# Effectiveness of protected areas in the deforestation fronts of Acre, Brazil

Teemu Koskimäki

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Department of Biology
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Deforestation in Amazonia is a high priority issue as it is intimately connected to global environmental problems, such as biodiversity loss and climate change. Contemporary threat from deforestation is concentrated on deforestation fronts, and protected areas (PAs) are the main tool used to combat the threat. PAs are an important investment of time and resources, making it important to know how effective they are in achieving their purpose. I selected PAs near the deforestation fronts in the state of Acre, Brazil, and applied a state-of-the-art matching method to study their effectiveness during 2011-2016, after Acre had begun the world's first jurisdictional REDD+ program. I calculated estimates of deforestation pressure and effectiveness for each PA separately and used these to calculate how much deforestation and $CO_2$ emissions each PA had avoided. I found substantial variation in the effectiveness estimates between individual PAs, corroborating previous matching studies, but the main protection types did not differ from each other in effectiveness in Acre. I found that three PAs currently exist in areas of very high pressure. During the study period, the PAs of Acre avoided a substantial amount of deforestation and carbon emissions and were therefore able to conserve much of the cherished services and functions the rainforests of Acre provide.

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Keywords: Amazon, arc-of-deforestation, biodiversity, climate change, counterfactual matching, impact evaluation, payments for ecosystem services

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# 1. Introduction

## *1.1. Tropical forests and a changing world*

Forests not only harbour the majority of terrestrial species (FAO, 2016), but also provide indispensable provisioning, regulating, supporting and cultural services for humanity (MEA, 2005; WWF, 2016). However, for thousands of years the process of economic development has pushed humankind to convert forests to agricultural use (FAO, 2016). Until the late 1800's, forest loss was most prevalent in the northern temperate zone, whereas these days most of the forest conversion to other land uses takes place in the tropical domain where intact forests still stand, particularly in South America and Africa (MacDicken *et al.*, 2015; FAO, 2016). Human influence has lead the global forest cover to reduce by 40% during historical times (Mace *et al.*, 2005) and today, forests cover around a third of the global land area, a third of which has remained as primary forests where ecological processes have not been significantly disturbed by human activities (FAO FRA, 2012; MacDicken *et al.*, 2015). Around half of these primary forests are located in the tropics, and the largest extent of primary forests can be found in South America (MacDicken *et al.*, 2015), although the extent to which these forests are truly without human influence has lately been under debate (Levis *et al.*, 2017). Presently, humans appropriate a quarter of the earth's net primary productivity (Krausmann *et al.*, 2013) and as we continue to appropriate ever more productive land area, habitats of other species are lost, presenting the greatest contemporary threat to biodiversity globally (MEA, 2005; WWF, 2016).

## *1.2. Deforestation*

The result of the historical and present forest loss on a global scale has been the fact that today, the planetary boundary for land-use change – measured by the amount of remaining forest cover – has been estimated to be in the zone of uncertainty and not inside the safe operating space within which human societies can develop and thrive (Steffen *et al.*, 2015). Deforestation, therefore, is an issue of high relevance and priority, especially in the tropics. For example, around 17.7 million hectares of the Amazon forest cover, roughly equal to the area of Austria and Portugal combined, were deforested between 2001-2012, mostly due to forest conversion for cattle ranching and soy production in Brazil (WWF, 2015b).

Deforestation is defined by the Food and Agriculture Organization of the United Nations (FAO) as “the conversion of forest to other land use or the permanent reduction of the tree canopy cover below the minimum 10 percent threshold” (FAO FRA, 2012). Similarly, the World Wide Fund for Nature (WWF) defines deforestation as “conversion of forest to another land use or significant long-term reduction of tree canopy cover. This includes conversion of natural forest to tree plantations, agriculture, pasture, water reservoirs and urban areas; but excludes logging areas where the forest is managed to regenerate naturally or with the aid of silvicultural measures” (WWF, 2015b). To define a “significant reduction”, WWF references the 10% threshold set by the FAO.

When writing about deforestation in his article in 1993, Norman Myers conveyed the importance of this issue eloquently: “This represents a largely irreversible loss of unique stocks of natural resources that could supply self-renewing goods and services to all Humankind in perpetuity. In view of the manifold benefits that are available from the forests, their progressive depletion must rank as one of the most impoverishing assaults which we humans are imposing upon The Biosphere and our descendants” (Myers, 1993).

### 1.2.1. Deforestation has diverse drivers

Though the ultimate antecedent of deforestation globally has been the process of economic development, varying underlying and proximate causes for deforestation have been identified between the world’s regions. (Geist and Lambin, 2002; FAO, 2016). Proximate causes are the immediate and usually intended consequences or human actions at the local level that directly impact forest cover, such as the expansion of agriculture or infrastructure (Geist and Lambin, 2002). Underlying drivers, such as demographic, economic or political factors, represent causes which underpin the proximate causes and their influence can stem anywhere from local to global processes (Geist and Lambin, 2002). If one factor needed to be selected as the main driver of deforestation, it would be systems dynamics, as different combinations and interactions of the underlying and proximate causes have been found to determine the decline in the tropical forest cover in varying geographical and historical contexts (Geist and Lambin, 2002). A meta-analysis by Geist and Lambin (2002) on the causes of deforestation found that in almost 30% of the case studies the full interplay of economic, institutional, technological, cultural, and demographic underlying variables synergistically affected deforestation. Geist and Lambin (2002) also identified that economic factors – including market growth, commercialization, demand/consumption, market failures, urbanindustrial growth, foreign exchange, production conditions and price changes – are the prominent underlying forces of tropical deforestation, affecting 81% of all cases worldwide.

The drivers of deforestation are different in different regions and countries. In Latin America, especially important underlying causes have been the growth of markets, formal policies and public attitudes (Geist and Lambin, 2002). As an example, the Amazon rainforest extends over multiple countries (Brazil, French Guiana, Suriname, Guyana, Venezuela, Colombia, Ecuador, Peru and Bolivia), which vary in the relative importance of the causes of deforestation. Cattle ranching is a dominant cause in many areas of Amazonia, and expanding small-scale agriculture is a primary threat in Bolivia, Colombia, Ecuador, Peru and the Guianas (WWF, 2015b). Today, the primary proximate causes of forest loss or severe degradation in the Brazilian Amazon are pastures and cattle ranching, mechanized agriculture, damns and roads (WWF, 2015b). Deforestation rates in Brazil are strongly related to local human population density and road access to regional markets, and agrarian settlements have been shown to consistently accelerate rates of deforestation and fires (Schneider and Peres, 2015). Along with cattle ranching, another major cause of deforestation in many Amazon regions is large scale soy production, largely grown for animal feed (WWF, 2014). In Brazilian and Bolivian Amazon, soy production has contributed to deforestation in two ways: through direct conversion of forests and through displacing cattle production to the forest frontier (WWF, 2014). Along with primary drivers, all regions have important secondary causes and usually multiple less important causes of forest loss and degradation (WWF, 2015b).

### 1.2.2. Deforestation fronts

Contemporary threat from deforestation is highly concentrated. Over 80% of the forest loss projected globally by 2030 will be accounted for by only eleven areas which have been termed “deforestation fronts” (WWF, 2015b). The concept dates back 25 years to an article by Myers (1993), but the term “deforestation front”, as used in this thesis, was defined in 2015 by the WWF to specifically indicate areas which are projected to have the largest concentrations of forest loss or severe degradation between 2010 and 2030, under business-as-usual scenarios and without further interventions to prevent losses (WWF, 2015b). The great majority of these fronts are situated in tropical areas around the world, with some in sub-tropical areas. These deforestation fronts are at imminent risk of large-scale deforestation and comprise some of the world’s most biologically diverse areas (WWF, 2015b). The Amazon has the world’s largest deforestation front, often called the “arc of deforestation”, with deforestation mostly concentrated into specific sub-fronts extending from Colombia to Peru and Bolivia along the Andes, and from the border between Peru and Brazil all the way to the coast of the Atlantic Ocean along the southern edge of the Amazon lowlands in Brazil (WWF, 2015b).

## *1.3. An issue interlinked with global change*

Deforestation, particularly in the tropics, contributes in multifaceted ways to multiple global environmental issues, such as to the ongoing sixth mass extinction event (Ceballos *et al.*, 2015) and to anthropogenic climate change (IPCC, 2014a), and the relevance of this issue has been recognized by several intergovernmental bodies which exist under the auspices of the United Nations. Both the Aichi Biodiversity Targets (included in the UN Strategic Plan for Biodiversity 2011-2020) and Sustainable Development Goals (SDG; included in the UN 2030 Agenda for Sustainable Development) include targets addressing the protection of forest habitats (CBD, 2014; UN, 2015). For example, one component of Aichi target 11 is that “At least 17 per cent of terrestrial and inland water areas are conserved”, and the SDG target 15.2 is to “By 2020, promote the implementation of sustainable management of all types of forests, halt deforestation, restore degraded forests and substantially increase afforestation and reforestation globally.” Despite these targets, biodiversity remains a challenging subject to study empirically, as only a fraction of the number of species estimated to inhabit this Earth have been described in the scientific literature (MEA, 2005) and of those, only 5% (91,523 species) have had their conservation status estimated by the IUCN (IUCN, 2017). However, as most terrestrial species are reliant on forest habitats, deforestation may be used as a proxy for roughly estimating the threats to biodiversity, especially in the tropical areas where the richness of species is unparalleled (Mace *et al.*, 2005). Tropical forests are also an important reservoir of sequestered carbon and hence protecting forests can help to mitigate both biodiversity loss and climate change (Soares-Filho *et al.*, 2010).

The diversity of living organisms and the global climate system represent two core planetary boundaries, i.e. proposed Earth system processes with estimated thresholds under which humanity can safely operate (Rockström *et al.*, 2009), and deforestation is intimately connected to them both. The Earth’s biosphere and climate system have coevolved for 4 billion years and provide the overarching planetary-level systems within which other earthly processes operate, and either of these systems alone could nudge the whole Earth system into a different state from the current geological epoch (Steffen *et al.*, 2015). Discussing the connections of deforestation to both biodiversity and climate change is therefore warranted in the context of this thesis.

### 1.3.1. Biodiversity under threat

Biodiversity, in the largest sense, means all the variety of life on Earth, and the term can be applied at different scales from genes to whole ecosystems to indicate all the "endless forms most beautiful and most wonderful [that] have been, and are being, evolved", to borrow the words of Charles Darwin (Darwin, 1859). In the Millennium Ecosystem Assessment (MEA 2005) it was evaluated, that humans have increased the recent species extinction rates by as much as 1,000 times the typical historical background rates (calculated from the fossil record) and projected that the future extinction rate will be over 10 times higher than the current rate. In a more recent paper Ceballos *et al*. (2015) conservatively re-analysed the issue using the highest plausible estimates of background extinction rates and the lowest estimates of modern extinctions, and their findings concurred the exceptionally rapid loss of biodiversity over the last few centuries, allowing them to conclude that indeed the sixth mass extinction is already under way.

According to the Convention on Biological Diversity, 70 percent of the projected loss of terrestrial biodiversity globally is due to drivers linked to agriculture (CBD, 2014). The most common threats to terrestrial populations are habitat loss and degradation, followed by overexploitation, whereas the importance of other threats such as pollution, invasive species, disease and climate change, vary according to taxonomic group (WWF, 2016). The threats to biodiversity are especially acute in the equatorial tropics, where species richness peaks and where the highest richness of families and endemism can also be found, even when accounting for area and productivity (Mace *et al.*, 2005). The tropical forest species living planet index (LPI), which is based on the trends of 369 populations of 220 vertebrate species and describes the average trends of these populations, has shown an overall decline of 41% between 1970 and 2009, consistent with the increase of tropical deforestation (WWF, 2016).

### 1.3.2. Carbon bound

In addition to providing habitats for species, tropical forests also have bound in them vast amounts of carbon, sequestered from the atmosphere. One climate change related key risk, meaning a factor which can have potentially severe impacts for natural and human systems, is the reduction of terrestrial carbon sinks (IPCC, 2014b). According to the Intergovernmental Panel for Climate Change (IPCC), around a quarter (~10–12 $GtCO_2eq/yr$) of the global net anthropogenic greenhouse gas emissions are accounted for by the Agriculture, Forestry and Other Land Use (AFOLU) sector, mainly resulting from deforestation, agricultural emissions from soil and nutrient management, and livestock (IPCC, 2014a). In low-income countries the AFOLU emissions are dominated by deforestation and forest degradation, and from 1990 to 2010 the total emissions from this source have been increasing (IPCC, 2014a).

The Amazon rainforest is a large-scale component of the Earth's climate system and a tipping point has been proposed to exist which anthropogenic forcing could exceed, resulting in an abrupt change of the region to a drier and less carbon dense system (Lenton *et al.*, 2008; IPCC, 2014b). The IPCC report indicated that current knowledge of this possible tipping point warrants low confidence, but the long term (2080-2100) risk even within the stringent mitigation scenarios is high, if societies do not undertake further adaptation efforts to reduce deforestation and fires (IPCC, 2014b). This hypothesized tipping point in the global climate system has been termed the "dieback of the Amazon rainforest" and, according to Lenton *et al*. (2008), land-use change alone may have the potential to bring the forest cover to a threshold at which a tiny perturbation could alter the state or development of the whole system. Understanding the future land cover change in the Amazon region in relation to the climate system is therefore important for maintaining biosphere integrity globally.

## *1.4. Protected areas as the main conservation tool*

Protected areas (PAs) are the main tool for combatting deforestation worldwide, as they are fundamental parts of virtually all national and international conservation strategies (Dudley, 2008; UNEP-WCMC and IUCN, 2016). A protected area is defined by the IUCN as "a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley, 2008). The World Database on Protected Areas (WDPA) has records of 202,467 terrestrial and inland water PAs, covering 14.7% (19.8 million $km^2$) of the world's extent (excluding Antarctica) (UNEP-WCMC and IUCN, 2016). Legally established PAs contain 17% of the world's forests, with the largest extent of protected forests being located in South America (MacDicken *et al.*, 2015).

### 1.4.1. Protected area types

PAs can be categorized based on the mode of governance (Table 1), or the type of management (Table 2). The IUCN provides a global framework for categorizing protected area management types which has been recognised by the Convention on Biological Diversity and is commonly used in protected area research (Dudley, 2008). The management categories presented in Table 2 help to expand the IUCN definition of a protected area (Dudley, 2008). These six categories are used together with four governance types, which describe who holds authority and responsibility over a given PA (Dudley, 2008). The IUCN categorization certainly does not encapsulate all possible management types in existence, but the categories are a reasonable approximation and a workable guide widely used in practical conservation (Dudley, 2008). To define a specific management category for a PA, the category definition should apply to at least 75 percent of its area, ensuring that the category is attributed based on the primary management objective(s) (Dudley, 2008). In several studies, PAs have also been categorized more roughly into three main types: strictly protected, sustainable use (or multiple use) and indigenous areas (For example: Soares-Filho *et al.*, 2010; Nolte *et al.*, 2013; Carranza *et al.*, 2014), partly utilizing these IUCN categories.

Table 1. A typology of PA governance types as defined by the IUCN (based on: Dudley, 2008). Governance indicates the actors who hold authority over management, are responsible and can be held accountable.

| | |
|---|---|
| Shared governance | Collaborative management (various degrees of influence); joint management (pluralist management board); transboundary management (various levels across international borders). |
| Private governance | By individual owner; by non-profit organisations (NGOs, universities, cooperatives); by for-profit organisations (individuals or corporate). |
| Governance by government | Federal or national ministry/agency in charge; sub-national ministry/agency in charge; government-delegated management (e.g. to NGO). |
| Governance by indigenous peoples and local communities | Indigenous peoples’ conserved areas and territories; community conserved areas – declared and run by local communities. |

Table 2. PA management types as defined by the IUCN (based on: Dudley, 2008). The categories classify protected areas according to their management objectives.

| IUCN management categories | | Definition |
|---|---|---|
| Ia | Strict nature reserve | Strictly protected for biodiversity and also possibly geological/ geomorphological features, where human visitation, use and impacts are controlled and limited to ensure protection of the conservation values. |
| 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. |
| 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. |
| III | Natural monument or feature | Areas set aside to protect a specific natural monument, which can be a landform, sea mount, marine cavern, geological feature such as a cave, or a living feature such as an ancient grove. |
| IV | Habitat/species management area | Areas to protect particular species or habitats, 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. |
| 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. |
| 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. |

### 1.4.2. Predicaments of protection

Extrapolations have shown, that the Aichi target 11 to conserve 17 percent of terrestrial areas by 2020 will be achieved, if existing commitments on designating PAs are implemented (CBD, 2014). However, during 2010-2015, the rate of increase in protected forest area has slowed globally (MacDicken *et al.*, 2015), and although the Aichi target 11 has been relatively successful, the status of most Aichi targets do not show such positive trends (CBD, 2014). The status of Aichi target 12 for instance, which by 2020 seeks to prevent the extinction of known threatened species while improving and sustaining the conservation status of those most in decline, was estimated to have had no significant overall progress for the first component and moving away from the target for the second, meaning the conservation status of the species most in decline has further deteriorated (CBD, 2014). In addition, caveats exist to the positive trend of protection, as the PA networks remain ecologically unrepresentative and many critical sites for biodiversity are poorly conserved (CBD, 2014). Even within the eleventh Aichi target, the other sub-targets which are focused on the management, connectedness and representativeness of PAs, showed an insufficient rate of progress to meet the target in time (CBD, 2014). This makes those PAs that have been established on critical biodiversity sites all the more important. PAs that exist today in the deforestation fronts of the tropics are, and will continue to be, of paramount importance since they are by default protecting the highly diverse and important tropical forest areas.

Nevertheless, PAs are not impermeable and deforestation sometimes does take place within them. For example, Bruner *et al*. (2001) used a questionnaire to assess 93 PAs from 22 tropical countries, and wrote that net clearing was reported in 17% of the PAs. Likewise, DeFries *et al*. (2005) wrote that of the 198 PAs they studied in moist and dry tropical and subtropical forests around the world, around a quarter experienced declines in forest cover within their administrative boundaries (91% of these had no more than 5% loss of forest area, however). A further problem of PAs is the fact that they are prone to be affected by changes in the political and economic landscape of the administrative area within which they are located. PAs can be downgraded, downsized, or even degazetted (meaning the complete loss of protection status) (Bernard *et al*. 2014).

The alarming rates of decline of the Living Planet Index (WWF, 2016) and in the declining conservation status of many threatened species (CBD, 2014; IUCN, 2017) require us to question the approaches used for establishing and managing PAs. Literature reveals, that there has existed a universal tendency to assign PAs to locations either remote or unsuitable for other productive land use (Joppa and Pfaff, 2009). Therefore, PAs have experienced lower levels of pressure, and compared to other areas have had low levels of deforestation. In other words, these PAs have not done much in terms of protecting the species and habitats that have been threatened. Not all PAs are like this of course, as some are specifically established on high pressure areas to prevent disproportionate damage or to protect specific conservation values (Nolte *et al.*, 2013).

The future of tropical biodiversity and the complex and far-reaching ecosystem services tropical forests provide will depend on the successful management of the deforestation threat, particularly in the deforestation fronts. This success in turn depends on the effectiveness of PAs. Not all PAs are equal, and their characteristics need to be considered when evaluating their effectiveness, which is something researchers have started increasingly to zero in on (see for example: Joppa & Pfaff 2011; Nolte *et al*. 2013; Eklund *et al*. 2016). Weak PAs, for instance with poor governance or few resources, may not withstand the intense deforestation pressures expected in the deforestation fronts.

### 1.4.3. Estimating PA effectiveness

PAs are an important investment of time and resources, making it important to know how effective they are in achieving their purpose. Numerous studies have developed and refined methods for estimating PA effectiveness, varying from basic inside-outside comparisons to comparisons between PA types, and to more refined approaches which account for several confounding factors and create artificial controls to study what would have happened without protection (Joppa and Pfaff, 2010; Soares-Filho *et al.*, 2010). Simple comparisons, which define buffers inside and outside PAs and calculate how these zones differ in the amount of deforestation, are not sufficient, because the buffer areas do not necessarily have the same landscape variables and can differ in their remoteness, influencing the likelihood of the zones for being deforested (Joppa and Pfaff, 2010; Soares-Filho *et al.*, 2010). A study by Joppa & Pfaff (2011) found that controlling for land characteristics reduced the significant positive impact of protection by 50 percent or more. They reported that PAs closer to roads and cities, and those on flatter land, can have higher impacts due to a larger deforestation pressure, which results in a larger potential for reducing deforestation. Spatial patterns in deforestation pressure were also exemplified by a study on tropical forest reserves from Africa, the Americas, and the Asia-Pacific region (20 PAs each), which found that for roughly 80 percent of the PAs considered, there was a gradual increase in deforestation from 5 km within the PAs to 10 km outside their borders (Lui and Coomes, 2016).

PA effectiveness is an estimate of how the existence of a PA, and not the confounding landscape variables, has helped to obtain the desired goals, with the goals usually represented by retained forest cover and measured by how much deforestation a PA prevented within a specified period of time. Contemporary approaches for estimating PA effectiveness use matching (Andam *et al.*, 2008; Gaveau *et al.*, 2009) to compare sample points within protected areas to similar control points in non-protected areas, with the similarity being determined by the values of several controlled variables in the point locations, such as elevation, precipitation, distance to markets and distance to forest edge. By using these artificial control groups, researchers can calculate how much deforestation would have been expected to happen within PAs, if they had not been protected (counterfactual approach). If the counterfactual deforestation pressure equals the observed deforestation within a PA, the PA has had no effect. If the observed amount deforestation is larger than the estimated amount, the PA in question has had a negative effect which has increased deforestation within its boundaries. The less deforestation is observed within PAs when compared to control points, the more effective the PAs are in reducing deforestation. By using this counterfactual matching method, the effectiveness scores of PAs can be estimated and compared, even if the PAs exist in areas with different levels of pressure.

## *1.5. Conservation in the Brazilian Amazon*

Brazil has the world's largest national terrestrial PA network, which accounts for half of the protected land in Latin America (UNEP-WCMC and IUCN, 2016). Over 40% of all Brazilian forests are covered by this network (MacDicken *et al.*, 2015), and in the Brazilian Amazon, the protected areas shield over half of the remaining forests (Soares-Filho *et al.*, 2010). This extensive designation of protection has been the result of decades worth of intense deforestation pressures which have threatened the indigenous cultures and natural riches of the forests. Brazil had record breaking increase in deforestation through the 1990s, which reached a peak in 2005 (Tollefson, 2012), making the administration react by increasing enforcement to cut down on the extremely high deforestation rates, while increasing the amount of land under protection (Nepstad *et al.*, 2009). There were also federal campaigns which cancelled credit for illegal land holdings and imprisoned operators who broke the law, and pressure towards buyers of Amazonian products (Nepstad *et al.*, 2009). Along with these policies, there was a retraction of cattle and soy industries which also played a part in reducing deforestation rates (Soares-Filho *et al.*, 2010). All of these measures resulted in a historical 75% decrease of deforestation from 2004 to 2010 in the Brazilian Amazon (INPE, 2018).

Two studies, for the entire Brazilian Amazon, have estimated the effectiveness PAs had in the years prior to 2010 (Soares-Filho *et al.*, 2010; Nolte *et al.*, 2013), and their findings suggest that all main protection types (strictly protected, sustainable use and indigenous areas) will help prevent forest loss in the high-pressure deforestation front areas of the Brazilian Amazon in the coming decade, indigenous and strict areas more than sustainable use areas (Soares-Filho *et al.*, 2010; Nolte *et al.*, 2013). Indigenous areas are decidedly relevant in the Amazon, where they represent nearly a third of the biome and the majority of existing PAs (WWF, 2015b).

### 1.5.1. Challenges for conservation in the current decade

Many changes have taken place in Brazil after 2010, with potential relevance for the conservation efforts. The new political climate which used hard command-and-control measures to cut down deforestation, generated to a rural backlash of agribusiness and small landowners opposing protection (Tollefson, 2011). An organized coalition of rural agricultural interests started to support a weakening to the country's forest code in 2010, eventually lobbying the proposal through the nation's House of Representatives and the National Congress in 2012, while rises in soy and beef prices increased demand for arable land and fuelled the tensions (Tollefson, 2011, 2012). Even at that time the new bill raised worry among researchers and was even called a "recipe for Amazon dieback", predicting a new wave of deforestation (Tollefson, 2011).

Since 2012, the rate of deforestation in Brazil has started to increase, raising fears that Brazil may be losing decades of progress in forest protection (Tollefson, 2016). Between 2010-2015, Brazil reported the world's greatest annual net loss of forest area globally (984 thousand ha annually), with a rate of 0.2% (MacDicken *et al.*, 2015). In 2015, Brazil had a remaining forest area of 493,538 thousand ha ($4.9\times10^6$ $km^2$), covering 59% of the land area and representing 12% of global forest area (MacDicken *et al.*, 2015). Between August 2015 and July 2016 the total deforested area in Brazil was 29% above the previous year and 75% above the 2012 level, when deforestation hit a historic low (Tollefson, 2016). This may have been reflected in the study by Pack *et al*. (2016), who wrote that the creation of PAs has stagnated in Brazil since 2009, with recent evidence suggesting an increasing rate of PADDD (Protected Area Downgrading, Downsizing, and Degazettement, see: Bernard *et al*. 2014). In addition to the changes in the forest code, Brazil was also rocked by the largest economic recession in the country's history between 2014 and 2016 (BBC, 2017) and president Dilma Rousseff was impeached following a time of great political turmoil in 2016 (Watts, 2016). It is not unreasonable to question whether these factors might have had an influence on the effectiveness of the Brazilian PA network during the current decade.

## *1.6. The state of Acre under focus*

The state of Acre is particularly interesting when considering the conservation of forest habitats and the carbon bound in them, because PAs cover approximately half of the state's area (Figure 1) and in 2010, around the time when the above-mentioned changes started to take place in Brazil, Acre launched the world's first jurisdictional PES-REDD+

program aiming to reduce greenhouse gas emissions from deforestation and forest degradation. Acre is situated in the western most part of the Brazilian Amazon, sharing border with both Peru and Bolivia (Figure 1).

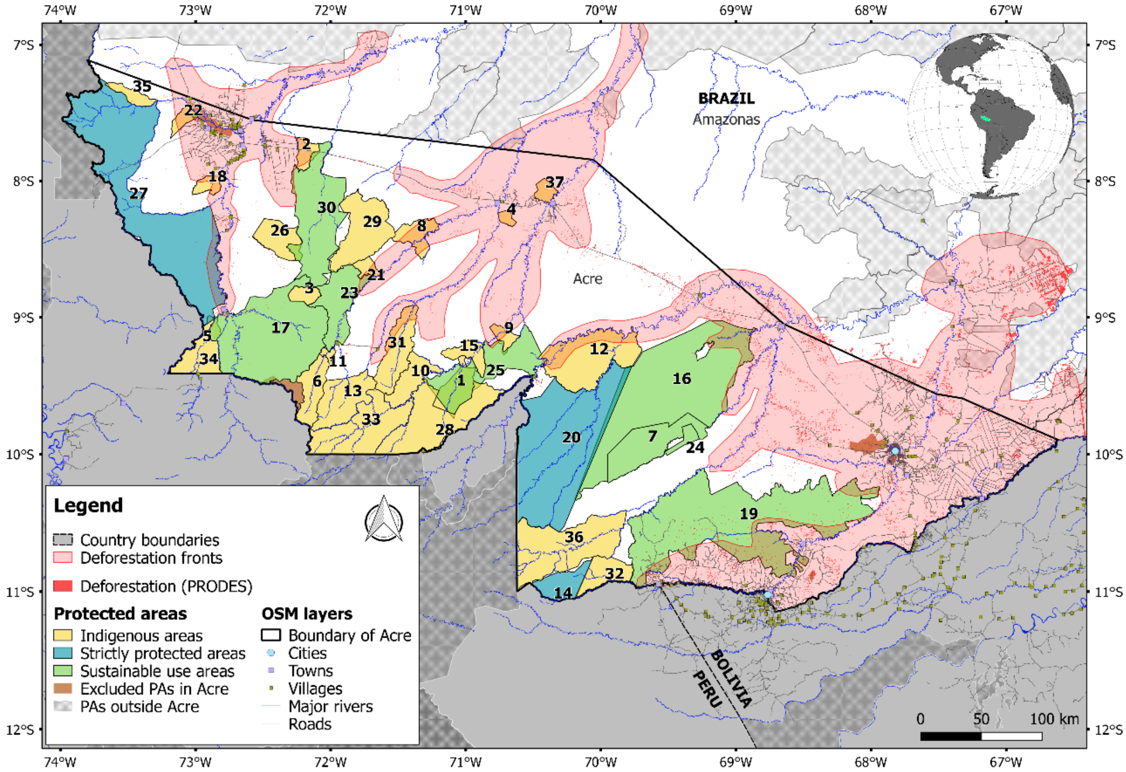


Figure 1. Deforestation fronts and protected areas in the state of Acre, Brazil. The protected areas which are included in this thesis are numbered and categorized into three main types. Red pixels indicate deforestation during 2011-2016 (based on PRODES data).

The state of Acre has a high total species richness for birds, mammals and amphibians relative to the rest of the Brazilian Amazon (Jenkins *et al.*, 2015), but three sub-fronts of the Amazon deforestation front threaten both the biodiversity and the forest carbon contained within Acre's PA network (Figure 1). Road building and paving within the state has been projected to lead to a large increase in deforestation by 2030 and 2050 (Soares-Filho *et al*. 2006). Acre is famous for its conservation history and is no stranger to conflicting interests between those incentivized to deforest and those with interests to conserve. In 1988, the now famous rubber tapper and rainforest activist Chico Mendes was murdered in Acre by ranchers, following his efforts to curb deforestation (Climate Focus, 2013), leading to a lasting social and political transformation which has favoured forest conservation and sustainable use ever since (Climate Focus, 2013).

### 1.6.1. World's first jurisdictional REDD+ program

Deforestation accounts for a great fraction of global $CO_2$ emissions and reducing deforestation is one of the most cost-effective mitigation options available (IPCC, 2014b). Limiting global warming to 2°C, as is the aim of the Paris Agreement (UNFCCC, 2015), will require 40-70% reductions in global anthropogenic greenhouse gas emissions by 2050 compared to 2010, and emissions levels near or below zero in 2100 (IPCC, 2014b). To achieve these reductions, the sectoral $CO_2$ emissions from AFOLU need to be near zero and substantial net sequestration may be needed (IPCC, 2014b). For these reasons, the Parties of the United Nations Framework Convention on Climate Change (UNFCCC) developed the REDD+ mechanism for financing forest related emissions reductions in developing countries (FAO, 2016). REDD+ name comes from "Reducing Emissions from Deforestation and forest Degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks", and these programs will be vital for global efforts to combat climate change (FAO, 2016).

Acre's State System of Incentives for Environmental Services (SISA in short, from Portuguese: "Sistema de Incentivos a Serviços Ambientais") is recognized as the world's first jurisdictional Payments for Ecosystem Services REDD+ program (PES-REDD+), which was created in 2010 through state law (WWF, 2011; Duchelle *et al.*, 2014). The REDD+ mechanism creates financial value for the carbon stored in forests by enabling developing countries to get result-based payments from other countries in return for demonstrated and verified emissions reductions (UN-REDD, 2015). In 2012, Acre was the first to receive results-based payments for verified emission reductions from the German REDD Early Movers (REM) program, which supports pioneers in forest protection (KfW Development Bank, 2017). The REDD+ program of Acre was made jurisdictional, instead of only selecting priority areas, to prevent the creation of new areas where deforestation would be more lucrative and to promote sustainable practices also in low-pressure areas (Sills *et al.*, 2014).

Acre's REDD+ program (SISA) includes carbon sequestration, maintenance of water and hydrological services, conservation of soils, conservation of biodiversity and valuation of traditional knowledge, but, apart from carbon sequestration, most do not yet have specific regulations in place (Duchelle *et al.*, 2014). First program to have been implemented was the "ISA-Carbono" carbon sequestration program, which is used to finance primarily the state's existing policies and programs to reduce deforestation (Duchelle *et al.*, 2014). ISA-Carbono promotes environmental compliance via command-and-control approaches and improved monitoring, including the utilization of satellite images in combination with the Rural Environmental Registry (CAR), to which all rural properties are required to be enrolled by law (Federative Republic of Brazil, 2012; Sills *et al.*, 2014). In addition, the program promotes sustainable production practices and actions in both forestry and agriculture in Acre (Sills *et al.*, 2014).

## *1.7. In this thesis*

In this thesis I will use a recently developed matching method (Eklund *et al.*, 2016) to study the effectiveness of PAs in and near the deforestation fronts of Acre, Brazil. As a result of the changes that have taken place in Brazil since 2010, the context in which PAs exist has changed, potentially affecting PA management and the incentives of deforesters. These changes, together with Acre's REDD+ program, may therefore have had consequences on deforestation pressures and the effectiveness of PAs in Acre. The unique situation of Acre, with its rich biodiversity, pioneering REDD+ program and high present and projected threats from deforestation, make it an internationally relevant area in which to study PA effectiveness. These characteristics also mean, that Acre could possibly represent the first of many similar areas to come, where climate change and biodiversity loss are held at bay by restraining deforestation with REDD+ incentives.

I will use the most recent available deforestation data from the past six years (2011-2016) in order to fill two important knowledge gaps: First, how effective have the individual PAs of Acre been since 2010? And, do differences in effectiveness exist between the main PA types in Acre? To tie the topic to larger global issues and to the ongoing REDD+ program of Acre, I will also estimate how much deforestation and carbon emissions each PA avoided during the past six years. A second goal for this thesis is to explore and discuss deforestation in the context of two major global environmental issues: biodiversity loss and climate change.

# 2. Materials and methods

## *2.1. Background information about the study area*

The state of Acre is home to over 816,000 people (Government of Acre, 2017). Today, more than 73% of the population of Acre lives in urban areas and 56% of the population is situated within the municipalities of Rio Branco and Cruzeiro do Sul (Government of Acre, 2017). The area of Acre is 164,124 $km^2$, and around 46% of the area is protected (Government of Acre, 2017). Indigenous areas are counted as one important conservation unit type and they cover 2,390,112 ha, or around 14.5% of the land. The combined population of the indigenous groups is 19,962 inhabitants, or around 2.5% of the population, and they have a total of 209 villages in Acre (Government of Acre, 2017). Sustainable use conservation units cover 3,569,818 ha, or roughly 22%, and integral protection conservation units (strictly protected areas) 1,563,769 ha which equals to around 9.5% of the area of Acre.

Acre has a humid equatorial climate with high temperatures all year round. Palm trees are dominant in over 85% of the rainforest area, whereas bamboos are dominant in 10% of the forests, mostly concentrated in the Purus and Juruá regions (Government of Acre, 2017). In the previous decade the rate of deforestation decreased in the state of Acre, very much the same way as in the rest of Brazil (Figure 2). Between 2000 and 2015, the amount of deforestation in Acre dropped from 547 $km^2$ to 264 $km^2$, resulting in a reduction of the annual rate from 0.33% to 0.16%. After 2010 the amount of deforestation has stayed relatively level, with some fluctuation between years (Figure 2).

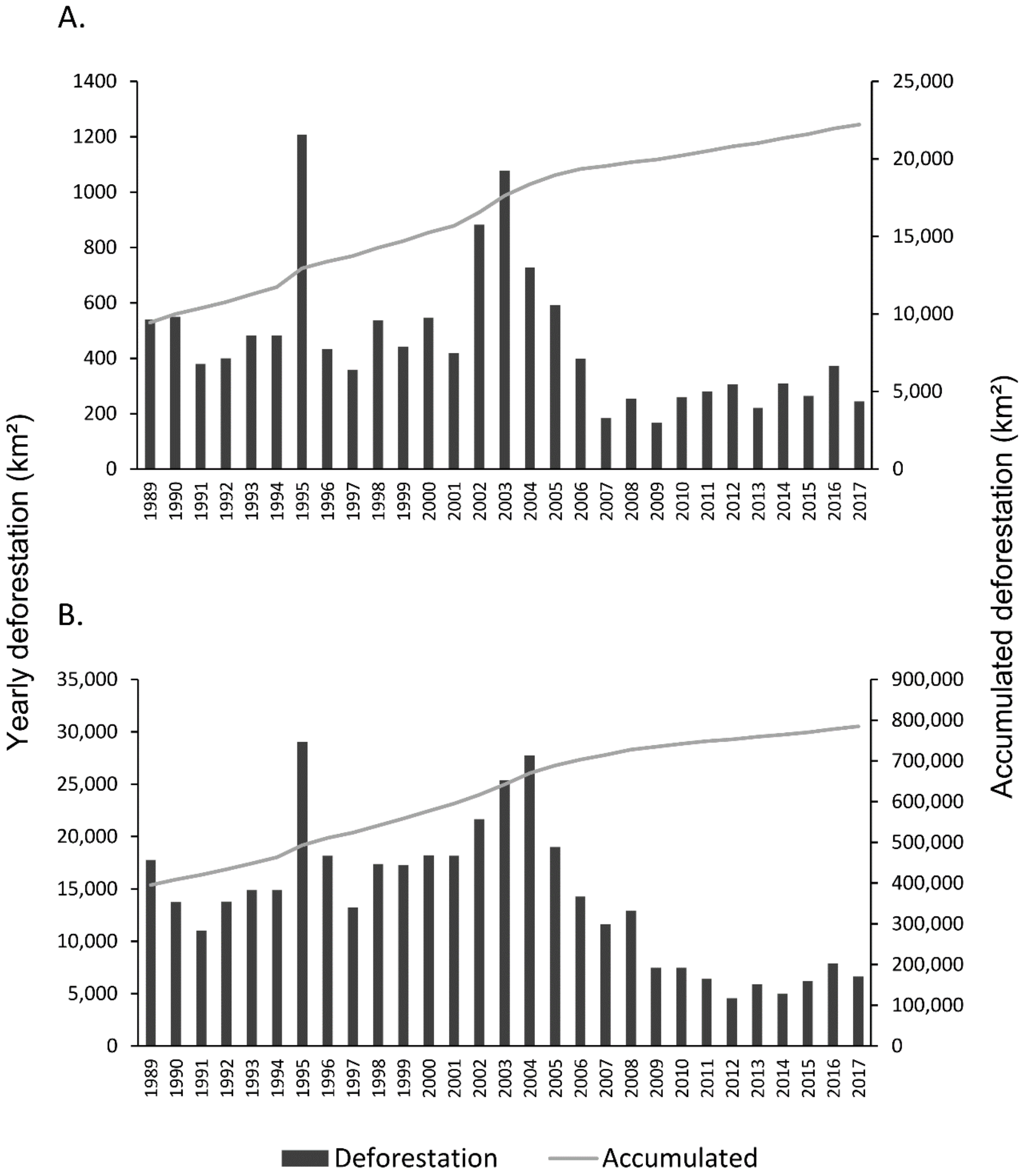


Figure 2. Deforestation in Acre (A) and the total Brazilian Amazon (B) between 1989 and 2017. Data from the Government of Acre (2017) and INPE (2013).

## *2.2. Datasets required to estimate protected area effectiveness*

Before the effectiveness could be estimated for the PAs of Acre, it was necessary to download the deforestation front and protected area polygons, along with the deforestation data from the years covered by this thesis (2011-2016). In addition, to find suitable control areas for the PAs, I needed to get several covariate datasets which I used as control variables to ensure that the PAs were compared to environmentally similar areas (Table 3).

I obtained the protected area polygons and characteristics for Brazil in November of 2017 from the World Database on Protected Areas (WDPA; UNEP-WCMC and IUCN, 2017), and the deforestation datasets (PRODES) for the years 2011-2016 I downloaded from the Brazilian Institute for Space Research (Instituto Nacional de Pesquisas Espaciais; INPE, 2013). The downloaded PRODES deforestation data was based on 30m resolution Landsat imagery, capable of detecting deforestation in small forests patches. To account for the areas that did not have forest cover in 2010, such as areas with cumulative prior deforestation, I used the 2010 Vegetation Continuous Fields (VCF) collection (DiMiceli *et al.*, 2011) as the baseline forest cover dataset. VCF includes proportional estimates for tree cover and was derived with the MODIS sensor on board NASA's Terra satellite.

I used several datasets as proxies for agricultural suitability of the land, which can influence the ability and incentives of people to deforest. I downloaded the median altitude and slope datasets for the year 2012 from the Global Agro-Ecological Zones platform (FAO/IIASA, 2010). Floodable areas, along with forest structure and land cover type, I controlled for with the 2009 GlobCover dataset provided by the European Space Agency (ESA, 2010). I controlled for the amount of precipitation by including the CHELSA Bioclim Annual precipitation (Bio12; Karger *et al.*, 2017) data as one covariate.

As deforestation is carried out by people, the most accessible areas are expected to have the greatest likelihood for deforestation. To control for this, I calculated a surface layer which had the shortest Euclidian distance to forest edge for each pixel in Acre. I used QGIS (v2.18.15) to perform all GIS spatial analyses. I defined forest edge to be either a road, a river of a pixel which had a baseline forest cover (VCF) value less than 45% (defined as non-forest in this thesis, see Figure 3). I downloaded OpenStreetMap (OSM) roads and rivers using the QuickOSM plugin (Trimaille, 2018; version 1.4.7), compiling the roads layer from the “motorway”, “trunk”, “primary”, “secondary”, “tertiary”, “unclassified”, “track” and “road” values and the rivers layer from the “river”, “riverbank” and “natural_water” values. Along with distance to forest edge, access to regional markets can be an important determinant of the true incentives to deforest, and therefore I used a layer of travel time estimates created by Nelson (2008) for the year 2000 as a covariate as well.

Table 3. Datasets used in this thesis. I used WGS84 projection for all layers. Resolutions are rounded.

| | Variable | Description | Resolution | Source |
|---|---|---|---|---|
| | Protected areas | Protected Area polygons for Brazil. | - | UNEP-WCMC and IUCN (2017), The World Database on Protected Areas (WDPA). |
| | Deforestation estimates | PRODES: Fine-scale dataset based on LandSat imagery. Years 2011-2016 combined. | 250 m | Brazilian Institute for Space Research (Instituto Nacional de Pesquisas Espaciais; INPE). |
| | Deforestation front | Polygons of the deforestation fronts in the Amazon. | - | WWF (2015a). |
| *Covariates* | Baseline forest cover | Percent tree cover. Vegetation Continuous Fields (VCF) dataset. Year 2010. | 250 m | Global Land Cover Facility, University of Maryland (DiMiceli *et al.*, 2011). |
| | Slope and terrain | Median altitude and terrain slope datasets. Year 2012. Covariate for agricultural suitability. | 9.3 km | International Institute for Applied Systems Analysis (FAO/IIASA, 2010): Global Acro-Ecological Zones (GAEZ). |
| | Floodable areas | GlobCover dataset. Year 2009. Covariate for agricultural suitability. | 300 m | European Space Agency GlobCover Portal (ESA, 2010). |
| | Precipitation | Annual precipitation data. Covariate for agricultural suitability. | 1 km | CHELSA Bioclim Annual precipitation (Bio 12) dataset (Karger et al., 2017). |
| | Distance to forest edge | Raster surface with shortest Euclidean distances to forest edge. Calculated based on baseline forest cover (VCF), OpenStreetMap (OSM) roads and OSM rivers layers. | 250 m | Individual datasets from: Open Street Map products downloaded in Feb 2018 with QuickOSM plugin in QGIS. VCF as above. |
| | Travel time to cities | Raster surface with fastest travel times to cities with 50,000 or more people. Calculated for the year 2000. | 1 km | Travel time to major cities: a global map of accessibility. Global Environment Monitoring Unit, Joint Research Centre of the European Commission (Nelson, 2008). |

### 2.2.1. Preparing the map layers

The protected area dataset from the WDPA contained all available information relating to protected areas, regardless if the PAs are designated or planned, or if the spatial information is available as polygons or points. It was therefore necessary to trim the PA layer to make it suitable for analyses. There were no points within Acre, so I excluded the point layer. From the polygon layer, I only included PAs with "Designated" status, as these areas are recognized or dedicated through legal means, implying a specific and binding commitment to long term conservation (UNEP-WCMC, 2017). I excluded PAs designated after 2010 to only include PAs which had been in force before the first year of the study period.

As the purpose of this thesis is to study PAs either within or in close proximity to the deforestation fronts, I created a 50 km buffer around the WWF deforestation front polygon layer and used the buffer area to select those PAs which had any overlap with this extended deforestation front area. To enable comparisons with the previous PA effectiveness studies done in Brazil (Soares-Filho *et al.*, 2010; Nolte *et al.*, 2013), and to see whether the main PA types differed in terms of effectiveness, I grouped the PAs into three main categories. First, strictly protected areas, which included state and national biological stations, national and state parks, ecological stations and biological reserves. Second, sustainable use areas, which included state and national forests, extractive reserves, sustainable development reserves and PAs categorized as "forest". Indigenous areas were considered to be the third main PA category. After cleaning and categorizing the PA dataset, I used several criteria to select PAs for this study, following the approach of Nolte *et al.* (2013). I excluded environmental protection areas as they have been reported to primarily consist of private lands without significant additional restrictions from the protected area designation (Nolte *et al.*, 2013), and I excluded areas of relevant ecological interest, because they may be either public or private lands and are usually small areas which conserve some specific natural features (Unidades de Conservação no Brazil, 2018). The state of Acre did not have any of the other types of PAs that were excluded in the study by Nolte *et al.* (2013).

Next, I applied two further criteria to enhance comparability with the previous study by Nolte *et al.* (2013): Only those PAs which had more than 800 forest parcels, in the 250 m resolution VCF layer of the year 2010, were included. This limit corresponded to the limit used by Nolte *et al.* (2013), as they used four times coarser resolution (1 km) data and a limit of 200 forest parcels. However, I decided to use a different definition for forested area in this thesis. Instead of >25% forest cover limit, I decided to use a >= 45% limit because for the matching I needed to have a limit which covered the non-forested areas well. I decided to use the 45% value based on visual comparison between Google satellite images and the VCF layer (Figure 3). I calculated the number of forest parcels for each PA, by first creating a new raster layer with the raster calculator tool, transforming all VCF cells with values equal to or above 45 to ones, and the rest to zeros. I used the Zonal Statistics tool to calculate the sum of pixels for each PA, and then selected those PAs which had a sum greater than 800. Second, I removed rivers, roads and nulls from the VCF layer and calculated the mean value from the VCF layer for each PA with the Zonal Statistics tool. I used this mean value to make sure all PAs had a mean forest cover value higher than 50%, following the criteria used by Nolte *et al.* (2013).

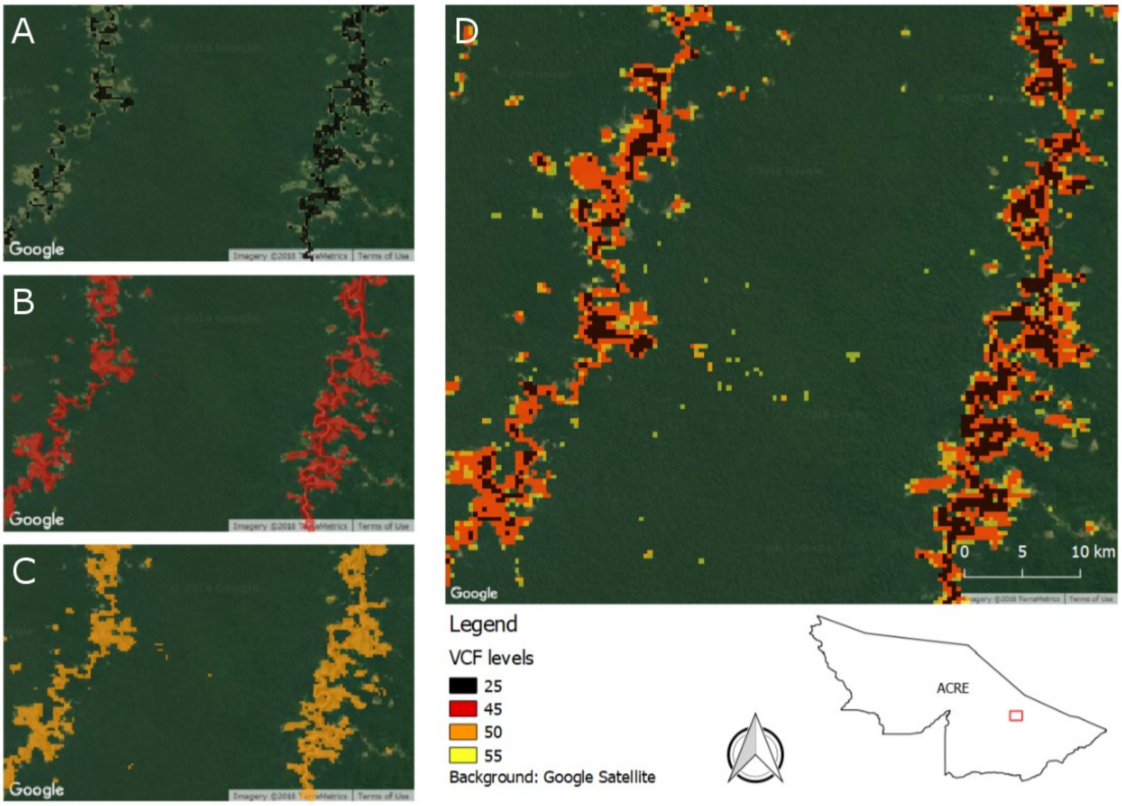


Figure 3. Comparison of different values of the Vegetation Continuous Fields (VCF) layer, to find an optimal forest cover edge value. The VCF values indicate forest cover defined as equal to or less than that specific value. Forest cover less than 25% was considered too low, as it did not adequately cover the deforestation visible in the satellite images. Values larger than 45% had too many erroneous pixels outside the cleared areas. A-C visualize the VCF layers individually, and D has all layers stacked. The same area is represented in each.

It was necessary for me to choose a new forest cover value, as the non-forested areas had to be excluded from the sampled area. This is because when a matching method is used to compare the likelihood of a specific sample point to experience deforestation, all points in the dataset from which the points of comparison are searched for must have been forested at the start of the study period. If they are not, a pixel may be compared to an area where the occurrence of deforestation was impossible since it had already happened prior. Based on visual observations, the 45% forest cover limit seemed to closely resemble the true forest edge based on the satellite images, maximizing the covered non-forest area and minimizing the number of erroneous forest cover edge pixels (Figure 3) which may be caused by individual disturbances within a forest or a naturally lower forest density, neither of which should be defined as the edge of the forest as they do not influence accessibility to the forest.

After selecting the PAs no further than 50 km of the deforestation fronts and applying the other above-mentioned criteria, a set of 37 PAs remained. Of these, seven relatively small PAs were located either mostly or completely within the boundaries of the original deforestation front layer (Figure 1), and the rest were located within the reach of the buffer area.

I calculated the distance to forest edge layer by merging the OSM road and river data, rasterising the resulting layer and then using the GRASS tool r.series (Clements and GRASS Development Team, 2018) to combine it to a copy of the VCF baseline forest cover layer, which first had all <45 cells transformed to ones. The resulting binary raster layer had cells with value 1 representing forest edge. Then, using the raster analysis tool Proximity to calculate distance to closest cell with a value of 1, I created the final raster surface which contained, for each pixel, the shortest distance to forest edge. I downloaded the yearly PRODES deforestation polygon layers for Brazil from 2011 to 2016 and combined them into one layer containing all deforestation that had taken place during those six years. I rasterized this layer to the same resolution as the VCF layer (~250m).

## *2.3. The matching method*

I applied a new matching method developed by Eklund *et al.* (2016) to estimate the effectiveness of the PAs in Acre’s deforestation fronts. Matching methods are a way to computationally compare a sample of pixels from within a PA to an artificial control area that is similar in all other aspects except for the assigned protection status. By calculating the deforestation in the control areas, the expected deforestation pressure and PA effectiveness can be counterfactually estimated. Because PAs are large scale tools for conservation, studying their effectiveness using satellite-based data means having to deal with large datasets, especially when a high spatial resolution is used. The new computationally efficient method Eklund *et al.* (2016) developed allows for larger sample sizes than previously used matching methods and is not limited to the comparison of pairs of pixels.

I have visualized my workflow in Chart 1 to illustrate the different sections of the work and the steps I took during those sections. The first part of the analyses was to sample pixels from each PA and the non-protected area and intersect these sample points with the covariate rasters. In the second part I used a script provided to me by Eklund *et al.* (2016) to compile the artificial controls used in counterfactual matching. I ran the data using the Taito supercluster which enabled parallel computation with a multitude of cores (computational resources available for research by CSC – IT Center for Science, Finland). After the runs, I used the output files to calculate estimates of deforestation pressure and two different PA effectiveness measures. Lastly, I used the PA effectiveness results to calculate the amount of deforestation and carbon emissions each PA had avoided during my study period.

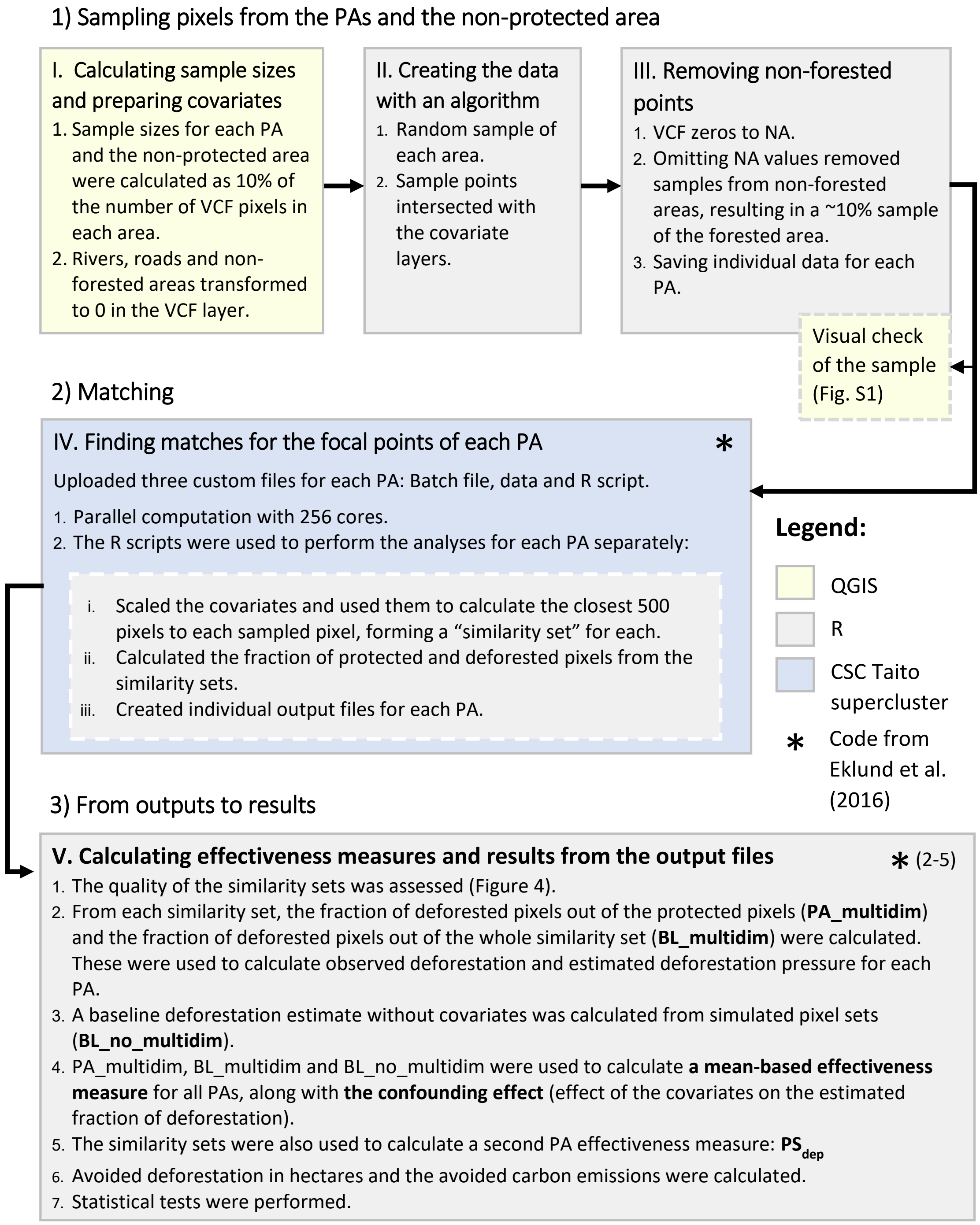


Chart 1. Flowchart of the work process, from input data to results. The programs used in each step of the work are indicated with colours. I performed the work using QGIS and R Studio, and the actual matching part was performed with R using the Taito supercluster which is available for research use by CSC – IT Center for Science, Finland.

### 2.3.1. Details of the matching

I performed the matching separately for each PA, which means that I had to take a separate sample for each PA. I also took one sample of the non-protected area of Acre and combined it to each PA dataset, because the samples from this area were used to find non-protected matches for the sampled pixels of each PA. I sampled 10% of the pixels in each PA and the non-protected area, which at 250m resolution resulted in total to over 300,000 sampled points. In the first step of the sampling, the non-forested areas were included to the sample size, and those points were removed later in the process as missing values. The final sample for the forested areas, after the missing values were removed, was approximately 10% for each PA and the non-protected area (Table S1, Figure S1). I performed the random sampling in R (R Studio, version 1.0.153), using the sp package (Pebesma and Bivand, 2005). I used the PA types that were excluded from this study to limit sampling, by excluding those areas from the non-protected area polygon (Figure S1).

Along with the unique datasets, I prepared unique R scripts and batch files for all PAs. The batch file contained commands for executing the runs in the Taito supercluster, such as the output file names, maximum duration of the run, a command to use parallel computing and a specified number of requested cores and memory. The batch file then commanded the use of R environment to run the R script. The run time with the Taito supercluster ranged roughly from 3 minutes (smallest sample) to 24 minutes (largest sample). The R script for performing the runs in the Taito supercluster was shared to me by the main author of Eklund *et al.* (2016), and I made no modifications to the way this script performed the matching (an official R-package for this method is being prepared). The matching process is described in detail in the original paper by Eklund *et al.* (2016). To describe it in short, the script scaled the covariates and applied a Mahalanobis transformation on them, and then calculated the Euclidean distance between the transformed pixels to get Mahalanobis distance. This distance measure was used, as it accounts for potential collinearity among the variables. Then, an iterative process was used in which the span of each covariate was calculated, restricted, and used to select the 500 environmentally most similar pixels to be used in the comparisons. These 500 pixels form a so called “similarity set” for each focal pixel (pixel included in the sample of a PA). In this new matching method, instead of only comparing protected pixels to non-protected, the environmental similarity is used to create a cloud of points which are used for the comparisons. The Rmpi library (Yu, 2002) in R was used for parallelization in the CSC Linux environment and the vegan library (Oksanen *et al.*, 2018) to calculate the Mahalanobis distances.

### 2.3.2. Quality checks for the matching output files

For visual quality checks of the matching process, I created histograms from the matching output files to visualize for each PA how many focal points had how large a fraction of the similarity sets either protected or deforested, and how long was the maximum environmental distance in each similarity set (Figure 4). The histograms confirmed that there were enough non-protected pixels in the similarity sets of the sample points for each PA, meaning that the matching found enough comparable pixels from the non-protected areas. In fact, nine PAs had a very large number of similarity sets with a low fraction of protected pixels compared to non-protected pixels, which turned out to be due to their small area and thus a smaller sample size.

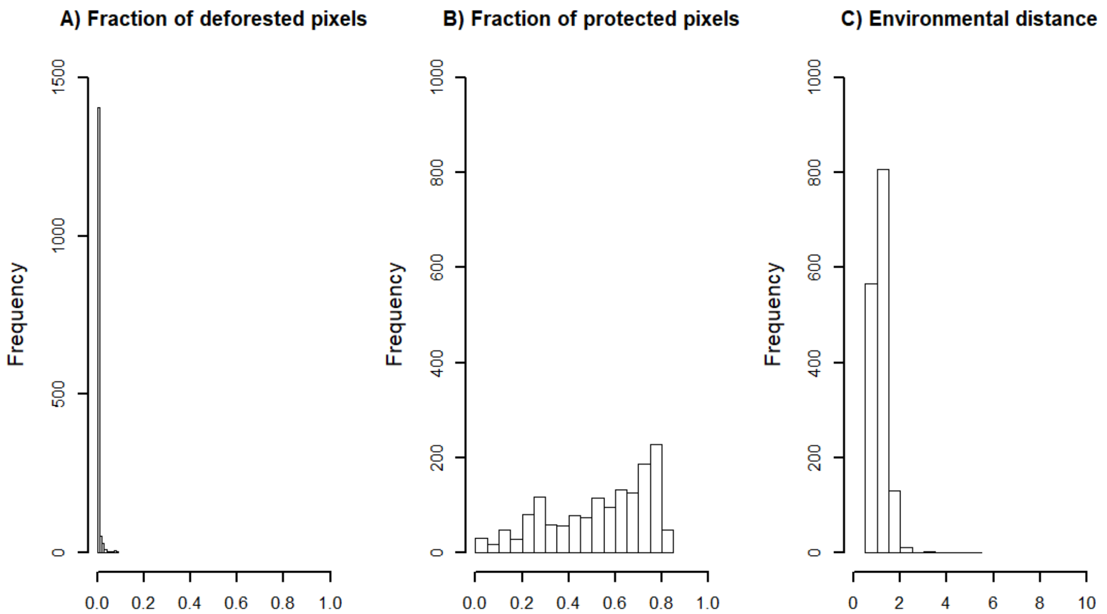


Figure 4. An example of histograms created for PA 10 from one of the matching output files, to visualize the frequencies protected and deforested points in the similarity sets, and the similarity of the points in the similarity sets (environmental distance). A high fraction of pixels with a frequency of 1 would mean that a high number of similarity sets had a fraction of either deforestation (A) or protection (B) that was close to 100%. For the environmental distance (C) a high frequency with a distance of 1 would mean that a high number of similarity sets had a maximum environmental distance close to 1, or in other words the similarity set of 500 closest pixels was environmentally very similar to the focal pixel.

### 2.3.3. Calculating the effectiveness measures

The matching created individual outputs for each PA, which I used to calculate the deforestation pressure and PA effectiveness estimates with a script provided by Eklund *et al.* (2016). I modified the script to suit my data, but otherwise no alterations were done to it. Only the similarity sets which included at least 10% non-protected control pixels were used to calculate the estimates of deforestation pressure for each PA, and consequently the mean-based effectiveness measure which was derived from the deforestation estimates. I generated a measure of the background deforestation rate disregarding protection and the covariates, termed baseline non-multidimensional deforestation estimate (BL_no-multidim) by Eklund *et al.* (2016), by sampling 500 pixels 100 000 times from the deforestation column of the original input data of each PA. This created simulated 500 pixel sets for which the environmental similarity was not considered. I visualized these non-aggregated results for each PA as density curves, in a similar way that was done by Eklund *et al.* (2016). The density curves showed the fractions of deforestation from all focal point similarity sets for each PA without aggregating the results, which is why I examined them to decide whether a mean or median measure would describe the deforestation estimates better (Figure 5).

The original authors of this method (Eklund *et al.*, 2016) used the medians of PA_multidim, BL_multidim and BL_no_multidim to calculate a median based effectiveness measure for all PAs. This decision was based on the fact that a large number of zeros exist in the data of the deforestation measures. However, the authors also noted that the large number of zero deforestation similarity sets heavily influenced the median measure of effectiveness, which therefore gave divergent results from the other effectiveness measure they used ($PS_{dep}$, see below). Based on this information and an investigation of the density plots created for each PA from the multidim datasets, a different measure of central tendency, the arithmetic mean, was opted for in this thesis (see Figure 5 for four selected plots which display different relationships between the mean and median values). If a little over half of the similarity sets have zero deforestation, the median will ignore the other half of the similarity sets, no matter how high deforestation fractions they have. The mean describes the data better and, unlike the median, never ignores the existence of focal point similarity sets with deforestation.

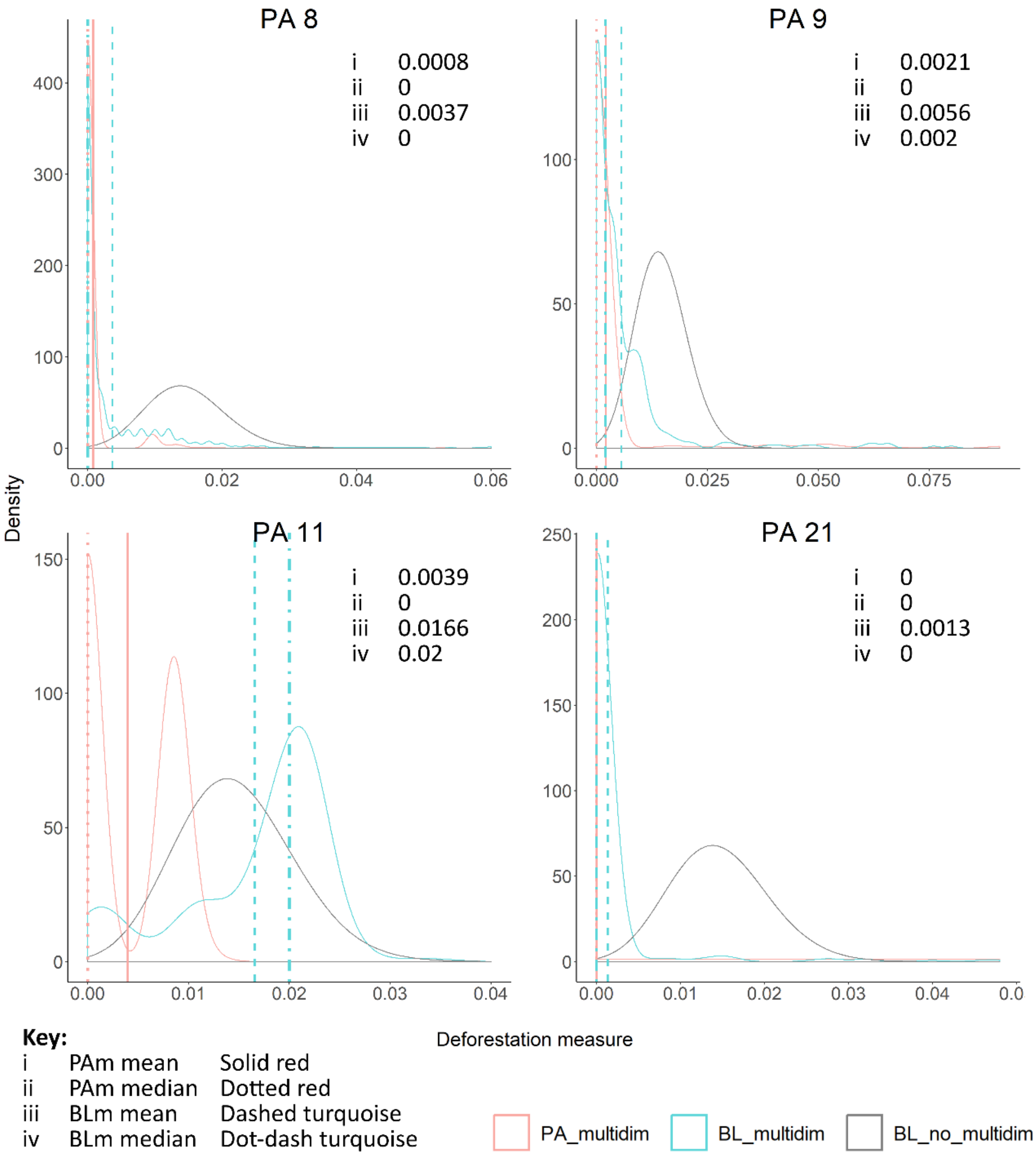


Figure 5. Density curves with vertical median and mean lines of the deforestation measures for selected PAs. The vertical lines were used to determine whether the mean or the median was better suited for aggregating the deforestation fractions in the similarity sets for each PA. Plots were created and investigated for all PAs, and these four PAs were selected here because they show different types of density curves. The means described the data better as the medians were zero in many cases even though deforestation was present in the similarity sets. Based partly on these plots, the mean was selected as the optimal measure for aggregating the results. Note the difference in scales.

In this thesis, PA_multidim and BL_multidim means were used to calculate effectiveness measures for all PAs. The mean PA effectiveness measure gives the fraction of forested area in each PA that would have been expected to be deforested if the area was not under protection. The difference between the baseline multidimensional deforestation (BL_multidim), which indicates how much deforestation existed on environmentally similar areas, and the protected area multidimensional deforestation (PA_multidim) indicating the deforestation present only within the protected areas, gives the measure of PA effectiveness (Mean PA effectiveness = Mean BL_multidim minus mean PA_multidim). The mean of PA_multidim values over the focal point similarity sets reveals if there were similarity sets with deforestation within a PA and how much on average the similarity sets of these focal pixels had deforestation. With the mean of BL_multidim, the average pressure (expected fraction of deforestation) towards the PA is given. If the median of PA_multidim was used, the PA effectiveness could be overestimated when the baseline pressure is high, because if deforestation is not present in most focal point similarity sets, the median would ignore all deforestation. In addition, the median could underestimate PA effectiveness when the baseline pressure is low, because it gives zero median values for many BL_multidim estimates, which feeds into the effectiveness calculations.

The effect of including the confounding factors (covariates) in the analyses is calculated as the difference between the baseline non-multidimensional deforestation estimate (BL_no_multidim) and the baseline multidimensional deforestation. In other words, the mean of the deforestation fractions over all focal point similarity sets was subtracted from the mean fraction of deforestation in the simulated 100,000 BL_no_multidim pixel sets (Confounding effect = Mean BL_no_multidim minus mean BL_multidim). Typically, covariates reduce the amount of deforestation that would be estimated for PAs if no covariates were used and therefore the result usually is a positive number which tells how much the deforestation would have been overestimated without covariates. However, if there are enough similarity sets with a high fraction of deforestation in the deforestation pressure estimate (BL_multidim), it is also possible to get a negative confounding effect using this measure, which would mean that based on the environmental characteristics of the pixels, more deforestation was expected than if covariates were not included. A positive confounding effect indicates that the mean effectiveness measure was reduced, whereas a negative confounding effect indicates that the effectiveness estimate was increased by the use of covariates.

To get a second estimate for PA effectiveness, another measure was also calculated: The probability of superiority for dependent groups, termed “$PS_{dep}$”. It is a non-parametric effect size statistic which is based on the number of similarity sets that indicate that PAs are more effective than expected (Figure 6). This statistic was calculated as:

$PS_{dep}$ = (sum.PAm.less.than.BLm + (0.5*sum.PAm.same.as.BLm))/total

Where PAm is PA_multidim, BLm is BL_multidim, and total is the number of sample points which had more than 10% non-protected points in the similarity sets. The result is a fraction of times when the PA pixels in each set had less deforestation than would have been expected. For example, a 0.75 $PS_{dep}$ value would mean that in 50% of the similarity sets of each focal point, there was less deforestation within the PA than in the control areas, and the other half had no difference when compared to baseline rates. In reality, variation can exist in these ratios. The same 0.75 value can be produced if 51% of the similarity sets showed less deforestation than baseline, 48% showed the same amount and 1% showed more deforestation than the baseline.

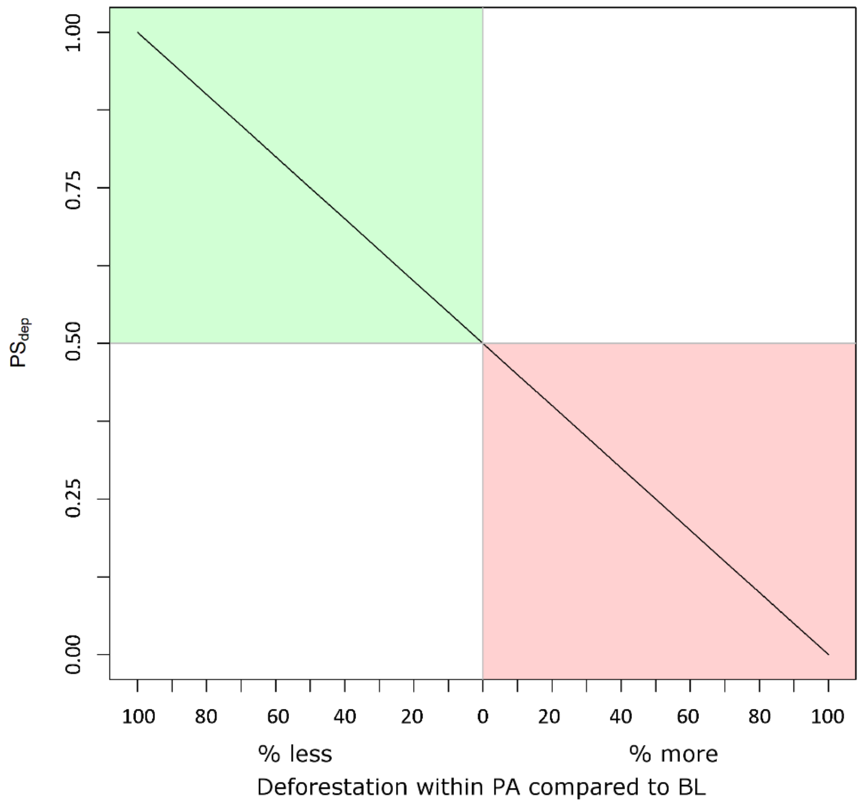


Figure 6. The concept of $PS_{dep}$. When $PS_{dep}$ is 0.5, deforestation is the same within and outside a protected area on average for all focal point similarity sets. Less than 0.5 mean that PAs had more deforestation than baseline (BL_multidim) and more than 0.5 values mean that PAs had less deforestation than the baseline. The diagonal black line represents the theoretical linear relationship between $PS_{dep}$ and the fractions of deforestation.

## *2.4. Calculating avoided deforestation and carbon emissions*

By using the percentage of avoided deforestation from the mean-based effectiveness measure to multiply the number of forested pixels in each PA, I was able to calculate the number of forested pixels in which deforestation was avoided for each PA. When this was multiplied with the exact pixel size of the deforestation layer (231.9267 * 231.9267 metres, the same resolution as the VCF layer), it was possible obtain the estimate of avoided deforestation in units of area. I calculated the number of forested pixels for each PA from the VCF raster, using the >=45% forest cover as the definition for forested area.

To calculate how much carbon emissions were curbed by the PAs of Acre during the study period, I used carbon density data of each PA which was reported in the supplementary materials (Dataset S1) of Soares-Filho *et al.* (2010), who used a carbon bookkeeping model to calculate emissions from deforestation by using forest carbon biomass estimates of Saatchi *et al.* (2007) with the assumptions that the carbon content of wood biomass was 50% and that deforestation releases 85% of the carbon bound in trees to the atmosphere. Soares-Filho *et al.* (2010) had reported 'mean_C_ton_ha' values for each PA in Brazil, meaning the mean amount of carbon per hectare of rainforest, which I used to calculate the avoided carbon emissions for each PA together with the estimated area of deforestation each PA had avoided, and using the same the assumption that deforestation releases 85% of the carbon bound in the forests. PAs 3 and 29 had to be omitted from the carbon calculations due to a lack of carbon density data. The contributions to avoided emissions by these three PAs were therefore not included in the totals calculated for Acre.

## 2.5. *Statistical tests*

To test if the PAs had overall less deforestation than the matched areas, I used the Wilcoxon signed rank test to compare the median deforestation fractions of the PAs, calculated from the similarity sets of each PA (mean of PA_multidim values), to the corresponding fractions calculated for the matched areas (mean of BL_multidim values). The multidim similarity sets are paired (matched) comparisons since the protected pixels were included in the estimation of background deforestation rates. To test whether PA effectiveness differed between the three PA types, I performed Mann-Whitney U tests between strictly protected, indigenous and sustainable use areas, using the Wilcoxon rank sum test for the $PS_{dep}$ effectiveness measures. In the Mann-Whitney U tests, binomial distribution was accounted for by allowing continuity correction, and the comparisons were not performed as paired since the PA types are not dependent or repeated measurements.

# 3. Results

## *3.1. Varying effectiveness of Acre's protected areas since 2010*

My findings showed that between 2011 and 2016, the PAs had overall less deforestation than the areas to which they were matched (Table 4), meaning that Acre's PA network successfully prevented additional deforestation from happening. Substantial variation existed between the effectiveness of individual PAs, however (Figure 7). The effectiveness of the PAs in avoiding forest loss (the mean-based effectiveness measure) varied from 0.02 to 1.57 percent, indicating the fraction of forested area that expectedly would have been deforested if the PA did not exist (Figure 7). The $PS_{dep}$ effectiveness measure varied from 0.53 to 0.98 between the PAs, corresponding respectively to almost 0% and 100% less deforestation than the baseline (Figure 7). Three out of the 37 PAs had a very low $PS_{dep}$ effectiveness, with less than 10% of the similarity sets showing less deforestation than the baseline (PAs 23, 3 and 29). Conversely, there were also three PAs which had more than 90% of the similarity sets showing less deforestation for the protected pixels, which is an indication that those PAs were particularly effective (PAs 5, 11 and 32). The order of effectiveness for the three most effective PAs was inverse for the two effectiveness measures so that PA 5 was the most effective based on $PS_{dep}$ and PA 32 with the mean-based effectiveness measure. All three most effective PAs were indigenous areas, and they all had prevented between 1.02% and 1.56% of additional deforestation relative to their respective areas. PAs 5 and 11 are small indigenous areas with rivers running through them and they had only a small amount of deforestation within their boundaries, whereas PA 32 had no deforestation during the study period (Figure 8). Two of the least effective PAs (3 and 29) had low effectiveness as a consequence of remoteness, whereas PA 23 had low effectiveness due to deforestation. A river crosses the southern part of PA 23 and aligns its eastern edge, which increased its susceptibility to deforestation.

Table 4. Wilcoxon signed rank test to test if the fraction of deforestation within the PA network differed from the control areas.

| Test | Comparison | Median PAm | Median BLm | p | V | Conf. low | Conf. high | (pseudo) median |
|---|---|---|---|---|---|---|---|---|
| Wilcoxon signed rank | PAm ~ BLm | 0.0002 | 0.0034 | <0.001 | 0 | -0.004 | -0.002 | -0.003 |

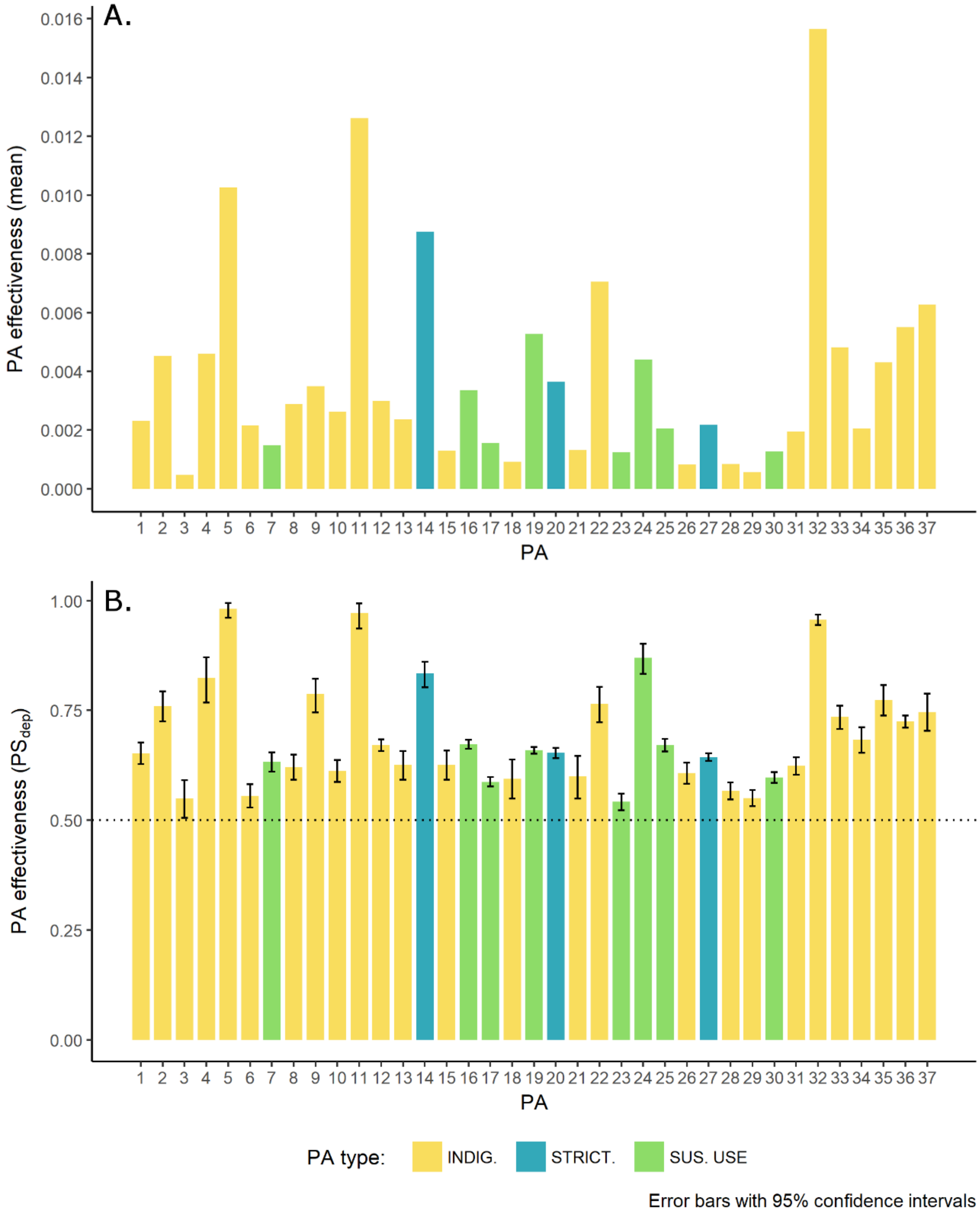


Figure 7. The effectiveness of individual PAs. A: Mean-based effectiveness measure (avoided deforestation as a fraction). B: $PS_{dep}$ effectiveness measure, where the dotted line at $PS_{dep}$ value 0.5 indicates that deforestation was, on average, the same within and outside a protected area for all focal point similarity sets. All PAs with bars extending above the 0.5 line had a positive effect. Great variability existed between the effectiveness of PAs, with both effectiveness measures. Because the two effectiveness measures describe different aspects of effectiveness, there was some differences in the results, but overall these two effectiveness measures showed a very similar pattern. Bars are coloured according to the three PA types: Indigenous, strictly protected and sustainable use areas. See the PAs on a map in Figure 1.

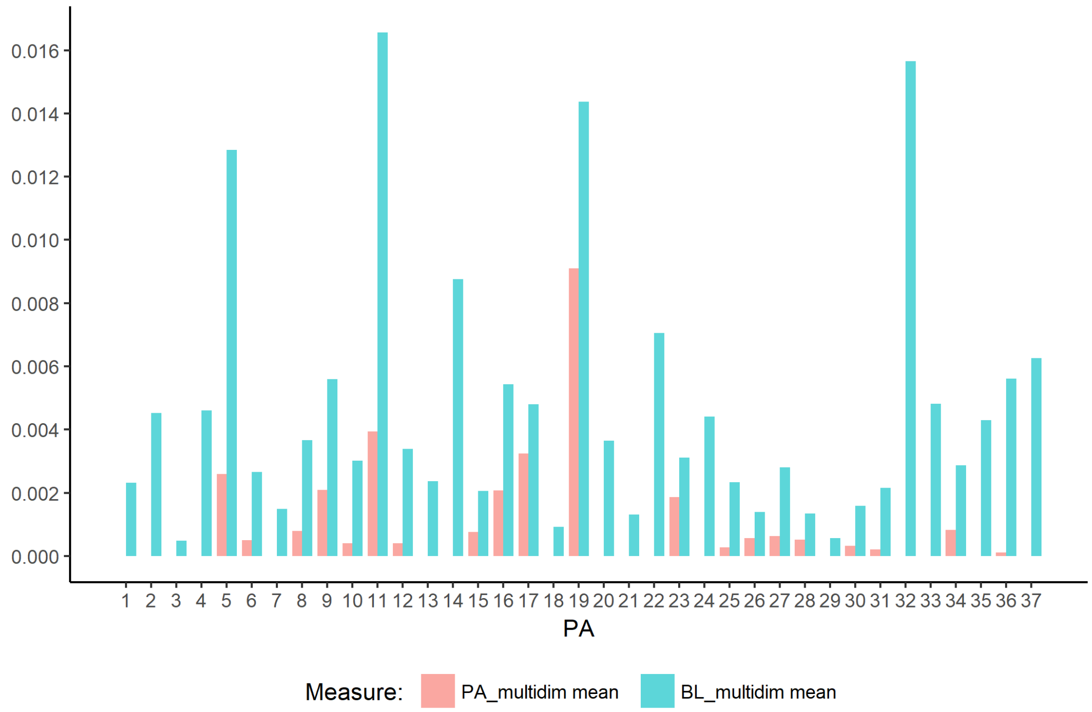


Figure 8. Deforestation observed within the PAs (PA_multidim) and the pressures each PA faced (BL_multidim). BL_multidim is the fraction of deforestation in the areas to which PA samples were compared using the matching method. Those PAs with little or no deforestation despite high pressures were assigned a high mean-based effectiveness value (Figure 7). 17 PAs had no deforestation in the ~10% sample.

### *3.2. No differences in effectiveness found between the main PA types*

The main protected area types did not differ from each other significantly in Acre during the period under study and this result was the same for both measures of effectiveness (Figure 9). I used Mann-Whitney U tests to statistically confirm this finding (Table 5). Even though the individual PAs had great variation between them in their baseline deforestation pressures (Figure 8), and therefore in their potential to avoid deforestation, I observed no considerable differences between the PA types in the pressures they faced (Figure S2).

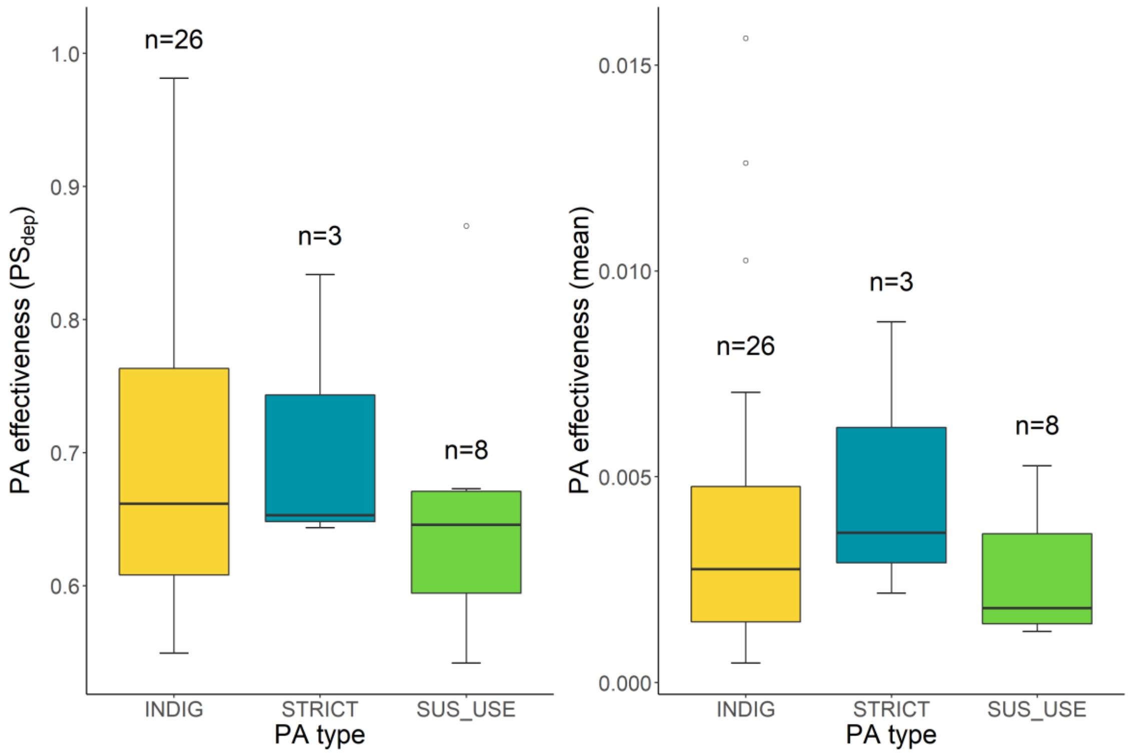


Figure 9. Boxplots of the PA effectiveness measures by PA type, with $PS_{dep}$ on the left and mean-based effectiveness measure on the right. The number of PAs in each group is marked on top of each box. The whiskers extend to the largest value no further than 1.5 times the distance between the first and third quartiles. Outliers as points.

Table 5. I compared the differences in the $PS_{dep}$ effectiveness estimates between PA types with Mann-Whitney U tests (Wilcoxon rank sum test in R; W statistic is equivalent to the U statistic in this test).

| Test | Comparison | p | W | Difference in location |
|---|---|---|---|---|
| Wilcoxon rank sum | Indig. ~ Strict | 0.56 | 30 | -0.030 |
| Wilcoxon rank sum | Indig. ~ Sus.use | 0.44 | 124 | 0.029 |
| Wilcoxon rank sum | Strict ~ Sus.use | 0.63 | 15 | 0.051 |

| **PA type** | **$PS_{dep}$ mean** | **$PS_{dep}$ SD** | **n** |
|---|---|---|---|
| Indigenous | 0.70 | 0.13 | 26 |
| Strict | 0.71 | 0.11 | 3 |
| Sustainable use | 0.65 | 0.10 | 8 |

### 3.3. *Effectiveness of the seven PAs within the deforestation fronts varied*

PAs 4 and 37 were located completely within one of the deforestation fronts (Table 6, Figure 10). PA 4 had high effectiveness with both measures, higher with $PS_{dep}$ (0.82) than with means (0.0045). It was situated close to an urbanized area, but this indigenous area had only three pixels worth of deforestation during the study period, not enough to be present in the sample. PA 37 also had high effectiveness with both measures, but unlike PA 4, it had a higher estimate based on means (0.0063) than with $PS_{dep}$ (0.75). PA 37 is geographically close to PA 4, but on the other side of the BR-364 inter-state highway, and it too had experienced some deforestation on the riverbank which defines its south-eastern border, but not enough to be present in the sample. Variation existed in the effectiveness of indigenous areas that existed mostly (PAs 22, 18, 2, 8 and 9) within the deforestation fronts (Table 6, Figure 10).

Table 6. The effectiveness of the seven PAs which were located either completely (PAs 4 & 37) or mostly (PAs 22, 18, 2, 8 & 9) inside the deforestation fronts. All of these PAs were indigenous areas.

| PA | Effectiveness | Notes |
|---|---|---|
| 4 | High effectiveness with both measures, higher with PSdep (0.82) than with means (0.0046). | Situated close to the town of Tarauacá and the BR-364 road. Completely within a deforestation front. |
| 37 | High effectiveness with both measures, higher with means (0.0063) than with PSdep (0.75). | Situated close to the town of Feijó and the BR-364 road. Completely within a deforestation front. |
| 22 | High effectiveness with both measures, higher with means (0.0071) than with PSdep (0.76). | Close to the town of Mâncio Lima, an urbanized centre. |
| 18 | Low effectiveness with both measures, lower with means (0.0009) than with PSdep (0.59). | The effectiveness was estimated to be surprisingly low, as there was large scale deforestation right on the edge of this indigenous area and none within. |
| 2 | High effectiveness with both measures, slightly higher with PSdep (0.76) than with means (0.0045). | The BR-364 road crosses straight through this indigenous area, but zero deforestation was observed within. |
| 8 | Medium effectiveness with both measures, a lot higher with means (0.0029) than with PSdep (0.62). | A river aligns and crosses this PA. Effectiveness affected by deforestation within the PA. |
| 9 | Medium effectiveness with means (0.0035), high effectiveness with PSdep (0.79). | A river aligns and crosses this PA. Effectiveness affected by deforestation within the PA. |

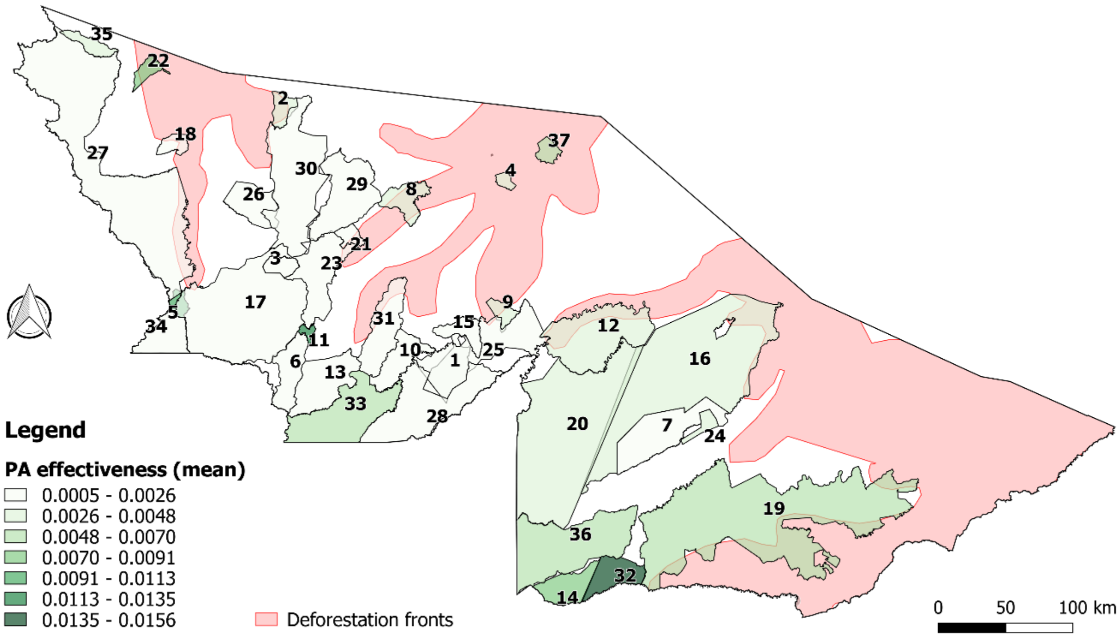


Figure 10. Mean-based effectiveness of the PAs considered in this thesis. The effectiveness of the seven PAs mostly or completely within the deforestation fronts varied.

### 3.4. *Three protected areas had negative confounding effects*

Part of the reason why PAs 11 and 32 had high effectiveness was the fact that the confounding effects for those PAs, along with PA 19, turned out to be negative (Figure 11), meaning that more deforestation, instead of less, was expected for these PAs when the covariates were used. PAs 11 and 19 had the highest fractions of deforestation within them compared to any other PA included in this study (Figure 8), whereas PA 32 had no deforestation within its borders. Both the PA effectiveness values and the confounding effects were different when calculated using medians, as should be expected. Out of all PAs, only PA 11 showed a negative confounding effect when medians were used, indicating that the negative confounding effect was clear enough to be seen even with that measure, albeit not in as much detail as when the means were used.

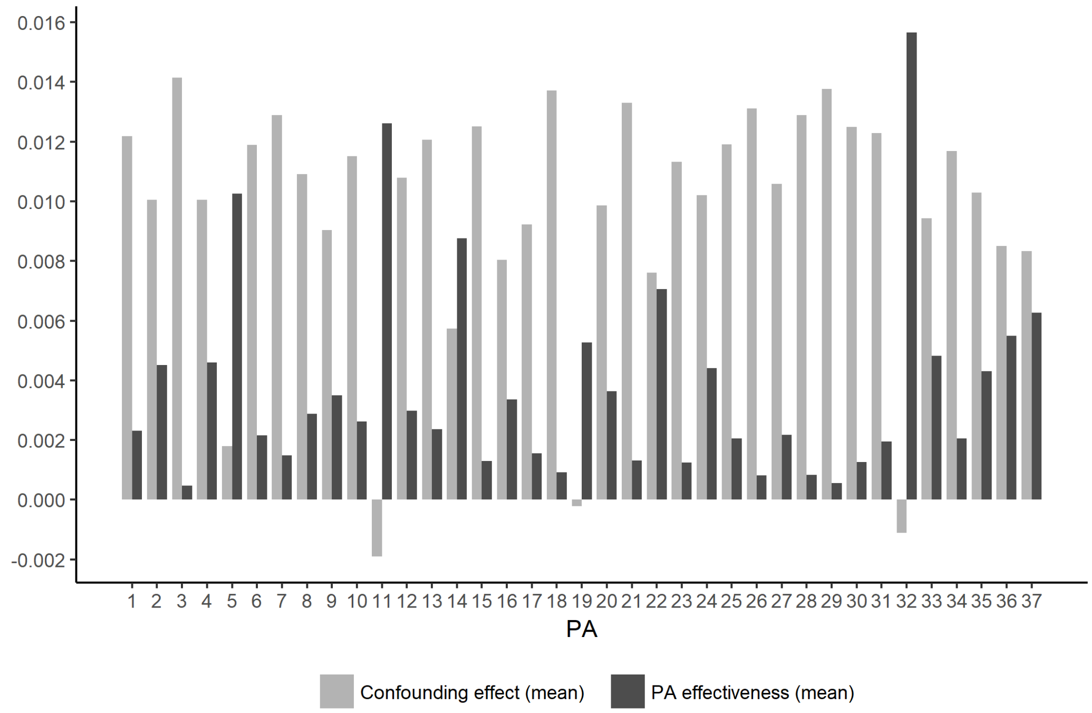


Figure 11. Mean-based confounding effects and the mean-based PA effectiveness measure for all PAs. Three PAs experienced a negative confounding effect indicating high pressures against these PAs. The y-axis for mean effectiveness is calculated as the fraction of pixels that would have been expected to experience deforestation if the PA had not existed. In other words, the avoided deforestation. For the confounding effect, the y-axis is the remaining fraction of deforestation that would have been expected without the confounding factors or protection. When the confounding effect is negative, more deforestation was expected for the PA based on the covariates (BL_multidim) as compared to the overall background rate estimated without them (BL_no_multidim).

PA 19 is an extractive reserve and the largest PA in Acre, and it had the largest fraction of deforestation out of all PAs within the time period of this study (Figure 8), more than two times as much as the PAs with the second and third largest fractions. Yet, this did not surpass the estimated baseline deforestation rate and therefore PA 19 was estimated to have had a high mean-based effectiveness. According to the $PS_{dep}$ measure, PA 19 was only intermediately effective as only around 30% of the similarity sets showed less deforestation than the baseline. The confounding variables did not reduce the effectiveness measure for PA 19 but instead increased it, the same being true for the other PAs with negative confounding effects.

## *3.5. Substantial amount of deforestation and carbon emissions avoided*

The PAs included in this study avoided approximately 2,255,000 ha of deforestation in Acre during the study period, totalling to 217,394 kilotons (0.22 gigatons) of avoided carbon emissions (Table 7). 23,041 ha of deforestation was not included in the emissions calculations, as the carbon density data was not available for two PAs (3 and 28). According to the World Development Indicators data by The World Bank (2018a), the $CO_2$ emissions in Brazil were 2,729,828 kilotons in total during the latest six years from which data are available (between 2009 and 2014). The PAs I included in the carbon calculations avoided around 8% of this total during the six-year period considered in this thesis. Most deforestation was avoided by PA 19, a sustainable use area, which avoided approximately 483,930 ha of deforestation and 47,922 kt of carbon emissions during the period under study (Figure 12, Table 7). PA 19 is also the largest PA in Acre, with over 93,901 hectares more area than the second largest PA 27. The next most effective PAs in terms of avoided deforestation were, in order, PA 16 (another sustainable use area), 20 (strictly protected area), 27 (strictly protected area), 36 (indigenous area) and 32 (indigenous area). The order of PAs was the same when sorted by most avoided carbon emissions.

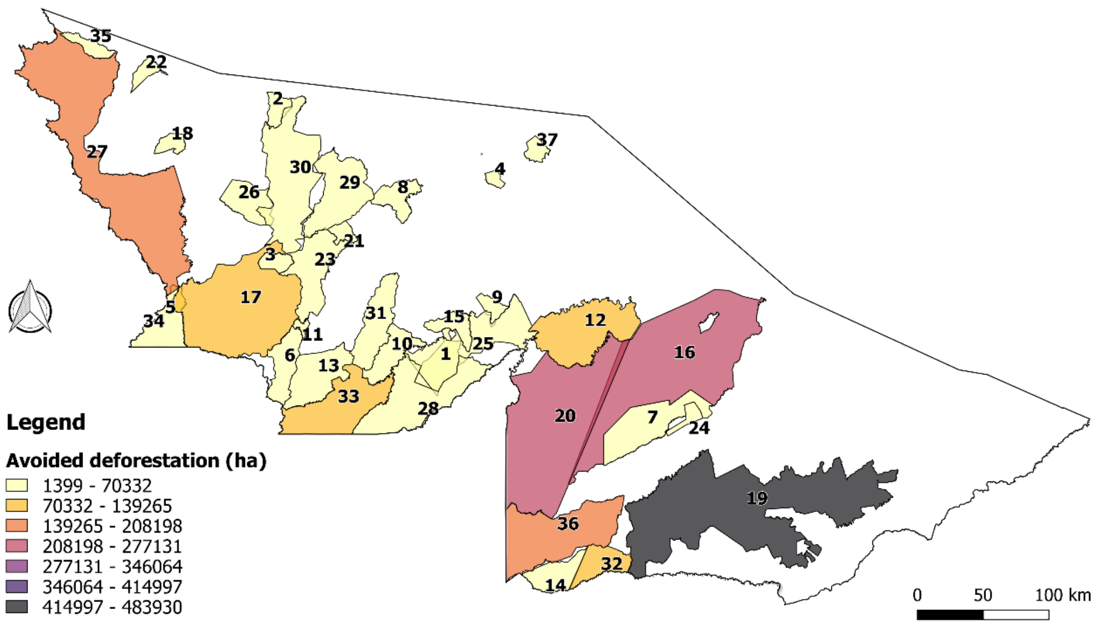


Figure 12. Avoided deforestation by the PAs in hectares. Both the extent and effectiveness of the PAs affected these estimates.

Table 7. The estimated amount of avoided deforestation and carbon emissions for individual PAs. Mean Carbon (C) ton/ha values were obtained from Soares-Filho et al. (2010). PAs 3 and 28 had to be omitted due to a lack mean C ton/ha data. It should be noted that the values are only approximations and they are based on conservative estimates of PA effectiveness. In addition, some PAs had overlap (see in Figure 12), which is not considered in the totals.

| PA | PA type | PA area (ha) | Mean C (ton/ha) | Avoided def. (pix) | Avoided def. (ha) | Avoided C (t) | Avoided C (kt) | Avoided C (GtC) |
|---|---|---|---|---|---|---|---|---|
| 1 | INDIG. | 80618.2 | 108.8 | 3452.3 | 18570.2 | 1717510.3 | 1717.5 | 0.0017 |
| 2 | INDIG. | 32623.6 | 138.9 | 2823.6 | 15188.3 | 1793841.1 | 1793.8 | 0.0018 |
| 3 | INDIG. | 28926.1 | - | 260.0 | 1398.5 | - | - | - |
| 4 | INDIG. | 12317.9 | 94.1 | 1027.0 | 5524.1 | 441790.6 | 441.8 | 0.0004 |
| 5 | INDIG. | 20764.0 | 114.4 | 3763.2 | 20242.1 | 1967585.8 | 1967.6 | 0.0020 |
| 6 | INDIG. | 87293.8 | 100.9 | 3532.8 | 19002.9 | 1630040.6 | 1630.0 | 0.0016 |
| 7 | SUS. USE | 176346.7 | 116.6 | 4962.2 | 26691.6 | 2646408.4 | 2646.4 | 0.0026 |
| 8 | INDIG. | 60698.7 | 116.3 | 3193.1 | 17175.9 | 1698444.0 | 1698.4 | 0.0017 |
| 9 | INDIG. | 27533.4 | 102.6 | 1643.1 | 8838.1 | 771076.2 | 771.1 | 0.0008 |
| 10 | INDIG. | 84364.6 | 98.4 | 3982.6 | 21422.6 | 1791008.7 | 1791.0 | 0.0018 |
| 11 | INDIG. | 8726.5 | 99.5 | 2002.0 | 10768.9 | 910905.4 | 910.9 | 0.0009 |
| 12 | INDIG. | 263129.8 | 114.2 | 14424.7 | 77590.6 | 7533793.2 | 7533.8 | 0.0075 |
| 13 | SUS. USE | 142619.1 | 103.7 | 6174.1 | 33210.5 | 2736775.5 | 2736.8 | 0.0027 |
| 14 | STRICT | 79092.6 | 120.5 | 12820.7 | 68962.6 | 7061944.3 | 7061.9 | 0.0071 |
| 15 | INDIG. | 45590.9 | 104.2 | 1089.8 | 5861.9 | 519267.2 | 519.3 | 0.0005 |
| 16 | SUS. USE | 750917.7 | 123.0 | 47775.1 | 256982.3 | 26862930.0 | 26862.9 | 0.0269 |
| 17 | SUS. USE | 537983.7 | 106.8 | 15449.6 | 83103.1 | 7543075.3 | 7543.1 | 0.0075 |
| 18 | INDIG. | 25651.6 | 125.9 | 443.6 | 2386.3 | 255295.8 | 255.3 | 0.0003 |
| 19 | SUS. USE | 931459.0 | 116.5 | 89966.5 | 483929.6 | 47922297.3 | 47922.3 | 0.0479 |
| 20 | STRICT | 693974.4 | 109.5 | 46750.4 | 251470.5 | 23400723.5 | 23400.7 | 0.0234 |
| 21 | INDIG. | 21987.2 | 105.9 | 543.2 | 2921.8 | 262889.2 | 262.9 | 0.0003 |
| 22 | INDIG. | 24499.1 | 119.1 | 3149.3 | 16940.3 | 1715131.4 | 1715.1 | 0.0017 |
| 23 | SUS. USE | 150923.2 | 103.7 | 3474.7 | 18690.3 | 1647687.6 | 1647.7 | 0.0016 |
| 24 | SUS. USE | 21147.7 | 114.9 | 1764.2 | 9489.8 | 927025.8 | 927.0 | 0.0009 |
| 25 | SUS. USE | 231556.1 | 101.0 | 8831.2 | 47503.1 | 4077715.4 | 4077.7 | 0.0041 |
| 26 | INDIG. | 87571.7 | 114.9 | 1357.7 | 7303.0 | 713136.7 | 713.1 | 0.0007 |
| 27 | STRICT | 837557.3 | 116.1 | 33145.4 | 178289.2 | 17597599.7 | 17597.6 | 0.0176 |
| 28 | INDIG. | 260970.0 | - | 4023.6 | 21642.7 | - | - | - |
| 29 | INDIG. | 187400.0 | 114.5 | 1973.3 | 10614.6 | 1033132.6 | 1033.1 | 0.0010 |
| 30 | SUS. USE | 324904.1 | 120.4 | 7727.1 | 41563.9 | 4252111.3 | 4252.1 | 0.0043 |
| 31 | INDIG. | 127383.6 | 98.7 | 4630.9 | 24909.8 | 2090043.4 | 2090.0 | 0.0021 |
| 32 | INDIG. | 78512.6 | 128.4 | 22871.3 | 123024.8 | 13424432.6 | 13424.4 | 0.0134 |
| 33 | INDIG. | 232795.0 | 99.5 | 20431.8 | 109902.7 | 9290920.8 | 9290.9 | 0.0093 |
| 34 | INDIG. | 87205.4 | 112.6 | 3193.6 | 17178.5 | 1644170.7 | 1644.2 | 0.0016 |
| 35 | INDIG. | 27263.5 | 117.9 | 2545.2 | 13690.6 | 1372028.8 | 1372.0 | 0.0014 |
| 36 | INDIG. | 313646.9 | 117.4 | 31358.2 | 168675.8 | 16838251.9 | 16838.3 | 0.0168 |
| 37 | INDIG. | 23474.0 | 106.9 | 2664.5 | 14332.1 | 1302607.0 | 1302.6 | 0.0013 |
| Totals: | | 7222289.7 | 3899.9 | 419652.0 | 2257307.7 | 217393598.3 | 217393.6 | 0.2174 |

# 4. Discussion

## *4.1. My results as a part of the larger conservation discourse*

My thesis considered a period of time from 2011 to 2016. Despite the economic recession and law changes which rocked Brazil during this time, all PAs included in my thesis had at least some level of effectiveness. This supports the well-established fact that, overall, PAs are effective at curbing deforestation (DeFries et al., 2005; Joppa and Pfaff, 2011; Lui and Coomes, 2016). I also found substantial variation in effectiveness between the individual PAs in Acre, corroborating previous matching studies which have shown similar patterns with different time periods and study areas, such as for the entire Brazilian Amazon (Nolte et al., 2013), the Brazilian Cerrado (Carranza et al., 2014), Peruvian Amazon (Schleicher et al., 2017) and the tropical protected areas in Africa (Bowker et al., 2017). However, I also observed some differences when comparing my results to these previously performed studies. Unlike the findings from the entire Brazilian Amazon or the Brazilian Cerrado, my findings did not show that the indigenous areas of Acre would have experienced more deforestation pressures compared to the other main PA types (Figure S2).

Based on my results, the main PA types cannot confidently be said to differ from each other in terms of PA effectiveness in Acre, even though previous research for Amazonian PAs has discovered that such differences do exist when the entire Brazilian Amazon is considered (Soares-Filho *et al.*, 2010; Nolte *et al.*, 2013). Differences have also been found in the Cerrado (Carranza et al., 2014) and the Peruvian Amazon (Schleicher et al., 2017). The fact that in Acre the PA types differed neither in the pressures they experienced nor in their effectiveness during my study period, is likely a consequence of the location of the individual PAs belonging to each category. In other words, none of the PA categories were predisposed to having higher or lower effectiveness by virtue of how the PAs belonging to those groups were located. This finding will likely not change in the future even if the pressures continue to increase in the deforestation fronts, unless new PAs are established or existing ones are degazetted. It might also well be that some relevant differences between the types do exist in their ability to avoid deforestation, but they may be too small to detect with the number of PAs Acre has.

### 4.1.1. Augmenting previous PA effectiveness studies in the Amazon

Prior to this thesis, two large studies on PA effectiveness in the Brazilian Amazon have been conducted, both using data from before the start of my study period. The first one of these was a study by Soares-Filho *et al.* (2010), who, instead of matching, used an adjusted odds ratio method to compare deforestation in 10 km buffers within and outside the PAs. The authors used data from all of the 595 PAs in the Brazilian Amazon between the years 1997 to 2008, to estimate the effectiveness of PAs in preventing carbon emissions.

Soares-Filho *et al.* (2010) found that, on average, strictly protected, sustainable use and indigenous areas showed an inhibitory effect, the most effective category being the indigenous lands. The authors also provided their effectiveness estimates for individual PAs, but these could not be reliably compared to my own effectiveness measures for the same PAs as there were different number of years of available data for each PA in the table provided by the authors, and no year had data for all PAs. When viewing their data, I noticed that the effectiveness measures estimated by Soares-Filho *et al.* (2010) were much higher than the estimates I derived for the same PAs. Soared-Filho had compared 10 km buffer zones immediately within and outside the PA border, comparing the zones to each other, without comparing individual pixels to each other. The authors wrote, that 453 of 571 of their buffer zone pairs turned out to have had significantly different distributions of deforestation probability, which may have affected their estimates of PA effectiveness, because they were not comparing like with like. The authors acknowledged this in their supplementary text, writing that the different distributions suggested that the differences should have been compensated for, for example by using matched samples. It is likely, that the matching and covariate choices I used in my thesis correctly reduced the effectiveness measure for most PAs, resulting in a more realistic measure of effectiveness. Some, if not most, of the differences between my estimates of PA effectiveness and the estimates calculated by Soares-Filho *et al.* (2010) were due to the differences in the selected methods, but a part of the differences must surely also be attributed to the different time-period considered, as deforestation in Acre has dropped substantially compared to the levels in the early 2000's (Figure 2). What fraction of the difference was due to the difference in methods and what was due to real changes in PA effectiveness between the time periods considered could not, however, be estimated.

The second major PA effectiveness study in the Brazilian Amazon prior to the start of my study period was performed by Nolte et al. (2013), who used a matching method to investigate the effectiveness of different PA types for two different time periods, 2000-2005 and 2006-2010. The findings of Nolte et al. (2013) showed that strictly protected areas avoided more deforestation than sustainable use areas in any given level of deforestation pressure, and indigenous lands were particularly effective in high pressure areas. In contrast to the results of Nolte et al. (2013), the results I acquired with the PAs of Acre did not show significant differences between the PA types in the years after 2010, or at least the difference in effectiveness between the three PA types was not so large as to be significant with the small sample sizes in Acre.

The effectiveness of the main PA types seems to be context dependent and largely driven by the placement of the different types of PAs. If many PAs of a specific category are established on high pressure areas, this will drive up the effectiveness of this PA category in the analyses, even if the other types could have been just as effective if similarly located. Indeed, Nolte et al. (2013) also reported shifting trends in the location of PAs, with strictly protected areas having been more likely to be established in high-pressure locations during 2000-2005 than sustainable use or indigenous areas. The results of Nolte et al. (2013) from the two periods of time they considered pointed to the conclusion that increased efforts of the Brazilian government during the second time period increased the overall effectiveness of all protection types. However, the writers also noted that protection is not guaranteed under any governance regime despite the consistent average patterns, as they observed individual cases with high and low deforestation rates for all protection types, pressure levels, and time periods (Nolte *et al.*, 2013). My results showed how the same continued to be true for the PAs in Acre, where the fraction of deforestation varied significantly between the individual PAs. This is one reason why it is important to carefully consider the context dependency of effectiveness estimates, especially if scientific research is used to inform management decisions for individual PAs.

### 4.1.2. Patterns in pressures and effectiveness

All PAs with the highest effectiveness scores had rivers flowing through them, increasing the ease of access and thus predisposing their areas to potential deforestation. This reality was reflected by the high baseline deforestation estimates for these PAs, which resulted in high estimated effectiveness. In contrast, the lowest effectiveness scores were found for PAs in remote locations which experienced low deforestation pressures and thus their effectiveness was not challenged. The low effectiveness estimates do not necessarily mean that the PAs would remain low in effectiveness if the pressures were to change. PA effectiveness is not, and indeed does not need to be, high in areas with low pressures. In addition to location, the effectiveness of a PA is partly determined by the period of time considered. This thesis only considered the state of Acre, using the latest six years of available data. A key thing to realize when interpreting these findings is the fact that the effectiveness measure is not an absolute measure, but instead a relative one.

Effectiveness of the seven PAs inside the deforestation fronts varied from low to high effectiveness based on my results. However, the estimate for PA 18, the one PA with low effectiveness, was surprisingly low considering that there was a large amount of deforestation both on the southern and northern edge outside of this indigenous area. Two of the indigenous areas within the deforestation fronts had a lower effectiveness measure due to deforestation near the rivers that crossed their areas (PAs 8 and 9). Nearly half of all indigenous areas included in my thesis experienced some deforestation. This finding reflects the results reported by Nolte *et al.* (2013), who wrote in their article that the indigenous areas in the entire Brazilian Amazon region exhibited slightly higher amounts of deforestation in low-pressure areas than other main PA types, suggesting that deforestation within indigenous areas might reflect internal pressures rather than external market-driven pressures which the covariates mostly control for. The internal subsistence-driven resource use of indigenous lands could explain why they have been found to be effective in high pressure areas and why they seem to exhibit deforestation regardless of the pressure levels.

I found that three PAs currently exist on areas of very high anthropogenic threat, indicated by negative confounding effects (PAs 11, 19 and 32). Such effects are expected to be found for PAs in high pressure locations regardless of the study site, but so far, I have not seen any study in this field of research reporting negative confounding effects for PAs. These negative confounding effects can reflect the fact that those PAs either have been established on high pressure areas to defy the deforestation threat, or the pressures in those areas have grown and the PAs now face great pressures. For PA 19, the negative confounding effect was a consequence of the latter (see case study below), and the same is likely true for PAs 11 and 32 which were established in 2002 and 1999, respectively (Table S2).

## *4.2. True protected area effectiveness is a series of actions*

When a PA is faced with deforestation pressures, its level of effectiveness is reliant of multiple factors, such as administration, planning, protection activities, monitoring, law enforcement, staff training, the security and reliability of funding, the involvement of communities and stakeholders, and the support of the external political and civil environment (Leverington *et al.*, 2010). Naturally, since the main PA types differ in their approach to conservation, the importance of these factors differ between them. For sustainable use areas the series of actions is especially complex, as these areas seek to accommodate conservation interests with the varying needs of different stakeholders. In order to illustrate this process, I will next examine the ganglion of conservation discourse in Acre, the Chico Mendes Extractive Reserve, as a case example.

### 4.2.1. A closer look at the Chico Mendes Extractive Reserve

Labled as PA 19 in this study, this sustainable use area had avoided the most deforestation and carbon emissions during the time period I studied, while also experiencing the largest losses of forest cover and carbon out of all considered PAs. The negative confounding effect for this PA suggested that the reserve exists in an area with very high deforestation pressures, and I estimated the reserve to have had an intermediate-to-high effectiveness. The reserve is situated a mere 30 kilometres from Rio Branco, the capital of Acre, and the vicinity of the reserve has an extensive road network, increasing the accessibility to the reserve. Several roads extend inside the reserve (Figure 13).

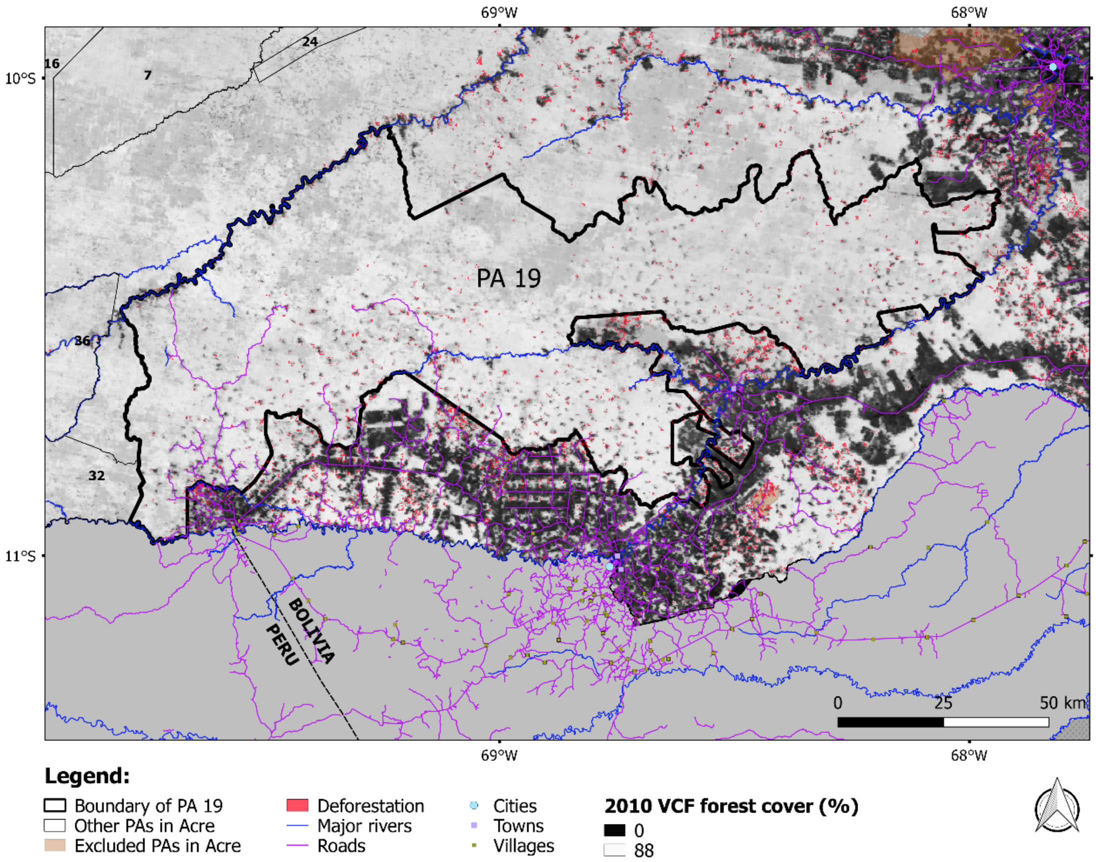


Figure 13. Map of the Chico Mendes Extractive Reserve (PA 19), with a baseline forest cover layer.

The reserve was established in 1990 and named to honour the famous rubber tapper and environmentalist Chico Mendes (ARPA, 2014), and it is managed cooperatively by the Brazilian Ministry of the Environment and the local communities with their representative organizations. Together the administrators form a management council, which agrees on a Management Plan, together with a Usage Plan and a Use Concession Agreement (IBAMA, 2006; WWF-Brazil, 2015). The reserve was designated to protect the sustainable use of the unit's natural resources by supporting the livelihoods and culture of rubber tapper families, and deforestation for the accomplishment of complementary activities is only allowed through a license obtained from the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA), and is legally limited to two hectares per year (IBAMA, 2006; CNUC, 2018).

The causes of deforestation within the Chico Mendes reserve have been due to an increase in the complementary activities, directly linked to the growing of livestock, fires and poor planning of agricultural production (IBAMA, 2006). As rubber is more difficult to gather, transport and sell than cattle, especially the younger generations today would rather choose the easier way, even if it harms the forests (Marcel, 2013). The management authorities of the reserve have even planned awareness raising campaigns for the youth of the reserve about the importance of restraining from deforestation for the sustainability of the unit (IBAMA, 2006). In the official Management Plan for the reserve, it is mentioned that the traditional extractive production of latex and Brazil nuts is directly associated with the reduction of deforestation rates, contributing to the provision of several types of environmental services, and carbon credit markets have been envisioned as a possible incentive for providing these services (IBAMA, 2006). Acre's new REDD+ program has been used to enable price mark-ups for the rubber products, incentivising traditional forest conserving practices in the reserve (KfW Development Bank, 2017), which may have increased the effectiveness I estimated for this PA.

The Chico Mendes Extractive Reserve has a particularly important symbolic relevance for conservation in Acre, and the ongoing transition from rubber production to cattle ranching has raised some worry (Otavio, 2013). However, local people seem to feel that the burden of environmental responsibility has been showed on their shoulders, even though they see their own actions as minuscule compared to the actions of the big oil and mining industries in the Amazon (Otavio, 2013). In a community forest monitoring study focusing on the reserve it was stated that 67% of households today raise cattle, with only 21% of households collecting rubber (Sabogal, 2015). Of the 551 people interviewed during the monitoring, 67% felt represented by the Management Council and 63% were aware of the Management Plan, but only 21% of those who were aware of the plan said it is working (Sabogal, 2015). This might reflect the growing pressures and changes taking place within the reserve. My results show that during recent years, many families have been incentivized, either by market logic or by the necessity to survive, to deforest within the reserve, as around 1% of the reserves vast area was estimated to have experienced deforestation. The community forest monitoring study also recorded 172 illegal invasive activities in the reserve, of which 41% were for hunting (Sabogal, 2015), reflecting the fact that true PA effectiveness is also influenced by factors other than forest cover loss, which is not seen when the effectiveness is only evaluated with remote sensing methods.

My results suggest that the management plan of the Chico Mendes Extractive Reserve has indeed had intermediate-to-high success in recent years, despite the deforestation that is happening in the reserve. This case example thereby exhibits how even in the extremely high-pressure environments of the agricultural frontier, sustainable use areas can survive and be effective, if a successful plan for management is created and implemented which reconciles conservation requirements with the needs of the people. Given the high pressures of this deforestation front, it seems likely that without this reserve the sustainable management of the forests resources, most of the carbon bound in the forests, along with the traditions, culture and way of life of the rubber tappers, might all have been lost from this extensive area.

The fact that rubber tappers have had the opportunity to continue their way of life may have produced better results in this area of high deforestation pressure than a strictly protected area like a national park would have done. This is because by allowing the small-scale deforestation resulting from the extractive activities, the rubber tappers did not have to migrate to other areas where the incentive to become cattle farmers might have been even higher. This process would have undeniably also created opposition towards environmental protection, affecting the larger conservation efforts in the long term. However, if the transition from the sustainable rubber tapper practices to cattle ranching will continue, the future managers of the park will have less incentives to maintain as stringent rules which prevent deforestation today. This might mean that in order for this historical reserve to retain its effectiveness in the future, the conservation status might eventually have to be fully or partially changed to a stricter protection status, possibly by modifying the existing zonation within the reserve.

## *4.3. Effectiveness beyond deforestation prevention*

PAs can differ in the reasons for which they were established, but one common unifying factor in rainforest environments is the intention to conserve forest habitats and the resources and services they provide. Deforestation can indicate the amount of remaining habitat, but not necessarily its quality. Studies that rely on measuring PA effectiveness via remote sensing methods, like the one I performed, have to depend on features that are visible to satellites, which mainly means the loss of forest cover. While the lack or presence of deforestation is a clear representation of PA effectiveness, it should not be considered as the sole determinant of it, as PAs can be effective along other dimensions as well. My findings thus provide one evaluation of effectiveness, but it must be kept in mind that PAs are established for varying reasons, and therefore to obtain the most policy relevant estimates, the measure of effectiveness should be selected to reflect the case specific objectives of each PA in question.

Besides clearing, the quality of forests may also be affected by degrading the standing forests. Forests may be degraded for example by selective logging of chosen species or by unsustainable levels of hunting for large-bodied vertebrates which can result in an "empty forest" (Redford, 1992). Hunting induced decline in species diversity can have reverberations on the ecosystem functions and services the forests provide, affecting both the composition of the forests and their carbon density (Lewis, Edwards and Galbraith, 2015). Increased hunting pressure is the consequence of a growing anthropogenic presence in the rainforests, with people either in search of subsistence or seeking to satisfy demand arising from larger towns or international trade (Redford, 1992). Selective logging and hunting may also occur together, as logging can provide road access to hunters. Schleicher *et al.* (2017) included an assessment of forest degradation to their estimations of PA effectiveness, and their findings showed that forest degradation affected a much larger extent of forest area than deforestation did, within all PA types they considered in the Peruvian Amazon, and the amount of forest degradation PAs avoided was also consistently higher than the avoided deforestation. Schleicher *et al.* (2017) assessed, that the main predictor for both forest degradation and deforestation was accessibility, in particular the distance to previously deforested areas and the travel time to urban centres. Especially PAs which exist near roads or major rivers in Acre may therefore have been subject to forest degradation, even if they had high effectiveness in preventing deforestation.

## *4.4. Evaluating the method and possible caveats*

I made several assumptions in this thesis which may affect the estimated effectiveness results and are therefore necessary to consider when the results are interpreted. The original idea for this thesis was to study PA effectiveness in and near the deforestation fronts of the Amazon, which is why the PAs were selected using the 50 km buffer area around the deforestation fronts. Only one PA was excluded in Acre by applying this criterion, and the other selection criteria that were used only left four additional PAs out of the analyses. In retrospect, it would have been more valuable to include all PAs of Acre to the analyses and point out the exceptions where necessary.

One possible caveat of the method used is the fact that when the code searches for control pixels for the sampled pixels, the controls can be searched from a distant location, which might lead to an underestimation of the effect local conditions can have on deforestation, for example through municipality or town specific rules or markets, which were not considered in this thesis. However, this effect is likely to be very small, as local pixels are expected to be more environmentally similar to the sampled pixels than faraway pixels and are therefore also expected to be present in the set of pixels to which the comparisons are done.

I also checked to see if sample size had any effect on the effectiveness measures, as PAs differed considerably in sample size, ranging from 158 to 17048 sample points (Table S1). I observed no relationship, as should be expected, since the similarity sets have a constant size of 500 pixels, regardless of the PA sample size, and the PA sample size is relative to forested extent of the PAs in the baseline year 2010 (approximately, see Table S1). Small PAs may however end up having larger effectiveness measures a consequence of the fact that a small sample size results from a small PA area (less forest in smaller PAs), which increases the likelihood that the PA can exist wholly or mostly within a high-pressure environment, thereby potentially increasing its measured effectiveness.

Leakage was not considered in this thesis. It means an effect where instead of preventing deforestation, a PA simply displaces it to another area. A commonly used approach in PA effectiveness studies is to include a buffer around each PA, usually 10 km from the PA boundary, to account for potential leakage (Joppa and Pfaff, 2011; Nolte *et al.*, 2013; Carranza *et al.*, 2014; Schleicher *et al.*, 2017). Leakage was not considered in this thesis because the matching method I used does not rely on comparing zones inside and outside PAs, and previous research has found little to no support for the leakage hypothesis (Soares-Filho *et al.*, 2010; Carranza *et al.*, 2014). In fact, instead of leakage it seems that PAs may sometimes reduce deforestation in their vicinity (Soares-Filho *et al.*, 2010). If this is the case, instead of increasing the estimated effectiveness, not including buffers may have even made my effectiveness estimates slightly more conservative.

The economic recession and political crises that took place in Brazil after 2010 may have had an effect on the estimated deforestation pressures and PA effectiveness, but their specific effects could not be determined in this thesis. When I visually compared the different GDP measures of Brazil (World Bank, 2018) to the PRODES deforestation data, there seemed to be no consistent relationship between different measures of GDP and deforestation in Brazil between the years 2000-2016. Therefore, it may be that changes in laws and policy have affected deforestation more than the economic recession in Brazil. PA effectiveness may have been influenced if resources for PA management have been scarce, but this could not be evaluated.

The 2012 Forest Code change in Brazil has been estimated to raise deforestation in non-protected areas by 10% (Roriz, Yanai and Fearnside, 2017), which means it had the potential to increase my baseline deforestation estimates. However, it may take time for people to become aware of the new law and change their patterns of behaviour. The REDD+ program and the history of forest conservation in Acre may have also helped buffer against the possible ramifications for deforestation pressures and PA effectiveness during my study period.

### 4.4.1. Including protected area to estimations of baseline deforestation

One very definitive pragmatic decision done in this thesis was to include the protected area into the calculation of the baseline deforestation rates. This decision was done following the original method as it has previously been used by Eklund *et al.* (2016). This influenced my results by slightly lowering the estimated deforestation pressure and therefore the estimated PA effectiveness, making my estimates of effectiveness and pressure slightly more conservative. Because the decision reduced the estimated PA effectiveness results, it also reduced the estimates of avoided deforestation and avoided carbon emissions calculated for each PA.

The baseline deforestation rate which disregarded covariates and protection (BL_no_multidim) was calculated separately for each PA by taking the area of only that PA and the whole non-protected area in Acre and calculating the mean deforestation rate from 100,000 simulated 500-pixel sets. The inclusion of the protected area was based on the argument that the same area was considered for the estimation, only disregarding the protection status. This may not have been the best approach, as protection is expected to reduce the deforestation rate within it, meaning that the resulting baseline deforestation rate will be estimated to be lower than it actually is if pixels are included from within the protected area. In the same way, the baseline effectiveness measure which considered covariates (BL_multidim), calculated from the similarity sets, used all of the pixels included in the similarity sets instead of only using the non-protected matched pixels. This means that the baseline pressure might actually be somewhat higher for the PAs than what was estimated, and therefore the effectiveness estimates are estimated to be lower.

In hindsight, the protected pixels should have been excluded when calculating the baseline deforestation rates, as including them influenced the results by making the effectiveness estimates conservative, unnecessarily. The magnitude of this decision was different for each PA, affected by the fractions of pixels in the similarity sets that were selected from within the PA. For most PAs, the size of this effect was small, as the non-protected area is vastly larger than most of the PAs in Acre, and therefore most similarity set pixels or the pixels in the simulated sets were found from the non-protected areas of Acre. This was seen from the histograms that were created before the analyses were done (Figure 4). However, PAs 7, 13, 14, 20, 28, 34 and 36 had particularly high frequencies of similarity sets with a high fraction of protected pixels within them, and therefore their effectiveness might have been underestimated. In a similar manner, the confounding effects calculated for each PA might have been estimated to be slightly too high.

In the previous study which introduced the method used in this thesis, Eklund *et al.* (2016) included the area under protection in the estimates of baseline deforestation pressure because the idea was to evaluate deforestation rates disregarding protection. Based on verbal communication with the author, their results did not vary much when the PA was not included. However, my study did not estimate PA effectiveness only for aggregated groups of PAs, and therefore it would have been better to exclude the protected area from the deforestation rate estimations.

### 4.4.2. Sub-optimal covariate layers

Including the VCF layer as a covariate may have affected the matching in an unintentional way, because similar pixels were searched also by comparing the density of forest cover. Hypothetically, the forest cover density might reflect the structure of the forest, and therefore affect the likelihood for deforestation, but this is not known for certain. In addition, the forests of Acre have two major forest types, bamboo forests and palm tree forests, which conceivably might have some detectable difference in the density of forest cover, and there might also be a difference in the likelihood for deforestation between these forest types. These hypotheticals have not been tested, however, and therefore cannot be used to support the inclusion of the VCF layer.

The median altitude and slope datasets from IIASA and FAO were not optimal choices due to their coarse resolution of around 9 km. In hindsight, a much better option would have been the SRTM 90 m Digital Elevation maps. These dataset choices reduced the ability to detect possibly agriculturally suitable areas in detail and may therefore have resulted in a less optimal selection of control points.

True accessibility may also differ from the one measured by the covariates used in this study, as some parts of rivers may be less navigable, or alternatively some tributaries or distributaries which were not included in the river layer may in reality have influence on the accessibility to forest resources.

The travel time layer was from the year 2000, a decade before the start of my study period, and after that roads have been paved, affecting the true travel times to markets. There were some shortcomings in the travel time raster that seemed not to reflect the true positions of some roads and rivers, but I decided to use the layer as a covariate because it was only one covariate among many, and as travel times have been evaluated to be important predictors of deforestation pressures. In addition, no better option was available, and time constrained me from calculating my own travel time layer. A new map of travel times and global accessibility was published in early 2018 (Weiss *et al.*, 2018), but it turned out to have even more errors than the previous layer when the area of Acre was concerned.

### 4.4.3. Impact of overlapping protected areas

Several PAs had some amount of overlap with each other (Figure 1), which may have affected the effectiveness of these areas with “double protection”. These overlapping areas might have slightly affected the avoided deforestation and carbon emission estimates. The overlap was observed to be minor for most PAs, except for PAs 1, 5 and 25. PA 1 is an indigenous area and almost all of its area overlaps with PA 25. a national forest. This dual protection can be expected to have increased the estimated effectiveness for PA 25, and possibly also for PA 1, as national forests in Brazil are semi-open to the public and have management plans. The effectiveness of PAs 1 was intermediate, and PA 25 had medium-to-high effectiveness. PA 5, a small indigenous area, was one of the PAs for which I estimated a very high effectiveness, and therefore it warrants a closer look to evaluate if the effectiveness estimate was correct or not.

A river crosses straight through PA 5, predisposing its area to the edge effect. The PA is almost completely surrounded by three other PAs, two of which overlap with it, covering in total more than half of its area (clearly visible in Figure 10). East from the river bank the overlap is with an extractive reserve categorized as a sustainable use area, and the northern part is overlapped by a strictly protected national park. In the non-protected areas to the north, downstream of the river, and to the east within the extractive reserve, a great amount of deforestation had taken place during the study period, but only a little had materialized within the borders of PA 5, where 7 pixels worth of forest had been lost. No deforestation had taken place within the parts of PA 5 which overlapped with the strictly protected area, but four deforested pixels were situated on the area overlapping with the extractive reserve near the eastern bank of the river. Overall, the deforestation fractions of the similarity sets for PA 5 might have been reduced by the overlap with the strictly protected area, but the deforestation in the area overlapped with the extractive reserve did not have an effect, since none of the deforested pixels happened to be included in the sample for PA 5. It is not possible to say for certain if there would have been more or less deforestation without the reserve on the east bank. Satellite images and the baseline forest cover layer (VCF) indicated that forests within PA 5 had the potential to be lost on the exclusively indigenous territory in the west bank as well. The effectiveness of PA 17 (the extractive reserve) was not likely affected much by the overlap with PA 5, as it represents only a small corner of the large reserve (Figure 10). The histograms from the CSC run output files showed that the similarity sets of PA 5 had a low fraction of protected pixels, which indicates that the baseline deforestation rates were not affected by the overlapping areas in any serious way.

The effectiveness estimate for PA 5 was further influenced by the fact that almost 25% of the samples in its area were omitted due to a lack of slope data in the edges of Acre, on the western part of the PA. Had these samples been included, there would have been 25% more focal points with zero deforestation, as no deforestation had taken place within the omitted area. If the 25% of omitted pixels with no deforestation had been included, there would have been deforestation on 0.255% of the sampled pixels, which corresponds rather well with the 0.259% mean PA_multidim deforestation result that was obtained with the actual data used for the calculations. Based on the above, neither the PA_multidim or the BL_multidim would have likely differed much in the end, most likely resulting in a similar effectiveness estimate for PA 5 during the period under study.

#### 4.4.4. The sampling design

In addition to PA 5, the missing data in the covariate rasters also affected the sample size of all other PAs which were located on the edge of Acre (Figure 1), as the samples with missing values were omitted from the analyses. No deforestation had taken place near Acre's edge, in the areas which didn't have the covariate data, meaning that the effectiveness estimates might have been slightly lower if the samples from these areas had not been omitted.

Another problem with the sampling turned out to be the way I omitted samples in the non-forested areas. I intersected the sample points with the VCF layer and then omitted the samples that were on non-forested areas, but the result of this was that most PAs the remaining forested areas did not have a perfect 10% sampling effort, depending on the amount of non-forested area that each PA had in the VCF layer. The same problem was true for the non-protected area. Despite this methodological error, the samples were approximately 10% and varied little between the areas (mean over all: 9.80, SD: 0.44; Table S1, Figure S1). Some of this variation was due to differences in the amount of rivers, roads and non-forested areas within the PAs, but the lack of slope data at the edges of Acre was the largest cause for why PAs deviated from the 10% sample. The sample sizes of PAs 5, 34, 32, 6 and 28 were most affected, and they were all situated near the edge of Acre. Only three PAs had a sample under 9% of the forested pixels (PAs 5, 34 and 32), of which the lowest was PA 5 (8.04%). A better option for sampling would have been to exclude all non-forested areas from the polygons within which the samples were randomly selected in the first place, and not try to omit them later. Due to time constraints, I was not able to perform sensitivity analyses to properly estimate to what extent the sampling related problems might have affected the results I obtained. Future research should take these issues into account.

### *4.5. My results in relation to global environmental issues*

#### 4.5.1. Links to biodiversity loss

The State of Acre contains a large fraction of the species richness of the whole Amazon area, as species richness has been estimated to be highest at the south-western parts of the rainforest region (Soares-Filho *et al.*, 2006; Ceballos, Ehrlich and Dirzo, 2017). While people in the State of Acre are not to blame for the world's biodiversity crises, they hold the power to provide immediate and highly valuable conservation solutions due to the state's extensive PA network, which my results showed to be effective, and due to the high expected deforestation rates in the deforestation fronts. It has been shown that a more efficient use of existing agricultural lands could accommodate for the high increase in agricultural production which has been projected for Brazil, reducing the need to clear additional forests and even allowing for their restoration (Strassburg *et al.*, 2014). This means that by developing rural economies, the states in the Brazilian Amazon, including Acre, could help reverse both biodiversity loss and climate change. Biodiversity loss is a global scale problem and solving it cannot be postponed without majorly impoverishing future generations, since no realistic way exists to undo extinctions (especially in mass). All regions of the world need to do their part in addressing both the proximate and ultimate causes behind biodiversity loss, and one important task is to collaboratively end deforestation in the biologically diverse tropical rainforests.

### 4.5.2. Links to climate change

During the period considered in this thesis, the PAs in Acre avoided a significant amount of $CO_2$ emissions, approximately 7.3% of the total $CO_2$ equivalent greenhouse gas emissions Brazil had in the year 2012. Both the size and the effectiveness of the PAs influenced the $CO_2$ emission reduction estimates. These results for Acre are intimately related to Brazil's national commitment to reduce emissions from the loss of forest carbon stocks in accordance with the Paris Agreement, which the government of Brazil ratified in late 2016 (Secom, 2016; UNFCCC, 2018). Brazil's Nationally Determined Contribution (NDC) pledged to reduce greenhouse gas emissions 37% by 2025 compared to 2005 levels, with actions on three fronts: Agriculture (mostly restoring degraded pastures), energy (mostly increasing the share of biofuels), and forestry (mainly preventing deforestation and increasing reforestation) (Federative Republic of Brazil, 2015; Secom, 2016). However, without pro-active measures to control for land cover change, the low-emission IPCC scenarios are associated with the loss of primary forest habitats at very high rates due to a substantial deployment of bioenergy (CBD, 2014). The fact that Brazil seeks to mitigate climate change by increasing the share of biofuels in the Brazilian energy mix, to approximately 18% by 2030 (Federative Republic of Brazil, 2015), might create additional pressures for deforestation and forest degradation in the Amazon in the coming decades, especially if international demand grows simultaneously.

Huge reductions of greenhouse gas emissions are needed in the coming decades globally, and substantial net sequestration may be needed from the AFOLU sector

(IPCC, 2014b). That is why the UN created the REDD+ mechanism, and my results for the PAs of Acre suggest that substantial sequestration can be achieved in tropical areas with this approach. It has been proposed by Nepstad *et al.* (2009), that it might be feasible to bring an end to deforestation in the Brazilian Amazon with a ten year program, resulting in 2-5% reduction in global carbon emissions with a cost of $7 to $18 billion beyond what Brazil's budget outlays were at around 2009. The authors wrote that the REDD+ mechanism might be able to provide this sum as compensation for the reduced emissions, which would also benefit the indigenous groups and traditional forest communities who have not been properly compensated for the carbon bound in their forests. Nepstad *et al.* (2009) estimated that the cost to end deforestation in Acre would total anywhere from 412 to 813 million USD. Given the increasing need to control forest cover loss in Brazil in the coming years, it is important to also consider what other approaches might be needed to achieve the conservation objectives. It has been suggested that command-and-control approaches for implementing legislation will likely not suffice, but instead higher environmental standards for beef, soy and other commodities may need to be imposed by international markets (Soares-Filho *et al.*, 2006). These studies show how climate change mitigation and the conservation of biodiversity are global issues that cannot be solved without cooperation and help from the citizens of other nations. The state of Acre is a pioneer in building such cooperation, with its unique REDD+ program.

## *4.6. My results in relation to Acre's PES-REDD+ program*

My findings identified which PAs have most effectively retained their forest cover, and thus the associated benefits accruing to humans and other species. My findings also indicated which PAs had prevented most carbon emission since the beginning of Acre's PES-REDD+ program. During the time period of my study, the financing provided by the REDD+ program has mostly supported existing programs and projects of traditional forest extractivists, indigenous communities, small-scale farmers and cattle ranchers, which seek to prevent deforestation and forest degradation (Sills *et al.*, 2014). This suggests that the REDD+ program may have affected the effectiveness results I obtained for the period from 2011 to 2016, especially since indigenous communities and extractivists may have received additional support in return for protecting the standing forests, increasing the estimated effectiveness of these areas. Likewise, the support provided for colonist farmers and cattle ranchers may have reduced the estimated baseline deforestation pressures, which may have had the antagonistic effect of reducing the effectiveness estimates for all PA types. If strictly protected areas have not received additional financing thorough the REDD+ program, their effectiveness estimate during my study period might have been reduced. Research on PA effectiveness typically covers large areas, and therefore it is more than likely that some local projects and programs exist that are practically impossible to control for, because researchers might not even be aware of their existence. This might also apply to this thesis, as the jurisdictional REDD+ program may not be the only initiative that seeks to prevent deforestation and forest degradation in Acre.

Challenges still face the innovative REDD+ program of Acre, including finding a way to conjoin Acre's subnational REDD+ program to the national level program, securing continued funding for the program not only through donations but also by utilizing carbon markets, and facilitating the transition of thousands of rural smallholders away from deforestation intensive land-use practices, without dispossessing people of their rights (Sills *et al.*, 2014). Some opposition has existed against the REDD+ program mainly due to the issue of rights (Carbon Trade Watch, 2011; Lang, 2012; The Union of Rural Workers of Xapuri, 2012). The right to use and manage forests is especially important to forest peoples and indigenous communities, who may view the decision-making process of the REDD+ program as top-down and unrepresentative of the various different forest communities (The Union of Rural Workers of Xapuri, 2012). The communities can fear for their sovereignty, and the concept of environmental or carbon "services" or "compensation" can seem to them as promotion of a reductionist and mercantile view of the forests, in contrast to the more holistic view traditionally held by indigenous peoples in Acre (The Union of Rural Workers of Xapuri, 2012).

The situation in Acre therefore seems to reflect the global patterns, as it was reported in the Global Biodiversity Outlook 4 (CBD, 2014) that the Aichi Target 14 – which seeks to ensure the provision of ecosystem services while taking into account the needs of women, indigenous people, local communities and the poor and vulnerable – seems to be moving away from the target, with the global situation getting worse rather than better. Indeed, several NGOs have now proposed principles that could ensure effective and culturally sustainable execution of REDD+ programs, which include respecting and recognizing the rights of indigenous and local communities by "promoting land tenure, self-determination, free, prior and informed consent for any REDD+ projects, and strong social safeguards" (WWF, 2011). REDD+ needs to be scaled up quickly to effectively preserve forests and fight climate change, but this needs to be reconciled with the need to make the decision processes inclusive, and the need to give proper time and respect for traditional decision-making processes (WWF, 2011).

## *4.7. Research integrity and ethical considerations*

Political decisions influencing nature conservation and human rights can be affected by the findings of policy relevant research. When conducting research on topics such as protected area effectiveness in curbing deforestation, as I did, researchers need to be careful and judicious in the way results are presented, especially with regards to how much detail is given. In my view, science is not simply about producing objective information which can be utilized by society according to the ethical principles of a given time and place. Rather, scientists should have an active role in informing how their findings should and should not be interpreted and used. In think that scientists should not refrain from publishing any results, so long as they have been accumulated by ethical means, but they should carefully consider the possibility for the findings to be misinterpreted or misused in ways that could be harmful for people or to the environment. If this likelihood is considered high, results should be presented in a way that restricts this possibility. If the likelihood is considered low, the results should not be restricted, but great care should nonetheless be given to the way results and conclusions are phrased. No results should be left unpublished merely because they may have possible real-life effects. The more scientific evidence is available for society to use, the better the outcome is for everyone. Careful phrasing can buffer against the fears researchers may feel if they do not wish to be claimed responsible for how their research is interpreted and used.

Consider the results of this thesis. If the effectiveness of PAs is linked to their locations and identity, there may be real life consequences on the people responsible for those specific areas, as deforestation is something governments are generally trying to limit. It must therefore be stated, that the findings of this thesis, which represent the findings of only one scientific study, should not be interpreted as an undeniable truth, but rather as an informed study on the effectiveness of the PAs during a specific time and place. Because all PAs had some effectiveness and no PA had a negative effect, I am confident that these findings, even if presented in the detail and manner which I chose, should not result in any repercussions which might be ethically problematic. As the majority of PAs in this thesis were indigenous areas, the findings of this thesis have links not only to nature conservation but also to human rights and cultural sustainability. However, indigenous areas are not established on the condition of avoiding deforestation, so even if indigenous areas would have low effectiveness, it would likely not lead to any consequences to the indigenous communities.

# 5. Conclusions

My results carry with them a positive message as they show that the PAs of Acre have been able to conserve the cherished tropical rainforests to a substantial degree, even in areas of high pressure. Despite the methodological limitations, I consider the general patterns of my results to be reliable and to fill the important gaps in knowledge I set out to fill – mainly showing that PAs in Acre have had varying effectiveness after 2010, and that the PA types did not differ significantly from each other in terms of effectiveness. Second, my results revealed that a large amount of deforestation was avoided by the PAs in Acre, meaning that the PAs have managed to conserve both the habitats of species and carbon bound in the forests.

My study provided a new and independent estimation of effectiveness for a subset of Brazilian PAs, with the latest data and a new method, and the results strengthen the case for the effectiveness of all main PA categories. In addition to corroborating previous findings for the Brazilian Amazon, my study is the first to present effectiveness estimations for individual PAs in more detail, and, as far as I am aware, the only matching based estimation of avoided forest carbon in Acre after 2010. Further, my study provides only the second case example of the new state-of-the-art matching method for estimating PA effectiveness, with the previous study having focused on the PAs of Madagascar (Eklund *et al.*, 2016). The method used in this thesis is easily scalable and has the potential to improve estimates of PA effectiveness if utilized in future matching studies. My study also had a specific focus on the deforestation fronts, which will be increasingly relevant areas for mitigating biodiversity loss and climate change in the coming decades. More empirical research is needed worldwide on these areas of high threat.

The results of this thesis should not be interpreted as evidence that since PAs work, no other conservation efforts are needed. PAs are limited to only avoid deforestation within their designated boundaries, while two thirds of projected forest loss in the Amazon is expected to happen outside of their reach (Soares-Filho *et al.*, 2006). Solutions to conserve rainforests outside the PA boundaries are therefore also very much required to conserve species, forest carbon and the multitude of ecosystem services provided by the Amazon rainforest.

Action to prevent deforestation in the tropics will be a key factor as humanity seeks to mitigate global climate change and the ongoing loss of biodiversity. Designated protected areas have successfully conserved rainforests, but protection alone does not solve the underlying economic and social structures which incentivize people to use ecosystems unsustainably. While these deeper solutions are discussed, researchers should continue to provide ever improving knowledge about the efforts societies take to conserve nature. This will help to ensure, that the world's forests prevail and thrive while humanity works towards more sustainable practices and equilibrium with the planetary boundaries.

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# References


Andam, K. S. *et al.* (2008) 'Measuring the effectiveness of protected area networks in reducing deforestation', *Proceedings of the National Academy of Sciences*, 105(42), pp. 16089–16094. doi: 10.1073/pnas.0800437105.

ARPA (2014) *ICMBio constrói modelo de sustentabilidade*. Available at: http://arpa.mma.gov.br/icmbio-constroi-modelo-de-sustentabilidade/ (Accessed: 6 April 2018).

BBC (2017) *Brazil emerges from recession as GDP grows 1%*. Available at: http://www.bbc.com/news/world-latin-america-40120364 (Accessed: 28 February 2018).

Bernard, E., Penna, L. A. O. and Araújo, E. (2014) 'Downgrading, Downsizing, Degazettement, and Reclassification of Protected Areas in Brazil', *Conservation Biology*, 28(4), pp. 939–950. doi: 10.1111/cobi.12298.

Bowker, J. N. *et al.* (2017) 'Effectiveness of Africa's tropical protected areas for maintaining forest cover', *Conservation Biology*, 31(3), pp. 559–569. doi: 10.1111/cobi.12851.

Bruner, A. G. *et al.* (2001) 'Effectiveness of Parks in Protecting Tropical Biodiversity', *Science*, 291(5501), pp. 125–128. doi: 10.1126/science.291.5501.125.

Carbon Trade Watch (2011) *Acre Letter - against REDD and the commodification of nature*. Available at: http://www.carbontradewatch.org/articles/acre-letter-against-redd-and-the-commodification-of-nature.html (Accessed: 8 April 2018).

Carranza, T. *et al.* (2014) 'Protected area effectiveness in reducing conversion in a rapidly vanishing ecosystem: The Brazilian Cerrado', *Conservation Letters*, 7(3), pp. 216–223. doi: 10.1111/conl.12049.

CBD (2014) *Global Biodiversity Outlook 4*. Montréal: Secretariat of the Convention on Biological Diversity.

Ceballos, G. *et al.* (2015) 'Accelerated modern human–induced species losses: Entering the sixth mass extinction', *Science Advances*, 1(5). doi: 10.1126/sciadv.1400253.

Ceballos, G., Ehrlich, P. R. and Dirzo, R. (2017) 'Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines', *Proceedings of the National Academy of Sciences*, p. 201704949. doi: 10.1073/pnas.1704949114.

Clements, G. and GRASS Development Team (2018) *GRASS GIS manual: r.series*. Available at: https://grass.osgeo.org/grass74/manuals/r.series.html (Accessed: 16 April 2018).

Climate Focus (2013) *Acre, Brazil: Subnational Leader in REDD+*.

CNUC (2018) *Reserva Extrativista Chico Mendes*, *Ministry of the Environment, Brazil. National Register of Conservation Units (CNUC).* Available at: http://sistemas.mma.gov.br/cnuc/index.php?ido=relatorioparametrizado.exibeRelatorio&relatorioPadrao=true&idUc=222 (Accessed: 4 April 2018).

Darwin, C. (1859) *On the origin of species by means of natural selection, or preservation of favoured races in the struggle for life*. London : John Murray, 1859.

DeFries, R. *et al.* (2005) 'Increasing Isolation of Protected Areas in Tropical Forests over the past Twenty Years', *Ecological Applications*, 15(1), pp. 19–26. doi: 10.1890/03-5258.

DiMiceli, C. M. *et al.* (2011) *Annual Global Automated MODIS Vegetation Continuous Fields (MOD44B) at 250 m Spatial Resolution for Data Year 2010, Collection 5, Percent Tree Cover, University of Maryland, College Park, MD, USA.* Available at: http://ftp.glcf.umd.edu/data/vcf/ (Accessed: 14 March 2018).

Duchelle, A. E. *et al.* (2014) 'Chapter 2: Acre's State System of Incentives for Environmental Services (SISA), Brazil', in Sills, E. O. (ed.) *REDD+ on the ground: A case book of subnational initiatives across the globe*. Bogor, Indonesia, pp. 33–50.

Dudley, N. (2008) *Guidelines for Applying Protected Area Management Categories*. Edited by N. Dudley. Gland: IUCN, Gland, Switzerland. doi: 10.2305/IUCN.CH.2008.PAPS.2.en.

Eklund, J. *et al.* (2016) 'Contrasting spatial and temporal trends of protected area effectiveness in mitigating deforestation in Madagascar', *Biological Conservation*. Elsevier B.V., 203, pp. 290–297. doi: 10.1016/j.biocon.2016.09.033.

ESA (2010) 'GlobCover 2009, Global Land Cover Map V2.3.' Available at: http://due.esrin.esa.int/page_globcover.php (Accessed: 9 November 2017).

FAO (2016) *State of the World's Forests 2016. Forests and agriculture: land-use challenges and opportunities.* Rome.

FAO/IIASA (2010) *Global Agro-ecological Zones (GAEZ v3.0), Terrain Resources, FAO, Rome, Italy and IIASA, Laxenburg, Austria.* Available at: http://gaez.fao.org/Main.html?ticket=ST-61503-MuLj2AhHK9LYI6EqpwiH-cas# (Accessed: 14 March 2018).

FAO FRA (2012) *Forest Resources Assessment 2015: Terms and Definitions*, *FAO report*. Available at: http://www.fao.org/docrep/017/ap862e/ap862e00.pdf.

Federative Republic of Brazil (2012) *Federal Law 12.727, 17 October 2012*. Available at: http://www.planalto.gov.br/ccivil_03/_Ato2011-2014/2012/Lei/L12727.htm (Accessed: 8 April 2018).

Federative Republic of Brazil (2015) *Intended Nationally Determined Contribution: Towards achieving the objective of the United Nations Framework Convention on Climate Change*. Available at: http://www4.unfccc.int/Submissions/INDC/Published Documents/Brazil/1/BRAZIL iNDC english FINAL.pdf.

Gaveau, D. L. A. *et al.* (2009) 'Evaluating whether protected areas reduce tropical deforestation in Sumatra', *Journal of Biogeography*, 36(11), pp. 2165–2175. doi: 10.1111/j.1365-2699.2009.02147.x.

Geist, H. J. and Lambin, E. F. (2002) 'Proximate Causes and Underlying Driving Forces of

Tropical Deforestation', *BioScience*, 52(2), p. 143. doi: 10.1641/0006-3568(2002)052[0143:PCAUDF]2.0.CO;2.

Government of Acre (2017) *Acre em Números 2017*.

IBAMA (2006) *Plano de Manejo Reserva Extrativista Chico Mendes*, *GOVERNO FEDERAL*. Available at: http://www.icmbio.gov.br/portal/biodiversidade/unidades-de-conservacao/biomas-brasileiros/amazonia/unidades-de-conservacao-amazonia/2016-resex-chico-mendes.html.

INPE (2013) 'Projeto PRODES Metodologia para o Cálculo da Taxa Anual de Desmatamento na Amazônia Legal', pp. 1–37.

INPE (2018) *Instituto Nacional de Pesquisas Espaciais (2018) PRODES: Assessment of deforestation in Brazilian Amazonia.* Available at: http://www.obt.inpe.br/prodes/ (Accessed: 28 February 2018).

IPCC (2014a) *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Edited by J. C. M. Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel. Cambridge: Cambridge University Press.

IPCC (2014b) *Climate Change 2014: Synthesis Report*. doi: 10.1017/CBO9781107415324.

IUCN (2017) *IUCN Red List Summary Statistics*, *The IUCN Red List of Threatened Species*. Available at: http://www.iucnredlist.org/about/summary-statistics (Accessed: 27 February 2017).

Jenkins, C. N. *et al.* (2015) 'Patterns of vertebrate diversity and protection in Brazil', *PLoS ONE*, 10(12), pp. 1–13. doi: 10.1371/journal.pone.0145064.

Joppa, L. N. and Pfaff, A. (2009) 'High and far: Biases in the location of protected areas', *PLoS ONE*, 4(12), pp. 1–6. doi: 10.1371/journal.pone.0008273.

Joppa, L. N. and Pfaff, A. (2011) 'Global protected area impacts', *Proceedings of the Royal Society B: Biological Sciences*, 278(1712), pp. 1633–1638. doi: 10.1098/rspb.2010.1713.

Joppa, L. and Pfaff, A. (2010) 'Reassessing the forest impacts of protection: The challenge of nonrandom location and a corrective method', *Annals of the New York Academy of Sciences*, 1185, pp. 135–149. doi: 10.1111/j.1749-6632.2009.05162.x.

Karger, D. N. *et al.* (2017) 'Climatologies at high resolution for the earth's land surface areas', *Scientific Data*. The Author(s), 4, p. 170122. Available at: 10.1038/sdata.2017.122.

KfW Development Bank (2017) *REDD+ in the State of Acre, Brazil: Rewarding a pioneer in forest protection and sustainable livelihood development.*

Krausmann, F. *et al.* (2013) 'Global human appropriation of net primary production doubled in the 20th century', *Proceedings of the National Academy of Sciences*, 110(25), pp. 10324–10329. doi: 10.1073/pnas.1211349110.

Lang, C. (2012) *Problems with REDD and payments for environment services in Acre, Brazil*. Available at: www.redd-monitor.org/2012/11/01/problems-with-redd-and-payments-for-environment-services-in-acre-brazil/ (Accessed: 8 April 2018).

Lenton, T. M. *et al.* (2008) 'Tipping elements in the Earth's climate system.', *Proceedings of the National Academy of Sciences of the United States of America*. National Academy of Sciences, 105(6), pp. 1786–93. doi: 10.1073/pnas.0705414105.

Leverington, F. *et al.* (2010) 'A Global Analysis of Protected Area Management Effectiveness', *Environmental Management*, 46(5), pp. 685–698. doi: 10.1007/s00267-010-9564-5.

Levis, C. *et al.* (2017) 'Persistent effects of pre-Columbian plant domestication on Amazonian forest composition', *Science*, 355(6328), pp. 925–931. doi: 10.1126/science.aal0157.

Lewis, S. L., Edwards, D. P. and Galbraith, D. (2015) 'Increasing human dominance of tropical forests', *Science*, 349(6250), pp. 827–832. doi: 10.1126/science.aaa9932.

Lui, G. V. and Coomes, D. A. (2016) 'Tropical nature reserves are losing their buffer zones, but leakage is not to blame'. Academic Press Inc., 147, pp. 580–589. doi: 10.1016/j.envres.2015.11.008.

MacDicken, K. *et al.* (2015) *FAO Global Forest Resources Assessment 2015, 2nd ed.*

Mace, G. *et al.* (2005) *Millennium Ecosystem Assessment Chapter 4, Biodiversity*.

Marcel, Y. (2013) *'Filho de seringueiro hoje já nasce querendo criar gado', diz extrativista*, *Globo G1 Acre*. Available at: http://g1.globo.com/ac/acre/noticia/2013/12/filho-de-seringueiro-hoje-ja-nasce-querendo-criar-gado-diz-extrativista.html (Accessed: 5 April 2018).

MEA (2005) *Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Biodiversity Synthesis.* World Resources Institute, Washington, DC. doi: 10.1057/9780230625600.

Myers, N. (1993) 'Tropical Forests: The Main Deforestation Fronts', *Environmental Conservation*, 20(1), pp. 9–16. doi: 10.1017/S0376892900037176.

Nelson, A. (2008) 'Travel time to major cities: A global map of Accessibility. Global Environment Monitoring Unit - Joint Research Centre of the European Commission, Ispra Italy. Available at http://forobs.jrc.ec.europa.eu/products/gam'.

Nepstad, D. *et al.* (2009) 'The End of Deforestation in the Brazilian Amazon', *Science*, 326(5958), pp. 1350–1351. doi: 10.1126/science.1182108.

Nolte, C. *et al.* (2013) 'Governance regime and location influence avoided deforestation success of protected areas in the Brazilian Amazon', *Proceedings of the National Academy of Sciences*, 110(13), pp. 4956–4961. doi: 10.1073/pnas.1214786110.

Oksanen, J. *et al.* (2018) 'Vegan: Community Ecology Package. R package version 2.4-6.' Available at: http://cran.r-project.org/%0Apackage=vegan.

Otavio, C. (2013) *Na terra de Chico Mendes, borracha perde espaço para pecuária e motosserras*, *O Globo*. Available at: https://oglobo.globo.com/brasil/na-terra-de-chico-mendes-borracha-perde-espaco-para-pecuaria-motosserras-11005552 (Accessed: 6 April 2018).

Pack, S. M. *et al.* (2016) 'Protected Area Downgrading, Downsizing, and Degazettement (PADDD) in the Amazon'. Elsevier Ltd, 197, pp. 32–39. doi: 10.1016/j.biocon.2016.02.004.

Pebesma, E. J. and Bivand, R. S. (2005) 'Classes and methods for spatial data in R'. R News 5 (2).

Redford, K. H. (1992) 'The empty forest', *BioScience*, 42(6), pp. 412–422. doi: 10.2307/1311860.

Rockström, J. *et al.* (2009) 'Planetary Boundaries: Exploring the Safe Operating Space for Humanity', *Ecology and Society*, 14(2), pp. 472–475. doi: 10.1038/461472a.

Roriz, P. A. C., Yanai, A. M. and Fearnside, P. M. (2017) 'Deforestation and Carbon Loss in Southwest Amazonia: Impact of Brazil's Revised Forest Code', *Environmental Management*. Springer US, 60(3), pp. 367–382. doi: 10.1007/s00267-017-0879-3.

Saatchi, S. *et al.* (2007) 'Distribution of aboveground live biomass in the Amazon basin', *Global Change Biology*, 13(4), pp. 816–837. doi: 10.1111/j.1365-2486.2007.01323.x.

Sabogal, D. (2015) *Community-based forest monitoring: experiences from the Chico Mendes Extractive Reserve.* Oxford.

Schleicher, J. *et al.* (2017) 'Conservation performance of different conservation governance regimes in the Peruvian Amazon', *Scientific Reports*, 7(1), pp. 1–10. doi: 10.1038/s41598-017-10736-w.

Schneider, M. and Peres, C. A. (2015) 'Environmental Costs of Government-Sponsored Agrarian Settlements in Brazilian Amazonia', *PLOS ONE*. Edited by R. Zang, 10(8), p. e0134016. doi: 10.1371/journal.pone.0134016.

Secom (2016) *Brazil at COP22 Fact sheet*.

Sills, E. O. *et al.* (2014) *REDD+ on the ground: A case book of subnational initiatives across the globe*. Bogor, Indonesia.

Soares-Filho, B. *et al.* (2010) 'Role of Brazilian Amazon protected areas in climate change mitigation', *Proceedings of the National Academy of Sciences*, 107(24), pp. 10821–10826. doi: 10.1073/pnas.0913048107.

Soares-Filho, B. S. *et al.* (2006) 'Modelling conservation in the Amazon basin', *Nature*, 440(7083), pp. 520–523. doi: 10.1038/nature04389.

Steffen, W. *et al.* (2015) 'Planetary boundaries: Guiding human development on a changing planet', *Science*, 347(6223), pp. 1259855–1259855. doi: 10.1126/science.1259855.

Strassburg, B. B. N. *et al.* (2014) 'When enough should be enough: Improving the use of current agricultural lands could meet production demands and spare natural habitats in Brazil', *Global Environmental Change*, 28(1), pp. 84–97. doi: 10.1016/j.gloenvcha.2014.06.001.

The Union of Rural Workers of Xapuri (2012) 'How REDD and Environmental Services Threaten the Lives of Forest People in Acre', pp. 1–4.

The World Bank (2018) *CO2 emissions (kt)*, *World Development Indicators*. Available at: https://data.worldbank.org/indicator/EN.ATM.CO2E.KT?locations=BR.

Tollefson, J. (2011) 'Brazil revisits forest code', *Nature*. Nature Publishing Group, 476(7360), pp. 259–260. doi: 10.1038/476259a.

Tollefson, J. (2012) 'Brazil set to cut forest protection', *Nature*, 485(7396), p. 19. doi: 10.1038/485019a.

Tollefson, J. (2016) 'Deforestation spikes in Brazilian Amazon', *Nature*, 540(7632), pp. 182–182. doi: 10.1038/nature.2016.21083.

Trimaille, E. (2018) *QuickOSM plugin*, *QGIS Python Plugins Repository*. Available at: https://plugins.qgis.org/plugins/QuickOSM/ (Accessed: 16 April 2018).

UN (2015) *Transforming our world: The 2030 agenda for sustainable development*, *A/RES/70/1*. doi: 10.1007/s13398-014-0173-7.2.

UN-REDD (2015) *Fact Sheet: About REDD+*. Available at: http://www.unredd.net/documents/redd-papers-and-publications-90/un-redd-publications-1191/fact-sheets/15279-fact-sheet-about-redd.html.

UNEP-WCMC (2017) *World Database on Protected Areas User Manual 1.5*. Cambridge, UK. Available at: http://wcmc.io/WDPA_Manual.

UNEP-WCMC and IUCN (2016) *Protected Planet Report 2016*, *UNEP-WCMC and IUCN*. Cambridge UK and Gland, Switzerland: UNEP-WCMC and IUCN. doi: 10.1017/S0954102007000077.

UNEP-WCMC and IUCN (2017) 'Protected Area Profile for Brazil from the World Database of Protected Areas, November 2017. Cambridge, UK: UNEP-WCMC and IUCN. Available at: www.protectedplanet.net'.

UNFCCC (2015) *The Paris Agreement*. Available at: http://unfccc.int/paris_agreement/items/9485.php (Accessed: 1 March 2018).

UNFCCC (2018) *Paris Agreement - Status of Ratification*. Available at: http://unfccc.int/paris_agreement/items/9444.php (Accessed: 8 April 2018).

Unidades de Conservação no Brazil (2018) *Área de Relevante Interesse Ecológico*. Available at: https://uc.socioambiental.org/uso-sustentável/área-de-relevante-interesse-ecológico (Accessed: 14 April 2018).

Watts, J. (2016) *Dilma Rousseff impeachment: what you need to know – the Guardian briefing*, *The Guardian*. Available at: https://www.theguardian.com/news/2016/aug/31/dilma-rousseff-impeachment-brazil-what-you-need-to-know (Accessed: 28 February 2018).

Weiss, D. J. *et al.* (2018) 'A global map of travel time to cities to assess inequalities in accessibility in 2015', *Nature*. Nature Publishing Group, 553(7688), pp. 333–336. doi: 10.1038/nature25181.

World Bank (2018) *Brazil*. Available at: https://data.worldbank.org/country/brazil (Accessed: 3 November 2017).

WWF (2011) *Forests and Climate: REDD+ at a Crossroads*, *Living Forests Report, Chapter 3*. Available at: http://wwf.panda.org/about_our_earth/deforestation/forest_publications_news_and_reports/living_forests_report/.

WWF (2014) *The Growth of Soy: Impacts and Solutions*, *WWF Report*. WWF International, Gland, Switzerland. Available at: http://issuu.com/wwfsoyreport/docs/wwf_soy_report_final_jan_19/1?e=10667775/6569194.

WWF (2015a) *Deforestation Fronts - Amazon*, *ArcGIS Online*. Available at: https://www.arcgis.com/home/item.html?id=b861d7e1446e429ea4cc925cf05abdfa (Accessed: 16 April 2018).

WWF (2015b) *Living Forests Report, Chapter 5: Saving Forests at Risk*.

WWF (2016) *Living Planet Report 2016. Risk and resilience in a new era.* WWF International, Gland, Switzerland.

WWF-Brazil (2015) *Guia Informativo da Gestão Participativa na Reserva Extrativista Chico Mendes*.

Yu, H. (2002) 'Rmpi: Parallel Statistical Computing in R', *R News*, 2(2), pp. 10–14. Available at: http://cran.r-project.org/doc/Rnews/Rnews_2002-2.pdf (Accessed: 18 March 2018).

# Appendix 1

Table S1. Sample sizes of each PA, including the percent of forested areas sampled after samples in non-forested areas had been omitted. Original sample size indicates 10% of all pixels in each area and final sample size has had points in non-forested areas omitted. PA 0 indicates the non-protected area.

| PA | Original sample size | Final sample size | Difference between original and final sample size (in points) | Difference between original and final sample size (in %) | Forested pixels | Non-forested pixels | Percent of forested area sampled |
|---|---|---|---|---|---|---|---|
| 0 | 176630 | 142534 | 34096 | 19.30 | 1436099 | 330205 | 9.93 |
| 1 | 1529 | 1485 | 44 | 2.88 | 14886 | 401 | 9.98 |
| 2 | 633 | 621 | 12 | 1.90 | 6249 | 84 | 9.94 |
| 3 | 547 | 547 | 0 | 0.00 | 5470 | 0 | 10.00 |
| 4 | 229 | 227 | 2 | 0.87 | 2233 | 58 | 10.17 |
| 5 | 392 | 295 | 97 | 24.74 | 3669 | 254 | 8.04 |
| 6 | 1687 | 1525 | 162 | 9.60 | 16419 | 452 | 9.29 |
| 7 | 3349 | 3341 | 8 | 0.24 | 33407 | 78 | 10.00 |
| 8 | 1146 | 1118 | 28 | 2.44 | 11070 | 393 | 10.10 |
| 9 | 489 | 471 | 18 | 3.68 | 4698 | 192 | 10.03 |
| 10 | 1554 | 1519 | 35 | 2.25 | 15224 | 315 | 9.98 |
| 11 | 166 | 158 | 8 | 4.82 | 1587 | 71 | 9.96 |
| 12 | 4969 | 4850 | 119 | 2.39 | 48356 | 1337 | 10.03 |
| 13 | 2702 | 2621 | 81 | 3.00 | 26165 | 850 | 10.02 |
| 14 | 1489 | 1411 | 78 | 5.24 | 14634 | 260 | 9.64 |
| 15 | 863 | 842 | 21 | 2.43 | 8397 | 237 | 10.03 |
| 16 | 14337 | 14236 | 101 | 0.70 | 142378 | 995 | 10.00 |
| 17 | 10195 | 9768 | 427 | 4.19 | 99212 | 2735 | 9.85 |
| 18 | 485 | 485 | 0 | 0.00 | 4846 | 0 | 10.01 |
| 19 | 17726 | 17048 | 678 | 3.82 | 170813 | 6447 | 9.98 |
| 20 | 13170 | 12702 | 468 | 3.55 | 128419 | 3282 | 9.89 |
| 21 | 415 | 411 | 4 | 0.96 | 4119 | 30 | 9.98 |
| 22 | 463 | 438 | 25 | 5.40 | 4467 | 167 | 9.81 |
| 23 | 2858 | 2793 | 65 | 2.27 | 27944 | 640 | 9.99 |
| 24 | 402 | 401 | 1 | 0.25 | 4006 | 16 | 10.01 |
| 25 | 4393 | 4262 | 131 | 2.98 | 42908 | 1023 | 9.93 |
| 26 | 1654 | 1654 | 0 | 0.00 | 16542 | 0 | 10.00 |
| 27 | 15680 | 14481 | 1199 | 7.65 | 152356 | 4446 | 9.50 |
| 28 | 4936 | 4526 | 410 | 8.31 | 48023 | 1338 | 9.42 |
| 29 | 3539 | 3538 | 1 | 0.03 | 35368 | 18 | 10.00 |
| 30 | 6143 | 6081 | 62 | 1.01 | 60932 | 495 | 9.98 |
| 31 | 2442 | 2386 | 56 | 2.29 | 23765 | 657 | 10.04 |
| 32 | 1504 | 1263 | 241 | 16.02 | 14615 | 422 | 8.64 |
| 33 | 4403 | 4051 | 352 | 7.99 | 42402 | 1630 | 9.55 |
| 34 | 1629 | 1338 | 291 | 17.86 | 15593 | 699 | 8.58 |
| 35 | 605 | 589 | 16 | 2.64 | 5916 | 137 | 9.96 |
| 36 | 5924 | 5625 | 299 | 5.05 | 57070 | 2166 | 9.86 |
| 37 | 438 | 428 | 10 | 2.28 | 4252 | 132 | 10.07 |

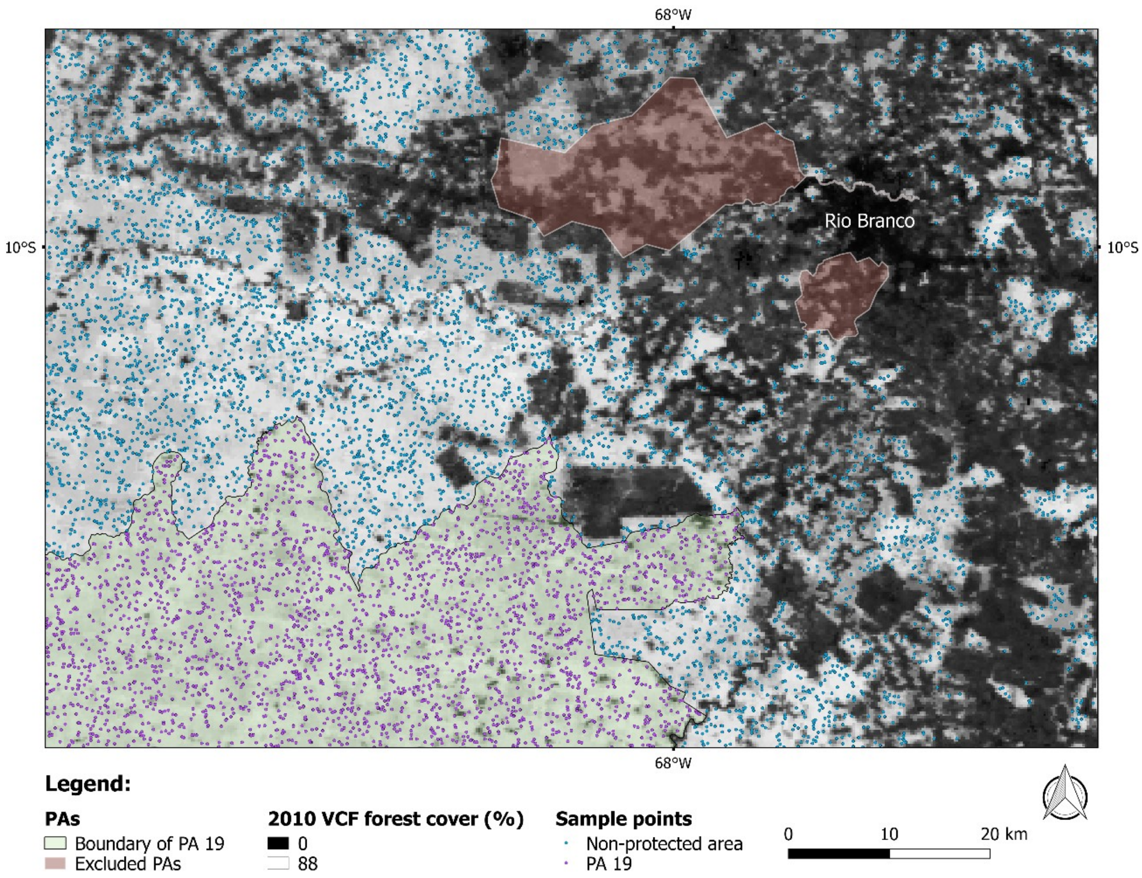


Figure S1. Samples of the forested areas visualized. Approximately 10% of the forested areas (VCF >=45%) in each PA were sampled. Excluded PAs were not sampled.

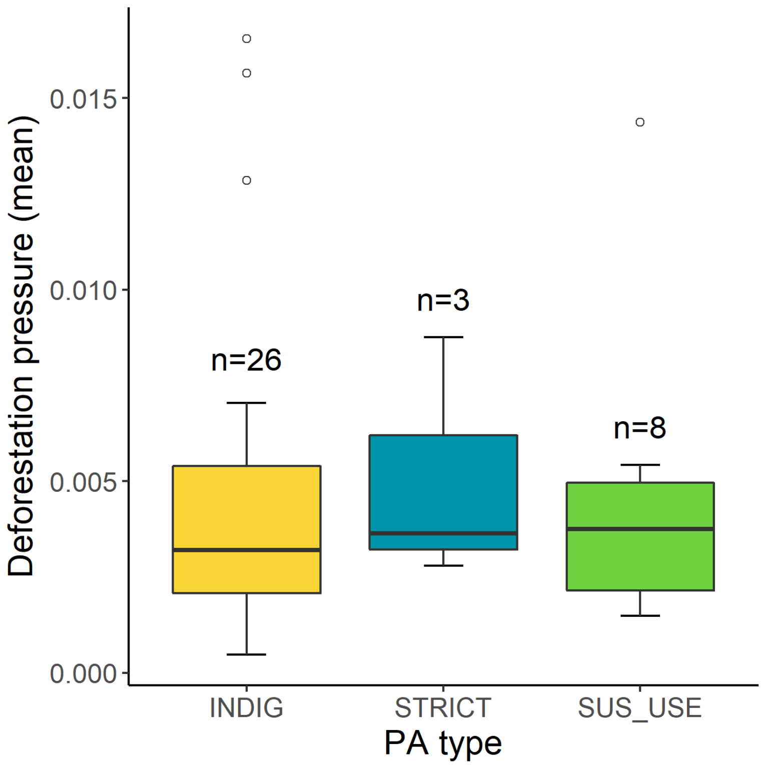


Figure S2. I found no significant differences between the protected area types in the pressures they faced (based on mean BL_multidim).

Table S2. Additional information on the PAs included in this thesis. FLONA = National forest; RESEX = extractive reserve; PARNA = national park; ESEC = Ecological Station; ICMBio = Instituto Chico Mendes de Conservação da Biodiversidade (Chico Mendes Institute for Biodiversity Conservation); SEMA = Secretaria de Estado de Meio Ambiente do Acre (Secretary of State for the Environment of Acre)

| PA | PA name (in Portuguese) | Designated | Type | IUCN category | Management authority |
|---|---|---|---|---|---|
| 1 | Jaminaua/Envira | 2003 | INDIG | - | Indigenous |
| 2 | Campinas/Katukina | 1999 | INDIG | - | Indigenous |
| 3 | Jaminawa Arara do Rio Bagé | 1999 | INDIG | - | Indigenous |
| 4 | Igarapé do Caucho | 1998 | INDIG | - | Indigenous |
| 5 | Arara do Rio Amônia | 2009 | INDIG | - | Indigenous |
| 6 | Kaxinawá do Rio Jordão | 1996 | INDIG | - | Indigenous |
| 7 | FLONA do Macauã | 1988 | SUS. USE | VI | ICMBio |
| 8 | Kaxinawá da Praia do Carapanã | 2002 | INDIG | - | Indigenous |
| 9 | Kaxinawá Nova Olinda | 2002 | INDIG | - | Indigenous |
| 10 | Kulina do Rio Envira | 1996 | INDIG | - | Indigenous |
| 11 | Kaxinawá do Baixo Rio Jordão | 2002 | INDIG | - | Indigenous |
| 12 | Alto Rio Purus | 2002 | INDIG | - | Indigenous |
| 13 | Alto Tarauacá | 2009 | INDIG | - | Indigenous |
| 14 | ESEC do Rio Acre | 1981 | STRICT | Ia | ICMBio |
| 15 | Kulina Igarapé do Pau | 2001 | INDIG | - | Indigenous |
| 16 | RESEX do Cazumbá-Iracema | 2002 | SUS. USE | VI | ICMBio |
| 17 | RESEX do Alto Juruá | 1990 | SUS. USE | VI | ICMBio |
| 18 | Jaminawa do Igarapé Preto | 1999 | INDIG | - | Indigenous |
| 19 | RESEX Chico Mendes | 1990 | SUS. USE | VI | ICMBio |
| 20 | Parque Estadual Chandless | 2004 | STRICT | II | SEMA |
| 21 | Kampa do Igarapé Primavera | 2002 | INDIG | - | Indigenous |
| 22 | Poyanawa | 2002 | INDIG | - | Indigenous |
| 23 | RESEX do Alto Tarauacá | 2000 | SUS. USE | VI | ICMBio |
| 24 | FLONA de São Francisco | 2001 | SUS. USE | VI | ICMBio |
| 25 | FLONA de Santa Rosa do Purus | 2001 | SUS. USE | VI | ICMBio |
| 26 | Arara do Igarapé Humaitá | 2006 | INDIG | - | Indigenous |
| 27 | PARNA da Serra do Divisor | 1989 | STRICT | II | ICMBio |
| 28 | Riozinho do Alto Envira | 2007 | INDIG | - | Indigenous |
| 29 | Rio Gregório | 2007* | INDIG | - | Indigenous |
| 30 | RESEX Riozinho da Liberdade | 2005 | SUS. USE | VI | ICMBio |
| 31 | Kaxinawá do Rio Humaitá | 1996 | INDIG | - | Indigenous |
| 32 | Cabeceira do Rio Acre | 1999 | INDIG | - | Indigenous |
| 33 | Kampa e Isolados do Rio Envira | 1999 | INDIG | - | Indigenous |
| 34 | Kampa do Rio Amonea | 1995 | INDIG | - | Indigenous |
| 35 | Nukini | 1997 | INDIG | - | Indigenous |
| 36 | Mamoadate | 1999 | INDIG | - | Indigenous |
| 37 | Katukina/Kaxinawá | 1999 | INDIG | - | Indigenous |