity Areas and improve national PA network connectivity on average by 7%. This thesis also shows that on average PPAs are effective at reducing deforestation however, there is high heterogeneity of the effectiveness of PPAs, and some can experience high levels of deforestation (deforestation rate range: 0% - 29.4%, average 0.89%). This thesis shows that on average PPAs are more effective than regional state PAs at reducing deforestation however, they are not as effective as strictly protected national PAs. Lastly, this thesis shows that the social outcomes of PAs are mixed, and not dissimilar to those of other PA governance types. PPAs can also have positive social impacts for landowners by improved social networks, increased property value, or a reduction in taxes. However, similarly to state PAs in some cases local communities surrounding PPAs may incur costs, including a reduction in social capital and loss of cultural identity.

This thesis contributes to wider debates on the roll back of the state and increase in neoliberal approaches and private actors in supplying public services. Whilst many are apprehensive about the outsourcing of functions traditionally performed by governments to the private sector, this thesis has shown that PPAs can have both environmental and social benefits and can outperform state actors in some circumstances. We also show that PPAs can have negative social outcomes and not and not always effective. However, this is also the case for PAs under other governance types. My research therefore suggests that the conservation needs to move away from a simple dichotomy of “state is good” and “private is bad” to a more nuanced discussion of how PA governance affects PA outcomes. This discussion needs to focus more on what the motivations are for why a certain PA has been established are and who is responsible for governing individual PAs.

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Chapter 7: Appendices

Appendix A: Format of data submitted to the WDPA

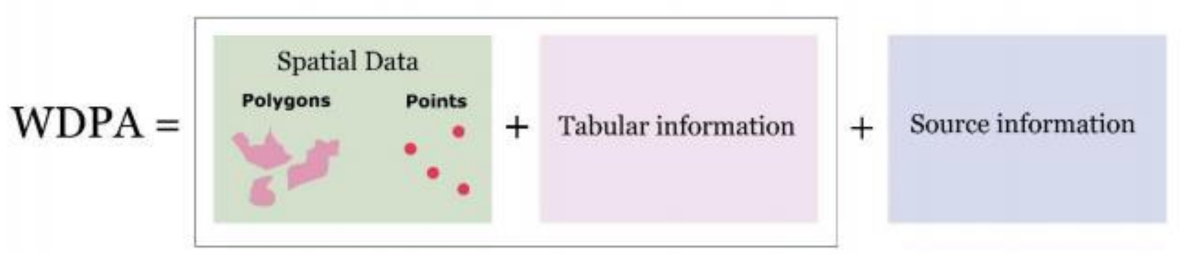

Figure A.1 Structure of the World Database of Protected Areas

Table A.1 Summarised description and allowed values for the WDPA attributes (Version 1.4) (Sourced from UNEP-WCMC, 2016)

No	Requirement	Provided by	Field Name	Type	Length	Accepted values
1	Minimum	UNEP-WCMC	WDPAID	Number (Double)	N/A	Assigned by UNEP-WCMC. Unique identifier for a protected Area.
2	Minimum	UNEP-WCMC	WDPA_PID	Text (String)	52	Assigned by UNEP-WCMC. Unique identifier for parcels or zones within a protected area.
3	Minimum	Data provider	PA_DEF	Text (String)	20	**Allowed values:** 1 (meets IUCN and/or CBD PA definition); 0 (does not meet IUCN and/or CBD PA definition (currently stored outside WDPA)).
4	Minimum	Data provider	NAME	Text (String)	254	Name of the protected area (PA) as provided by the data provider.
5	Minimum	Data provider	ORIG_NAME	Text (String)	254	Name of the protected area in original language.
6	Minimum	Data provider	DESIG	Text (String)	254	Name of designation.
7	Complete	Data provider	DESIG_ENG	Text (String)	254	Designation in English. **Allowed values for international-level designations:** Ramsar Site, Wetland of International Importance; UNESCO-MAB Biosphere Reserve; World Heritage Site. **Allowed values for regional-level designations:** Baltic Sea Protected Area (HELCOM); Specially Protected Area (Cartagena Convention); Marine Protected Area (CCAMLR); Marine Protected Area (OSPAR); Site of Community Importance (Habitats Directive); Special Protection Area (Birds Directive); Specially Protected Areas of Mediterranean Importance (Barcelona Convention). No fixed values for protected areas designated at a national level.
8	Minimum	Data provider	DESIG_TYPE	Text (String)	20	**Allowed values:** National, Regional, International, Not Applicable
9	Complete	Data provider	IUCN_CAT	Text (String)	20	**Allowed values:** Ia, Ib, II, III, IV, V, VI, Not Applicable, Not Assigned, Not Reported
10	Minimum	UNEP-WCMC	INT_CRIT	Text (String)	100	Assigned by UNEP-WCMC. For World Heritage and Ramsar sites only.
11	Minimum	Data provider	MARINE	Text (String)	20	**Allowed values:** 0 (100% Terrestrial PA), 1 (Coastal: marine and terrestrial PA), and 2 (100 % marine PA).
12	Minimum	Data provider	REP_M__AREA	Number (Double)	N/A	Marine area in square kilometers.
13	Minimum	UNEP-WCMC	GIS_M_AREA	Number (Double)	N/A	Assigned by UNEP-WCMC.
14	Minimum	Data provider	REP_AREA	Number (Double)	N/A	Area in square kilometers.
15	Minimum	UNEP-WCMC	GIS_AREA	Number (Double)	N/A	Assigned by UNEP-WCMC.
16	Complete	Data provider	NO_TAKE	Text (String)	50	**Allowed values:** All, Part, None, Not Reported, Not Applicable (if Marine field = 0).

No	Requirement	Provided by	Field Name	Type	Length	Accepted values
17	Complete	Data provider	NO_TK_AREA	Number (Double)	N/A	Area of the no-take area in square kilometers
18	Minimum	Data provider	STATUS	Text (String)	100	**Allowed values:** Proposed, Inscribed, Adopted, Designated, Established.
19	Minimum	Data provider	STATUS_YR	Number (Long Integer)	12	Year of enactment of status (STATUS field).
20	Complete	Data provider	GOV_TYPE	Text (String)	254	**Allowed values:** Federal or national ministry or agency, Sub-national ministry or agency, Government-delegated management, Transboundary governance, Collaborative governance, Joint governance, Individual landowners, Non-profit organisations, For-profit organisations, Indigenous peoples, Local communities, Not Reported.
21	Complete	Data provider	OWN_TYPE	Text (String)	254	**Allowed values:** State, Communal, Individual landowners, For-profit organisations, Non-profit organisations, Joint ownership, Multiple ownership, Contested, Not Reported.
22	Complete	Data provider	MANG_AUTH	Text (String)	254	Individual or group that manages the protected area.
23	Complete	Data provider	MANG_PLAN	Text (String)	254	Link or reference to the protected area's management plan.
24	Minimum	UNEP-WCMC	VERIF	Text (String)	20	Assigned by UNEP-WCMC. Fixed values: State Verified, Expert Verified, Not Reported (for unverified data that was already in the WDPA prior to the inclusion of the 'Verification' field).
25	Minimum	UNEP-WCMC	RESTRICT	Text (String)	20	Not publicly available, for UNEP-WCMC use only.
26	Minimum	UNEP-WCMC	METADATAID	Number (Long Integer)	12	Assigned by UNEP-WCMC. Link to source table.
27	Complete	Data provider	SUB_LOC	Text (String)	100	**Allowed values:** ISO 3166-2 sub-national code where the PA is located.
28	Minimum	Data provider	PARENT_ISO3	Text (String)	20	**Allowed values:** ISO 3166-3 character code of country where the PA is located.
29	Minimum	Data provider	ISO3	Text (String)	20	**Allowed values:** ISO 3166-3 character code of country or territory where the PA is located.

Appendix B: Articles Reviewed for Chapter 3

Papers reviewing the environmental or social impacts of PPAs have been highlighted as:

● Environmental Impacts Evaluated

♦ Social impacts of PPAs Evaluated

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Appendix C: Supplementary Tables and Figures from Chapter 4

Table A.3 Area protected by PAs and percent share of the total area protected for case study countries

Country	Area Protected PPAs (Km²)	Protected area protected by PPAs (%)	Area protected by state PAs (Km²)	Protected area protected by state PAs (%)	Area protected by co-managed PAs (Km²)	Protected area protected by co-managed PAs (%)	Area protected by community governed protected area (Km²)	Protected area protected by community governed PAs (%)	Land protected by any type of PA (%)
Australia	87,350	6.4	617,055	45	-	-	666,324	49	18
Belize	1,254	16	4,503	56	2,211	27	139	1.7	35
Brazil	4,207	0.2	1,428,704	59	-	-	988,363	41	28
Canada	225	0.02	903,497	96	40,058	4.2	973	0.10	9
Chile	12,550	9.1	122,848	89	2122	1.5	-	-	18
Colombia	1,220	0.8	154,390	99	-	-	-	-	14
Finland	548	1.1	48,501	99	-	-	-	-	15
Guatemala	3,851	22	12,328	70	1,411	8.02	5	0.03	16
Honduras	13	0.1	12,786	100	-	-	-	-	11
Kenya	3,644	4.4	49,053	60	-	-	29,411	36	14
Mexico	3,365	1.4	244,511	98	507	0.2	-	-	13
Namibia	53,629	15	143,232	40	-	-	163,926	45	44
Peru	29,388	14	177,041	85	80	0.04	2,647	1.3	16
South Africa	24,219	25	72,134	74	-	-	592	0.61	8
USA	21,123	1.8	629,481	54	525,889	45	-	-	12
Total	246,586	3.4	4,620,065	63	572,278	7.8	1,852,381	25	16

Table A.4 KBA analysis

Country	Total Area of KBAs (Km^2)	KBA protected by state PAs (%)	State PAs within KBAs (%)	KBA protected by co-managed PAs (%)	Co-managed PAs area within KBAs (%)	KBA protected by PPAs (%)	PPAs area within a KBAs (%)	KBA protected by community governed PAs (%)	Community governed PAs within KBAs (%)
Australia	459,112	27	20	0.0	0.0	4.9	26	4.1	2.8
Belize	29,990	14	95	7.6	100	4.1	99	0.5	100
Brazil	1,002,749	50	35	-	-	0.1	16	7.7	7.8
Canada	354,756	34	13	0.0	0.4	0.0	3.6	-	0.0
Chile	53,720	60	26	1.5	39	4.5	19	-	-
Colombia	185,593	15	18	-	-	0.1	13	-	-
Finland	28,045	66	38	-	-	0.0	2.2	-	-
Guatemala	54,088	20	87	2.6	98	6.9	98	0	0.0
Honduras	23,852	42	78	-	-	0	1.3	-	-
Kenya	73,393	46	69	-	-	1.0	20	4.2	10
Mexico	416,268	22	38	0.0	32	0.3	41	-	-
Namibia	105,883	91	67	-	-	0	0.0	7.1	4.6
Peru	231,420	60	78	0.0	99	1.4	11	0.4	33
USA	828,124	3.01	4.01	4.6	7.3	0.1	3.3	-	-
South Africa	218,004	30	35	0.3	0.0	5.2	53	-	-
Total	**4,064,997**	**32**	**28**	**1.1**	**1.01**	**1.2**	**20**	**2.6**	**5.8**

Table A.5 Percentage of area in different Human Footprint (HF) categories within PPAs and 1 km, 5 km and 10 km buffers surrounding PPAs

		Percentage of total area with each HF category		
HF	**PPAs**	**1 km**	**5 km**	**10 km**
0	25	8.6	6.5	6.9
1 – 2	28	25	22	30
3 – 5	28	38	38	33
6 – 11	14	20	23	20
12 – 50	5	8	10	9.8

Table A.6 Number and percentage of PPAs within 0 m, 30 m, 100 m and 500 m of a PA under state, shared or community governance regime

Country	No. of PPAs	No. of PPAs within 0 m of a PA under state, community or co-managed governance regime	Percent of PPAs within 0m of a PA under state, community or co-managed governance regime	No. of PPAs within 30 m of a PA under state, community or co-managed governance regime	Percent of PPAs within 30 m of a PA under state, community or co-managed governance regime	No. of PPAs within 100 m of a PA under state, community or co-managed governance regime	Percent of PPAs within 100 m of a PA under state, community or co-managed governance regime	No. of PPAs within 500 m of a PA under state, community or co-managed governance regime	Percent of PPAs within 500 m of a PA under state, community or co-managed governance regime
Australia	1616	444	27	485	30	514	32	614	38
Belize	10	6	60	6	60	6	60	8	80
Brazil	675	66	9.9	72	11	77	11	96	14
Canada	388	0	0.0001	0	0.0001	0	0.0001	0	0.0002
Chile	287	22	7.7	22	7.7	23	8.0	28	9.7
Colombia	580	25	4.3	29	5.0	34	6.0	55	9.4
Finland	4293	716	17	795	19	838	20	1031	24
Guatemala	149	27	18	28	19	29	19	33	22
Honduras	11	0	0.00	0	0.00	1	9.01	1	9.1
Kenya	28	13	46	13	46	14	50	14	50
Mexico	342	10	2.9	11	3.2	13	4	14	4.1
Namibia	27	12	44	12	44	12	44	12	44
Peru	94	10	11	13	14	13	14	16	17
South Africa	927	124	16	166	18	171	19	178	20
USA	8130	461	5.7	562	7.0	652	8	1031	13
Total	**17557**	**1960**	**11**	**2214**	**13**	**2397**	**14**	**3131**	**18**

Table A.7 Percentage of protected connected land including or excluding PPAs and change in protected connected land at a dispersal distance of 10 km for case study countries

Country	Percent of protected connected land excluding PPAs	Percent of protected connected land including PPAs	Percent change in the area of protected connected land at 10 km dispersal
Australia	6.4	6.6	0.9
Belize	27	30	11
Brazil	13	13	0.008
Canada	0.12	0.12	0.0001
Chile	7.1	7.1	1.2
Colombia	0.6	0.6	0.02
Finland	7.2	7.4	2.4
Guatemala	4.6	5.6	23
Honduras	6.0	6.0	0.02
Kenya	6.6	8.5	29
Mexico	3.0	3.0	0.08
Namibia	32	36	12
Peru	5.0	5.3	6.7
South Africa	1.9	2.2	20
USA	0.88	0.88	0.06
Total	**8.64**	**9.4**	**7.05**

Table A.8 Percent change in the area of connected land at 1 km, 10 km, 30 km and 100 km dispersal distances when including PPAs

Country	Percent change in the area of connected land at 1km dispersal	Percent change in the area of connected land at 10km dispersal	Percent change in the area of connected land at 30km dispersal	Percent change in the area of connected land at 100km dispersal
Australia	0.4	0.9	1.3	1.5
Belize	3.8	11	12	12
Brazil	0.006	0.01	0.01	0.3
Canada	0.00001	0.0001	0.0003	0.0003
Chile	0.8	1.2	5.2	9.2
Colombia	0.07	0.02	1.6	1.8
Finland	0.07	2.44	1.02	1.1
Guatemala	9.3	23	24	25
Honduras	0.0002	0.02	0.03	0.07
Kenya	24	29	36	49
Mexico	0.02	0.08	0.2	8.4
Namibia	12	12	12	12
Peru	5	6.7	7.2	20
South Africa	14	20	28	33
USA	0.04	0.06	0.1	1.2
Average	**4.5**	**7.06**	**8.6**	**11.65**

Table A.9 Additional PPA boundaries

Country	No. of additional boundaries found outside of the WDPA	Area (Km^2)	Source	Legitimacy
Australia	60	126	Collaborative Australian Protected Area Database (CAPAD) http://www.environment.gov.au/land/nrs/science/capad/2016	National government led reporting database for Australian protected areas. Every two years information on PAs is collected from state and territory governments and other protected areas managers.
Belize	2	6	Biodiversity and Environmental Resource Data System of Belize (BERDS) http://www.biodiversity.bz/	Non-governmental organizations recording protected areas
Brazil	465	3,568	Chico Mendes Institute for Biodiversity Conservation (ICMBio)	The Chico Mendes Institute for Biodiversity Conservation is a special regime municipality. Created on August 28, 2007, by Law 11,516 , ICMBio is linked to the Ministry of Environment and is part of the National Environment System.
Canada	20	3.6	Conservation Areas Reporting and Tracking System (CARTS) https://www.ccea.org/carts/	A geodatabase which contains data from all federal, provincial and territorial jurisdictions in Canada, which update their protected areas data to CARTS on an annual basis.
Chile	274	12,893	Asociacion de Iniciativas de Conservacion en Areas Privada y de Pueblos Originarios http://asiconservachile.cl/acch/	Network of private land owners across Chile
Colombia	54	48	Parques Nacionales Naturales de Colobmbia (RUNAP)	National government agency which records protected areas
Finland	0	0		

Guatemala	46	125	Consejo Nacional de Areas Protegias (CONAP) http://www.conap.gob.gt/AreasProtegidas.aspx	National Council of Protected Areas which records and tracks protected areas within Guatemala
Honduras	3	2.1	Department of Protected Areas Institute of Forest Conservation (ICF) https://portalunico.iaip.gob.hn/portal/index.php?portal=349	National government agency recording protected areas in Honduras
Kenya	17	2,296	Data obtained from in country expert Board member of The Center for Sustainable Dryland Ecosystems and Societies (CSDES)	
Mexico	9	10	Comisión Nacional de Áreas Naturales Protegidas (CONAP) https://www.gob.mx/conanp	National Government Agency for monitoring of protected areas
Namibia	25	50,786	Namibian Chamber of Environment (NCE) http://www.n-c-e.org/	Non-government organisation that works outside the Namibian government to monitor protected areas
Peru	0	0	None found	
South Africa	63	376	Department of Environmental Affairs https://egis.environment.gov.za/data_egis/data_download/current	National database of protected areas

USA	0	0	https://www.usgs.gov/core-science-systems/science-analytics-and-synthesis/gap	Natioanl database of protected areas
Total	**1,038**	**70,230**		

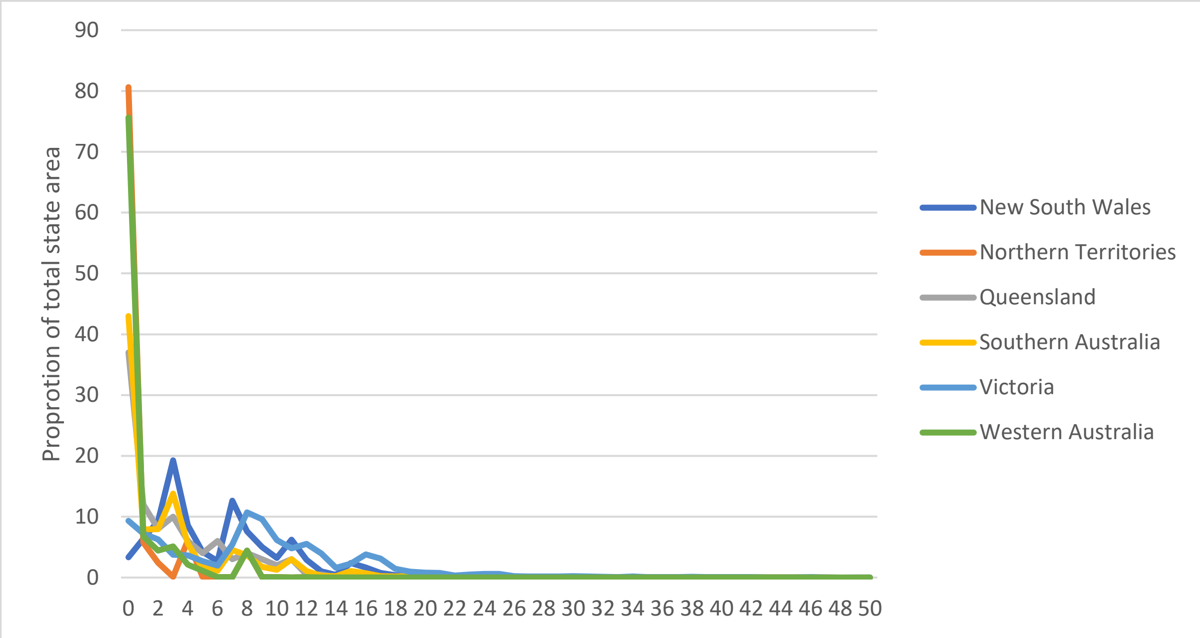

Figure A.2 Proportion of total area of each Australian state within Global Human Footprint Categories

The majority of individual state's area is located in areas with human footprint scores <3 (our cut off point for human induced change). Similarly, a very limited proportion of the total area in each state is located in areas with human footprints >12 (very high human pressure).

Appendix D: Supplementary Tables and Figures from Chapter 5

Appendix D – Sampling Design

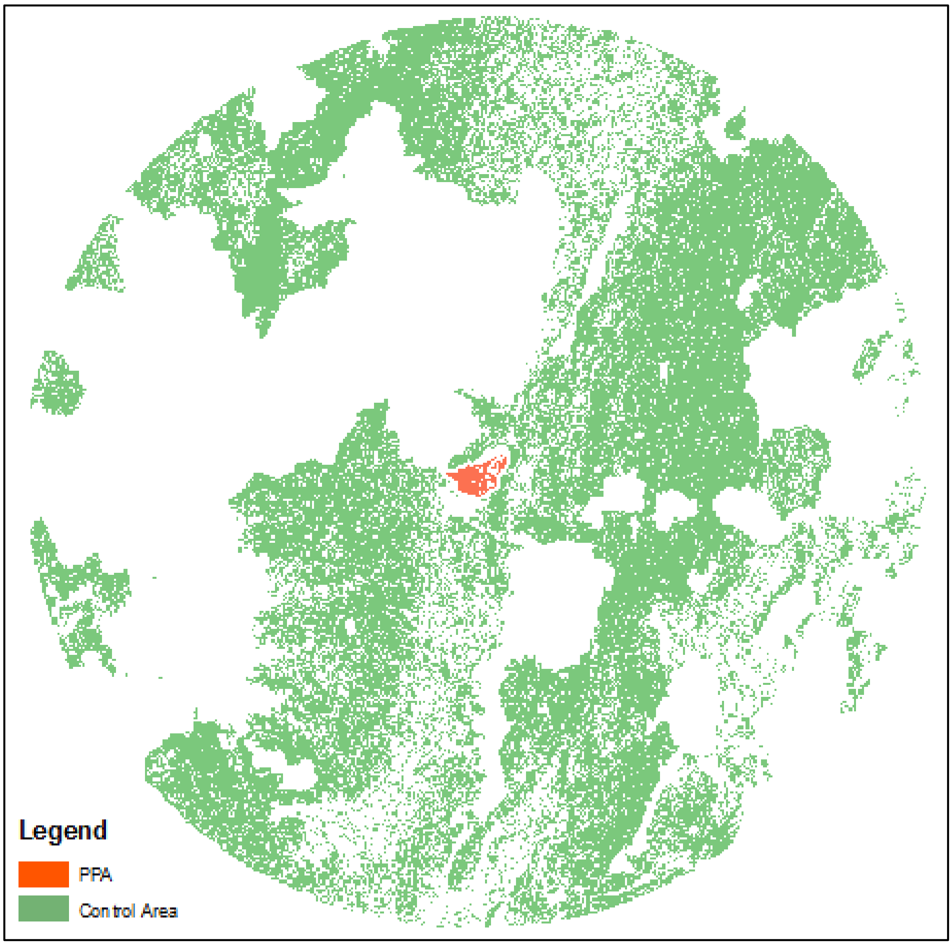

Figure A.3 Example of potential pixels which could be matched for an individual PPA (in red) and the potential pixels which could be used for the control area (in green).

Table A.10 Predictor variables included in the analysis, mapped or resampled (bilinear interpolation) at 30m resolution and reprojected to MAGNA_TRANSVERSE

Predictor variable	**Description**	**Source**
Distance (km) to: (i) Main Roads (ii) Main and vicinal roads	Euclidean distance to (i) main (national) and all roads.	Geographic Institute Agustin Codazzi - IGAC
Distance to main waterways (km)	Euclidean distance to main navigable layers.	Institute of Hydrology, Meteorology and Environmental Studies – IDEAM
Travel time to markets (h)	Estimated time travel to nearest city of at least 50,000 people in 2000, based on population centres, transportation networks, topography, land cover and political boundaries. The accessibility map was provided at 30 arc-seconds resolution and was resampled to 30m.	Global Environment Monitoring Unit (available at: http://forobs.jrc.ec.europa.eu/products/gam)
Distance to forest edge (km)	Euclidean distance to nearest forest edge in 2000 at 30m resolution based on the 2000 forest map from Hansen et al., (2013).	Hansen et al., (2013)
Elevation (m)	Elevation was based on the Shuttle Radar Topographic Mission (STRM) 30m digital elevation data.	STRM30 (available at: http://dwtkns.com.strm30m/)
Slope (degrees)	Determined the slope using the ArcMap’s slope tool, based on the STRM digital elevation data.	STRM30 (available at: http://dwtkns.com.strm30m/)
Rainfall (mm)	Mean annual precipitation data were obtained from the WorldClim Global Climate data (~1950 – 2000) provided at 30 arc-seconds resolution and resampled to 30m resolution.	WORLDCLIM http://worldclim.org/data/worldclim21.html

Population density	Population Density data was obtained from World Pop. Data was provided at 1km resolution and resampled to 30m.	WorldPop (http://worldpop.org),
Land use designations: (i) Regional state PAs (ii) Privately protected areas (iii) All other state PAs (iv) Community & Indigenous PAs	All state PAs were obtained from the World Database of Protected Areas (WDPA) and El Registro Único Nacional de Áreas Protegidas (RUNAP). indigenous reserves and Afro-Colombian lands from the Colombian Geographic Information System for Planning (Sistema de informacion geografica para la planacion y el ordenamiento territorial; SIGOT)	WDPA (Available at: http://www.protectedplanet.net /) RUNAP (available at: http://runap.parquesnacionales.gov.co/). Indigenous reserves and Afro-Colombian lands from the Colombian Geographic Information System for Planning (Sistema de informacion geografica para la planacion y el ordenamiento territorial; SIGOT)

Appendix D - Matching Testing

There were 16 scenarios used for testing which parameters to use within the matching process. The variations were:

- 0m minimum distance between sampled pixels
- 50m minimum distance between sampled pixels
- 0km inner buffer for control area surrounding the PPA
- 1km inner buffer for control area surrounding the PPA
- 25km outer buffer for control area surrounding the PPA
- 50km outer buffer for control area surrounding the PPA
- 100km outer buffer for control area surrounding the PPA
- 500km outer buffer for control area surrounding the PPA

These 16 variations are subsequently referred to as:

- 0km_25km_0m
- 0km_25km_50m
- 1km_25km_0m
- 1km_25km_50m
- 0km_50km_0m
- 0km_50km_50m
- 1km_50km_0m
- 1km_50km_50m
- 0km_100km_0m
- 0km_100km_50m
- 1km_100km_0m
- 1km_100km_50m
- 0km_500km_0m
- 0km_500km_50m
- 1km_500km_0m
- 1km_500km_50m

500 PPA pixels and 250,000 control pixels were requested for each test. Four PPAs were able to provide enough pixels for all 16 variations:

Table A.11 PPAs used in matching testing

Year Established	ID number	Name
2005	555555922	Tulcan Los Canelos 2
2014	555592795	Jabiru
2014	555592802	Hacienda El Triunfo
2014	555636253	Mesetas de Versalles

Matching covariates

The following is a list of all possible matching covariates were available to be selected:

- Access_to_market
- Average_precipitation
- Distance_to_all_roads
- Distance_to_forest_edge
- Distance_to_main_roads
- Distance_to_water
- Elevation
- Population_density
- Slope

For each of the four testing PPAs, all possible combinations of six covariates from this group were tested.

The input data used for testing was 1km_100km_50m, i.e. a minimum of 50m between sampled pixels, with control pixels being sampled from the area between 1km and 100km from the edge of the PPA.

For each combination of the covariates, the Std. Mean Diff. from the matching output was recorded, and the mean of the absolute standard mean difference for the covariates was recorded, and used to work out which combination of covariates produced the best matches, with lower values indicating better matches.

For each of the four PPA test areas, the combination of covariates which produced the best matches:

2005, 555555922, Tulcan Los Canelos 2

access_to_market + average_precipitation + distance_to_all_roads + distance_to_water + elevation + slope

2014, 555592795, Jabiru

distance_to_all_roads + distance_to_forest_edge + distance_to_main_roads + elevation + population_density + slope

2014, 555592802, Hacienda El Triunfo

access_to_market + distance_to_main_roads + distance_to_water + elevation + population_density + slope

2014, 555636253, Mesetas de Versalles

access_to_market + average_precipitation + distance_to_all_roads + elevation + population_density + slope

Overall

To work out which combination of covariates produced the best matches overall, the mean of the mean standard mean difference for each combination of covariates was compared, and the following combination produced the lowest average values:

access_to_market + distance_to_main_roads + distance_to_water + elevation + population_density + slope

Matching tests

For each of the four tests PPA area, or each of the 16 variations of input data, the *MatchIt* matching process was run, using the following method:

matched <- matchit(group ~ access_to_market + distance_to_main_roads + distance_to_water + elevation + population_density + slope', data=matching_input, method='nearest', distance='glm',)

This method will perform 1:1 matching and find a matching control pixel for each of the PPA pixels.

For each test, Moran testing was performed where possible, using functions from both the spdep and ape libraries, to check for spatial autocorrelation of the value of forest cover the year in which the PPA was established plus five years for all matched pixels and also separately for the PPA pixels and the control pixels.

Results

Standard Mean Difference

The standard mean difference values for the matched pixels.

Table A.12 Standard Mean Difference values for 555555922 (2005)

	matched_smd_min	matched_smd_max	matched_smd_mean
0km_25km_0m	0.0001	0.21	0.06
0km_25km_50m	0.00007	1.19	0.41
1km_25km_0m	0.001	0.89	0.33
1km_25km_50m	0.0004	0.90	0.32
0km_50km_0m	0.00003	..56	0.54
0km_50km_50m	0.00004	2.09	0.78
1km_50km_0m	0.00002	2.18	0.77
1km_50km_50m	0.00003	2.20	0.78
0km_100km_0m	0.00006	0.71	0.28
0km_100km_50m	0.00007	2.21	0.48
1km_100km_0m	0.0002	0.69	0.18
1km_100km_50m	0.00007	0.71	0.25
0km_500km_0m	0.003	1.69	0.54
0km_500km_50m	0.003	1.39	0.38
1km_500km_0m	0.003	2.30	0.52
1km_500km_50m	0.003	1.40	0.50

Table A.13 Standard Mean Difference values for 555592795 (2014)

	matched_smd_min	matched_smd_max	matched_smd_mean
0km_25km_0m	0.002	0.21	0.09
0km_25km_50m	0.002	0.16	0.05
1km_25km_0m	0.003	0.36	0.13
1km_25km_50m	0.003	0.28	0.13
0km_50km_0m	0.01	0.52	0.19
0km_50km_50m	0.06	0.51	0.17
1km_50km_0m	0.02	1.34	0.36
1km_50km_50m	0.01	1.14	0.34
0km_100km_0m	0.04	0.33	0.13
0km_100km_50m	0.03	0.28	0.12
1km_100km_0m	0.04	0.63	0.20
1km_100km_50m	0.05	0.56	0.18
0km_500km_0m	0.02	0.60	0.22
0km_500km_50m	0.01	1.16	0.27
1km_500km_0m	0.0005	1.15	0.28
1km_500km_50m	0.01	0.44	0.17

Table A.14 Standard Mean Difference values for 555592802 (2014)

	matched_smd_min	matched_smd_max	matched_smd_mean
0km_25km_0m	0.00003	0.08	0.3
0km_25km_50m	0.000005	0.06	0.04
1km_25km_0m	0.0003	0.12	0.06
1km_25km_50m	0.00009	0.14	0.05
0km_50km_0m	0.00004	0.11	0.04
0km_50km_50m	0.00007	0.10	0.05
1km_50km_0m	0.0001	0.14	0.04
1km_50km_50m	0.0002	0.12	0.04
0km_100km_0m	0.00001	0.09	0.06
0km_100km_50m	0.00002	0.22	0.09
1km_100km_0m	0.00001	0.08	0.03
1km_100km_50m	0.00003	0.09	0.04
0km_500km_0m	0.0001	0.14	0.06
0km_500km_50m	0.00007	0.14	0.06
1km_500km_0m	0.00007	0.17	0.07
1km_500km_50m	0.00008	0.23	0.07

Table A.15 Standard Mean Difference values for 555636253 (2014)

	matched_smd_min	matched_smd_max	matched_smd_mean
0km_25km_0m	0.00001	0.59	0.13
0km_25km_50m	0.000009	1.10	0.22
1km_25km_0m	0.000003	0.53	0.13
1km_25km_50m	0.00002	0.70	0.12
0km_50km_0m	0.000005	0.52	0.18
0km_50km_50m	0.00002	1.63	0.48
1km_50km_0m	0.000000006	0.26	0.11
1km_50km_50m	0.00001	0.28	0.11
0km_100km_0m	0.00002	0.88	0.28
0km_100km_50m	0.00002	0.39	0.15
1km_100km_0m	0.00002	1.32	0.43
1km_100km_50m	0.00004	0.27	0.12
0km_500km_0m	0.00008	1.71	0.45
0km_500km_50m	0.00003	0.96	0.19
1km_500km_0m	0.00002	1.20	0.52
1km_500km_50m	0.00005	1.52	0.32

All

Table A.16 Mean of standard mean difference values across the 4 PPAs

	matched_smd_min	matched_smd_max	matched_smd_mean
0km_25km_0m	0.0006	0.27	0.08
0km_25km_50m	0.0004	0.63	0.18
1km_25km_0m	0.001	0.47	0.16
1km_25km_50m	0.0008	0.50	0.16
0km_50km_0m	0.003	0.68	0.24
0km_50km_50m	0.02	1.08	0.37
1km_50km_0m	0.005	0.38	0.32
1km_50km_50m	0.002	0.94	0.32
0km_100km_0m	0.01	0.50	0.19
0km_100km_50m	0.007	0.78	0.21
1km_100km_0m	0.01	0.68	0.21
1km_100km_50m	0.01	0.41	0.14
0km_500km_0m	0.01	1.04	0.32
0km_500km_50m	0.01	0.91	0.24
1km_500km_0m	0.008	1.20	0.35
1km_500km_50m	0.01	0.90	0.27

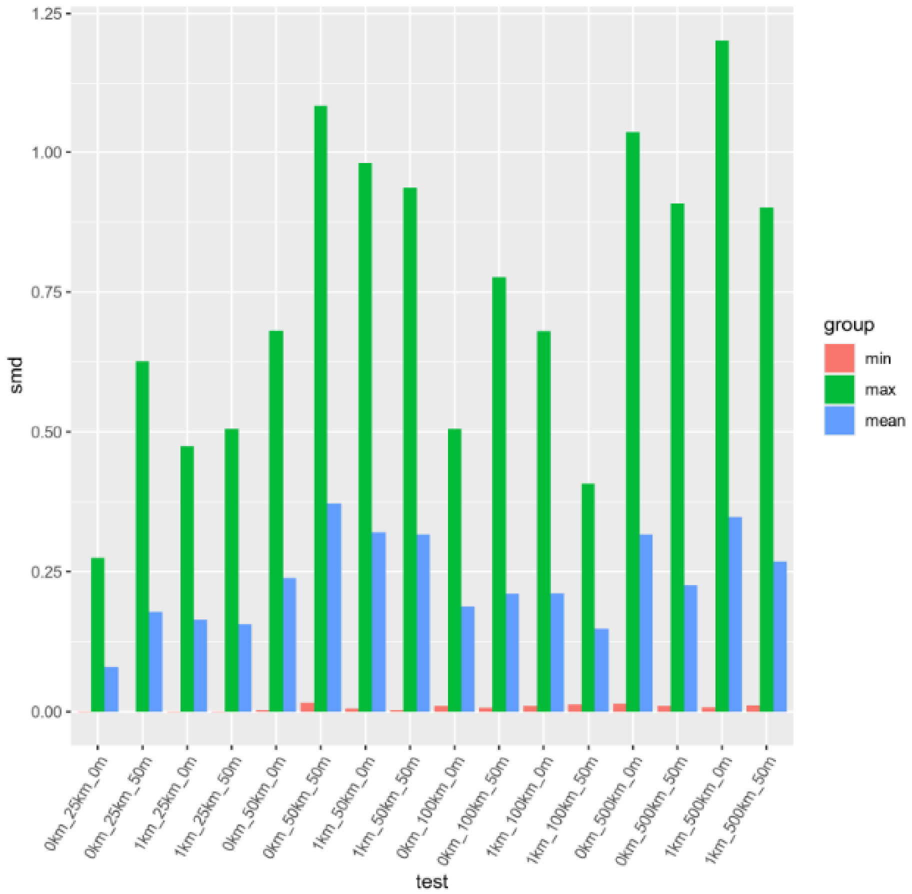

Figure A.4 Mean of standard mean difference values across the 4 PPAs

Deforestation

The number of pixels which had been deforested in the year of PPA establishment plus five years.

Table A.17 Deforestation values for 555555922 (2015)

	All _deforested	PPA_deforested	Control_deforested
0km_25km_0m	2	0	2
0km_25km_50m	1	0	1
1km_25km_0m	0	0	0
1km_25km_50m	2	0	2
0km_50km_0m	3	0	3
0km_50km_50m	3	0	3
1km_50km_0m	2	0	2
1km_50km_50m	2	0	2
0km_100km_0m	2	0	2
0km_100km_50m	1	0	1
1km_100km_0m	1	0	1
1km_100km_50m	5	0	5
0km_500km_0m	2	0	2
0km_500km_50m	2	0	2
1km_500km_0m	1	0	1
1km_500km_50m	1	0	1

Table A.18 Deforestation values for 555592795 (2014)

	All _deforested	PPA_deforested	Control_deforested
0km_25km_0m	13	4	9
0km_25km_50m	11	3	8
1km_25km_0m	5	4	1
1km_25km_50m	3	3	0
0km_50km_0m	16	4	12
0km_50km_50m	12	3	9
1km_50km_0m	4	4	0
1km_50km_50m	3	3	0
0km_100km_0m	7	4	3
0km_100km_50m	7	3	4
1km_100km_0m	6	4	2
1km_100km_50m	5	3	2
0km_500km_0m	14	4	10
0km_500km_50m	23	3	20
1km_500km_0m	12	4	8
1km_500km_50m	12	3	9

Table A.19. Deforestation values for 555592802 (2014)

	All _deforested	PPA_deforested	Control_deforested
0km_25km_0m	11	6	5
0km_25km_50m	10	6	4
1km_25km_0m	9	6	3
1km_25km_50m	15	6	9
0km_50km_0m	10	6	4
0km_50km_50m	13	6	7
1km_50km_0m	11	6	5
1km_50km_50m	15	6	9
0km_100km_0m	12	6	6
0km_100km_50m	15	6	9
1km_100km_0m	14	6	8
1km_100km_50m	16	6	10
0km_500km_0m	14	6	8
0km_500km_50m	17	6	11
1km_500km_0m	20	6	14
1km_500km_50m	12	6	6

Table A.20 Deforestation values for 555636253 (2014)

	All _deforested	PPA_deforested	Control_deforested
0km_25km_0m	8	1	7
0km_25km_50m	1	1	0
1km_25km_0m	7	1	6
1km_25km_50m	8	1	7
0km_50km_0m	7	1	6
0km_50km_50m	9	1	8
1km_50km_0m	9	1	8
1km_50km_50m	8	1	7
0km_100km_0m	11	1	10
0km_100km_50m	7	1	6
1km_100km_0m	11	1	10
1km_100km_50m	8	1	7
0km_500km_0m	16	1	15
0km_500km_50m	9	1	8
1km_500km_0m	12	1	11
1km_500km_50m	11	1	10

All

Table A.21 Mean of deforestation values across all 4 PPAs

	All _deforested	PPA_deforested	Control_deforested
0km_25km_0m	8.50	2.75	5.75
0km_25km_50m	5.75	2.50	3.25
1km_25km_0m	5.25	2.75	2.50
1km_25km_50m	7.00	2.50	4.50
0km_50km_0m	9.00	2.75	6.25
0km_50km_50m	9.25	2.50	6.75
1km_50km_0m	6.50	2.75	3.75
1km_50km_50m	7.00	2.50	4.50
0km_100km_0m	8.00	2.75	5.25
0km_100km_50m	7.50	2.50	5.00
1km_100km_0m	8.00	2.75	5.25
1km_100km_50m	8.50	2.50	6.00
0km_500km_0m	11.50	2.75	8.75
0km_500km_50m	12.75	2.50	10.25
1km_500km_0m	11.25	2.75	8.50
1km_500km_50m	9.00	2.50	6.50

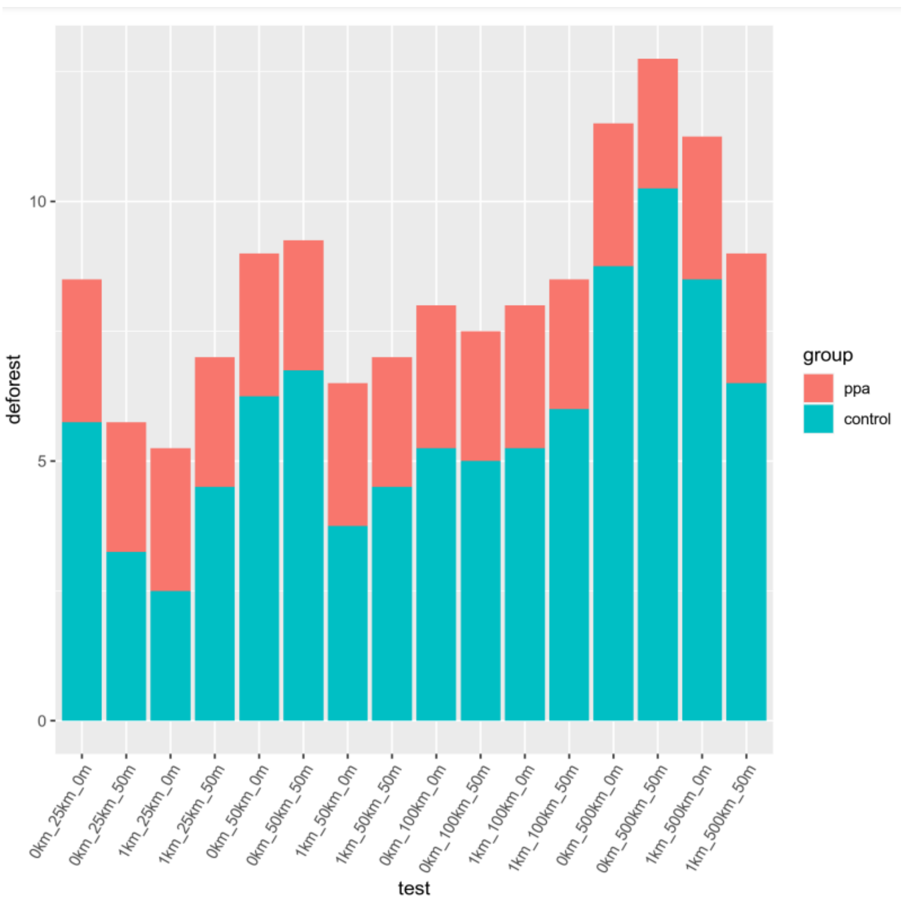

Figure A.5 Mean of deforestation values across all 4 PPAs

Moran Testing (ape)

Moran test of deforestation values for year of establishment plus five using ape library.

Table A.22 Moran testing (ape) values for 555555922 (2005)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	0.0010	N/A	0.003
0km_25km_50m	0.0008	N/A	0.002
1km_25km_0m	NA	N/A	N/A
1km_25km_50m	0.0002	N/A	0.001
0km_50km_0m	0.00006	N/A	0.003
0km_50km_50m	0.005	N/A	0.002
1km_50km_0m	0.0002	N/A	0.002
1km_50km_50m	0.0006	N/A	0.002
0km_100km_0m	0.0008	N/A	0.003
0km_100km_50m	0.0008	N/A	0.002
1km_100km_0m	0.001	N/A	0.002
1km_100km_50m	0.0009	N/A	0.006
0km_500km_0m	0.001	N/A	0.004
0km_500km_50m	0.001	N/A	0.004
1km_500km_0m	0.0009	N/A	0.003
1km_500km_50m	0.0009	N/A	0.003

Table A.23 Moran testing (ape) values for 555592795 (2014)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	0.02	0.004	0.07
0km_25km_50m	0.03	0.0004	0.09
1km_25km_0m	0.03	0.004	-0.01
1km_25km_50m	0.02	0.0004	N/A
0km_50km_0m	0.06	0.004	0.2
0km_50km_50m	0.03	0.0004	0.1
1km_50km_0m	0.006	0.004	N/A
1km_50km_50m	0.002	0.0004	N/A
0km_100km_0m	0.004	0.004	0.02
0km_100km_50m	0.002	0.0004	0.02
1km_100km_0m	0.001	0.004	-0.003
1km_100km_50m	-0.001	0.0004	-0.003
0km_500km_0m	0.01	0.004	0.008
0km_500km_50m	0.02	0.0004	0.02
1km_500km_0m	0.08	0.004	0.1
1km_500km_50m	0.02	0.0004	0.03

Table A.24 Moran testing (ape) values for 555592802 (2014)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	0.005	0.007	0.006
0km_25km_50m	0.003	0.006	0.001
1km_25km_0m	0.004	0.007	-0.002
1km_25km_50m	0.03	0.006	0.06
0km_50km_0m	0.002	0.007	-0.005
0km_50km_50m	0.01	0.006	0.03
1km_50km_0m	0.009	0.007	0.02
1km_50km_50m	0.001	0.006	0.004
0km_100km_0m	0.003	0.007	0.0003
0km_100km_50m	0.003	0.006	0.0004
1km_100km_0m	0.05	0.007	0.1
1km_100km_50m	0.03	0.006	0.05
0km_500km_0m	0.002	0.007	-0.05
0km_500km_50m	0.008	0.006	0.009
1km_500km_0m	0.03	0.007	0.05
1km_500km_50m	0.002	0.006	-0.005

Table A.25 Moran testing (ape) values for 555636253 (2014)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	0.0008	0.002	0.005
0km_25km_50m	0.001	0.002	N/A
1km_25km_0m	0.004	0.002	0.01
1km_25km_50m	0.0006	0.002	-0.002
0km_50km_0m	0.0003	0.002	-0.003
0km_50km_50m	-0.0003	0.002	-0.006
1km_50km_0m	00.0004	0.002	-0.005
1km_50km_50m	0.0008	0.002	-0.003
0km_100km_0m	0.005	0.002	0.002
0km_100km_50m	0.00002	0.002	-0.003
1km_100km_0m	0.009	0.002	0.006
1km_100km_50m	0.0004	0.002	-0.005
0km_500km_0m	0.02	0.002	0.01
0km_500km_50m	0.01	0.002	0.01
1km_500km_0m	0.01	0.002	0.008
1km_500km_50m	0.005	0.002	-0.001

All

Table A.26 Mean of absolute Moran test values across all 4 PPAs

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	0.008	0.004	0.02
0km_25km_50m	0.008	0.003	0.03
1km_25km_0m	0.004	0.004	0.005
1km_25km_50m	0.007	0.003	0.02
0km_50km_0m	0.01	0.004	0.05
0km_50km_50m	0.01	0.003	0.04
1km_50km_0m	0.004	0.004	0.009
1km_50km_50m	0.001	0.003	0.003
0km_100km_0m	0.003	0.004	0.007
0km_100km_50m	0.001	0.003	0.006
1km_100km_0m	0.01	0.004	0.03
1km_100km_50m	0.009	0.003	0.02
0km_500km_0m	0.007	0.004	0.007
0km_500km_50m	0.02	0.003	0.01
1km_500km_0m	0.03	0.004	0.05
1km_500km_50m	0.007	0.003	0.01

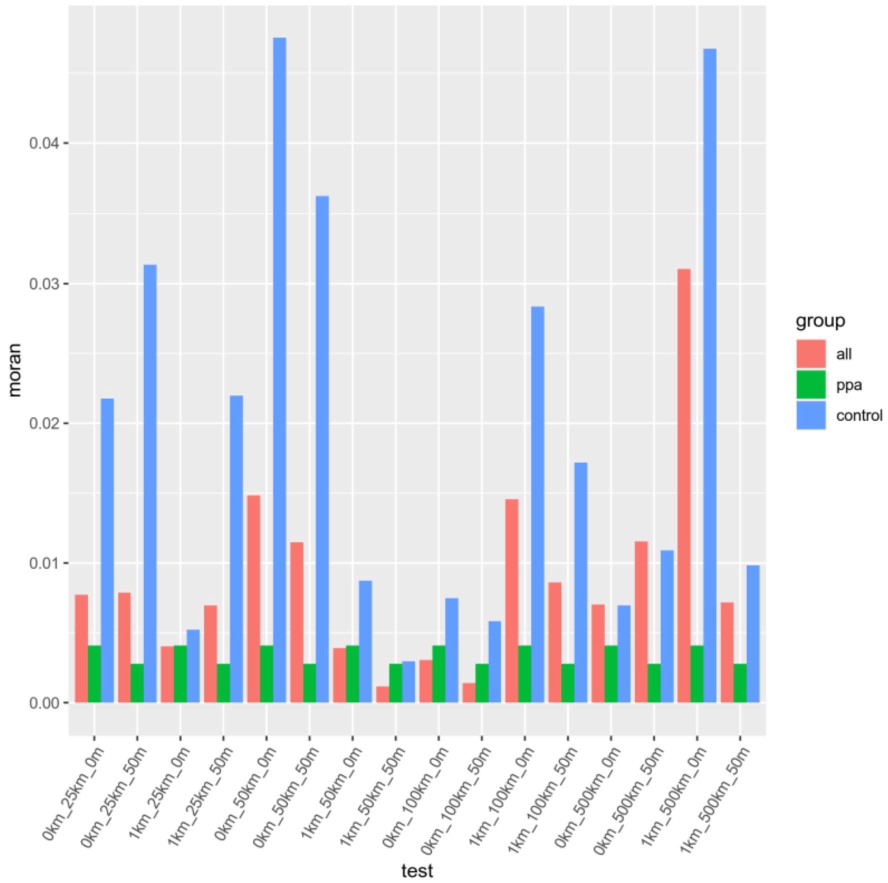

Figure A.6 Mean of absolute Moran test values across all 4 PPAs

Moran Testing (spdep)

Moran test of deforestation values for year of establishment plus five using spdep library.

Table A.27 Moran testing (spdep) values for 555555922 (2005)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	-0.002	N/A	-0.004
0km_25km_50m	-0.001	N/A	-0.002
1km_25km_0m	N/A	N/A	N/A
1km_25km_50m	-0.001	N/A	-0.003
0km_50km_0m	-0.001	N/A	0.005
0km_50km_50m	0.009	N/A	0.006
1km_50km_0m	-0.002	N/A	-0.004
1km_50km_50m	-0.002	N/A	-0.003
0km_100km_0m	-0.002	N/A	-0.004
0km_100km_50m	-0.001	N/A	-0.002
1km_100km_0m	-0.001	N/A	-0.002
1km_100km_50m	-0.0007	N/A	-0.006
0km_500km_0m	-0.002	N/A	-0.004
0km_500km_50m	-0.002	N/A	-0.004
1km_500km_0m	-0.001	N/A	-0.002
1km_500km_50m	-0.001	N/A	-0.002

Table A.28 Moran testing (spdep) values for 555592795 (2014)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	-0.0004	0.01	0.0002
0km_25km_50m	0.01	0.02	0.05
1km_25km_0m	0.002	0.01	N/A
1km_25km_50m	0.002	0.02	0.03
0km_50km_0m	0.009	0.01	-0.000005
0km_50km_50m	-0.004	0.02	N/A
1km_50km_0m	0.01	0.01	N/A
1km_50km_50m	0.009	0.02	-0.004
0km_100km_0m	-0.002	0.01	-0.003
0km_100km_50m	-0.002	0.02	-0.004
1km_100km_0m	-0.003	0.01	-0.004
1km_100km_50m	-0.003	0.02	-0.004
0km_500km_0m	-0.006	0.01	-0.001
0km_500km_50m	0.009	0.02	-0.004
1km_500km_0m	0.004	0.01	0.003
1km_500km_50m	0.00005	0.02	-0.003

Table A.29 Moran testing (spdep) values for 555592802 (2014)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	-0.002	0.02	-0.002
0km_25km_50m	-0.002	0.01	-0.001
1km_25km_0m	-0.003	0.02	-0.006
1km_25km_50m	0.009	0.01	0.02
0km_50km_0m	-0.003	0.02	-0.006
0km_50km_50m	-0.003	0.01	-0.008
1km_50km_0m	-0.002	0.02	-0.003
1km_50km_50m	-0.002	0.01	-0.003
0km_100km_0m	0.002	0.02	0.005
0km_100km_50m	0.001	0.01	0.002
1km_100km_0m	0.004	0.02	0.01
1km_100km_50m	0.02	0.01	0.04
0km_500km_0m	0.001	0.02	0.004
0km_500km_50m	0.01	0.01	0.02
1km_500km_0m	0.03	0.02	0.03
1km_500km_50m	0.0006	0.01	0.003

Table A.30 Moran testing (spdep) values for 555636253 (2014)

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	-0.004	-0.002	-0.0003
0km_25km_50m	-0.001	-0.001	N/A
1km_25km_0m	0.006	-0.002	0.003
1km_25km_50m	0.0006	-0.001	0.0003
0km_50km_0m	0.004	-0.002	0.001
0km_50km_50m	-0.007	-0.001	-0.01
1km_50km_0m	-0.009	-0.002	-0.02
1km_50km_50m	0.003	-0.001	-0.001
0km_100km_0m	-0.0007	-0.002	-0.0006
0km_100km_50m	0.01	-0.001	0.01
1km_100km_0m	0.02	-0.002	0.02
1km_100km_50m	0.004	-0.001	0.0002
0km_500km_0m	0.06	-0.002	0.05
0km_500km_50m	0.02	-0.001	0.01
1km_500km_0m	0.03	-0.002	0.02
1km_500km_50m	0.01	-0.001	0.006

All

Table A.31 Mean of absolute Moran test values (spdep) across all 4 PPAs

	All _moran_i	PPA_moran_i	Control_moran_i
0km_25km_0m	0.002	0.01	0.002
0km_25km_50m	0.004	0.01	0.02
1km_25km_0m	0.004	0.001	0.004
1km_25km_50m	0.003	0.01	0.007
0km_50km_0m	0.004	0.001	0.01
0km_50km_50m	0.005	0.01	0.006
1km_50km_0m	0.006	0.001	0.008
1km_50km_50m	0.002	0.01	0.002
0km_100km_0m	0.001	0.001	0.005
0km_100km_50m	0.003	0.01	0.005
1km_100km_0m	0.008	0.001	0.009
1km_100km_50m	0.008	0.01	0.01
0km_500km_0m	0.02	0.001	0.01
0km_500km_50m	0.01	0.01	0.02
1km_500km_0m	0.02	0.001	0.02
1km_500km_50m	0.004	0.01	0.004

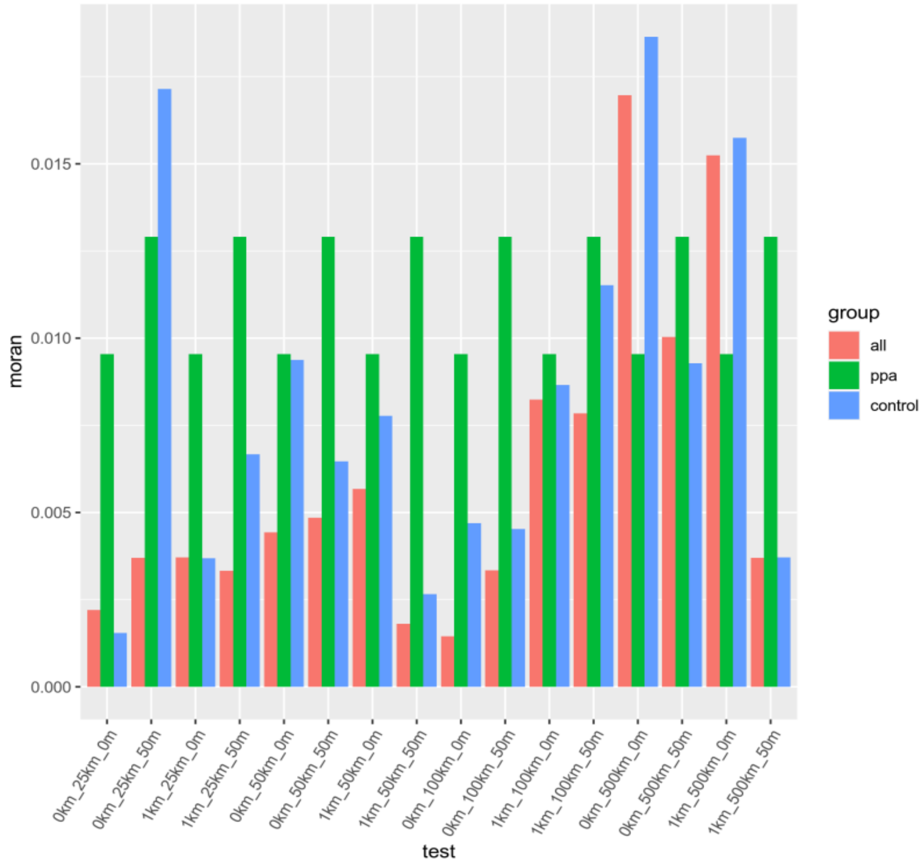

Figure A.7 Mean of absolute Moran test values (spdep) across all 4 PPAs

Appendix D - Covariate Balance

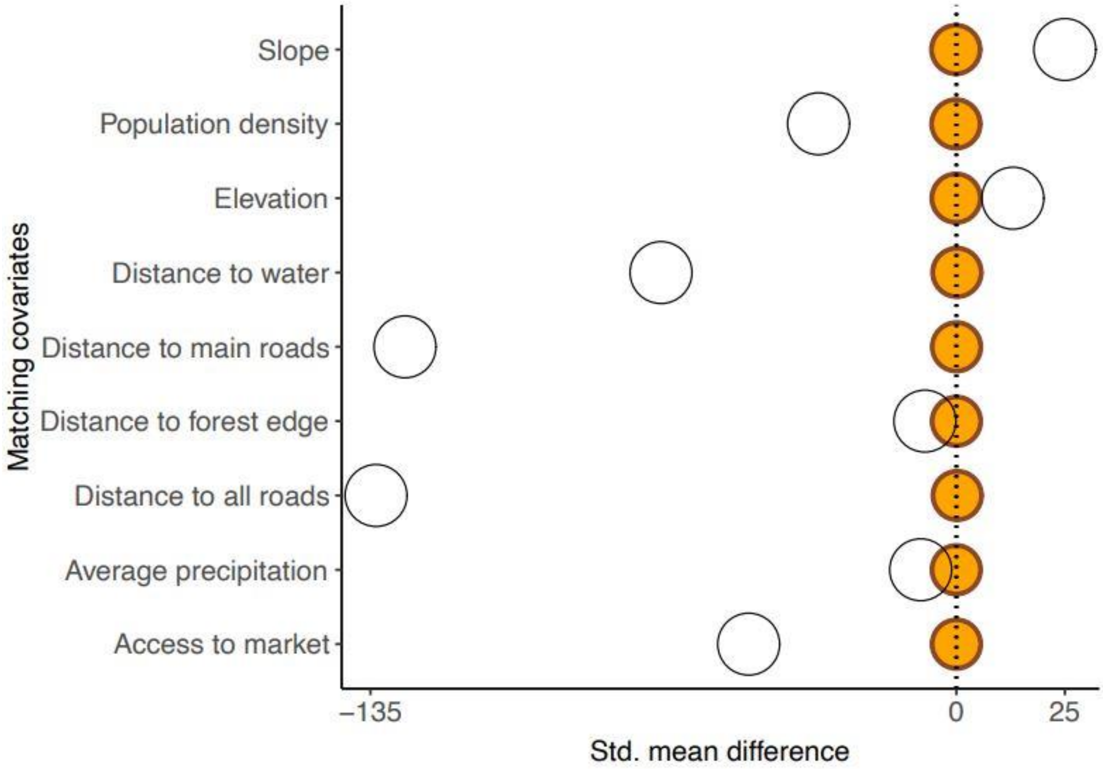

Figure A.8 Average covariate balance before and after 1:1 matching using land designated as a PPA as treatment

Standardized mean difference (averaged across 271 individual PPAs) for all matching covariates before (open circles) and after matching (orange circles).

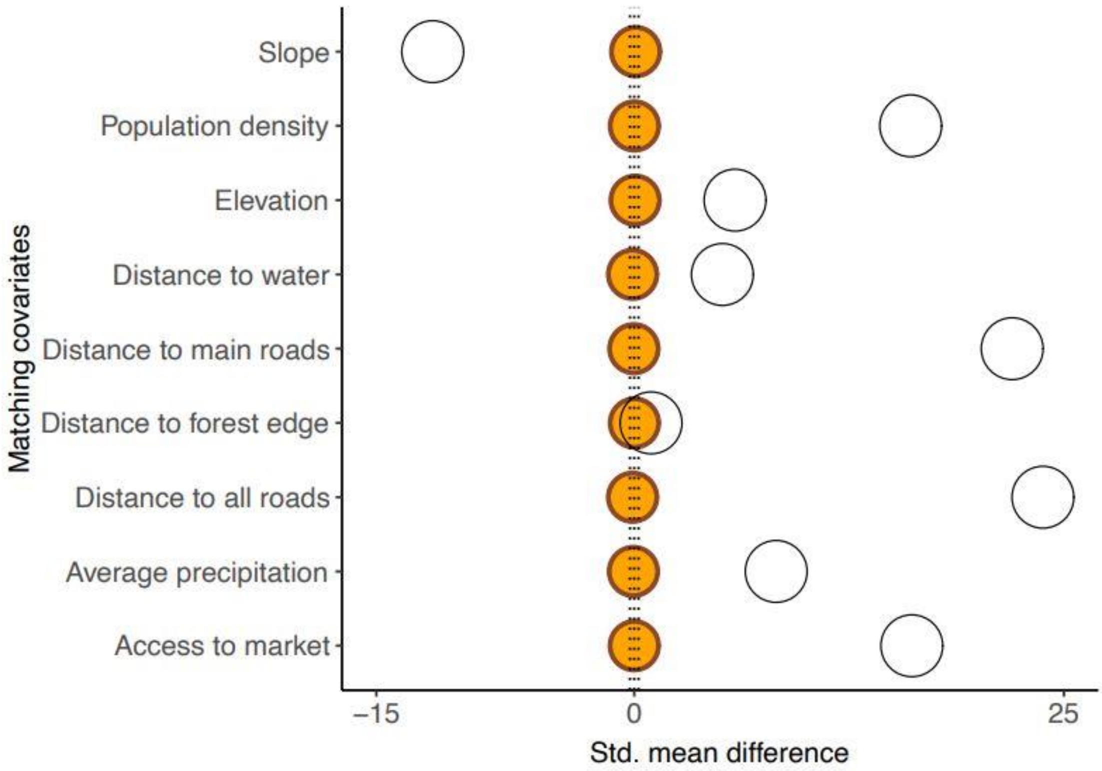

Figure A.9 Average covariate balance before and after 1:1 matching using land designated as a regional state PA as treatment

Standardized mean difference (averaged across 34 regional state PAs) for all matching covariates before (open circles) and after matching (orange circles).

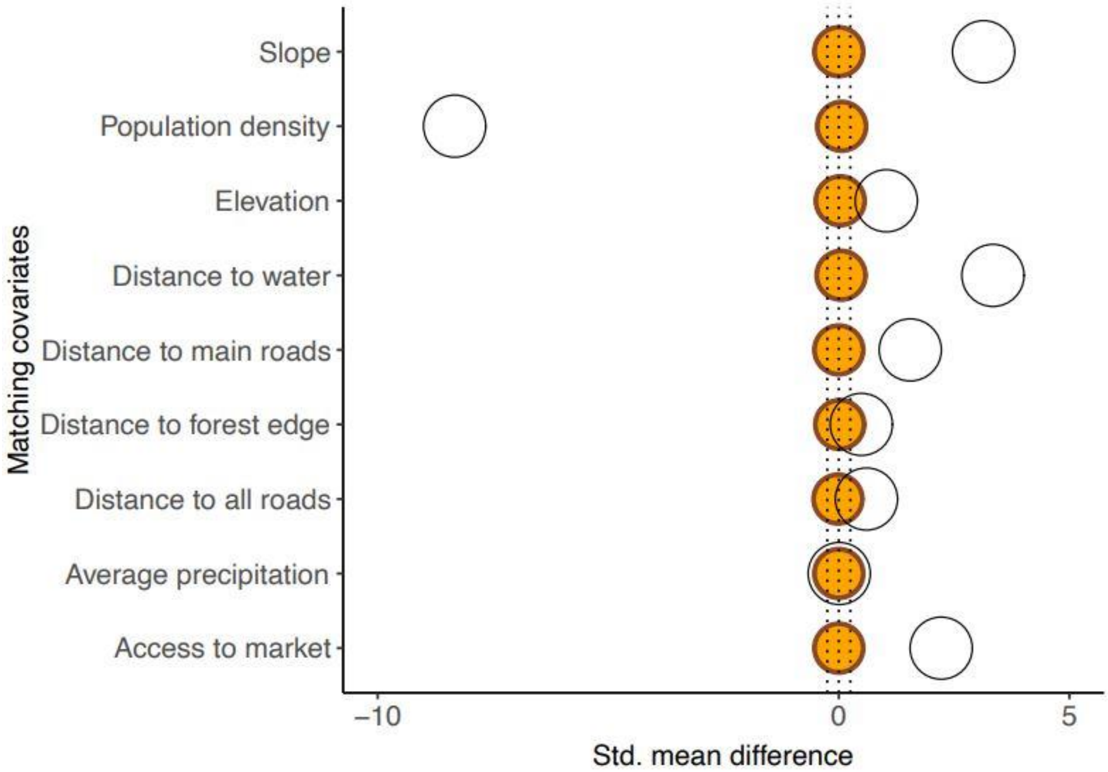

Figure A.10 Average covariate balance before and after 1:1 matching using land designated as a regional state PA as treatment

Standardized mean difference (averaged across 5 individual National PAs) for all matching covariates before (open circles) and after matching (orange circles).

Appendix D - Robustness check

Table A.32 Mean Coefficients and mean standard errors for each PA Governance Type

Governance Type	Mean Coefficient	Mean Standard Error
PPAs	-60.51	24.17
National PAs	2.68	3.69
Regional State PAs*	2.45	1.6

** Results exclude 1 extreme outlier with a value of -5.78e+14*

*** Results exclude 31 extreme outliers with an average value of -6.88e+14*

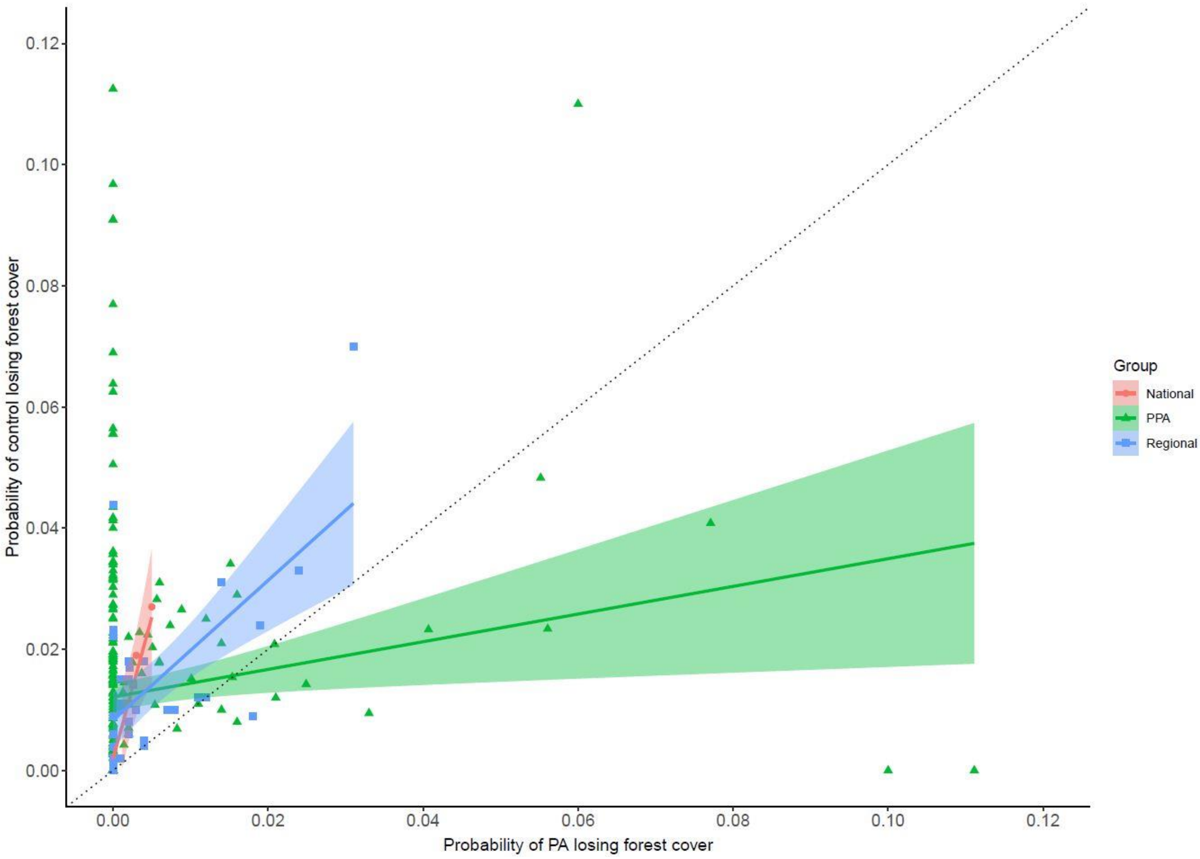

Figure A.11 Predicted forest loss in PAs and their corresponding matched unprotected controls.

Data points below the x = y line are PAs where forest loss inside the PA is less than forest loss in the PAs matched unprotected control. Data points above the x = y line are PAs where forest loss inside the PA is equal to or greater than forest loss in the PA's matched unprotected control

Appendix D - Deforestation rates

Table A.33 Statistics of 272 PPAs using a 100km^2 control area buffer: Number of sample pixels in PPAs (N_{ppa}), number of sample pixels in Control Areas (N_{ca}) and N_{ca}/N_{ppa} ratio before matching; Number of sample pixels in PPAs (n_{ppa}) and number of sample pixels in Control Areas (n_{ca}) after matching; Number of deforestation sample pixels in PPAs (n_{ppad}) and number of deforestation sample pixels in Control Areas (n_{cad}) after matching; Deforestation rate in NRs (DR_{ppa}), deforestation rate in Control Areas (DR_{ca}) and monitoring period. Sample pixel size: 30m × 30m

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
1	Cundinamarca	0.62	438	219000	500.00	438	438	0	3	0.0	0.7	2012 – 2017
2	Tolima	0.03	25	50000	2000.00	25	25	0	0	0.0	0.0	2009 – 2014
3	Cauca	0.14	30	50000	1666.67	30	30	0	0	0.0	0.0	2003 - 2008
4	Cauca	0.03	17	50000	2941.18	17	17	0	0	0.0	0.0	2003 – 2008
5	Valle de Cauca	0.06	63	50000	793.65	63	63	0	0	0.0	0.0	2012 – 2017
6	Boyaca	1.09	711	355500	500.00	711	711	1	3	0.1	0.4	2005 – 2010
7	Cauca	1.6	830	415000	500.00	830	830	0	10	0.0	1.2	2003 – 2008
8	Boyaca / Santander	14.22	15076	50000	33.17	1000	1000	16	29	1.6	2.9	2005 – 2010
9	Cundinamarca	0.51	502	251000	500.00	502	502	3	9	0.6	1.8	2007 – 2012
10	Cauca	0.09	25	50000	2000.00	25	25	0	0	0.0	0.0	2003 – 2008

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
11	Cauca	0.04	12	50000	4166.67	12	12	0	0	0.0	0.0	2003 – 2008
12	Cauca	0.01	16	50000	3125.00	16	16	0	0	0.0	0.0	2007 – 2012
13	Valle de Cauca	0.14	153	76500	500.00	153	153	0	3	0.0	2.0	2014 – 2019
14	Cauca	0.08	66	50000	757.58	66	66	0	0	0.0	0.0	2003 – 2008
15	Cauca	0.68	319	159500	500.00	319	319	0	6	0.0	1.9	2003 – 2008
16	Huila	0.6	671	335500	500.00	671	671	0	7	0.0	1.0	2006 – 2011
17	Tolima	0.3	319	159500	500.00	319	319	0	5	0.0	1.6	2014 - 2019
18	Cundinamarca	3.49	2837	50000	176.24	1000	1000	0	1	0.0	0.1	2013 – 2018
19	Cundinamarca	1.14	1141	499742	437.99	1000	1000	0	15	0.0	1.5	2005 – 2010
20	Cundinamarca	0.18	197	98500	500.00	197	197	1	4	0.5	2.0	2005 – 2010
21	Cundinamarca	0.48	145	72500	500.00	145	145	8	7	5.5	4.8	2007 – 2012
22	Cauca	0.14	101	50500	500.00	101	101	0	0	0.0	0.0	2006 – 2011
23	Valle de Cauca	1.33	1393	50000	358.94	1000	1000	11	11	1.1	1.1	2001 – 2006
24	Cundinamarca	2.93	169	84500	500.00	169	169	1	3	0.6	1.8	2002 – 2007
25	Valle de Cauca	1.25	483	241500	500.00	483	483	0	1	0.0	0.2	2014 2019

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
26	Antioquia	0.2	223	111500	500.00	223	223	0	7	0.0	3.1	2013 – 2018
27	Boyaca	0.39	382	191000	500.00	382	382	0	1	0.0	0.3	2014 – 2019
28	Valle de Cauca	0.14	112	56000	500.00	112	112	0	3	0.0	2.7	2010 – 2015
29	Antioquia	10.19	8436	50000	59.27	1000	1000	0	5	0.0	0.5	2006 – 2011
30	Santander	5.23	407	197187	484.49	407	407	0	14	0.0	3.4	2014 – 2019
31	Cauca	1.57	1743	50000	286.86	1000	1000	0	4	0.0	0.4	2010 – 2015
32	Valle de Cauca	2.29	2445	50000	204.50	1000	1000	2	6	0.2	0.6	2001 – 2006
33	Cauca	0.02	13	50000	3846.15	13	13	0	0	0.0	0.0	2003 – 2008
34	Cauca	0.08	48	50000	1041.67	48	48	0	3	0.0	6.3	2006 – 2011
35	Valle de Cauca	2.68	1338	50000	373.69	1000	1000	0	7	0.0	0.7	2013 – 2018
36	Caqueta	0.22	218	109000	500.00	218	218	0	9	0.0	4.1	2014 – 2019
37	Cauca	0.01	15	50000	3333.33	15	15	0	0	0.0	0.0	2003 – 2008
38	Cauca	0.11	54	50000	925.93	54	54	0	1	0.0	1.9	2003 – 2008
39	Valle de Cauca	0.12	133	66500	500.00	133	133	0	0	0.0	0.0	2014 – 2019
40	Valle de Cauca	0.14	55	50000	909.09	55	55	0	0	0.0	0.0	2014 – 2019

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
41	Cauca	0.04	29	50000	1724.14	29	29	0	2	0.0	6.9	2005 - 2010
42	Cauca	0.08	63	50000	793.65	63	63	0	0	0.0	0.0	2006 – 2011
43	Huila	0.04	45	50000	1111.11	45	45	0	1	0.0	2.2	2005 – 2010
44	Valle de Cauca	0.57	281	140500	500.00	281	281	7	4	2.5	1.4	2008 – 2013
45	Huila	0.03	26	50000	1923.08	26	26	0	0	0.0	0.0	2005 – 2010
46	Cauca	0.23	145	72500	500.00	145	145	0	0	0.0	0.0	2006 – 2011
47	Huila	0.11	124	62000	500.00	124	124	0	7	0.0	5.6	2010 – 2015
48	Cauca	0.17	185	92500	500.00	185	185	1	2	0.5	1.1	2003 – 2008
49	Huila	0.17	185	50000	694.44	72	72	0	0	0.0	0.0	2005 – 2010
50	Huila	0.06	72	50000	4166.67	12	12	0	0	0.0	0.0	2005 – 2010
51	Cauca	0.02	12	329500	500.00	659	659	0	5	0.0	0.8	2003 – 2008
52	Huila	0.61	659	50000	704.23	71	71	0	1	0.0	1.4	2005 – 2010
53	Valle de Cauca	0.07	71	148500	500.00	297	297	0	1	0.0	0.3	2014 – 2019
54	Huila	0.27	297	50000	500.00	45	45	0	1	0.0	2.2	2005 – 2010
55	Valle de Cauca	0.11	100	50000	2777.78	100	100	6	11	6.0	11.0	2014 – 2019

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
56	Cauca	0.13	18	55000	500.00	18	18	0	1	0.0	5.6	2003 – 2008
57	Huila	0.1	110	50000	1562.50	110	110	0	3	0.0	2.7	2005 – 2010
58	Huila	0.03	32	50000	1111.11	32	32	0	0	0.0	0.0	2005 – 2010
59	Huila	0.05	49	50000	1020.41	49	49	1	0	2.0	0.0	2005 – 2010
60	Huila	0.02	23	50000	2173.91	23	23	0	1	0.0	4.3	2003 – 2008
61	Cauca	0.29	284	142000	500.00	284	284	0	9	0.0	3.2	2003 - 2008
62	Cauca	0.59	441	220500	500.00	441	441	34	18	7.7	4.1	2007 – 2012
63	Cauca	1.06	811	405500	500.00	811	811	3	13	0.4	1.6	2003 – 2008
64	Huila	0.35	389	194500	500.00	389	389	0	3	0.0	0.8	2005 - 2010
65	Cauca	0.12	12	50000	4166.67	12	12	0	0	0.0	0.0	2003 – 2008
66	Huila	0.61	543	271500	500.00	543	543	4	13	0.7	2.4	2005 – 2010
67	Cauca	0.13	63	50000	793.65	63	63	0	1	0.0	1.6	2006 – 2011
68	Cundinamarca	0.11	12	50000	4166.67	12	12	0	0	0.0	0.0	2007 – 2012
69	Cundinamarca	0.48	532	266000	500.00	532	532	0	2	0.0	0.4	2007 – 2012

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
70	Huila	0.14	160	80000	500.00	160	160	1	1	0.6	0.6	2005 – 2010
71	Cauca		35	50000	1428.57	35	35	0	0	0.0	0.0	
72	Cundinamarca	0.02	12	50000	4166.67	12	12	0	0	0.0	0.0	2007 – 2012
73	Cauca	0.02	24	50000	2083.33	24	24	0	1	0.0	4.2	2012 – 2017
74	Cauca	0.13	69	50000	724.64	69	69	0	2	0.0	2.9	2012 – 2017
75	Cauca	0.23	120	60000	500.00	120	120	0	2	0.0	1.7	2005 – 2010
76	Huila	0.1	106	53000	500.00	106	106	0	2	0.0	1.9	2005 – 2010
77	Cundinamarca	0.13	75	50000	666.67	75	75	0	0	0.0	0.0	2013 – 2018
78	Cundinamarca	0.05	52	50000	961.54	52	52	5	0	9.6	0.0	2002 – 2007
79	Cauca	0.49	215	107500	500.00	215	215	0	0	0.0	0.0	2003 – 2008
80	Huila	0.4	441	220500	500.00	441	441	3	0	0.7	0.0	2005 – 2010
81	Valle de Cauca	1.93	1435	50000	348.43	1000	1000	0	3	0.0	0.3	2014 – 2019
82	Cauca	0.06	32	50000	1562.50	32	32	0	0	0.0	0.0	2003 – 2008
83	Narino	0.21	177	88500	500.00	177	177	1	5	0.6	2.8	2006 – 2011
84	Huila	0.26	285	142500	500.00	285	285	4	2	1.4	0.7	2005 – 2010

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
85	Huila	0.13	113	56500	500.00	113	113	0	3	0.0	2.7	2005 – 2010
86	Valle de Cauca	0.95	373	186500	500.00	373	373	0	1	0.0	0.3	2013 – 2018
87	Huila	0.14	158	79000	500.00	158	158	0	2	0.0	1.3	2005 – 2010
88	Valle de Cauca	0.27	49	50000	1020.41	49	49	3	0	6.1	0.0	2008 – 2013
89	Huila	0.19	212	106000	500.00	212	212	0	2	0.0	0.9	2005 – 2010
90	Valle de Cauca	0.15	44	50000	1136.36	44	44	2	0	4.5	0.0	2010 – 2015
91	Huila	0.21	228	114000	500.00	228	228	0	1	0.0	0.4	2005 – 2010
92	Cauca	0.05	42	50000	1190.48	42	42	0	0	0.0	0.0	2003 – 2008
93	Huila	0.4	418	209000	500.00	418	418	0	8	0.0	1.9	2005 – 2010
94	Cauca	0.1	47	50000	1063.83	47	47	0	1	0.0	2.1	2006 – 2011
95	Valle de Cauca	0.16	29	50000	1724.14	29	29	0	0	0.0	0.0	2014 – 2019
96	Huila	0.03	24	50000	2083.33	24	24	0	1	0.0	4.2	2014 – 2019
97	Valle de Cauca	3.19	3115	50000	160.51	1000	1000	0	10	0.0	1.0	2013 – 2018
98	Huila	4.14	2660	50000	187.97	1000	1000	0	15	0.0	1.5	2008 – 2013
99	Huila	0.01	11	50000	4545.45	11	11	0	0	0.0	0.0	2014 – 2019

Name of PPA	**State**	**Area (km^2)**	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	**DR_{ppa} (%)**	**DR_{ca} (%)**	**Monitoring period**
100	Santander	2.39	685	342500	500.00	685	685	0	22	0.0	3.2	2014 – 2019
101	Santander	0.18	176	88000	500.00	176	176	0	1	0.0	0.6	2012 – 2017
102	Tolima	74.20	63812	50000	7.84	1000	1000	12	25	1.2	2.5	2014 – 2019
103	Cundinamarca	1.48	1003	50000	498.50	1000	1000	2	7	0.2	0.7	2012 – 2017
104	Meta	0.1	37	50000	1351.35	37	37	0	0	0.0	0.0	2014 – 2019
105	Cauca	0.32	172	86000	500.00	172	172	7	4	4.1	2.3	2003 – 2008
106	Cundinamarca	0.06	67	50000	746.27	67	67	0	1	0.0	1.5	2007 - 2012
107	Tolima	6.37	3242	50000	154.23	1000	1000	14	10	1.4	1.0	2014 – 2019
108	Huila	0.24	264	132000	500.00	264	264	4	9	1.5	3.4	2012 – 2017
109	Cauca	0.09	95	50000	526.32	95	95	0	2	0.0	2.1	2003 – 2008
110	Huila	0.09	99	50000	505.05	99	99	0	3	0.0	3.0	2005 – 2010
111	Cauca	0.08	79	50000	632.91	79	79	0	0	0.0	0.0	2005 – 2010
112	Huila	0.03	28	50000	1785.71	28	28	0	0	0.0	0.0	2005 – 2010
113	Valle de Cauca	0.13	116	58000	500.00	116	116	0	3	0.0	2.6	2014 – 2019
114	Cauca	0.52	401	200500	500.00	401	401	0	11	0.0	2.7	2003 – 2008

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
115	Cauca	0.27	198	99000	500.00	198	198	0	1	0.0	0.5	2003 – 2008
116	Cauca	0.28	214	107000	500.00	214	214	12	5	5.6	2.3	2011 – 2016
117	Huila	0.07	78	50000	641.03	78	78	0	0	0.0	0.0	2005 – 2010
118	Cauca	0.06	52	50000	961.54	52	52	0	0	0.0	0.0	2003 – 2008
119	Meta	0.14	61	50000	819.67	61	61	0	1	0.0	1.6	2014 – 2019
120	Cauca	0.04	39	50000	1282.05	39	39	0	0	0.0	0.0	2007 – 2012
121	Cauca	0.02	18	50000	2777.78	18	18	0	1	0.0	5.6	2007 – 2012
122	Valle de Cauca	0.04	26	50000	1923.08	26	26	0	0	0.0	0.0	2010 – 2015
123	Valle de Cauca	0.02	25	50000	2000.00	25	25	0	0	0.0	0.0	2014 – 2019
124	Cauca	0.09	52	50000	961.54	52	52	0	1	0.0	1.9	2003 – 2008
125	Cundinamarca	1.46	1479	5.00E+05	338.07	1000	1000	2	6	0.2	0.6	2005 – 2010
126	Huila	0.44	487	243500	500.00	487	487	0	9	0.0	1.8	2005 – 2010
127	Cundinamarca	0.26	285	142500	500.00	285	285	0	1	0.0	0.4	2007 – 2012
128	Huila	0.05	60	50000	833.33	60	60	0	0	0.0	0.0	2014 – 2019
129	Cauca	0.12	92	50000	543.48	92	92	0	1	0.0	1.1	2006 – 2011

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
130	Cauca	0.11	58	50000	862.07	58	58	0	0	0.0	0.0	2006 – 2011
131	Cauca	0.03	24	50000	2083.33	24	24	0	1	0.0	4.2	2005 – 2010
132	Cundinamarca	0.13	141	70500	500.00	141	141	0	9	0.0	6.4	2004 – 2009
133	Cundinamarca	0.09	64	50000	781.25	64	64	0	1	0.0	1.6	2004 – 2009
134	Valle de Cauca	0.72	67	50000	746.27	67	67	0	0	0.0	0.0	2013 – 2018
135	Valle de Cauca	0.02	22	50000	2272.73	22	22	0	0	0.0	0.0	2011 – 2016
136	Narino	0.2	218	109000	500.00	218	218	0	4	0.0	1.8	2010 – 2015
137	Valle de Cauca	1.02	804	402000	500.00	804	804	0	12	0.0	1.5	2014 – 2019
138	Tolima	0.83	878	439000	500.00	878	878	3	20	0.3	2.3	2014 – 2019
139	Cauca	0.23	212	106000	500.00	212	212	7	2	3.3	0.9	2003 – 2008
140	Valle de Cauca	1.5	727	363500	500.00	727	727	6	5	0.8	0.7	2009 – 2014
141	Cauca	0.27	143	71500	500.00	143	143	2	3	1.4	2.1	2003 – 2008
142	Cauca	0.8	611	305500	500.00	611	611	0	9	0.0	1.5	2006 – 2011
143	Huila	0.2	223	111500	500.00	223	223	1	5	0.4	2.2	2005 – 2010
144	Huila	0.02	22	50000	2272.73	22	22	0	0	0.0	0.0	2005 - 2010

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
145	Cauca	0.08	88	50000	568.18	88	88	0	1	0.0	1.1	2014 – 2019
146	Valle de Cauca	0.25	237	118500	500.00	237	237	6	0	2.5	0.0	2008 – 2013
147	Valle de Cauca	0.07	22	50000	2272.73	22	22	0	0	0.0	0.0	2009 – 2014
148	Cauca	0.16	14	50000	3571.43	14	14	0	0	0.0	0.0	2005 – 2010
149	Cauca	0.12	12	50000	4166.67	12	12	0	0	0.0	0.0	2005 – 2010
150	Cauca	0.12	53	50000	943.40	53	53	0	0	0.0	0.0	2005 – 2010
151	Cauca	0.09	72	50000	694.44	72	72	0	0	0.0	0.0	2006 – 2011
152	Cundinamarca	1.16	1277	50000	391.54	1000	1000	2	7	0.2	0.7	2007 – 2012
153	Cauca	0.03	28	50000	1785.71	28	28	0	1	0.0	3.6	2007 – 2012
154	Valle de Cauca	0.07	58	50000	862.07	58	58	12	2	20.7	3.4	2008 – 2013
155	Cauca	0.13	101	50500	500.00	101	101	0	1	0.0	1.0	2003 – 2008
156	Huila	0.01	11	50000	4545.45	11	11	0	0	0.0	0.0	2005 – 2010
157	Valle de Cauca	0.29	111	55500	500.00	111	111	0	0	0.0	0.0	2013 – 2018
158	Cundinamarca	0.03	24	50000	2083.33	24	24	3	0	12.5	0.0	2004 – 2009
159	Huila	0.03	31	50000	1612.90	31	31	0	0	0.0	0.0	2005 – 2010

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
160	Valle de Cauca	0.26	83	50000	602.41	83	83	0	3	0.0	3.6	2008 – 2013
161	Valle de Cauca	0.08	88	50000	568.18	88	88	0	0	0.0	0.0	2009 – 2014
162	Tolima	0.22	206	103000	500.00	206	206	0	7	0.0	3.4	2003 – 2008
163	Valle de Cauca	0.19	104	52000	500.00	104	104	0	0	0.0	0.0	2013 – 2018
164	Cauca	0.08	60	50000	833.33	60	60	0	0	0.0	0.0	2011 – 2016
165	Cauca	0.09	99	50000	505.05	99	99	0	5	0.0	5.1	2005 – 2010
166	Huila	0.03	28	50000	1785.71	28	28	0	0	0.0	0.0	2014 – 2019
167	Valle de Cauca	0.32	110	55000	500.00	110	110	3	0	2.7	0.0	2010 – 2015
168	Tolima	0.1	80	50000	625.00	80	80	0	9	0.0	11.3	2004 – 2009
169	Cauca	0.08	32	50000	1562.50	32	32	0	0	0.0	0.0	2003 – 2008
170	Valle de Cauca	0.04	39	50000	1282.05	39	39	0	0	0.0	0.0	2012 – 2017
171	Huila	0.01	16	50000	3125.00	16	16	0	0	0.0	0.0	2005 – 2010
172	Cauca	0.08	14	50000	3571.43	14	14	2	0	14.3	0.0	2003 – 2008
173	Cauca	0.05	31	50000	1612.90	31	31	0	3	0.0	9.7	2012 -2017
174	Valle de Cauca	0.69	697	348500	500.00	697	697	0	12	0.0	1.7	2008 – 2013

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
175	Cauca	0.06	56	50000	892.86	56	56	0	0	0.0	0.0	2006 – 2011
176	Cauca	0.19	204	102000	500.00	204	204	0	7	0.0	3.4	2003 – 2008
177	Huila	0.02	21	50000	2380.95	21	21	0	0	0.0	0.0	2005 – 2010
178	Cauca	0.06	41	50000	1219.51	41	41	0	0	0.0	0.0	2006 – 2011
179	Cauca	0.16	124	62000	500.00	124	124	0	0	0.0	0.0	2006 -2011
180	Valle de Cauca	0.27	284	142000	500.00	284	284	0	9	0.0	3.2	2014 – 2019
181	Cundinamarca	0.06	70	50000	714.29	70	70	0	0	0.0	0.0	2009 – 2014
182	Narino	0.3	328	164000	500.00	328	328	0	0	0.0	0.0	2014 – 2019
183	Huila	0.03	10	50000	5000.00	10	10	1	0	10.0	0.0	2005 – 2010
184	Cauca	0.04	36	50000	1388.89	36	36	0	0	0.0	0.0	2003 – 2008
185	Valle de Cauca	0.36	158	79000	500.00	158	158	2	0	1.3	0.0	2009 – 2014
186	Cundinamarca	0.06	21	50000	2380.95	21	21	0	0	0.0	0.0	2014 – 2019
187	Casanare	4.15	3534	5.00E+05	141.48	1000	1000	2	22	0.2	2.2	2014 – 2019
188	Cauca	0.07	53	50000	943.40	53	53	3	0	5.7	0.0	2003 – 2008
189	Huila	0.17	130	65000	500.00	130	130	0	0	0.0	0.0	2013 – 2018

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
190	Cauca	0.08	18	50000	2777.78	18	18	0	0	0.0	0.0	2003 – 2008
191	Boyaca	0.04	11	50000	4545.45	11	11	0	0	0.0	0.0	2010 – 2015
192	Huila	0.04	44	50000	1136.36	44	44	0	4	0.0	9.1	2005 – 2010
193	Valle de Cauca	2.92	1463	50000	341.76	1000	1000	16	8	1.6	0.8	2008 – 2013
194	Cauca	0.11	77	50000	649.35	77	77	0	1	0.0	1.3	2003 – 2008
195	Cundinamarca	0.02	21	50000	2380.95	21	21	0	0	0.0	0.0	2014 – 2019
196	Cundinamarca	0.32	332	166000	500.00	332	332	0	2	0.0	0.6	2013 – 2018
197	Cauca	0.27	220	110000	500.00	220	220	0	4	0.0	1.8	2003 – 2008
198	Narino	0.34	369	184500	500.00	369	369	1	0	0.3	0.0	2011 – 2016
199	Huila	0.21	226	113000	500.00	226	226	2	6	0.9	2.7	2005 – 2010
200	Boyaca	0.08	44	50000	1136.36	44	44	0	1	0.0	2.3	2007 – 2012
201	Santander	3.25	2637	93611	35.50	1000	1000	0	3	0.0	0.3	2014 – 2019
202	Cauca	0.12	75	50000	666.67	75	75	0	0	0.0	0.0	2005 – 2010
203	Cauca	0.05	55	50000	909.09	55	55	0	0	0.0	0.0	2005 – 2010
204	Cauca	0.04	38	50000	1315.79	38	38	0	0	0.0	0.0	2005 – 2010

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
205	Cauca	1.02	1067	50000	468.60	1000	1000	0	33	0.0	3.3	2005 – 2010
206	Cauca	0.14	119	59500	500.00	119	119	0	3	0.0	2.5	2005 – 2010
207	Cauca	0.04	30	50000	1666.67	30	30	0	0	0.0	0.0	2005 – 2010
208	Cauca	0.05	47	50000	1063.83	47	47	0	0	0.0	0.0	2005 – 2010
209	Cauca	0.08	18	50000	2777.78	18	18	0	0	0.0	0.0	2005 – 2010
210	Cauca	0.04	25	50000	2000.00	25	25	0	1	0.0	4.0	2005 – 2010
211	Cauca	0.03	10	50000	5000.00	10	10	0	0	0.0	0.0	2005 – 2010
212	Cauca	0.12	17	50000	2941.18	17	17	5	0	29.4	0.0	2005 – 2010
213	Cauca	0.09	48	50000	1041.67	48	48	1	1	2.1	2.1	2005 – 2010
214	Cauca	0.04	39	50000	1282.05	39	39	0	0	0.0	0.0	2005 – 2010
215	Cauca	0.07	72	50000	694.44	72	72	3	1	4.2	1.4	2005 – 2010
216	Cauca	0.38	198	99000	500.00	198	198	2	3	1.0	1.5	2006 – 2011
217	Cauca	0.12	72	50000	694.44	72	72	0	0	0.0	0.0	2005 – 2010
218	Cundinamarca	1.8	1666	497483	298.61	1000	1000	0	12	0.0	1.2	2004 – 2009
219	Valle de Cauca	0.08	70	50000	714.29	70	70	0	1	0.0	1.4	2014 – 2019

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
220	Cundinamarca	0.64	394	197000	500.00	394	394	1	7	0.3	1.8	2012 – 2017
221	Valle de Cauca	0.17	120	60000	500.00	120	120	0	3	0.0	2.5	2009 – 2014
222	Valle de Cauca	0.04	31	50000	1612.90	31	31	4	0	12.9	0.0	2009 – 2014
223	Valle de Cauca	0.76	324	162000	500.00	324	324	0	1	0.0	0.3	2009 – 2014
224	Cauca	0.02	18	50000	2777.78	18	18	0	1	0.0	5.6	2007 – 2012
225	Huila	0.01	14	50000	3571.43	14	14	0	0	0.0	0.0	2005 – 2010
226	Huila	0.0.2	18	50000	2777.78	18	18	0	0	0.0	0.0	2005 – 2010
227	Huila	0.01	15	50000	3333.33	15	15	0	0	0.0	0.0	2005 -2010
228	Huila	0.02	24	50000	2083.33	24	24	0	0	0.0	0.0	2005 – 2010
229	Huila	0.02	21	50000	2380.95	21	21	0	0	0.0	0.0	2005 - 2010
230	Huila	0.01	15	50000	3333.33	15	15	0	0	0.0	0.0	2005 – 2010
231	Huila	0.01	14	50000	3571.43	14	14	0	0	0.0	0.0	2005 – 2010
232	Narino	5.51	1702	50000	293.77	1000	1000	21	12	2.1	1.2	2003 – 2008
233	Cauca	0.14	109	54500	500.00	109	109	1	0	0.9	0.0	2003 – 2008
234	Narino	0.18	87	50000	574.71	87	87	0	0	0.0	0.0	2003 – 2008

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
235	Cauca	0.26	65	50000	769.23	65	65	1	1	1.5	1.5	2003 – 2008
236	Valle de Cauca	0.06	51	50000	980.39	51	51	0	0	0.0	0.0	2013 – 2018
237	Huila	0.03	30	50000	1666.67	30	30	0	0	0.0	0.0	2005 – 2010
238	Valle de Cauca	0.69	689	344500	500.00	689	689	1	10	0.1	1.5	2010 – 2015
239	Antioquia	11.58	12453	5.00E+05	40.15	1000	1000	6	31	0.6	3.1	2005 – 2010
240	Cundinamarca	0.02	19	50000	2631.58	19	19	0	0	0.0	0.0	2005 – 2010
241	Boyaca	0.29	274	137000	500.00	274	274	0	2	0.0	0.7	2003 – 2008
242	Valle de Cauca	0.28	309	154500	500.00	309	309	0	1	0.0	0.3	2012 – 2017
243	Valle de Cauca	1.97	669	334500	500.00	669	669	0	12	0.0	1.8	2014 – 2019
244	Cauca	0.06	48	50000	1041.67	48	48	0	0	0.0	0.0	2006 – 2011
245	Valle de Cauca	0.49	111	55500	500.00	111	111	0	0	0.0	0.0	2009 - 2014
246	Boyaca	1.55	1702	414674	243.64	1000	1000	0	0	0.0	0.0	2012 – 2017
247	Cauca	0.03	28	50000	1785.71	28	28	1	1	3.6	3.6	2003 – 2008
248	Valle de Cauca	0.22	222	111000	500.00	222	222	0	1	0.0	0.5	2011 – 2016
249	Cauca	0.41	75	50000	666.67	75	75	0	0	0.0	0.0	2003 – 2008

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
250	Valle de Cauca	0.2	129	64500	500.00	129	129	0	1	0.0	0.8	2014 – 2019
251	Cauca	0.17	66	50000	757.58	66	66	2	1	3.0	1.5	2006 – 2011
252	Cauca	0.05	46	50000	1086.96	46	46	0	1	0.0	2.2	2003 – 2018
253	Valle de Cauca	0.29	314	157000	500.00	314	314	0	4	0.0	1.3	2003 – 2018
254	Cauca	0.16	63	50000	793.65	63	63	0	2	0.0	3.2	2003 – 2008
255	Cauca	0.29	108	54000	500.00	108	108	1	0	0.9	0.0	2006 – 2011
256	Cauca	0.16	31	50000	1612.90	31	31	0	1	0.0	3.2	2006 – 2011
257	Cauca	0.18	153	76500	500.00	153	153	0	3	0.0	2.0	2006 - 2011
258	Huila	0.14	11	50000	4545.45	11	11	0	0	0.0	0.0	2014 – 2019
259	Valle de Cauca	0.35	18	50000	2777.78	18	18	2	0	11.1	0.0	2009 – 2014
260	Valle de Cauca	0.08	11	50000	4545.45	11	11	0	1	0.0	9.1	2009 – 2014
261	Valle de Cauca	0.13	57	50000	877.19	57	57	0	0	0.0	0.0	2013 – 2018
262	Huila	12.81	14230	50000	35.14	1000	1000	0	9	0.0	0.9	2005 – 2010
263	Huila	0.1	69	50000	724.64	69	69	0	0	0.0	0.0	2014 – 2019
264	Cauca	0.02	25	50000	2000.00	25	25	5	0	20.0	0.0	2010 – 2015

Name of PPA	State	Area (km^2)	N_{ppa}	$N_{ca\ (100km^2)}$	N_{ca}/N_{ppa}	n_{ppa}	n_{ca}	n_{ppad}	n_{cad}	DR_{ppa} (%)	DR_{ca} (%)	Monitoring period
265	Cauca	0.17	87	50000	574.71	87	87	0	1	0.0	1.1	2006 – 2011
266	Santander	0.03	26	50000	1923.08	26	26	0	2	0.0	7.7	2010 – 2015
267	Huila	0.2	214	107000	500.00	214	214	2	1	0.9	0.5	2010 – 2015
268	Huila	0.73	775	387500	500.00	775	775	1	10	0.1	1.3	2005 – 2010
269	Huila	0.13	138	69000	500.00	138	138	0	3	0.0	2.2	2014 - 2019
270	Valle de Cauca	0.11	61	50000	819.67	61	61	0	0	0.0	0.0	2008 – 2013
271	Cundinamarca	0.37	276	138000	500.00	276	276	0	4	0.0	1.4	2001 – 2006

Table A.34 Statistics of 34 regional state PAs using a 100km² control area buffer: Number of sample pixels in PAs (N_{ppa}), number of sample pixels in Control Areas (N_{ca}) and N_{ca}/N_{pa} ratio before matching; Number of sample pixels in PAs (n_{pa}) and number of sample pixels in Control Areas (n_{ca}) after matching; Number of deforestation sample pixels in PAs (n_{ppd}) and number of deforestation sample pixels in Control Areas (n_{cad}) after matching; Deforestation rate in NRs (DR_{ppa}), deforestation rate in Control Areas (DR_{ca}) and monitoring period. Sample pixel size: 30m × 30m

Name of PA	State	Area (km²)	N_{pa}	N_{ca}	N_{ca}/N_{pa}	N_{pa}	n_{ca}	N_{pad}	n_{cad}	DR_{pa} (%)	DR_{ca} (%)	Monitoring period
Bosques de Misiguay	Santander	28.05	1000	50000	500	1000	1000	7	10	0.70	1.00	2014 – 2019
Cerro Banderas Ojo Blanco	Huila	249.45	1000	50000	500	1000	1000	8	10	0.80	1.00	2007 – 2012
Cerro la Judia	Santander	35.21	1000	50000	500	1000	1000	2	18	0.20	1.80	2009 – 2014
Cerro Paramo de Miraflores Rigoberto Urriago	Huila	316.47	1000	50000	500	1000	1000	2	8	0.20	0.80	2005 – 2010
Complejo Cienagas Papayal	Santander	28.05	1000	50000	500	1000	1000	24	33	2.40	3.30	
Corredor Biologico Guacharos Purace	Huila	711.09	1000	50000	500	1000	1000	2	15	0.20	1.50	2007 – 2012
Cuchillas Negra y Guanaque	Boyaca	193.05	1000	50000	500	1000	1000	0	1	0.00	0.10	
Del Nima	Valle de Cauca	16.98	1000	50000	500	1000	1000	4	5	0.40	0.50	2006 – 2011
El Vinculo	Valle de Cauca	0.84	905	452500	500	905	905	0	21	0.00	2.32	2006 – 2011
Esperanza del Mayo	Cauca	1.59	463	231500	500	463	463	0	1	0.00	0.22	2014 – 2019

Name of PA	State	Area (km^2)	N_{pa}	N_{ca}	N_{ca}/N_{pa}	N_{pa}	n_{ca}	N_{pad}	n_{cad}	DR_{pa} (%)	DR_{ca} (%)	Monitoring period
Humedales entre los Rios Leon y Suriqui	Antioquia	3.38	1000	50000	500	1000	1000	18	9	1.80	0.90	2009 – 2014
La Montana	Antioquia	19.07	1000	500000	500	1000	1000	31	70	3.10	7.00	2014 – 2019
La Selva	Antioquia	0.63	180	90000	500	180	180	0	4	0.00	2.22	2014 – 2019
La Sierpe	Valle de Cauca	182.93	1000	500000	500	1000	1000	1	15	0.10	1.50	2008 – 2013
Las Areas Naturales la Siberia y Parte de la Cuenca Alta del Rio las Ceibas	Huila	283.56	1000	500000	500	1000	1000	12	12	1.20	1.20	2007 – 2012
Laureles, Maracaibo y las Delicias	Cundinamarca	0.93	778	389000	500	778	778	0	3	0.00	0.39	2014 – 2019
Los Besotes	Cesar	0.55	433	216500	500	433	433	0	19	0.00	4.39	2013 – 2018
Metropolitano Cerro el Volador	Antioquia	1.04	941	470500	500	941	941	2	16	0.21	1.70	2009 – 2014
Pan de Azucar el Consuelo	Santander / Boyaca	97.2	1000	500000	500	1000	1000	1	2	0.10	0.20	2012 – 2017
Paramo de Rabanal	Boyaca	45.3	777	388500	500	777	777	2	11	0.26	1.42	2009 – 2014
Paramo de Santurban	Santander	43.61	1000	500000	500	1000	1000	0	1	0.00	0.10	2013 – 2018
Paramo del Duende	Valle de Cauca	145.38	1000	500000	500	1000	1000	0	0	0.00	0.00	2005 – 2010
Rionegro	Risaralda	1.97	1000	500000	500	1000	1000	0	6	0.00	0.60	2011 – 2016

Name of PA	**State**	**Area (km^2)**	**N_{pa}**	**N_{ca}**	**N_{ca}/N_{pa}**	**N_{pa}**	**n_{ca}**	**N_{pad}**	**n_{cad}**	**DR_{pa} (%)**	**DR_{ca} (%)**	**Monitoring period**
San Miguel de los Farallones	Casanare	33.79	1000	500000	500	1000	1000	3	10	0.30	1.00	2011 – 2016
Santa Emilia	Risaralda	5.29	1000	500000	500	1000	1000	1	11	0.10	1.10	2011 - 2016
Santurban Salazar de las Palmas	Norte de Santander	190.88	1000	500000	500	1000	1000	1	10	0.10	1.00	2013 – 2018
Serrania de las Quinchas	Santander	140.66	1000	500000	500	1000	1000	14	31	1.40	3.10	2009 – 2014
Serrania de las Quinchas	Boyaca	212.28	1000	500000	500	1000	1000	19	24	1.90	2.40	2008 – 2013
Serrania de Minas	Huila	290.92	1000	500000	500	1000	1000	11	12	1.10	1.20	2006 – 2011
Serrania el Peligro	Boyaca	24.27	1000	500000	500	1000	1000	2	6	0.20	0.60	2009 – 2014
Sisavita	Norte de Santander	112.73	1000	500000	500	1000	1000	4	4	0.40	0.40	2008 – 2013
Unidad Biogeografica de Siscunci Oceta	Boyaca	126.99	1000	500000	500	1000	1000	4	18	0.40	1.80	2008 – 2013
Verdeyaco el Oxigeno	Cauca	2.94	1000	500000	500	1000	1000	2	11	0.20	1.10	2012 - 2017
Verdum	Risaralda	5.75	1000	500000	500	1000	1000	0	9	0.00	0.90	2011 - 2016

Table A.35 Statistics of 34 regional state PAs using a 100km^2 control area buffer: Number of sample pixels in PAs (N_{ppa}), number of sample pixels in Control Areas (N_{ca}) and N_{ca}/N_{pa} ratio before matching; Number of sample pixels in PAs (n_{pa}) and number of sample pixels in Control Areas (n_{ca}) after matching; Number of deforestation sample pixels in PAs (n_{ppd}) and number of deforestation sample pixels in Control Areas (n_{cad}) after matching; Deforestation rate in NRs (DR_{ppa}), deforestation rate in Control Areas (DR_{ca}) and monitoring period. Sample pixel size: 30m

Name of PA	State	Area (km^2)	N_{pa}	N_{ca}	N_{ca}/N_{pa}	N_{pa}	n_{ca}	N_{pad}	n_{cad}	DR_{pa} (%)	DR_{ca} (%)	Monitoring period
Serrania de los Churumbelos Auka Wasi	Cauca	930.35	1000	500000	500	1000	1000	0	5	0.00	0.50	2007 - 2012
Serrania de los Yariguies	Santander	596.99	1000	500000	500	1000	1000	5	27	0.50	2.70	2005 - 2010
Plantas Medicinales Orito Ingi Ande	Narino	8.68	1000	500000	500	1000	1000	3	10	0.30	1.00	2008 - 2013
Complejo Volcanico Dona Juana Cascabel	Cauca	631.5	1000	500000	500	1000	1000	0	0	0.00	0.00	2007 - 2012
Alto Fragua Indi Wasi	Caqueta	762.04	1000	500000	500	1000	1000	3	19	0.30	1.90	2002 - 2017

Appendix D - Absolute and Relative Effects

Table A.36 Effectiveness of the 271 PPAs in maintaining forest cover during the period 2005–2019 using a control area buffer of 100km^2. Since the DR_{ca} of 113 PPAs equals zero (Supplementary Table S23), it is impossible to calculate the relative effects for them, which are indicated as N/As in the 'Relative effect' column.

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
1	Cundinamarca	0.62	0.68	100
2	Tolima	0.03	0.00	N/A
3	Cauca	0.14	0.00	N/A
4	Cauca	0.03	0.00	N/A
5	Valle de Cauca	0.06	0.00	N/A
6	Boyaca	1.09	0.28	66.67
7	Cauca	1.6	1.20	100
8	Boyaca / Santander	14.22	1.30	44.83
9	Cundinamarca	0.51	1.20	66.67
10	Cauca	0.09	0.00	N/A
11	Cauca	0.04	0.00	N/A
12	Cauca	0.01	0.00	N/A
13	Valle de Cauca	0.14	1.96	100
14	Cauca	0.08	0.00	100
15	Cauca	0.68	1.88	100.00
16	Huila	0.6	1.04	100.00
17	Tolima	0.3	1.57	100.00
18	Cundinamarca	3.49	0.10	100.00
19	Cundinamarca	1.14	1.50	100.00
20	Cundinamarca	0.18	0.10	75.00
21	Cundinamarca	0.48	1.50	-14.29

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
22	Cauca	0.14	1.52	N/A
23	Valle de Cauca	1.33	-0.69	0.00
24	Cundinamarca	2.93	0.00	66.67
25	Valle de Cauca	1.25	0.00	100.00
26	Antioquia	0.2	1.18	100.00
27	Boyaca	0.39	0.21	100.00
28	Valle de Cauca	0.14	3.14	100.00
29	Antioquia	10.19	0.26	100.00
30	Santander	5.23	2.68	100.00
31	Cauca	1.57	0.50	100
32	Valle de Cauca	2.29	3.44	66.67
33	Cauca	0.02	0.40	N/A
34	Cauca	0.08	0.40	100
35	Valle de Cauca	2.68	0.00	100
36	Caqueta	0.22	6.25	100
37	Cauca	0.01	0.70	N/A
38	Cauca	0.11	4.13	100
39	Valle de Cauca	0.12	0.00	N/A
40	Valle de Cauca	0.14	1.85	N/A
41	Cauca	0.04	0.00	100
42	Cauca	0.08	0.00	N/A
43	Huila	0.04	6.90	100

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
44	Valle de Cauca	0.57	0.00	-75.00
45	Huila	0.03	2.22	N/A
46	Cauca	0.23	-1.07	N/A
47	Huila	0.11	0.00	100.00
48	Huila	0.17	0.00	50.00
49	Huila	0.06	5.65	N/A
50	Cauca	0.02	0.54	N/A
51	Huila	0.61	0.00	100.00
52	Valle de Cauca	0.07	0.00	100.00
53	Huila	0.27	0.76	100.00
54	Valle de Cauca	0.11	1.41	45.45
55	Cauca	0.13	0.34	100.00
56	Huila	0.1	5.00	100.00
57	Huila	0.03	5.56	N/A
58	Huila	0.05	2.73	N/A
59	Huila	0.02	0.00	100.00
60	Cauca	0.29	-2.04	100.00
61	Cauca	0.59	4.35	-88.89
62	Cauca	1.06	3.17	76.92
63	Huila	0.35	-3.63	100.00
64	Cauca	0.12	1.23	N/A
65	Huila	0.61	0.77	69.23
66	Cauca	0.13	0.00	100.00
67	Cundinamarca	0.11	1.66	N/A

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
68	Cundinamarca	0.48	1.59	100.00
69	Huila	0.14	0.00	0.00
70	Cauca	0.14	0.38	N/A
71	Cundinamarca	0.02	0.00	N/A
72	Cauca	0.02	0.00	100.00
73	Cauca	0.13	0.00	100.00
74	Cauca	0.23	4.17	100.00
75	Huila	0.1	2.90	100.00
76	Cundinamarca	0.13	1.67	N/A
77	Cundinamarca	0.05	1.89	N/A
78	Cauca	0.49	0.00	N/A
79	Huila	0.4	-9.62	N/A
80	Valle de Cauca	1.93	0.00	100.00
81	Cauca	0.06	-0.68	N/A
82	Narino	0.21	0.30	80.00
83	Huila	0.26	0.00	-100.00
84	Huila	0.13	2.26	100.00
85	Valle de Cauca	0.95	-0.70	100.00
86	Huila	0.14	2.65	100.00
87	Valle de Cauca	0.27	0.27	N/A
88	Huila	0.19	1.27	100.00
89	Valle de Cauca	0.15	-6.12	N/A
90	Huila	0.21	0.94	100.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
91	Cauca	0.05	-4.55	N/A
92	Huila	0.4	0.44	100.00
93	Cauca	0.1	0.00	100.00
94	Valle de Cauca	0.16	1.91	N/A
95	Huila	0.03	2.13	100.00
96	Valle de Cauca	3.19	0.00	100.00
97	Huila	4.14	4.17	100.00
98	Huila	0.01	1.00	N/A
99	Santander	2.39	1.50	100.00
100	Santander	0.18	0.00	100.00
101	Tolima	74.20	3.21	52.00
102	Cundinamarca	1.48	0.57	71.43
103	Meta	0.1	1.30	N/A
104	Cauca	0.32	0.50	-75.00
105	Cundinamarca	0.06	0.00	100.00
106	Tolima	6.37	-1.74	-40.00
107	Huila	0.24	1.49	55.56
108	Cauca	0.09	-0.40	100.00
109	Huila	0.09	1.89	100.00
110	Cauca	0.08	2.11	N/A
111	Huila	0.03	3.03	N/A
112	Valle de Cauca	0.13	0.00	100.00
113	Cauca	0.52	0.00	100.00
114	Cauca	0.27	2.59	100.00
115	Cauca	0.28	2.74	-140.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
116	Huila	0.07	0.51	N/A
117	Cauca	0.06	-3.27	N/A
118	Meta	0.14	0.00	100.00
119	Cauca	0.04	0.00	N/A
120	Cauca	0.02	1.64	100.00
121	Valle de Cauca	0.04	0.00	N/A
122	Valle de Cauca	0.02	5.56	N/A
123	Cauca	0.09	0.00	100.00
124	Cundinamarca	1.46	0.00	66.67
125	Huila	0.44	1.92	100.00
126	Cundinamarca	0.26	0.40	100.00
127	Huila	0.05	1.85	N/A
128	Cauca	0.12	0.35	100.00
129	Cauca	0.11	0.00	N/A
130	Valle de Cauca	0.03	1.09	100.00
131	Cauca	0.03	0.00	100.00
132	Cundinamarca	0.13	4.17	100.00
133	Cundinamarca	0.09	6.38	100.00
134	Valle de Cauca	0.72	1.56	N/A
135	Valle de Cauca	0.02	0.00	N/A
136	Narino	0.2	0.00	100.00
137	Valle de Cauca	1.02	1.83	100.00
138	Tolima	0.83	1.49	85.00
139	Cauca	0.23	1.94	-250.00
140	Valle de Cauca	1.5	-2.36	-20.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
141	Cauca	0.27	-0.14	33.33
142	Cauca	0.8	0.70	100.00
143	Huila	0.2	1.47	80.00
144	Huila	0.02	1.79	N/A
145	Cauca	0.08	0.00	100.00
146	Valle de Cauca	0.25	1.14	N/A
147	Valle de Cauca	0.07	-2.53	N/A
148	Cauca	0.16	0.00	N/A
149	Cauca	0.12	0.00	N/A
150	Cauca	0.12	0.00	N/A
151	Cauca	0.09	0.00	N/A
152	Cundinamarca	1.16	0.00	71.43
153	Cauca	0.03	0.50	100.00
154	Valle de Cauca	0.07	3.57	-500.00
155	Cauca	0.13	-17.24	100.00
156	Huila	0.01	0.99	N/A
157	Valle de Cauca	0.29	0.00	N/A
158	Cundinamarca	0.03	0.00	N/A
159	Huila	0.03	-12.50	N/A
160	Valle de Cauca	0.26	0.00	100
161	Valle de Cauca	0.08	3.61	N/A
162	Tolima	0.22	0.00	100.00
163	Valle de Cauca	0.19	3.40	N/A
164	Cauca	0.08	0.00	100.00
165	Cauca	0.09	0.00	100.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
166	Huila	0.03	5.05	N/A
167	Valle de Cauca	0.32	0.00	N/A
168	Tolima	0.1	-2.73	100.00
169	Cauca	0.08	11.25	N/A
170	Valle de Cauca	0.04	0.00	N/A
171	Huila	0.01	0.00	N/A
172	Cauca	0.08	0.00	N/A
173	Cauca	0.05	-14.29	100.00
174	Valle de Cauca	0.69	9.68	100.00
175	Cauca	0.06	1.72	N/A
176	Cauca	0.19	0.00	100.00
177	Huila	0.02	3.43	N/A
178	Cauca	0.06	0.00	N/A
179	Cauca	0.16	0.00	N/A
180	Valle de Cauca	0.27	0.00	100.00
181	Cundinamarca	0.06	3.17	N/A
182	Narino	0.3	0.00	N/A
183	Huila	0.03	0.00	N/A
184	Cauca	0.04	-10.00	N/A
186	Cundinamarca	0.06	-1.27	N/A
187	Casanare	4.15	0.00	90.91
188	Cauca	0.07	2.00	N/A
189	Huila	0.17	-5.66	N/A
190	Cauca	0.08	0.00	N/A
191	Boyaca	0.04	0.00	N/A

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
192	Huila	0.04	0.00	100.00
193	Valle de Cauca	2.92	9.09	-100.00
194	Cauca	0.11	-0.80	100.00
195	Cundinamarca	0.02	1.30	N/A
196	Cundinamarca	0.32	0.00	100.00
197	Cauca	0.27	0.60	100.00
198	Narino	0.34	1.82	N/A
199	Huila	0.21	-0.27	66.67
200	Boyaca	0.08	1.77	100.00
201	Santander	3.25	2.27	100.00
202	Cauca	0.12	0.30	N/A
203	Cauca	0.05	0.00	N/A
204	Cauca	0.04	0.00	N/A
205	Cauca	1.02	0.00	100.00
206	Cauca	0.14	3.30	100.00
207	Cauca	0.04	2.52	N/A
208	Cauca	0.05	0.00	N/A
209	Cauca	0.08	0.00	N/A
210	Cauca	0.04	0.00	N/A
211	Cauca	0.03	4.00	0.00
212	Cauca	0.12	0.00	N/A
213	Cauca	0.09	-29.41	-200.00
214	Cauca	0.04	0.00	33.33
215	Cauca	0.07	0.00	N/A
216	Cauca	0.38	-2.78	100.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
217	Cauca	0.12	0.51	100.00
218	Cundinamarca	1.8	0.00	85.71
219	Valle de Cauca	0.08	1.20	N/A
220	Cundinamarca	0.64	1.43	0.00
221	Valle de Cauca	0.17	1.52	100.00
222	Valle de Cauca	0.04	2.50	N/A
223	Valle de Cauca	0.76	-12.90	100.00
224	Cauca	0.02	0.31	100.00
225	Huila	0.01	5.56	N/A
226	Huila	0.02	0.00	N/A
227	Huila	0.01	0.00	N/A
228	Huila	0.02	0.00	N/A
229	Huila	0.02	0.00	N/A
230	Huila	0.01	0.00	N/A
231	Huila	0.01	0.00	N/A
232	Narino	5.51	0.00	-75.00
233	Cauca	0.14	-0.90	N/A
234	Narino	0.18	-0.92	N/A
235	Cauca	0.26	0.00	0.00
236	Valle de Cauca	0.06	0.00	N/A
237	Huila	0.03	0.00	N/A
238	Valle de Cauca	0.69	0.00	90.00
239	Antioquia	11.58	1.31	80.65
240	Cundinamarca	0.02	2.50	N/A
241	Boyaca	0.29	0.00	100.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
242	Valle de Cauca	0.28	0.73	100.00
243	Valle de Cauca	1.97	0.32	100.00
244	Cauca	0.06	1.79	N/A
245	Valle de Cauca	0.49	0.00	N/A
246	Boyaca	1.55	0.00	N/A
247	Cauca	0.03	0.00	0.00
248	Valle de Cauca	0.22	0.00	100.00
249	Cauca	0.41	0.45	N/A
250	Valle de Cauca	0.33	0.00	100.00
251	Cauca	0.17	0.78	-100.00
252	Cauca	0.05	-1.52	100.00
253	Valle de Cauca	0.29	2.17	100.00
254	Cauca	0.16	1.27	100.00
255	Cauca	0.29	3.17	N/A
256	Cauca	0.16	-0.93	100.00
257	Cauca	0.18	3.23	100.00
258	Huila	0.14	1.96	N/A
259	Valle de Cauca	0.35	0.00	N/A
260	Valle de Cauca	0.08	-11.11	100.00
261	Valle de Cauca	0.13	9.09	N/A
262	Huila	12.81	0.00	100.00
263	Huila	0.1	0.90	N/A
264	Cauca	0.02	0.00	N/A
265	Cauca	0.17	-20.00	100.00
266	Santander	0.03	1.15	100.00

Name of PPA	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
267	Huila	0.2	7.69	-100.00
268	Huila	0.73	-0.47	90.00
269	Huila	0.13	1.16	100.00
270	Valle de Cauca	0.11	2.17	N/A
271	Cundinamarca	0.37	0.00	100.00

Table A.37 Effectiveness of the 34 Regional State PAs in maintaining forest cover during the period 2005–2019 using a control area buffer of 100km^2.

Regional state PA ID	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
1	Santander	28.05	0.3	30.00
2	Huila	249.45	0.2	20.00
3	Santander	35.21	1.6	88.89
4	Huila	316.47	0.6	75.00
5	Santander	28.05	0.9	27.27
6	Huila	711.09	1.3	86.67
7	Boyaca	193.05	0.1	100.00
8	Valle de Cauca	16.98	0.1	20.00
9	Valle de Cauca	0.84	2.32	100.00
10	Cauca	1.59	0.22	100.00
11	Antioquia	3.38	-0.9	-100.00
12	Antioquia	19.07	3.9	55.71
13	Antioquia	0.63	2.22	100.00
14	Valle de Cauca	182.93	1.4	93.33
15	Huila	283.56	0.0	0.00
16	Cundinamarca	0.93	0.4	100.00
17	Cesar	0.55	4.4	100.00
18	Antioquia	1.04	1.5	87.50
19	Santander / Boyaca	97.2	0.1	50.00
20	Boyaca	45.3	1.2	81.82
21	Santander	43.61	0.1	100.00
22	Valle de Cauca	145.38	0.0	0.00
23	Risaralda	1.97	0.6	100.00
24	Casanare	33.79	0.7	70.00

Regional state PA ID	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
25	Risaralda	5.29	1.0	90.91
26	Norte de Santander	190.88	0.9	90.00
27	Santander	140.66	1.7	54.84
28	Boyaca	212.28	0.5	20.83
29	Huila	290.92	0.1	8.33
30	Boyaca	24.27	0.4	66.67
31	Norte de Santander	112.73	0.0	0.00
32	Boyaca	126.99	1.4	77.78
33	Cauca	2.94	0.9	81.82
34	Risaralda	5.75	0.9	100.00

Table A.38 Effectiveness of the 5 National PAs in maintaining forest cover during the period 2005–2019 using a control area buffer of 100km2. Since the DRca of 1 national PA equals zero (Supplementary Table S25), it is impossible to calculate the relative effects for them, which are indicated as N/As in the 'Relative effect' column.

Regional state PA ID	State	Area (km^2)	Absolute effect (%)	Relative effect (%)
Serrania de los Churumbelos Auka Wasi	Cauca	930.35	0.5	100.00
Serrania de los Yariguies	Santander	596.99	2.2	81.48
Plantas Medicinales Orito Ingi Ande	Narino	8.68	0.7	70.00
Complejo Volcanico Dona Juana Cascabel	Cauca	631.5	0.0	N/A
Alto Fragua Indi Wasi	Caqueta	762.04	1.6	84.21

Table A.39 Descriptive and inferential statistics of Protected Areas relative effects

Protected Area	n	Mean (%)	Min (%)	Max (%)	SD (%)	SE (%)	Lower bound of 95% CI (%)	Upper bound of 95% CI (%)
PPAs	271	69.57	-500	100	76.66	6.16	57.25	81.88
Regional State PAs	34	61.10	-100	100	45.12	7.74	45.62	76.58
National PAs	5	83.92	70	100	12.36	6.18	71.56	96.28

Appendix D - PA Management Category by PA Governance Type

Table A.40 Break down of PA management Category by PA Governance Type for all Protected Areas reported to the WDPA in Colombia (March 2022)

PA Governance Type	PA Management Category	Count	Percentage (%)
National State PA	Ia	2	2
	II	35	32
	IV	10	9
	VI	59	54
	N/A	3	2
Regional State PA	II	58	21
	V	26	9
	VI	196	70
Privately Protected Area	VI	912	100

Appendix D - Proportion of each PA governance type within each probability of deforestation level

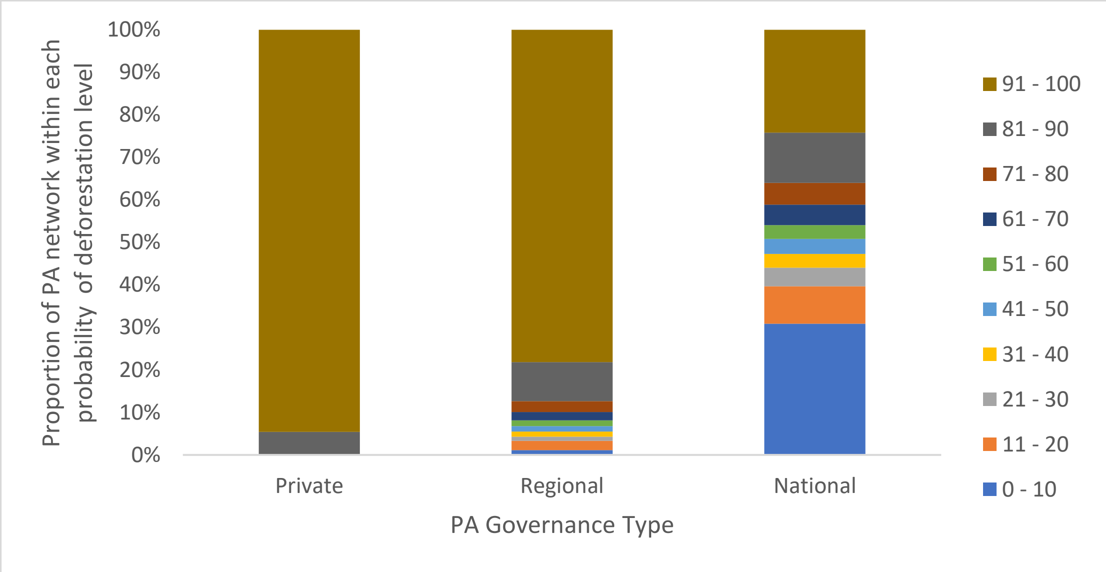

Figure A.12 Proportion of each PA governance type within each probability of deforestation level

This map was produced by overlaying PA polygon boundaries on a deforestation probability map from (Negret et al., 2019). The total area of each PA governance type within each deforestation probability score (0 – 100) was calculated in ArcMap10.4. Probabilities were calculated by dividing the total area of each PA governance type within each probability of deforestation score by the total area of each PA governance type.

Reference:

Negret, P. J., Sonter, L., Watson, J. E. M., Possingham, H. P., Kendall, R., Suarez, C., Ochoa-Quintero, J. M., Maron, M. (2019) Emerging evidence that armed conflict and coca cultivation influence deforestation patterns, PANGAEA, https://doi.org/10.1594/PANGAEA.899573,

Appendix D - Uses of Private Protected Areas

Table A.41 Area and % of PPA's dedicated to conservation, buffer zones, agricultural systems and intensive use and infrastructures

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
1	23.04	23.04	7.83	7.83	57.60	57.60	11.52	11.52
2	28.63	28.63	0.00	0.00	42.75	42.75	28.63	28.63
3	86.36	86.36	0.00	0.00	13.18	13.18	0.45	0.45
4	33.33	33.33	0.00	0.00	63.33	63.33	3.33	3.33
5	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
6	40.00	40.00	10.00	10.00	50.00	50.00	0.00	0.00
7	33.08	33.08	0.00	0.00	66.15	66.15	0.77	0.77
8	90.00	90.00	5.00	5.00	0.00	0.00	5.00	5.00
9	81.25	81.25	0.00	0.00	17.86	17.86	0.89	0.89
10	16.67	16.67	0.00	0.00	81.67	81.67	1.67	1.67
11	33.33	33.33	0.00	0.00	63.33	63.33	3.33	3.33
12	97.63	97.63	0.00	0.00	2.30	2.30	0.08	0.08

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
13	51.10	51.10	4.70	4.70	44.16	44.16	0.04	0.04
14	20.00	20.00	0.00	0.00	79.67	79.67	0.33	0.33
15	40.00	40.00	10.00	10.00	50.00	50.00	0.00	0.00
16	94.00	94.00	0.00	0.00	6.00	6.00	0.00	0.00
17	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
18	78.28	78.28	14.79	14.79	0.00	0.00	6.93	6.93
19	99.04	99.04	0.00	0.00	0.00	0.00	0.96	0.96
20	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
21	36.94	36.94	14.33	14.33	48.73	48.73	0.00	0.00
22	39.14	39.14	0.00	0.00	58.71	58.71	2.15	2.15
23	43.00	43.00	43.00	43.00	13.98	13.98	0.02	0.02
24	76.06	76.06	23.94	23.94	0.00	0.00	0.00	0.00
25	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
26	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
27	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
28	33.28	33.28	14.09	14.09	51.24	51.24	1.39	1.39
29	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
30	99.98	99.98	0.00	0.00	0.00	0.00	0.02	0.02
31	94.93	94.93	0.00	0.00	0.00	0.00	5.07	5.07
32	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
33	14.29	14.29	84.76	84.76	0.00	0.00	0.95	0.95
34	83.33	83.33	6.67	6.67	8.83	8.83	1.17	1.17
35	9.65	9.65	20.35	20.35	69.83	69.83	0.17	0.17
36	40.76	40.76	19.15	19.15	33.67	33.67	6.42	6.42
37	16.67	16.67	0.00	0.00	83.17	83.17	0.17	0.17
38	25.00	25.00	0.00	0.00	74.75	74.75	0.25	0.25
39	23.17	23.17	0.00	0.00	76.83	76.83	0.00	0.00
40	44.05	44.05	0.00	0.00	55.95	55.95	0.00	0.00

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
41	34.48	34.48	0.00	0.00	55.17	55.17	10.34	10.34
42	17.10	17.10	0.00	0.00	76.97	76.97	5.93	5.93
43	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
44	45.16	45.16	1.86	1.86	52.98	52.98	0.00	0.00
45	33.33	33.33	0.00	0.00	66.67	66.67	0.00	0.00
46	21.45	21.45	0.00	0.00	77.69	77.69	0.86	0.86
47	50.00	50.00	17.86	17.86	29.46	29.46	2.68	2.68
48	66.66	66.66	0.00	0.00	33.34	33.34	0.00	0.00
49	37.50	37.50	25.00	25.00	37.50	37.50	0.00	0.00
50	11.96	11.96	0.00	0.00	83.25	83.25	4.78	4.78
51	65.26	65.26	10.85	10.85	23.47	23.47	0.42	0.42
52	36.51	36.51	3.94	3.94	59.25	59.25	0.30	0.30
53	83.33	83.33	16.67	16.67	0.00	0.00	0.00	0.00
54	14.38	14.38	6.47	6.47	79.06	79.06	0.09	0.09

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
55	46.15	46.15	0.00	0.00	53.85	53.85	0.00	0.00
56	50.00	50.00	0.00	0.00	50.00	50.00	0.00	0.00
57	16.67	16.67	16.67	16.67	63.33	63.33	3.33	3.33
58	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
59	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
60	39.80	39.80	0.00	0.00	60.15	60.15	0.05	0.05
61	2.68	2.68	27.53	27.53	69.70	69.70	0.09	0.09
62	42.86	42.86	14.29	14.29	42.86	42.86	0.00	0.00
63	9.09	9.09	0.00	0.00	90.82	90.82	0.09	0.09
64	33.11	33.11	1.99	1.99	64.90	64.90	0.00	0.00
65	19.61	19.61	0.00	0.00	78.51	78.51	1.87	1.87
66	90.00	90.00	0.00	0.00	10.00	10.00	0.00	0.00
67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
68	53.85	53.85	15.38	15.38	30.77	30.77	0.00	0.00

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
69	25.25	25.25	0.00	0.00	70.76	70.76	3.99	3.99
70	87.78	87.78	5.99	5.99	5.99	5.99	0.25	0.25
71	11.00	11.00	0.00	0.00	89.00	89.00	0.00	0.00
72	73.33	73.33	0.00	0.00	23.08	23.08	3.60	3.60
73	6.25	6.25	0.00	0.00	89.58	89.58	4.17	4.17
74	30.00	30.00	20.00	20.00	50.00	50.00	0.00	0.00
75	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
76	60.00	60.00	40.00	40.00	0.00	0.00	0.00	0.00
77	63.64	63.64	0.00	0.00	36.36	36.36	0.00	0.00
78	54.05	54.05	0.00	0.00	45.93	45.93	0.01	0.01
79	41.22	41.22	0.00	0.00	58.31	58.31	0.47	0.47
80	16.39	16.39	0.00	0.00	72.79	72.79	10.82	10.82
81	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
82	13.51	13.51	8.95	8.95	77.48	77.48	0.05	0.05

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
83	50.00	50.00	20.00	20.00	30.00	30.00	0.00	0.00
84	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
85	83.33	83.33	0.00	0.00	16.67	16.67	0.00	0.00
86	36.20	36.20	0.00	0.00	63.45	63.45	0.35	0.35
87	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
88	50.00	50.00	0.00	0.00	47.50	47.50	2.50	2.50
89	45.47	45.47	6.06	6.06	45.47	45.47	3.00	3.00
90	33.52	33.52	0.00	0.00	61.45	61.45	5.03	5.03
91	11.97	11.97	1.95	1.95	85.26	85.26	0.82	0.82
92	14.00	14.00	0.00	0.00	71.00	71.00	15.00	15.00
93	81.85	81.85	2.06	2.06	16.06	16.06	0.03	0.03
94	19.53	19.53	9.41	9.41	67.53	67.53	3.53	3.53
95	29.00	29.00	0.00	0.00	64.00	64.00	7.00	7.00
96	5.90	5.90	31.46	31.46	40.16	40.16	22.48	22.48

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
97	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
98	62.45	62.45	6.72	6.72	30.67	30.67	0.17	0.17
99	32.54	32.54	24.14	24.14	41.78	41.78	1.54	1.54
100	54.36	54.36	44.48	44.48	0.00	0.00	1.16	1.16
101	9.94	9.94	0.00	0.00	89.73	89.73	0.33	0.33
102	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
103	36.73	36.73	12.06	12.06	49.81	49.81	1.40	1.40
104	33.33	33.33	13.33	13.33	53.33	53.33	0.00	0.00
105	10.31	10.31	0.00	0.00	89.59	89.59	0.10	0.10
106	16.67	16.67	0.00	0.00	81.25	81.25	2.08	2.08
107	14.58	14.58	29.16	29.16	55.97	55.97	0.29	0.29
108	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
109	37.74	37.74	10.44	10.44	51.48	51.48	0.34	0.34
110	24.92	24.92	0.00	0.00	74.77	74.77	0.31	0.31

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
111	50.00	50.00	0.00	0.00	49.72	49.72	0.28	0.28
112	39.50	39.50	0.00	0.00	60.50	60.50	0.00	0.00
113	66.67	66.67	0.00	0.00	33.33	33.33	0.00	0.00
114	38.17	38.17	0.00	0.00	61.07	61.07	0.76	0.76
115	30.27	30.27	9.42	9.42	36.43	36.43	23.89	23.89
116	13.42	13.42	0.00	0.00	85.34	85.34	1.24	1.24
117	29.20	29.20	0.00	0.00	70.80	70.80	0.00	0.00
118	9.99	9.99	0.00	0.00	35.49	35.49	54.52	54.52
119	22.00	22.00	3.00	3.00	72.00	72.00	3.00	3.00
120	35.19	35.19	0.00	0.00	64.52	64.52	0.29	0.29
121	29.56	29.56	26.60	26.60	30.05	30.05	13.79	13.79
122	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
123	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
124	40.65	40.65	0.25	0.25	49.38	49.38	9.72	9.72

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
125	17.35	17.35	0.00	0.00	79.22	79.22	3.43	3.43
126	16.27	16.27	0.00	0.00	81.56	81.56	2.17	2.17
127	30.30	30.30	30.30	30.30	30.30	30.30	9.09	9.09
128	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
129	42.20	42.20	57.80	57.80	0.00	0.00	0.00	0.00
130	59.80	59.80	6.75	6.75	33.29	33.29	0.16	0.16
131	89.34	89.34	4.18	4.18	0.27	0.27	6.21	6.21
132	84.10	84.10	0.00	0.00	15.79	15.79	0.10	0.10
133	41.80	41.80	0.00	0.00	54.80	54.80	3.40	3.40
134	51.77	51.77	6.79	6.79	41.34	41.34	0.10	0.10
135	41.02	41.02	0.00	0.00	58.71	58.71	0.27	0.27
136	2.08	2.08	0.00	0.00	97.92	97.92	0.00	0.00
137	23.03	23.03	0.00	0.00	76.78	76.78	0.19	0.19
138	60.34	60.34	0.00	0.00	38.62	38.62	1.04	1.04

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
139	78.95	78.95	0.00	0.00	21.05	21.05	0.00	0.00
140	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
141	98.20	98.20	0.28	0.28	1.50	1.50	0.02	0.02
142	79.77	79.77	0.00	0.00	15.11	15.11	5.11	5.11
143	10.46	10.46	0.00	0.00	89.54	89.54	0.00	0.00
144	100.00	100.00	0.00	0.0	0.00	0.00	0.00	0.00
145	26.35	26.35	0.73	0.73	69.09	69.09	3.82	3.82
146	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
147	16.63	16.63	0.00	0.00	83.37	83.37	0.00	0.00
148	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
149	2.28	2.28	0.00	0.00	96.15	96.15	1.57	1.57
150	30.60	30.60	2.15	2.15	65.91	65.91	1.34	1.34
151	9.69	9.69	0.00	0.00	90.07	90.07	0.24	0.24
152	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
153	15.25	15.25	8.19	8.19	76.56	76.56	0.00	0.00
154	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
155	16.67	16.67	8.33	8.33	75.00	75.00	0.00	0.00
156	46.67	46.67	21.12	21.12	32.21	32.21	0.00	0.00
157	18.83	18.83	0.00	0.00	79.58	79.58	1.59	1.59
158	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
159	47.16	47.16	1.22	1.22	50.52	50.52	1.10	1.10
160	15.60	15.60	0.00	0.00	83.60	83.60	0.80	0.80
161	36.27	36.27	10.36	10.36	25.91	25.91	27.46	27.46
162	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
163	2.58	2.58	2.77	2.77	87.16	87.16	7.49	7.49
164	34.48	34.48	0.00	0.00	0.00	0.00	0.00	0.00
165	65.57	65.57	0.00	0.00	32.79	32.79	1.64	1.64
166	58.78	58.78	5.31	5.31	35.30	35.30	0.60	0.60

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
167	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
168	9.57	9.57	0.00	0.00	89.95	89.95	0.48	0.48
169	20.51	20.51	0.00	0.00	72.82	72.82	6.67	6.67
170	65.39	65.39	4.73	4.73	29.01	29.01	0.87	0.87
171	16.70	16.70	0.00	0.00	73.82	73.82	9.47	9.47
172	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
173	9.80	9.80	0.00	0.00	88.24	88.24	1.96	1.96
174	19.58	19.58	0.00	0.00	76.80	76.80	3.62	3.62
175	24.05	24.05	0.00	0.00	72.30	72.30	3.64	3.64
176	66.92	66.92	0.00	0.00	31.57	31.57	1.51	1.51
177	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
178	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
179	34.48	34.48	34.48	34.48	31.03	31.03	0.00	0.00
180	9.62	9.62	0.00	0.00	86.54	86.54	3.85	3.85

PPA ID	Area for conservation (Has)	Percentage of area for conservation (%)	Area for buffer zones (Has)	Percentage of area for buffer zones (%)	Area for Agricultural systems (Has)	Percentage of area for Agricultural Systems (%)	Area for Intensive use and Infrastructures (Has)	Percentage of area for Intensive use and Infrastructures (%)
181	9.52	9.52	0.00	0.00	90.48	90.48	0.00	0.00
182	41.80	41.80	7.62	7.62	47.07	47.07	3.51	3.51
183	45.23	45.23	39.35	39.35	15.42	15.42	0.00	0.00
184	40.82	40.82	0.00	0.00	54.42	54.42	4.76	4.76
185	77.28	77.28	0.00	0.00	19.77	19.77	2.95	2.95
186	59.06	59.06	0.00	0.00	39.37	39.37	1.57	1.57
187	8.00	8.00	32.00	32.00	35.00	35.00	25.00	25.00
188	33.33	33.33	0.00	0.00	66.67	66.67	0.00	0.00
189	15.91	15.91	79.37	79.37	4.05	4.05	0.67	0.67
190	32.91	32.91	0.00	0.00	64.98	64.98	2.11	2.11
191	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
192	91.06	91.06	8.94	8.94	0.00	0.00	0.00	0.00
193	16.38	16.38	0.00	0.00	83.38	83.38	0.24	0.24
194	98.50	98.50	0.00	0.00	1.50	1.50	0.00	0.00

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195	15.00	15.00	20.00	20.00	62.50	62.50	2.50	2.50
196	87.10	87.10	0.00	0.00	12.90	12.90	0.00	0.00
197	32.82	32.82	65.02	65.02	2.16	2.16	0.00	0.00
198	15.86	15.86	9.41	9.41	74.56	74.56	0.17	0.17
199	75.94	75.94	10.91	10.91	11.94	11.94	1.21	1.21
200	42.62	42.62	15.17	15.17	42.21	42.21	0.00	0.00
201	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
202	48.36	48.36	10.92	10.92	40.52	40.52	0.19	0.19
203	11.06	11.06	2.81	2.81	79.51	79.51	6.62	6.62
204	25.28	25.28	0.00	0.00	74.72	74.72	0.00	0.00
205	2.27	2.27	0.00	0.00	93.07	93.07	4.67	4.67
206	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
207	20.41	20.41	0.00	0.00	79.59	79.59	0.00	0.00
208	9.10	9.10	66.04	66.04	24.86	24.86	0.00	0.00

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209	16.14	16.14	40.84	40.84	43.02	43.02	0.00	0.00
210	12.47	12.47	0.00	0.00	87.53	87.53	0.00	0.00
211	29.91	29.91	7.03	7.03	62.89	62.89	0.16	0.16
212	19.13	19.13	0.00	0.00	80.87	80.87	0.00	0.00
213	29.99	29.99	31.93	31.93	38.08	38.08	0.00	0.00
214	83.33	83.33	10.42	10.42	0.00	0.00	6.25	6.25
215	81.11	81.11	0.00	0.00	14.73	14.73	4.16	4.16
216	72.59	72.59	0.00	0.00	0.00	0.00	27.41	27.41
217	32.56	32.56	0.00	0.00	61.89	61.89	5.56	5.56
218	9.81	9.81	0.00	0.00	87.80	87.80	2.39	2.39
219	25.89	25.89	3.75	3.75	70.36	70.36	0.00	0.00
220	1.73	1.73	2.97	2.97	93.23	93.23	2.07	2.07
221	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
222	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00

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223	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
224	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
225	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
226	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
227	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
228	0.33	0.33	0.00	0.00	74.75	74.75	24.92	24.92
229	11.08	11.08	0.00	0.00	88.64	88.64	0.28	0.28
230	25.00	25.00	35.00	35.00	35.00	35.00	5.00	5.00
231	23.04	23.04	0.00	0.00	76.80	76.80	0.15	0.15
232	65.00	65.00	0.00	0.00	0.00	0.00	35.00	35.00
234	33.33	33.33	0.00	0.00	66.67	66.67	0.00	0.00
235	82.86	82.86	0.00	0.00	14.93	14.93	2.21	2.21
236	36.33	36.33	42.15	42.15	21.07	21.07	0.44	0.44
237	31.45	31.45	0.00	0.00	0.00	0.00	68.55	68.55

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238	31.03	31.03	17.24	17.24	51.72	51.72	0.00	0.00
239	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
240	18.24	18.24	1.00	1.00	80.75	80.75	0.01	0.01
241	33.06	33.06	0.00	0.00	61.18	61.18	5.76	5.76
242	99.90	99.90	0.00	0.00	0.00	0.00	0.10	0.10
243	92.98	92.98	7.02	7.02	0.00	0.00	0.00	0.00
244	33.33	33.33	0.00	0.00	66.67	66.67	0.00	0.00
245	24.47	24.47	20.16	20.16	55.00	55.00	0.38	0.38
246	46.54	46.54	0.00	0.00	44.47	44.47	9.00	9.00
247	79.79	79.79	12.56	12.56	7.66	7.66	0.00	0.00
248	13.75	13.75	0.00	0.00	84.19	84.19	2.06	2.06
249	24.39	24.39	0.00	0.00	60.98	60.98	14.63	14.63
250	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
251	11.30	11.30	0.00	0.00	84.75	84.75	3.95	3.95

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252	14.59	14.59	0.00	0.00	84.26	84.26	1.15	1.15
253	17.24	17.24	0.00	0.00	80.72	80.72	2.04	2.04
254	86.11	86.11	3.97	3.97	7.94	7.94	1.98	1.98
255	40.87	40.87	0.00	0.00	55.22	55.22	3.91	3.91
256	18.01	18.01	1.88	1.88	80.12	80.12	0.00	0.00
257	25.38	25.38	4.69	4.69	69.93	69.93	0.00	0.00
258	41.60	41.60	15.00	15.00	42.60	42.60	0.80	0.80
259	100.00	100.00	0.00	0.00	0.00	0.00	0.00	0.00
260	44.13	44.13	0.00	0.00	44.18	44.18	11.69	11.69
261	3.30	3.30	32.74	32.74	62.54	62.54	1.43	1.43
262	20.03	20.03	0.00	0.00	76.93	76.93	3.03	3.03
263	99.38	99.38	0.00	0.00	0.00	0.00	0.62	0.62
264	84.77	84.77	0.00	0.00	13.12	13.12	2.11	2.11
265	78.95	78.95	1.91	1.91	19.14	19.14	0.00	0.00

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266	39.00	39.00	0.00	0.00	56.00	56.00	5.00	5.00
267	29.18	29.18	4.91	4.91	64.64	64.64	1.27	1.27
268	36.78	36.78	50.41	50.41	10.90	10.90	1.91	1.91
269	55.56	55.56	44.44	44.44	0.00	0.00	0.00	0.00