Three Essays on the Economics of Conservation Policy

by

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Defense Date: March 31, 2025

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Dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy in the University Program in Environmental Policy in The Graduate School of

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ABSTRACT

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Abstract

The challenges of sustainable development require land use which supports healthy ecosystems. As two sides of the same process, restoring degraded lands and protecting lands play a dual role in the path towards reaching those goals. In three essays, I use methods of economic and political analysis to understand the policy challenges faced by conservation practice, including restoration planning, protected area governance, and protected area evaluation.

Priorities of where to invest in ecosystem restoration are of major policy interest. While shifting ecological and land use dynamics have been shown to affect restoration plans, the impact of changing conditions on the economic value of restoration sites remains less explored. Climate change can be expected to affect both the benefits and costs of both a particular restoration project as well as large-scale restoration planning decisions, through changing crop yields, shifting vegetation, and urbanization, among other factors, potentially leading to abandonment or forgone opportunity. Here, I first present a framework for how economic value calculations for a restoration project are affected by projected future changes. I then apply the model to an empirical context of ecological interest: forest replanting in Brazil, compiling a dataset which includes spatial variation in opportunity costs, habitat protection benefits, biodiversity intactness, and projected changes in each of those variables as of 2050. I focus on the implications of two specific drivers of change: changing crop yields predicted by economic projections of climate models, and changing habitat driven by shifts in forest cover. I find that targeted restoration can be justified in a majority of areas across the country at present by habitat value alone, yet that conditions in 2050 would yield far more restoration benefits than present conditions would suggest, implying

the need for greater present-day investment. I also find sizeable geographic changes to restoration priorities at multiple scales, including a shift towards central regions of the country, away from at-risk forests in the Amazon and improving agricultural regions of the south. The extent of changing priorities differs with the extent of restoration, with a greater degree of changes in priorities at low levels of restoration investment.

The expansion of Marine Protected Areas (MPAs) is a major conservation priority, providing vital services for local populations and ecosystems alike. However, MPA effectiveness varies widely, often depending on governance—the rules and practices that regulate resource use. In Eastern Indonesia, the State University of Papua (UNIPA) and the World Wildlife Fund have been collecting data from 230 settlements across 12 MPAs to investigate patterns of governance and understand the factors behind MPA success. This study applies Ostrom's Design Principles for common pool resource governance, aggregating survey data to measure alignment with governance principles and conducting robustness checks to validate the framework. I use cluster analysis (Latent Profile Analysis, LPA) to identify common governance patterns among settlements. The analysis reveals substantial variation in governance across and within MPAs. While some MPAs demonstrate uniformly high levels of governance alignment, others show significant heterogeneity, potentially impacting their effectiveness. Cluster analysis identified four general categories of governance among settlements, highlighting areas of both strong and weak alignment with principles. Notably, the role of higher-level governance exhibited a distinct distribution among good governance principles, suggesting external management influences. These findings demonstrate the variation within single MPAs, emphasizing the need for context-specific approaches to governance. The framework provides a template for evaluating

governance practices in MPAs, both by operationalizing Ostrom's principles and using LPA to identify patterns of governance. The analysis also yields context-specific insights on the nature of governance in the MPAs of Eastern Indonesia and its alignment with principles of good governance.

Estimating the impact of protected areas (PAs) on ecological outcomes is central to conservation goals, yet non-random site selection and effects which vary across space and time present challenges. I construct a dataset of PA creation and land cover change across 1km grid cells spanning over thirty years along the coast of Brazil, focusing on a common governance type, strictly protected PAs, and ecosystem, mangrove forests. I use a difference-in-differences estimator with staggered implementation, enabled by high temporal resolution data on mangrove forest cover. After matching, yearly estimates suggest a decrease in mangrove cover prior to PA creation followed by an increase following creation, a pattern which eventually attenuates, yet is more present in federal and strict PAs. Given the possibility of selection effects rather than anticipation, to construct a balanced sample I aggregate outcomes and treatment to four-year intervals based on federal planning cycles. Across all aggregated specifications, I do not find a significant effect of protection status. In tandem, the annual and four-year results suggest the possibility of time-specific effects across the study period, but at a minimum, align with the literature on limited PA effects in the aggregate.

Dedication

To my family, including Camille Reale, Janice Reale, Peter Hatem, and Olivia Reale-Hatem, for their inspirational commitment to education and public service.

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1. Introduction: The past, present, and future of conservation policy

While there are many paths to reversing the worldwide decline of nature, all require significant changes in land use (Díaz et al. 2019). To ensure that land use patterns support ecological flourishing, conservation policy plays a central role in addressing the challenges of sustainable development. In three essays, I study two central priorities in conservation policy: protected areas (PAs) and ecosystem restoration.

Methodologically, understanding what determines the success of conservation programs requires diverse tools. While each of my chapters uses distinct methodologies, they are thematically linked by the goal of understanding how conservation programs can better reach their goals in practice. I draw primarily on the methods of economics and political science, yet use an approach informed by the nature of conservation science as an interdisciplinary and practice-driven field (Kareiva and Marvier 2012).

My first chapter integrates an array of ecological research and data on restoration with economic data on its costs and benefits to answer a basic question: how will economic priorities of restoration projects be affected by shifting conditions under climate change? Recent papers on restoration priorities have been situated in the broader scientific literature, with both natural science and social science methods playing a role (e.g., Jakovac et al. 2019). I explore the implications of two main projections of change across Brazil: shifting agricultural opportunity costs driven by changing crop yields and forest cover change under climatic shifts. I aim to provide both a conceptual economic framework that can be applied to other restoration contexts, as well as practical insights on restoration priorities in the forests of Brazil, which are themselves of disproportionate ecological interest. The paper is

similarly positioned in the conservation science literature, a field grounded in the explicit connections between human and natural systems (Kareiva and Marvier 2012).

My second chapter, in contrast, is focused on present political dynamics, examining governance practices in protected areas. Two key findings emerge from the literature on protected area effects: first, impacts vary widely, and second, governance has an indelible effect on both social and ecological outcomes. I draw on Ostrom's Design Principles of common pool resource governance to understand governance practices in a network of Marine Protected Areas (MPAs) in Eastern Indonesia. I aim to first provide a novel approach for operationalizing Ostrom's framework, and to then apply that framework towards understanding the array of governance that exists across 11 recently created MPAs in the region. I use a cluster analysis methodology, Latent Profile Analysis (LPA), to examine what patterns of governance exist in the settlements within MPAs, analyzing the variation in governance practices and the clusters which emerge as they relate to the geography of the MPAs. Variation in governance practices is in part driven by cultural norms of resource governance, with important implications for other contexts in which PA rules interact with longstanding traditional practices (Boli et al. 2014, McLeod et al. 2009).

My third chapter is retrospective, applying causal inference methods to estimate the historical long-run aggregate impact of PAs in the mangrove forests of Brazil. It follows a familiar pattern of impact evaluation papers in environmental economics (Ferraro 2009), updated to apply recent advances in difference-in-differences methods which allow for staggered treatment rollout (Gardner et al. 2024). The central contributions and limitations of this particular impact evaluation study are determined by its scope. I look at impacts over a long time period (35 years), sizeable geographic scale (the entire coast of Brazil), and at

detailed spatiotemporal resolution (30m satellite data yearly, aggregated to 1km grid cells and four-year government planning cycles), to answer a broad question of whether protected areas have had an overall impact. With few studies that look at staggered PA creation, this methodology allows for a detailed and comprehensive examination of the overall effect of PAs across a common biome, yet raises important questions about how the effects of PAs might vary geographically and temporally.

In sum, my dissertation can be taken as exploring the future, present, and past of conservation practice. I apply various methodologies to develop our understanding of two key conservation tools, driven by a desire to understand how such initiatives can reverse the decline of nature. While the limitations and goals of each methodology are distinct, they share a common aim of informing conservation practice and promoting ecological flourishing. Expanding protected area coverage and scaling up restoration projects are both necessary to meet those goals, and this research aims to help inform that development.

2. Climate projections affect restoration priorities: application to Brazil

2.1 Introduction

In the conservation sphere, interest and funding for ecological restoration projects are steadily increasing. With the UN declaring the 2020s the "Decade on Ecosystem Restoration", there is a recognition that to meet sustainable development targets, large-scale restoration initiatives are necessary (Jones et al. 2018). While the body of ecological research on restoration is sizeable, the decision of where to invest limited funds for a restoration project is fundamentally an economic one. Recent analyses have focused on the prioritization question from the perspective of present-day costs and benefits, an important step in understanding the economic planning decision (Jakovac et al. 2020, Busch et al. 2024), while others have used optimization methods to choose optimal sets of sites under current conditions (Beyer et al. 2016), often modeling landscape connectivity (Viani et al. 2018, Antongiovanni et al. 2022). In practice, however, programs have encountered challenges that present obstacles to long-run success. In particular, the literature has highlighted the long time period involved and the high cost of projects, both affecting optimal behavior (Brancalion et al. 2021). While some papers have looked at certain shifting ecological dynamics in isolation (Koch and Kaplan 2022), the way in which changing conditions, particularly those driven by climate change, affects the economic decision remains underexplored.

To understand these impacts, a model that accounts for the temporal dimension of restoration planning has the ability to improve economic outcomes for decision makers. With the wide-ranging body of literature on dynamic planning for the conservation of

existing ecosystems (Polasky et al. 2008), there is ample opportunity to apply similar tools to the restoration context, adapted to the particularities that a restoration program entails. In this paper, we aim to shed light on the planning decision in ecological restoration by presenting a generalized economic framework that can be applied to various restoration contexts, and through an empirical application, exploring how incorporating the projected effects of climate change might inform the planning decision.

At the most basic level, a restoration project aims to increase the economic value of an ecosystem, spending money today for the sake of an increased stream of benefits in the future, both as the project progresses and once it is complete (McBride et al. 2010). Restoration brings use and nonuse benefits for local communities and outsiders alike, including recreation, agriculture, biodiversity protection, carbon sequestration, and many others (Murdoch et al. 2007). On the other side of the equation, a restoration project includes two main categories of costs. First is the opportunity cost, the alternative economic value of a site being dedicated to restoration. The other is implementation costs, which are incurred over the period of time over which restoration is actively underway.

Both costs and benefits vary spatially and temporally. For example, biodiversity benefits are affected by local ecological conditions and surrounding land use patterns, while other categories of benefits have analogous variation. Opportunity costs are affected by agricultural conditions and demand for urban land uses. Perhaps most relevant, global climate change affects both benefits and costs (Koch and Kaplan 2022). Shifting habitat boundaries and vegetation density affect benefits, while human migration and changing crop choices and yields affect opportunity costs. Climate models provide some predictions as to the nature of these trajectories.

The economic decision faced by restoration planners has parallels with research on dynamic site selection for conservation, spanning the economics and ecology literatures (Costello and Polasky 2004, Strange et al. 2006). Planners have an array of sites available for restoration, each of which has its own characteristics. They have a limited budget, and aim to maximize some conservation objective, typically a combination of benefits that each site brings. Like restoration, conservation decisions are linked across time via the possibility of sites being developed (Costello and Polasky 2004), habitat dynamics (Van Teeffelen et al. 2012), or other factors. Work has also explored uncertainty in habitat dynamics (Gren et al. 2014), and population viabilities (Newbold and Siikamaki 2009), each of which has a parallel with the choice of restoration sites. In a sense, many of the models in conservation are the inverse of the restoration question: instead of preventing a site from facing future losses, the restoration planner chooses to spend on implementation today for the sake of benefits down the road.

Economic research on restoration itself has focused on specific contexts, prompting calls for a unified economic approach (Bodin et al. 2022). In the larger scientific literature, restoration analyses have primarily focused on spatial planning to maximize ecological goals (Rodrigues et al. 2011), along with work that considers the net-benefit maximization problem (Jakovac et al. 2020, Strassburg et al. 2019, Busch et al. 2024). Forward-looking applications have included planning for scenarios under climate change, with tropical forest regrowth (Koch and Kaplan 2022), and the intensity margin (the optimal level of restoration) for habitat restoration investments (Fonner et al. 2021).

From a mathematical perspective, the restoration planning decision has been formalized in McBride et al. (2010), while the choice between multiple restoration scenarios

is outlined in Metzger et al. (2017). Nevertheless, the economic planning problem for a forward-looking, dynamic actor has important differences from these models, which inform the theoretical framework we present. The temporal dynamics of restoration programs are affected by when costs are incurred. While protecting an existing area and implementing a restoration program both require an upfront investment, active restoration requires greater continued spending, with benefits being less certain and further in the future. Significantly, this increases the potential for path dependence and lock-in effects: if we have already spent years investing in one site, we may be unlikely to switch, since the benefits are closer, even if better alternatives exist (Drechler and Watzold 2020). Similarly, if a site happens to be well-suited for restoration in the present, due to randomness in opportunity costs or benefits, initiating restoration may beneficially commit a decision maker to remain in that site as conditions change. The decision to continue restoration investment does not occur at a single point in time, but is rather continued over a long time period with many opportunities for abandonment, heightening this effect. In this paper, we explore the implication of that dynamic, first presenting a generalized model of the restoration investment decision based on changing conditions over time.

2.2 Theoretical framework

2.2.1 Model of a single site

In this section, we aim to formalize the value to be found in a restoration project. The relevant decision faced by a planner is when and whether to initiate restoration at a given site, as a function of that site's characteristics. We first consider the decision at a single site, and define the following site-specific variables:

- Opportunity costs of the land per unit area, expressed as a flow of benefits the land would provide in an alternative use at each point in time $C(t)$. In the context we examine, the opportunity cost is agricultural land values. Changing crop yields through time are the assumed main driver of changes in agricultural opportunity costs in this context, but where other land uses predominate, other drivers of change may be present, such as urbanization.
- The maximum possible stream of benefits $B(t)$ per unit area that could be produced by a site, were it to be fully restored instantaneously, also expressed as a flow of benefits over time. For biodiversity benefits, drivers of change of this metric might be changing species distributions or habitat viability.

Each of these variables is likely to vary across space, and as such is indexed by the area *i* for which data are available, where each area contains sites available for restoration. In practice, their trajectories as a function of time can be interpolated from future projections at each site: i.e., if we have data on present conditions and data at a future point, we can interpolate to construct the path of each. We also define the following characteristics for a given restoration methodology:

- The recovery pathway $R(t)$ for restoration benefits as a function of time under restoration, as a fraction of full benefits.
- The implementation costs per unit area for each point in time $I(t)$, which can include planting, seedlings, fencing, and other costs associated with the restoration project
- The social discount rate δ

The decision faced by the planner is whether and when to initiate restoration. There are two periods over which land-use benefits are accrued: before restoration is initiated, and after. Given the optimal choice of initiation time, the total value of a site is the sum of the present value of benefits in each time period, expressed by its respective integral. For simplicity, we assume that each potential site has independent characteristics, such that there is no "spillover" across sites. While such spillovers may exist, we assume that whatever spillover effects exist are accounted for in the site-specific data projections available. Explicit functional forms for the nature of such spillovers could be integrated into this framework via the application of linear programming methods to select an array of sites (See Beyer et al. 2016, Strassburg et al. 2019), but are outside the scope of this paper.

Given an unrestored site, the planner chooses the initiation time t_0, to maximize the total gross value of a site, given by:

$$V_i = \int_0^{t_0} (C_i(t)e^{-\delta t})dt + \int_{t_0}^{\infty} (B_i(t)\,R(t) - I(t))e^{-\delta t}dt$$

This expression represents the value of the site over its infinite lifetime, given the initiation of restoration at the optimal time t_0. We can differentiate it with respect to t_0 to obtain the marginal condition that determines the date at which restoration should be initiated:

$$B_i(t_0)R(t_0) - I_i(t_0) - C_i(t_0) = 0$$

The second order condition is thus:

$$B_i'(t_0)R(t_0) + B_i(t_0)R'(t_0) - I_i'(t_0) - C_i'(t_0) < 0$$

While the marginality conditions are not used in our empirical application below, the validity of the second order condition will depend on the functional form of *R(t),* *B(t),* and *C(t).*

This model leaves out the possibility of planned abandonment of restoration, which would simply add an additional term. While in practice, reforestation reversals are quite common (Schwartz et al. 2020), given that the vast majority of ecological benefits of restoration come far down the line, restoration abandonment is likely to be due to changing conditions, rather than a reversal that was economically beneficial from the start. Since it is these unexpected abandonments that are cited as a primary challenge of restoration, rather than planned abandonment (Brancalion et al. 2019), this assumption is well served in empirical contexts.

Equivalent to maximizing V_i is to maximize an expression for net benefits, which subtracts from the expression above the present value of $C_i(t)$, i.e., the opportunity costs of restoration:

$$\int_{t_0}^{\infty}(B_i\,(t)R(t) - I_i(t) - C_i(t)e^{-\delta t})dt,$$

Section 2 of this paper estimates the value of each of these terms empirically for potential restoration sites across Brazil, with data changing between 2020 and 2050 for the first and third terms.

2.2.2 Prioritizing across sites

Many sites available for restoration might have a positive value for the preceding expression, which leads to the question of how to choose among them in a given year t_i'. If all variables in the expression are defined per standard unit area, then one option is to first choose the site that maximizes the expression below, i.e., offers the highest value per unit area:

$$NPV_i = \int_{t_i'}^{\infty} (B_i(t)R(t) - I_i(t) - C_i(t)e^{-\delta t})dt$$

Alternatively, we can rank the sites in that year using the benefit-cost ratio (BCR). In contrast to choosing the greatest NPV, which will yield the greatest benefits per site, choosing the set of sites with the greatest BCR until a budget is exhausted will maximize total net benefits, which is more relevant to a social planner (Nas 2016):

$$BCR_i = \int_{t_i'}^{\infty} [(B_i(t)R(t))/(I_i(t) + C_i(t))]e^{-\delta t}dt$$

In practice in the conservation sphere, there is often a target amount of land to restore $\bar{A}$. For example, Brazil's Native Vegetation Protection Law requires landowners to restore certain areas of their holdings, with a specific 20% threshold legislated in the Atlantic Forest region (Lima et al. 2020, Brancalion et al. 2016). International agreements also set overall goals, such as in the Aichi Biodiversity Targets. Given a vector of site priorities, we can choose the highest value sites until the target is met. Given sites $A_1 \ldots A_n$ listed in order of BCR_i, we choose:

$$Min\ j \mid \Sigma_{\{i=0\}}^{\{i=j\}} A_i \geq \bar{A}$$

We assume targets are not modified between years, and that there are no local economies of scale in terms of *B* or *I*: restoration targets and prioritization decisions are made across a wide spatial scale, (perhaps a country or region), but are based on local data.

2.3 Empirical application

We next move to investigating the implications of future trajectories in a particular restoration context of significant conservation interest. Forest restoration in Brazil has gained much funding and attention, due to high rates of degradation and endemism. The federal government has passed multiple versions of a natural revegetation law, starting with the 1965 Brazilian Forest Code (Rosário et al. 2019) and most recently updated in 2012. This law requires landowners to restore particular areas, with special attention to riparian zones, and meet minimum forest cover requirements. Due to the presence of sizeable tracts of deforested land without an active seed source, this often requires investing in active replanting. Active restoration, on which we base the empirical section, is distinguished from passive restoration by the necessity of planting seedlings, instead of depending on a natural seed source from nearby forests. There are a variety of active restoration methods, but in general, active restoration is higher in cost than passive methods, and is typically undertaken in places where passive restoration would not be possible.

There have been large-scale investments in restoration programs across the country, including the Atlantic Forest Restoration Pact. As a result, there is a large body of research spanning the conservation science and ecology literatures on the implementation of tree planting projects in the region, including measures of costs (Assunção and Chein 2016), biodiversity benefits, carbon sequestration, predictions of passive restoration (Crouzeilles et al. 2020), and models of priorities using present data (Rosário et al. 2019). Compliance with

the restoration requirements of the forest code has been sporadic (Brancalion et al. 2016), underlining the need for a better understanding of how conditions affect restoration decisions.

Our empirical application makes two main contributions. First, it demonstrates the value of a model that incorporates future trajectories, providing an illustrative case study for our theoretical framework. Second, we add to the literature on planning for restoration in Brazil, adding several components relevant to the value of restoration with a number of local data sources in tandem, all of which affect the restoration decision: spatial variation in restoration value, ecological intactness, projected changes in restoration value, agricultural opportunity costs, and projected changes in agricultural value. The goal of our analysis is to find the value of investment for each site with independent characteristics across the country, and thereby to yield a relative ranking of the priorities of sites. We calculate each site-level value as a BCR, discounting benefits and costs in perpetuity.

We define two scenarios, based on two separate sets of information: conditions in 2020, and conditions in 2050, defined with *0* and *f* subscripts respectively. The approach aims to illustrate the comparison between present decisions under two different sets of information. In each, we take both benefits and costs as fixed, using solely 2020 and 2050 in their respective scenario, and imagine the value of restoration initiated immediately ($t=0$) using either 2020 or 2050 values.

This approach is based on the availability of projections in 2050 at present. Given the framework presented above, further simulations could interpolate the trajectory between the present and a future point to examine how site selection progresses under different

decision criteria. In this context, however, the proximity of 2050 under the recovery trajectory used means that choosing sites at present based on information on conditions in 2050 is likely to lead to "full" (>90%) restoration under 2050 conditions, making 2050 conditions a relevant decision criteria at present.

2.3.1 Estimating benefits

We first aim to quantify the relevant conservation benefits. We assume the maximum potential restoration value is proportional to the level of biodiversity at risk within a given site, a "hotspots" approach (Myers et al. 2000). All data sources are presented in Table 1, and variables are defined in Table 2. Following work that uses species at risk to inform restoration priorities (Etter et al. 2020), we sum the number of threatened bird and mammal species by grid cell and normalize by the mean and standard deviation across all sites in the entire country. This measure is a proxy for expected avoided extinctions by a given habitat increase from restored land. Being built from species range maps, this measure generally aims to be a metric of the potential threatened species which can exist in a particular grid cell, given suitable restored habitat (Jenkins et al. 2013). A median grid cell has 18 threatened species (See Table 3).

Table 1: Data Sources

Variable	Name	Description	Varies spatially?	Temporally?

Biodiversity protection value of a restored tract	D	The sum of threatened mammal and bird species in each grid cell, from Jenkins et al. 2013	Yes, at 10km resolution, with 2050 values differing from 2020 values due to 1km resolution projections of forest risk (L)	Yes, in 2020 and 2050
Biodiversity intactness Index	I	A measure of the average proportion of biodiversity remaining in a given grid cell, from Newbold et al. 2016.	Yes, at 1km resolution	Yes, but with no spatial specificity: at the country scale in 2050 using Phillips et al. (2021).

Agricultural opportunity costs	A	Gridded crop production value per hectare from MapSPAM (Ru et al. 2022), multiplied by an estimate of the input share for cropland (Souza et al. 2021).	Yes, at 10km resolution	Yes, adjusted for changes in productivity in 2050 by municipality level Z (Assunção and Chein 2015)

Table 2: Definition of Empirical Variables

Variable name	**Definition**
R_i	Threatened birds in each tract *i*
$\mathrm{M_i}$	Threatened mammals in each tract *i*
I_i	Biodiversity intactness in each tract
n	Number of grid cells
$\overline{D_i}$	Normalized biodiversity value, $\overline{D_i} = \frac{R_i+M_i}{\Sigma(R_i+M_i)/n}$

$\overline{I_i}$	Normalized biodiversity intactness gap, $\overline{I_i} = \frac{1-I_i}{\Sigma(1-\frac{I_i}{n})}$

Table 3: Summary Statistics of Variables and Restoration Value Outcomes

Variable	**Minimum**	**25th Percentile**	**Mean**	**Median**	**75th Percentile**	**Maximum**
Threatened Species	2.0	11.0	16.9	18.0	23.0	35.0
Forest Change	-98.5%	-29.2%	-10.5	-6.2%	+10.5%	+88.7%
Agricultural Opportunity Costs	0	$11,413	$20,509	$15,646	$24,159	$334,897
Cost change	-100%	-46.4%	-32.1%	-35.0%	-17.7%	+57.3%
BCR 2020	0	.54	1.44	1.05	2.02	20.08
BCR 2050	0	.60	2.04	1.32	3.67	Inf
BCR Change	0	-1.3%	37.9%	25.2%	63.1%	Inf
BCR Abs. Percentile Change	0	2.06	8.04	5.52	11.27	98.48

The normalization of biodiversity value relative to the mean level of threatened species across all tracts implies that a value of 1 represents the number of threatened species in an average parcel, while a value of 2 represents double the mean number of threatened species. This value represents the impact that a given site has as restored habitat, relative to the average habitat benefits brought by a particular tract. The basic assumption is linear, taking as given that if we restore a site which can provide habitat for double the number of threatened species than the average, then that site has double the habitat value for biodiversity protection.

In general, there is more to be marginally gained in habitat protection from restoring sites which are further from their intact state due to human pressure. While implementation costs may also vary with degree of intactness, that variation is not the focus of this analysis (See "Conclusions and Policy Implications"). Newbold et al. (2016) constructs a spatial data layer that gives estimates a global index of intactness at 1km resolution, expressed as a proportion of the potential biodiversity which could exist in that grid cell. We are interested in the gap between the index for a given site and its value for potential fully intact ecosystem, which is 1. This gap is analogous to measures of human pressure which have been used in past measures of "restoration opportunity"—sites with a greater degree of degradation present larger marginal benefits of restoration (Reytar 2014). We normalize the gap relative to the intactness level in an average tract (Table 2). This normalized measure provides a valuable benchmark for a restoration planner, who is interested in the marginal benefit to be gained from active restoration of a site to a level which can support its full potential biodiversity benefits.

We monetize the benefits using habitat protection value estimates H for a typical area of forest from a meta-analysis by Brouwer et al. (2022). Their meta-analysis provides a range of estimated values given for different ecosystem services, but given that we use only their estimate for habitat protection value as most relevant to biodiversity protection, our monetized estimate of value can be taken as a lower bound on the total value of restoration. While true restoration values are likely far higher when accounting for other benefits, absent data on spatial and temporal variation in those other benefits, spatial and temporal priorities will remain the same.

We thus have the following basic equation for potential ecological benefits per hectare in each grid cell in the present, were the unrestored land in the grid cell to be restored immediately:

$$B_i^0 = H * \overline{D_\iota} * \overline{I_\iota}$$

As defined in Table 2, $\overline{D_\iota}$ represents normalized biodiversity value while $\overline{I_\iota}$ represents the normalized biodiversity intactness gap.

The current values of B_i^0 are then adjusted for the future using projected changes in forest potential under climate change from Bastin et al. (2019), such that B_i^0 is multiplied by a risk factor L_i in each site i, which can take on positive or negative values. The change represents risk in the potential habitat value of the ecosystem within a given grid cell, given expected climatic shifts which affect the habitat integrity of forests and other biomes. The biodiversity intactness index is adjusted across all sites using a constant Brazil-wide estimate, *BR*, of changes to 2050 given in Phillips et al. (2021). This adjustment accounts for

increasing anthropic pressure in the coming years. In sum, we define future values V_i^f given instantaneous restoration as follows, using tract-specific forest risk estimates L_i:

$$B_i^f = B_i^0 * (1 - L_i) * BR$$

2.3.2 Recovery trajectory and discounting

To project benefits and account for time under restoration, we estimate an ecosystem recovery curve over time. A recent meta-analysis of restoration in the Atlantic Forest has estimated that, according to median values in the studies reviewed, 50% recovery occurs at approximately ten years (Romanelli et al. 2022). While this estimate varies by study and the biodiversity metric used, the hyperbolic form remains consistent (Romanelli et al. 2022). Using the estimate of 50% recovery at ten years and the standard hyperbolic growth function, we use the recovery curve $R(t) = 1 - 0.5^{\frac{t}{10}}$, where 1 represents the maximum level of restoration.

The social discount rate δ is taken as 2 percent, a consensus value from a survey of economists (Drupp et al. 2018). Adding continuous discounting at this rate yields an overall expression for capitalized benefits, $\widehat{B_i^0}$, from time t_0 to ∞ for the present information scenario and future information scenario defined above:

$$\widehat{\mathrm{B}_\mathrm{i}^0} = \int_{t_0}^{\infty} \left(1 - 0.5^{t/10}\right) B_i^0 (e^{-0.02t}) \mathrm{dt}$$

$$\widehat{B_i^f} = \int_{t_0}^{\infty} \left(1 - 0.5^{t/10}\right) B_i^f (e^{-0.02t}) \mathrm{dt}$$

2.3.3 Estimating costs

The cost term of our model, C_i, includes two components: agricultural opportunity costs, estimated using site-specific yearly agricultural land values A_i, and spatially invariant implementation costs CI_i. . While in some contexts other opportunity costs may predominate, such as with urban land uses, this study focuses on the bulk of active restoration efforts in Brazil, which have centered on degraded agricultural land. These lands are likely to be too far from forest remnants and have too much soil degradation for natural regeneration to be feasible (Brancalion et al. 2019). Our agricultural land values are derived from IFPRI's gridded estimates of area of cropland harvested and value of crop production (Ru et al. 2019). These are used in tandem to calculate value per harvested area, converted to annual land rent by multiplying by an estimate of the land input share for cropland in Brazil (Evenson and Fuglie 2010), and capitalized to land value using the 2% social discount rate. While the land input share estimate may not account for local variation in crops grown at the 10-km scale of the data on agricultural crop production, disaggregated estimates are not available. Site-specific changes in agricultural productivity as of 2050 based on climatic shifts Z_i are taken on a municipality level from Assunção and Chein (2015), allowing us to define future agricultural values $A_i^f = A_i^0 * (1 + Z_i)$.

We assume implementation costs are constant across sites at CI. We draw an estimate of *CI* from Brancalion et al. (2019) which estimates total costs of restoration and an average timeline of those costs (approximately 5 years). We discount these costs by assuming they are equally distributed across the 5 years.

We thus have expressions for capitalized costs under 2020 conditions and 2050 conditions:

$$\widehat{C_i^0} = \int_{t_0}^{\infty} A_i^0 e^{-0.02t}\, dt + CI$$

$$\widehat{C_i^f} = \int_{t_0}^{\infty} A_i^f\, e^{-0.02t}\, dt + CI$$

This yields defined expressions for benefits and costs which each include discounted values over the infinite time horizon. The BCR and thereby the relative ranking for each tract i in the present and future are as follows:

$$V_i^0 = \frac{\widehat{B_i^0}}{\widehat{C_i^0}}$$

$$V_i^f = \frac{\widehat{B_i^f}}{\widehat{C_i^f}}$$

2.4 Results

Given the data constructed above, we first examine the main spatial patterns across the two drivers of changing conditions, changing habitat and agricultural land value. In Figure 1, we show the patterns of habitat change. Many forests in the south and east of the country are expected to see gains, due to changing precipitation patterns (Bastin et al. 2019). However, climatic shifts in the Amazon region and far south present risks to habitats in those regions.

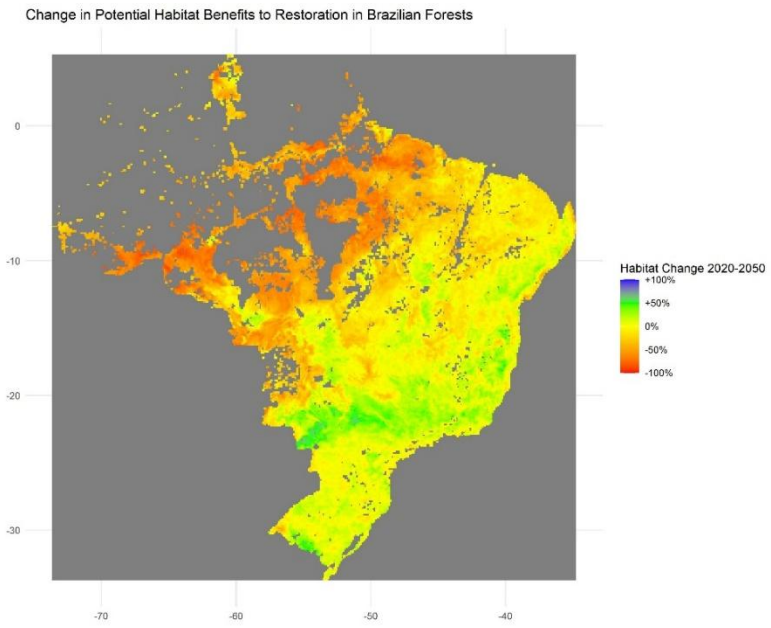


Figure 1: Change in Potential Benefits (Forest Habitat), 2020-2050

In Figure 2, we show changes in agricultural land value. The vast majority of the country faces climatic risks from decreased precipitation and warming patterns, with the exception of the far south, which may see an increase in productivity (Assunção and Chein 2015). The question in considering the impact on restoration planning is whether these changes are offset by corresponding risks to biodiversity.

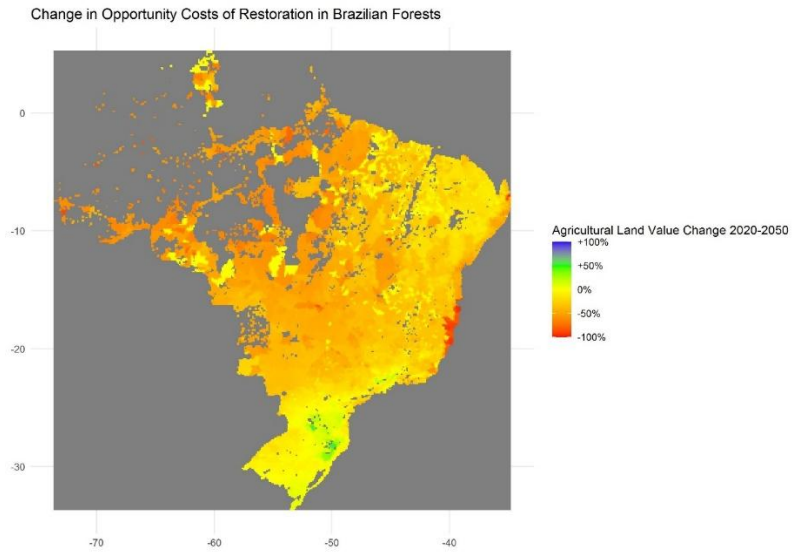


Figure 2: Change in Opportunity Costs (Agricultural Land Value), 2020-2050

Using the methodology described above, we next calculate the overall value of initiating restoration for every 1km grid cell in Brazil for which agricultural and biodiversity data are available (n=4.2million) given estimates of benefits and costs from 2020, were restoration initiated to be in that year. Data availability limits our analysis to those grid cells which include some agricultural land which could be restored. These results are visualized in Figure 3, showing the BCR for each grid cell across the country where ecological data is available. In general, the highest-priority sites are in the interior, with higher opportunity costs and lower species density along the more populated coast.

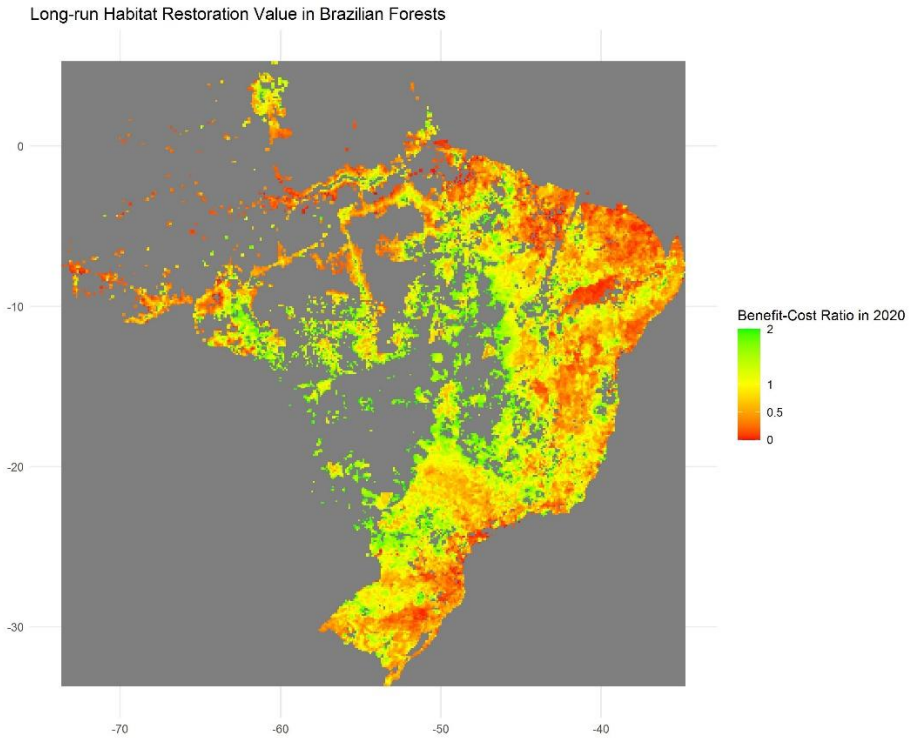


Figure 3: Benefit-Cost Ratio by Tract in 2020

We next calculate corresponding benefit-cost ratios for each grid cell in 2050, based on site-specific changes to habitat value and agricultural opportunity costs, and uniform changes to biodiversity intactness. These results are shown in Figure 4. Visually, the overall pattern of the coast and interior remains the same, but with some trends being exacerbated: some formerly low-value areas become even more so, as do high-value areas. The overall increase from 2020 to 2050, which is summarized in Table 3, is not immediately apparent, but looks to be mainly in sites of high value.

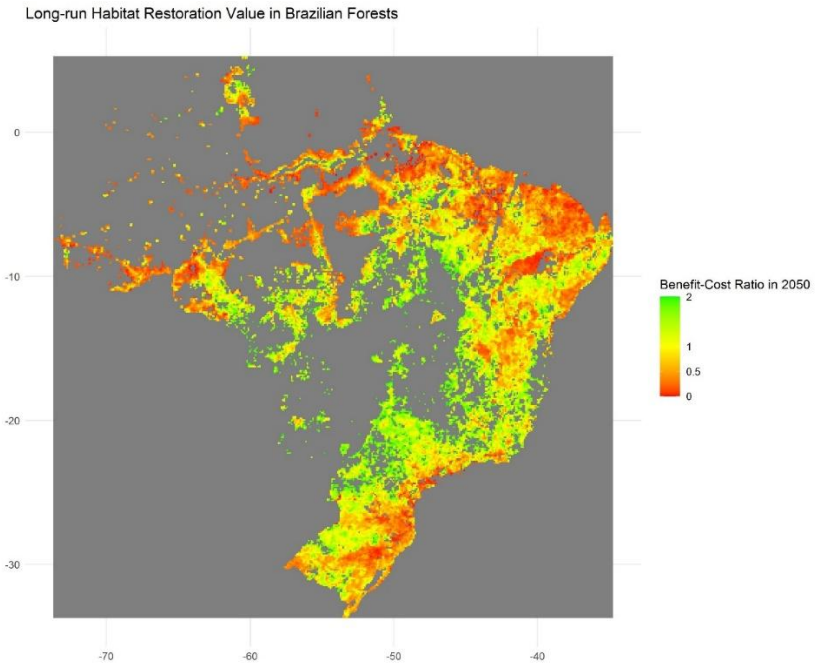


Figure 4: Benefit-Cost Ratio by Tract in 2050

We are also interested in the spatial patterns of changes in the BCR across the country, given the results observed in each year. We plot these changes in Figure 5 as a percentage of the BCR, showing which areas of the country are expected to gain in restoration value over the next 30 years, and which are expected to lose value. In general, sites in the far interior, primarily the Amazon region, lose value, likely due to the comparatively higher forest risk in these regions. In the far south value also goes down, with small forest gains being largely offset by a larger increase in agricultural opportunity costs. Meanwhile, the central region, including both the interior and the coast, appears to gain.

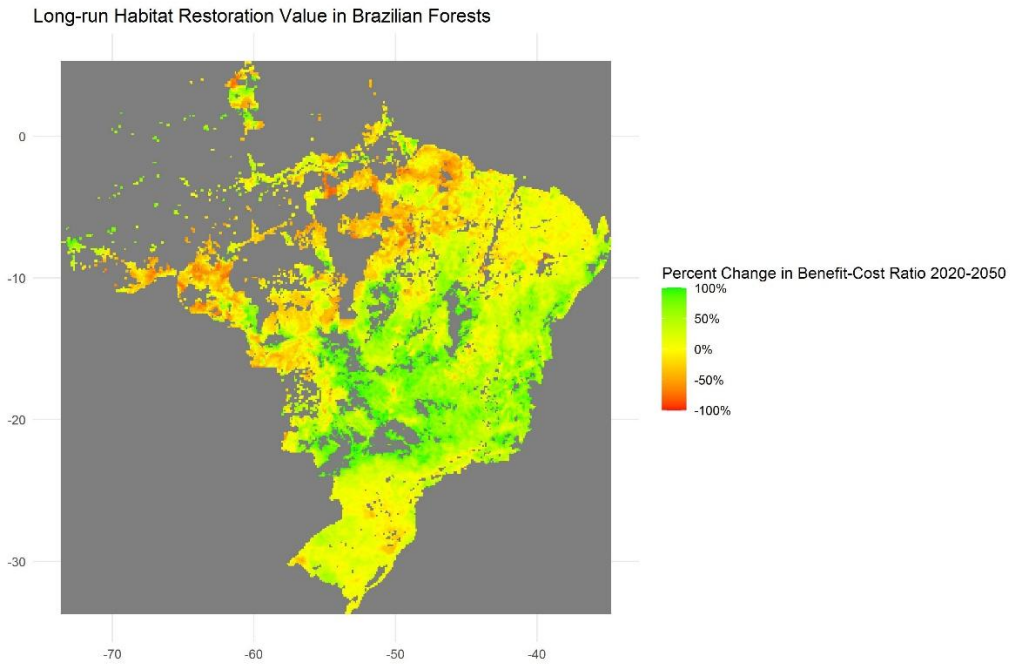


Figure 5: Percent Change in Benefit-Cost Ratio by Tract, 2020-2050

Perhaps more relevant for the policy decision, we would want to know the extent to which our relative rankings of potential restoration areas might change. For example, it could be the case that although significant changes in the value of restoration take place from 2020 to 2050, these changes may occur uniformly, and thus would not affect our choice of sites over time. In each scenario, grid cells have a percentile ranking of their BCR relative to all other sites. In Figure 6, we show the change in these percentile rankings from 2020-2050, illustrating the extent to which sites gain or lose relative to others. Spatial patterns generally follow the absolute changes observed in Figure 5, with some coastal areas in the population core observing a particularly large relative gain. Summary statistics in Table 3 show absolute changes in percentiles in either direction.

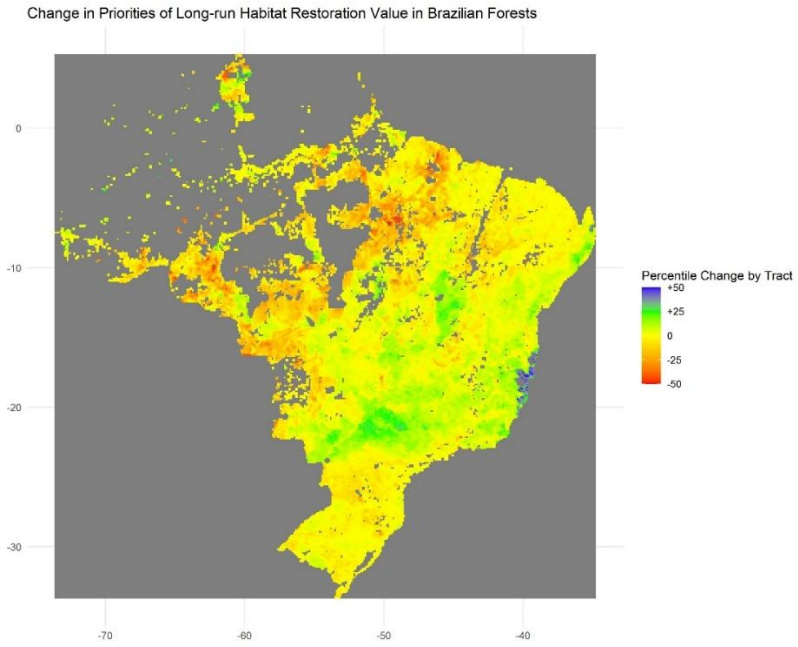


Figure 6: Change in Relative Rankings (Percentile) of Sites, 2020-2050

Further, given a particular target of the amount of land to restore, we can visualize the geographic distribution in those sites which would be selected under that target. For example, if we set an aim of fully restoring 25% of the land across the country, all grid cells that are in greater than the 75th percentile would be chosen. However, the particular grid cells which would be chosen change from 2020 to 2050, with a certain set of grid cells abandoned and a set of equal size initiated. We plot the geographic distribution of this change in priorities under a 25% target in Figure 7, with the 25% aligning with a goal of the Atlantic Forest Restoration Pact (Melo et al. 2013).

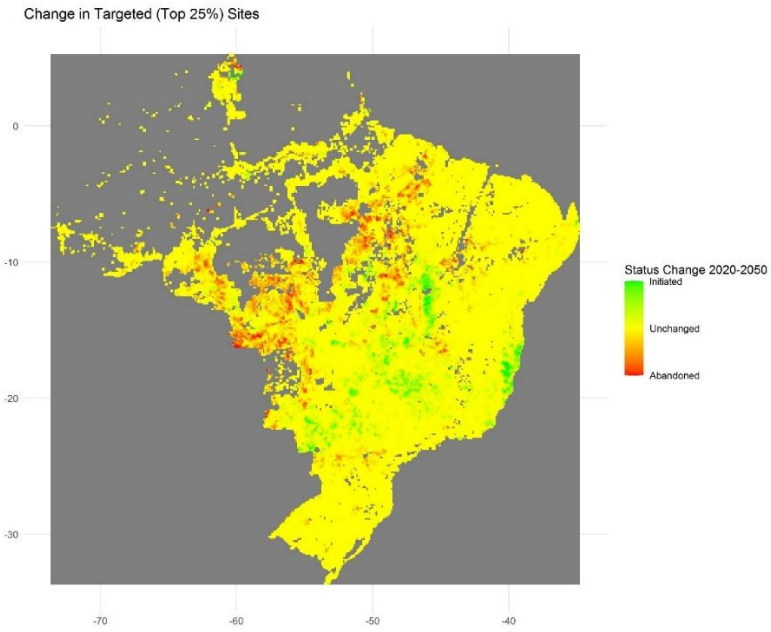


Figure 7: Change in Site Status 2020-2050, With 25% Restoration Target

In addition, the extent of this switch might vary with the level of commitment to restoration. We plot in Figure 8 the degree of change in priorities as a fraction of the land to be restored for each possible target from 0 to 99 percent. For example, if we select the best 10% of grid cells for restoration, and 5% of the total grid cells were to undergo a change in status from 2020 to 2050, then 50% of the selected grid cells have undergone a change. We observe a generally decreasing trend, with high fractions of switching at the low levels of restoration characteristic of the status quo.

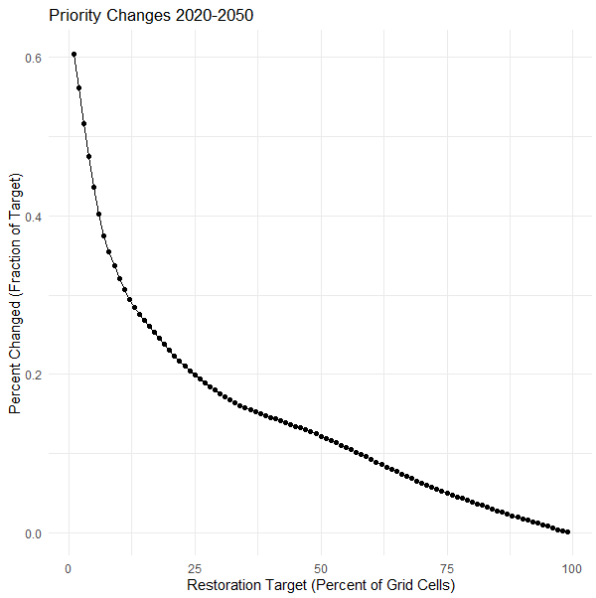


Figure 8: Changes in Selected Grid Cells by Restoration Target

Lastly, as discussed in the theoretical framework, the decision to initiate or abandon a restoration site may be based not on a target area of land but rather on an economic cutoff. Given the significant change in overall value of restoration observed from 2020-2050 (See Table 3 for summary statistics), we might expect restoration in a number of grid cells to become newly worthwhile. In Figure 9, we plot the fraction of total grid cells which become worthwhile over time, along with those that become unselected over the 30 year period. We show each value along with their sum, the total changes, for all possible decision criteria between a BCR of 0.75 and 1.75. Total selection changes range from approximately 12% to 15% of all sites, of which the majority (81% on average) is initiated restoration, with some abandonment. For reference, ~52% of sites in 2020 would be selected based on a BCR cutoff of 1, while in 2050 60% of sites would be selected, meaning that approximately 25% of sites initially unselected are eventually chosen. Changes in the distribution of benefit-cost ratios are also shown in Figure 10.

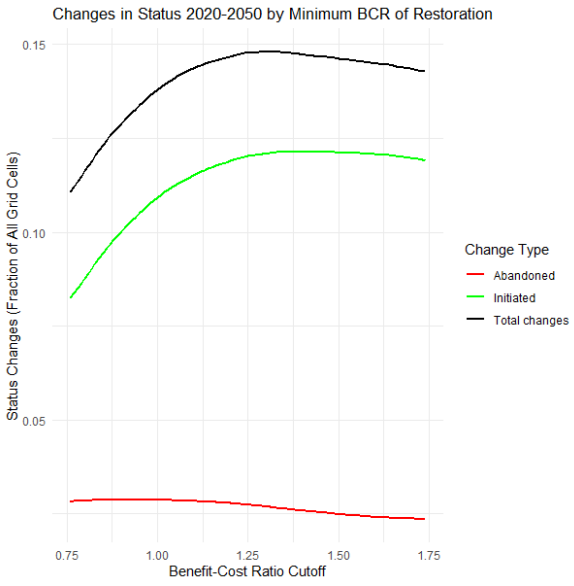


Figure 9: Changes in Status Based on a Minimum Benefit-Cost Ratio of Restoration

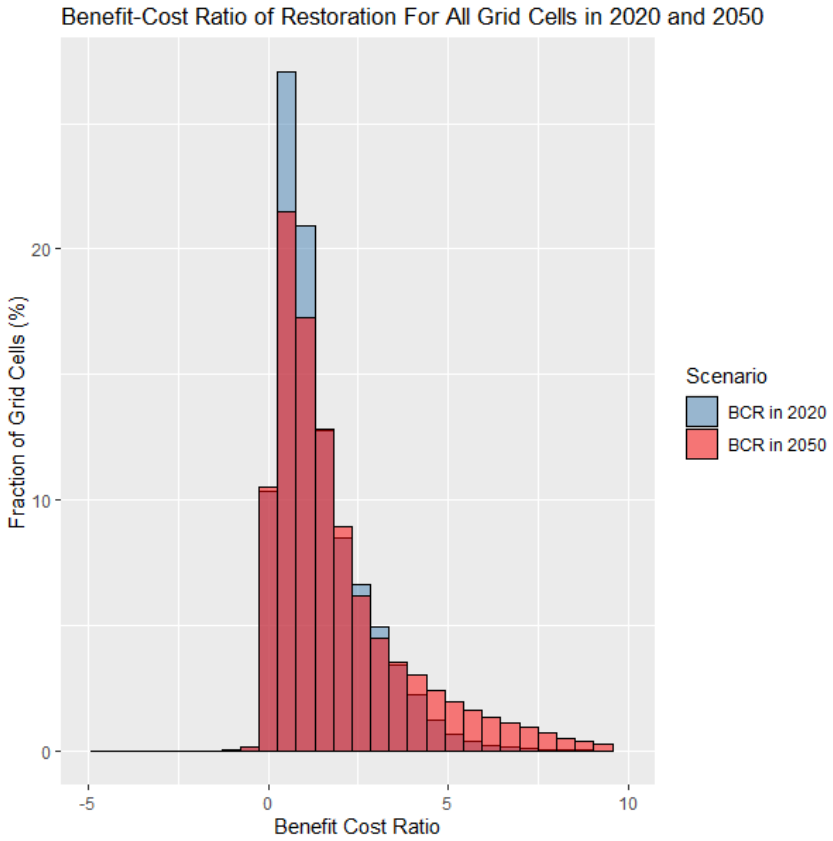


Figure 10: Changes in the Distribution of Benefit-Cost Ratios, 2020-2050

2.5 Conclusions and policy implications

Our first major finding is that in many cases, indeed across a majority of the country, active restoration can be justified based on habitat value alone. This finding is informed by the data we integrate on a variety of characteristics on cost and benefit data in Brazil, a novel finding in and of itself. The overall BCR is over one for the majority of grid cells, and across a wide range of BCR cutoff points, a significant fraction of available land in the country yields a positive value for the restoration project. In addition, the distribution of sites is significantly skewed right, as shown in Figure 10. This means that there are numerous sites with benefit-cost ratios far above 1, likely with higher numbers of threatened species and lower opportunity costs. As such, we see that active restoration can be highly profitable when it is targeted, supporting the previous literature on the topic (Crouzeilles et al. 2019). As seen in Figure 10, there are even more such cases in 2050, underlining the importance of investing early in those sites.

At the same time, the monetization of our priority estimates rests on a single meta-analysis of valuation studies, which has a large degree of uncertainty. The 95% confidence interval reported in Brouwer et al. (2022) is $109-$800 per hectare, a wide range around the mean of $455 used. While this average estimate may provide some benchmark at which restoration can be justified based on habitat protection alone, the degree of uncertainty leads us to focus on the relative changes.

Along these lines, the differences between 2020 and 2050 are apparent. Both forest potential and agricultural opportunity costs are expected to decrease in the aggregate by 2050 (See Table 3), and a priori, it is unclear which of these effects would be more significant. The median benefit-cost ratios are close, but the mean increases significantly in

the future. This shift reflects areas which are hard hit by climate change in their agricultural productivity, while retaining high forest cover.

The spatial patterns in the temporal changes observed also yield insights. In Figure 5, we see there is a significant portion of the country that sees a major increase in its value of restoration, primarily in the agricultural heartland, areas that will see significant challenges to agriculture under climatic changes, yet retain their habitat value. Meanwhile, some areas in the Amazon are initially worthwhile for restoration, but become less so. The overall variation in the BCR is increased, a common effect of climate change. Across both regions, the impact of these changes on how we plan restoration in the present is apparent, resulting in both delayed restoration in a site which is eventually highly worthwhile, and forgone value from choosing an initial site which may remain worthwhile, but far less so than others. It is clear that in these regions, the challenges to agriculture under climate change appear to far outweigh the risk to habitat, under current projections.

There are also meaningful changes in relative rankings of sites which differ from the absolute shift. Many sites along the coast, including in the north, gain significantly in relative terms due to their low initial value. While they are not the highest-value sites, they improve significantly relative to others, primarily driven by expected positive changes to habitat in this region, as made clear by the similar pattern in the northeast between Figures 1 and 6.

After choosing a particular target, there are similar spatial patterns of sites which undergo a change in status. In the 25% target scenario shown in Figure 7, the two areas in which sites become comparatively higher value are in the central region and along the coast, reflecting what we observe above. However, the total number of such sites will depend on

the target chosen, as seen in Figure 8. The pattern of switching is even more present when we consider the decision to initiate restoration in absolute terms in Figure 9. The fact that over 25% of sites which are initially unselected eventually become profitable is strong evidence that expectations of the future should influence our decision to restore land in the present.

Overall, our results provide strong support for the use of climate projections in restoration planning. Many factors remain unaccounted for in this analysis, as many anthropic land use changes are also expected to have an effect on the value of restoration. Shifting patterns of urbanization and residential land use may affect both sides of the equation: opportunity costs in rapidly growing areas may increase (and vice versa in depopulating or deindustrializing areas), while human pressure on biodiversity will affect the marginal benefits derived from restored land (Bezerra et al. 2022, Bezerra et al. 2022). Further, projected natural regeneration in certain regions may make active replanting unnecessary, and we expect to observe spatial variation in those expectations of regrowth (Crouzeilles et al. 2020). Implementation costs may also vary spatially and temporally, depending on changes in soil conditions as well as economic factors (Brancalion et al. 2021). While each source of spatial variation may affect our initial estimates of value, uncertainty in such projections can lead to compounding error across the outcomes we study.

As such, while each of these drivers of variation could be included in future work, here we aim to focus on perhaps the single largest driver of country-scale land use change in the coming years, climate change. Our use of climate projections is based solely on estimates of 2050 conditions, which brings with it inherent limitations: while we may not reach 2050 conditions immediately, the timeline for restoration benefits to be realized makes 2050

conditions relevant for decisions at present. Further, while we lack information for this analysis on changes in years beyond 2050, if trends continue, that will only exacerbate the shifts we observe. There are clear effects of climate change on both the costs and benefits side of the restoration value equation. While climate change undoubtedly affects other aspects of the human-ecological system in ways not explicitly modeled here, we aim to use the two climate-related sources of variation—changes to habitat value and changes in agricultural opportunity costs in Brazil--as a case study to illustrate the general impact projections of the future can have on restoration planning. The restoration benefits estimates are certainly a lower bound, with ample opportunity to apply a more comprehensive measure of biodiversity protection value, and other values beyond habitat protection. Nevertheless, the scale and extent of changes we observe suggest that climate projections ought to be taken into account under a range of circumstances and economic conditions. More fundamentally, our findings provide comprehensive support for the value of targeted active restoration aimed at providing habitat protection to species at risk, in a majority of grid cell areas across one of the world's megadiverse countries.

3. Governance Practices in the Marine Protected Areas of the Sunda Banda and Bird's Head Seascapes of Eastern Indonesia

3.1 Introduction

Around the world, over 775 million people, primarily in Asia and West Africa, depend on marine resources for their livelihood or their protein needs (Selig et al. 2019). Small-scale fisheries form the economic backbone of many communities, often with a long history of managing those resources (Stacey et al. 2021, Hauzer et al. 2013). At the same time, the intensity of commercial fishing operations steadily increased for much of recent history, before flattening out in the late 20th century (Selig et al. 2019). As a result, there is significant ecological pressure on the ecological health of the world's oceans, threatening the human economies that depend on it (Díaz et al. 2019).

At the same time, commercial fishing is not the only threat to the health of the sea. With the twin crises of global biodiversity loss and climate change, there is ever increasing urgency to meeting conservation goals. The UN's sustainable development goal of conserving and sustainably using the world's oceans includes increasing the area under protection, and the Convention on Biological Diversity aimed to increase the fraction of the oceans under protection from 8 to 30 percent (Albers and Ashworth 2022). Marine Protected Areas (MPAs) are a clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (IUCN 2012). A central research topic brought by this expansion is understanding how these MPAs can reach their ecological goals while ensuring continued access to the resources of the communities that live there. While progress has been made towards MPA coverage goals, a longstanding challenge of marine reserves has been the social dimension (Mascia 2004), and significant

work remains to be done towards creating equitably managed PAs (Zafra-Calvo et al. 2019). Understanding the equity and effectiveness questions requires a detailed investigation of how those MPAs operate, in particular how they are governed. This paper aims to understand MPA governance from a holistic perspective, and in doing so, better our understanding of how MPAs can meet their social and ecological goals in tandem.

3.2 Background and literature

3.2.1 Economic impacts and methods

The question of what makes marine protected areas effective is a wide-ranging one. While there is a large body of research on the topic, a few recent meta-analyses summarize the state of our empirical knowledge, on MPAs in particular. On the social side specifically, a recent analysis found half of documented well-being outcomes to be positive, while one third were negative (Ban et al. 2019).

In economics, there is a body of literature on both the effects of MPAs and the decision underlying their creation. From a political economy perspective, the opportunity costs involved with the implementation of an MPA affect the incentives of the actors involved (Smith et al. 2010). One mechanism through which MPAs can have a positive effect is by providing a biological repository for species that are fished elsewhere, mitigating the tragedy of the commons on the high seas (Lynham et al. 2020). A significant challenge for studies that aim to isolate specific ecological or socioeconomic effects of MPAs is disentangling multiple causal mechanisms which interact with both biological and social factors (Reimer and Haynie 2018). Analogously to in fisheries, actors in the MPA face economic incentives across different margins, which interact in ways that make identifying a causal effect challenging (Smith 2012). Spatial interactions also play a major role (Kroetz and Sanchirico 2015).

Beyond isolating specific biological outcomes or socioeconomic modeling, there are few evaluations that aim to evaluate costs and benefits of MPAs in tandem (Albers and Ashworth 2022). Some work, for example, has exploited expansions in existing MPAs to measure the associated benefits through ecological changes, quantifying the measured changes using other economic valuation studies (Stefanski and Shimshack 2016). Another recent study looks at the spatial distribution of benefits, relevant to a small-scale fishery in particular (Grantham et al. 2022). Other recent approaches aim to estimate economic benefits more broadly by estimating ecological changes and monetizing (Davis et al. 2019).

3.2.2 Synergies and governance

One important driver of protected area outcomes is governance. In the context of common pool resources, governance is defined as the set of rules and practices which determine how users access that resource (Sestakova and Plichtova 2019), rules and practices which exist to further well-being and ecological sustainability. An important question in the literature is the extent to which these two outcomes, promoting social benefits and ecological improvements, work in tandem or in conflict. If the former, then we would be more likely to see synergies between the two due to positive effects of each on the other, but on the other hand, if they are in conflict it might present a tradeoff. Some existing studies delve into this question, where Oldekop et al. 2016 found a generally positive relationship, with PAs that find positive socioeconomic outcomes being more likely to report positive conservation outcomes as well. Other studies have found mixed results (Smallhorn-West et al. 2020). Local perceptions of MPAs also have an effect on their success: if MPAs do not support local livelihoods, this is a major obstacle for governance (Bennett and Dearden 2014a), while recent analyses find local support to be strongly associated with perceptions of success (Bennett et al. 2018). In this vein, MPAs that have a greater degree of institutional

diversity (governances structures tailored to a particular context) are often more effective (Jones et al. 2013).

In particular, governance practices are known to be significantly related to the effectiveness of PAs, in meeting both social and ecological goals (Stacey et al. 2021). One known driver of positive ecological outcomes is management capacity for the PA (Gill et al. 2017). On land and in the opposite direction, a recent study found a positive association between organizational capacity and vertebrate abundance (Geldmann et al. 2018).

Further, both structures of governance and management practices have also been linked to synergy outcomes. Oldekop (2016) found that sustainable use plans are more associated with ecological and social benefits in tandem, more so than strict no-take regulations, while Edgar et al. (2014) found the opposite. High participation has been connected to both improved well-being and ecosystem health trends (Ban et al. 2017).

Other studies look at specific features of MPA governance and management. As with terrestrial PAs, capacity is viewed as a major constraint to effectiveness, with lower capacity MPAs being associated with worse ecological outcomes (Gill et al. 2017). This is particularly salient given the expense of enforcing regulations in a large area of ocean. In general, MPAs that are well-enforced, long lasting, large, and isolated are associated with a greater degree of success (Edgar et al. 2014).

More broadly, marine ecosystems have unique features compared to their terrestrial counterparts that affect how MPAs function. First, MPAs are more likely to produce ecological and social spillovers outside of their boundaries, since ocean areas are harder to fence in. Regulations are also often different: a majority of MPAs have sustainable use, rather than no-take regulations (Lubchenco and Grorud-Colvert 2015), due to the

dependence of communities on marine resources, despite the fact that no-take MPAs are shown to be more likely to reach their ecological goals (Sala and Giokoumi 2017). Lastly, local buy-in has been seen as particularly important for the success of MPA. While this reflects the importance of buy-in in terrestrial PAs, MPAs often have a high degree of economic dependence on marine resources, yet face negative perceptions in the local community (Bennett et al. 2019, Bennett and Dearden 2014b).

3.2.3 Theoretical framework

In particular, our understanding of governance in protected areas, including MPAs, is informed by the economic literature on effective common pool resource (CPR) governance, and it is this theoretical framework that guides the approach of this paper. Elinor Ostrom's theory of CPR management enumerates specific components of governance practices that are associated with effective governance, known as Ostrom's design principles (McGinnis and Ostrom 1992). The principles (See Table 6 below) provide a canonical starting place for understanding effective governance practices, and include clearly defined boundaries, graduated sanctions, and other characteristics that can lead to a well-managed resource (Ostrom 1990). However, in recent years, the scholarship has explored some of the complexity behind these interactions. Accounting for some of the complexities of real-world institutions, the institutional analysis and development (IAD) framework is one such widely used approach (McGinnis 2011). However, these social institutions are indelibly linked to biological interactions of the natural world. Building on IAD, the socio-ecological systems (SES) framework, in particular, has emerged as a tool for understanding the specifics of how ecological interactions relate to human social systems (Ostrom 2009, Nagel and Partelow 2022). From this research, specific principles have been identified for MPAs in particular (Di Franco et al. 2016). Since its publication, lines of

critique of Ostrom's framework have developed, including on its limited acknowledgement of state capacity, and the limited role of privatization, in fisheries specifically (Viet Thang 2018). However, these responses have also been criticized as misunderstanding Ostrom's claims (Sarker and Blomquist 2019). Even after these updated frameworks and advances in our understanding, Ostrom's principles remain widely cited as a starting point for understanding social institutions and norms of governance, and play a key role in linking social and ecological systems within the SES framework. This study uses the SES approach and Ostrom's principles to better understand the relationship between specific practices of governance and the socio-ecological system that they interact with, in a critical context for conservation and human development.

Empirically, there is also a literature which aims to quantify Ostrom's framework in practice. A 2010 meta-analysis found that across 91 studies that attempted to evaluate the framework empirically, the design principles remain well-supported, more so with the theoretical advances of the SES and IAD schema. (Cox et al. 2010). More recently, a review highlighted the need for work evaluating new CPR institutions, given their observed fragility and variability, along with work exploring geographic variation in the design principles (Gari et al. 2017). Other studies have applied the design principles to MPAs in particular, finding inconsistent alignment (Tebet et al. 2018).

3.3 Research context

The State University of Papua (UNIPA) has an ongoing project in eastern Indonesia, aimed at monitoring the ecological health of critical coral reef habitats in the region through the creation of marine protected areas, and researching their social and ecological effectiveness. The network of marine protected areas was created by the Government of Indonesia with these twin aims. To implement these goals, a collaboration of conservation

practitioners and policymakers exists between the WWF and the UNIPA in the marine ecosystems of the region, which collects social and ecological data in those MPAs. Since implementation of their joint study in 2010, the study aims to, in their own words, "solve the mystery of MPA performance" (Glew, Mascia, and Pakiding 2012) including studying different components of MPA governance. They implement a coordinated protocol of evaluation methods to better understand MPA effects in the region. In this chapter, I use that dataset to delve into what governance means in this context, exploring the landscape of governance practices across the study area.

3.4 Marine governance in practice

While governance practices and rules can mean many things in different settings, in the seascapes studied, one critical aspect of the MPA zonation is a participatory planning and management process, in which residents of the settlements had the ability to influence how the MPA was instituted (Glew, Mascia, and Pakiding 2012). Many of the settlements have customary arrangements of governance which influenced how this participatory process led to different rules (McLeod, Szuster, and Salm 2009). These customary norms restrict and shape the behavior of community members in accessing marine resources. In eastern Indonesia, two key cultural practices around governance are the practices of Sasi and Adat. From an anthropological study of resource use practices in the region, "(Adat) refers...to the customary norms, rules, interdictions, and injunctions that guide an individual's conduct as a member of the community, and the sanctions and forms of address by which these norms and rules are upheld. Adat also includes the set of local and traditional laws and dispute resolution systems by which society is regulated (Boli et al. 2014)." It is thus a broad term that includes a range of local laws and proscriptions around conduct, and a social mechanism that influences the governance of the MPA through the participatory planning process (Boli

et al. 2014). Sasi refers specifically to rules and prohibitions surrounding marine resource use. The same study continues: "Sasi is the spatial and temporal closure of an area of natural resources in the form of agricultural fields (gardens), forests, coral reefs, and fishing locations." In some settlements of the region, residents describe how practices of enforcement can work. One study describes a process: "The majority of those interviewed in Fafanlap (a settlement) said that if they witnessed someone breaking Sasi, they would notify the head of the village (kepala kampung) while less frequent responses included traditional leaders (Adat leaders), police, army, or the pearl company. Those that identified the police, army, or pearl company were specifically talking about infractions involving the use of destructive fishing techniques." (McLeod, Szuster, and Salm 2009)." These cultural practices shape how social interactions relate to practices of natural resource management. As such, the role of local cultural practices in the participatory planning process would be likely to affect how the MPA is run. While we do not attempt to measure these cultural processes directly, it is hypothesized that variation in these practices across settlements will lead to variation in observed governance practices, where sasi exists. Examining the nature of that variation in governance is a key part of this study.

Further, beyond specific practices such as Sasi and Adat, open questions remain as to the relationship between components of governance. Understanding these relationships and operationalizing their measurement through particular survey questions across a wide study area allows us to better understand the landscape of governance that exists.

3.5 Knowledge gaps and research questions

While many studies exist that explore particular indicators of governance, there are few which explore variation within a protected area, or in this case across a network of PAs with data spanning within each. With the effectiveness of PAs often varying widely within its

area, this is a critical gap. By exploring this level of variance, we aim to better understand why PA impacts may have significant heterogeneity, both across MPAs and within them, particularly for the latter in those MPAs which span a large geographic area. While Ostrom's framework has been operationalized in isolated cases elsewhere (Tebet et al. 2018), here, in contrast, we aim to do so across a wide network, examining the disparities in alignment with good governance practices that exist both within and across MPAs. In addition, we aim to operationalize each of Ostrom's principles, as opposed to a few in isolation.

This approach allows for us to examine the relationship between different components of Ostrom's framework. We aim to understand what patterns of governance exist across the study area, and to identify common practices between and across settlements.

This leads us to our main research questions. First, how can we operationalize Ostrom's indicators through survey instruments? While many studies look at particular components of governance, e.g., indicators which may or may not align with particular design principles, this study aims to explore governance from a holistic perspective using a wide variety of survey questions, while simultaneously being guided by and designing our aggregation process using Ostrom's theoretical framework. I describe the process behind the approach used here, and evaluate its strengths and weaknesses.

Second, we aim to understand the relationship between different components of governance that we observe. Are certain patterns of governance more likely to be observed in tandem? Do governance characteristics cluster geographically? The study explores how governance practices vary across the settlements of the region, aiming to better understand their nature.

3.6 Data sources

The data available includes two monitoring periods at each site: the baseline period, and at least one follow-up. Both measurements happened between 2010 and 2019, depending on the settlement. In the below analysis, we use the baseline data, with the earliest available year in which data exists taken for each settlement. The social monitoring data spans 230 settlements across the Sunda Banda and Bird's Head Seascapes, across 12 MPAs. Data from the Birds Head Seascape was coordinated by the World Wildlife Fund (WWF) US and UNIPA, while data from the Sunda Banda Seascape was coordinated by the WWF Indonesia in tandem (Setyawan et al. 2018). Seascapes are an area of marine ecosystems with a common ecology, in a specific region. Characteristics of each MPA and the settlements within are summarized below (Purwanto et al. 2021):

Table 4: Birds' Head Seascape MPAs

MPA	**Area**	**Number of Settlements**	**Management Authority**
Teluk Mayalibit	49,451	14	Provincial
Teluk Cenderawasih	1,453,000	27	Federal
Kofiau dan Pulau Boo	149,208	8	Provincial

Misool Selatan Timur	346,189	17	Provincial
Kaimana	503,194	15	Provincial

Table 5: Sunda Banda Seascape MPAs

MPA	Area	Management Authority
Selat Pantar	271,262	National
Flores Timur	149,903	National
Kei Kecil	150,000	National
Koon	9,901	National
Yamdena	783,800	National
Sulawesi Tenggara	276,043	National

Two primary tools were used to collect data: Focus Groups (FG) and Key Informant Interviews (KII). The following information on data collection is from the field manual for the study (Glew et al. 2012). In each settlement, a common procedure was used for both sets of surveys. For the FGs, which occurred first, field coordinators (the individuals responsible for conducting the surveys) would meet local officials, and ask them to set up a representative group of individuals for the focus group, with a common set of questions to those individuals, generally focusing on their use and practices surrounding marine resources

in the community. A standardized set of steps for finding those individuals and conducting the interviews was given to each interviewer, and the completeness of each FG was checked.

Following the FG, KII participants were identified in three ways: consultation of residents, selecting FG participants whose responses indicated a further interview would be informative, and asking KII participants to suggest others at the end of their own interview. As such, multiple KIIs were done in each settlement.

Both the FG survey and KII survey instruments included multiple questions directly relevant to specific Ostrom design principles. As such, they form the basis for the way in which Ostrom's Design Principles were operationalized in the study. From the two surveys, questions were selected which were seen to be particularly relevant to that specific design principle. See table 6 below for a summary of the process of going from specific survey instruments to measuring design principles and processing the data for each question, and see Appendix A for a full list of questions used.

3.7 Data processing

The goal of the data analysis process is to take data collected across a number of settlements and identify types of governance that exist between the settlements, what we refer to as clusters. We will then be able to categorize the settlements into each clusters, and examine patterns within each.

In order to aggregate data to the settlement level, data from the survey instrument was cleaned to generate an aggregated mean indicator for each survey question. For each question, the average response was taken across the settlement, for all FGs or KIIs in that settlement. Where no data was available for that settlement on a question (i.e., all of the KIIs or FGs yielded NA for that question), an NA response was assigned to that settlement on

the whole. The end result is a settlement-level average for each individual question on the FG and KII.

3.8 Aggregating data to Ostrom's indicators

The next step in the data processing is to aggregate questions to Ostrom's principles. Alignment in this context means conforming with Ostrom's definition of what constitutes good governance for that particular design principle. For each principle, I explain the relationship between the indicators which comprise it, how it was collected, and how it measures the design principle. The end result is normalized indicators which cover seven of the eight design principles, with one principle broken down into multiple components. The process of aggregating questions for each indicator is summarized in the below table, with the text of questions referenced in Appendix A. Appendix B also includes a list of limitations and robustness checks associated with each governance principle. All indicators are normalized by dividing by the mean of that indicator across all settlements, so that the resulting indicator calculated represents the average value of the indicator within that settlement relative to all settlements. E.g., a value of 1 in accountable monitoring for a particular settlement would mean that particular settlement has average levels of accountable monitoring across all components of the indicator, while a value of .5 or 2 would mean that particular settlement has an accountable monitoring metric that is half or twice as high as the mean across settlements respectively.

Each of the indicators calculated below corresponds with a single Ostrom's Design Principle, with the exception of accountable monitoring. Accountable monitoring is a process which requires that conditions of the resource be observed by the community, and that those observers have accountability measures in place which incentivize them to effectively enforce the rules established by the resource users (McGinnis and Ostrom 1992).

As such, the basic model of this process requires three independent steps: monitoring, penalties for failure to monitor, and enforcement, and accountability in the monitoring itself. As such, I split the indicator into three separate principles: active monitoring, compliance and enforcement, and accountability for monitors. All indicators (each bullet point in the final two columns) are normalized and equally weighted in the calculation of each indicator.

3.9 Governance principles aggregation

Table 6: Governance Principle Aggregation

Governance Principle	Meaning (McGinnis and Ostrom 1992)	Components	Scale	Survey questions
Clearly Defined Boundaries	The question of which households or individuals are able to access which resources must be clearly delineated, along with the definition of the resource itself which can be harvested.	• Number of ways in which boundaries were communicated, of 10 possible	• Ordinal 0-10	• FG-11
Participation in Decision Making	Individuals affected by the rules of the common pool resource must be able to	• Whether each participant was involved in MPA decision making	• Binary • Binary	• FG-14, FG-15

	participate in the modification of these operational rules	processes Whether each participant was involved in rule making processes		• FG-16, FG-17
Active Monitoring	There exists a system to observe the conditions of both the common pool resource and the participants who harvest the resource	• Frequency of monitoring of condition of marine resources (ecological) • Frequency of monitoring of well-being of the resource users (social) • Frequency of monitoring of the compliance of resource users	• Ordinal 0-5 • Ordinal 0-5 • Ordinal 1-5	• FG-18 • FG-19 • FG-20
Compliance and enforcement	There exists a system of enforcement mechanisms to promote compliance with resource rules.	• Frequency of penalties being received	• Ordinal 1-5	• KII-18

		• Frequency of penalties being handed out	• Ordinal 1-5	• FG-22
Accountability for Monitors (Sanctioning)	The system of monitoring and observation of the behavior of resource users is subject to a set of sanctions to encourage compliance with the monitoring itself.	• Number of sanctions for failure to monitor ecological conditions • Number of sanctions for failure to monitor social conditions • Number of sanctions for failure to monitor compliance • Number of sanctions for failing to penalize rulebreakers	• Cardinal 0-8 • Cardinal 0-8 • Cardinal 0-8 • Cardinal 0-8	• KII-23 • KII-22 • KII-21 • FG-17 (a-g)

Congruence with Local Conditions	Rules which govern resource access and use are determined by local conditions.	• The extent to which MPA rules change based on social conditions • The extent to which MPA rules change based on ecological conditions • The fraction of user groups which have specific rules	• Ordinal 0-5 • Ordinal 0-5 • Percentage	• KII-8 • KII-7 • KII-6
Support of Higher Governments	The ability of resource users to define their practices for resource use is not subject to challenge by external government authorities	• Ability of higher government entities to support resource rights for different members of the community	• Ordinal 0-5	• KII-14
Nested Governance	The appropriation, provision, monitoring, enforcement, conflict resolution, and	• Binary: if multiple zones exist, with different organizations in each, and if and only if so, whether the different	• Binary	• KII-2a-d

	governance activities are all nested in multiple layers of institutions.	organizations coordinate across zones (See Appendix C)		
Graduated Sanctions	The resources users who break rules of resource access and used are subject to multiple levels of penalties for breaking those rules.	• Mean of total number of sanctions used (FG), total number of different sanctions used if levels exist (KII) • Number of relevant factors that determine penalties	• Cardinal 0-6 • Cardinal -3 to 4	• FG 21a-f, KII-20a-g • KII-19

3.10 Approach to quantifying Ostrom's indicators

A central aim of this methodology is to approach governance holistically, while at the same time being able to describe different aspects of governance with specificity tailored to Ostrom's theory of common pool resource management. As discussed above, many of the indicators have multiple components, all of which are necessary in tandem for good governance practices to be implemented. With others, multiple survey questions deal with different ways of conceptualizing that indicator, and thus provide further information than a single question which attempts to quantify the indicator in its entirety. The methodology of normalizing and weighting aims to provide an imperfect metric of how well aligned with Ostrom's principles governance practices are in a particular settlement, relative to other settlements. Since all metrics are normalized, the metrics are all relative in nature, across a large area in which governance practices vary significantly across all indicators.

However, in those that do align, in many cases we see a high degree of congruence among the components of the indicators, while other measure more distinct components (See Appendix B). On the whole, this approach provides one way to both operationalize and measure Ostrom's indicators of good common pool resource management. Governance is a necessarily multidimensional concept, and as such, can be measured in many different ways, requiring a variety of specific survey questions. This then requires many decisions to be made in how to interpret those survey results, especially when attempting to quantify aspects of governance. The large extent of the survey area, including many different social, geographic, ethnic, and political contexts, would be expected to contain significant variation in governance, and thus requires a nuanced approach. Further, the imperfections of field

collected data, with missing data and many surveys with no response given, requires that we use all available information.

3.11 Patterns in indicators

I first examine the overall distribution of governance indicators across the settlements in the study. At a glance, the indicators exhibit different distributions (Figure 11). This may be indicative of differing factors which are driving the variation across the study area. A few examples of different distributions are given below. As above, the x-axis represents the normalized value of the indicator, relative to the mean across settlements, such that the mean is 0 for all principles.

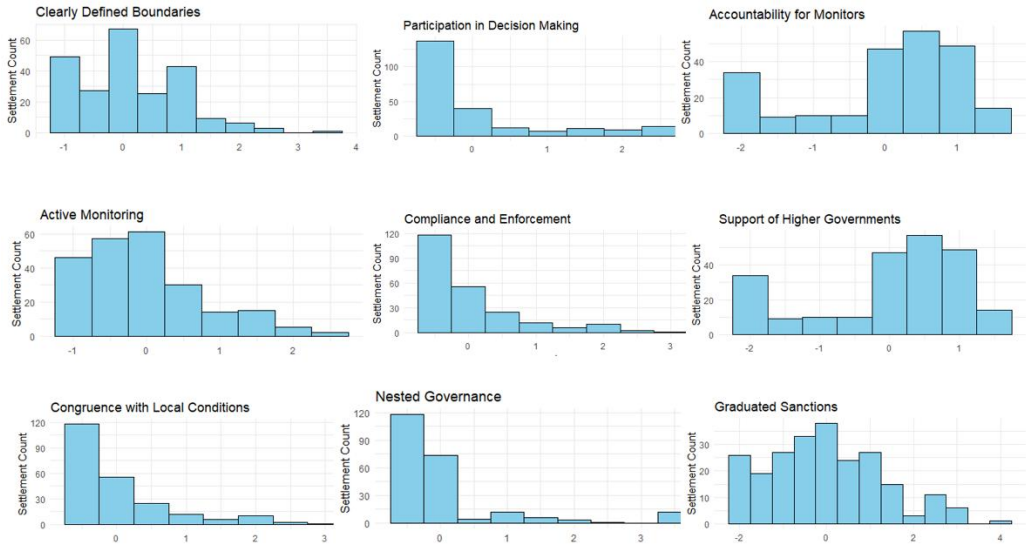
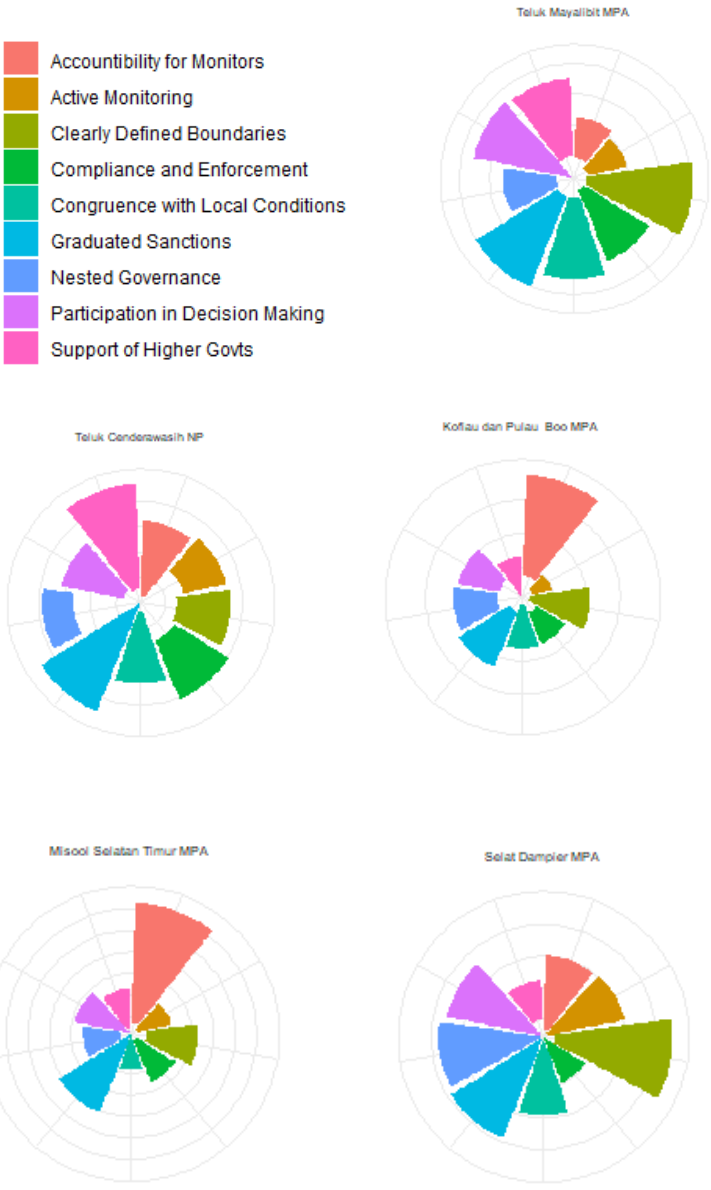


Figure 11: Normalized Indicators

While nested governance is driven by a few outliers, with the remainder clustered around 0, active monitoring has a more uniform spread. Part of these differing distributions may be caused by regression to the mean: indicators with more than one component are likely to have a smaller variance. For those distributions with a strong right skew, such as

congruence, compliance and enforcement, participation in decision making, and nested governance, the majority of settlements are below the mean, due to a few settlements being particularly high. Meanwhile, accountability for monitors and support of higher government exhibit a left skew, due to the presence of outliers on the low end. Due the process of normalization, we cannot make any absolute conclusions about the degree of good governance in settlements, only relative to other settlements. While this is a disadvantage in interpreting the results of each indicator, for the process of aggregation, given the fact that we are using many different questions with wording that may be interpreted differently in absolute terms, the process of normalization allows us to more readily aggregate to the level of indicators.

I next look at how governance indicators vary by MPA. Rose diagrams for each MPA are shown below (Figure 12). Distance from the center represents the normalized value of each indicator, relative to the mean. While scales are different due to different levels of variation, each is centered about the normalized mean such that the middle tick mark represents the average settlement value for that indicator for all MPAs. The opposite aggregation, with variation across MPAs for each indicator, is shown in Appendix D.


Accountibility for Monitors
Active Monitoring
Clearly Defined Boundaries
Compliance and Enforcement
Congruence with Local Conditions
Graduated Sanctions
Nested Governance
Participation in Decision Making
Support of Higher Govts
Teluk Mayalibit MPA
Teluk Cenderawasih NP
Kofiau dan Pulau Boo MPA
Misool Selatan Timur MPA
Selat Dampier MPA

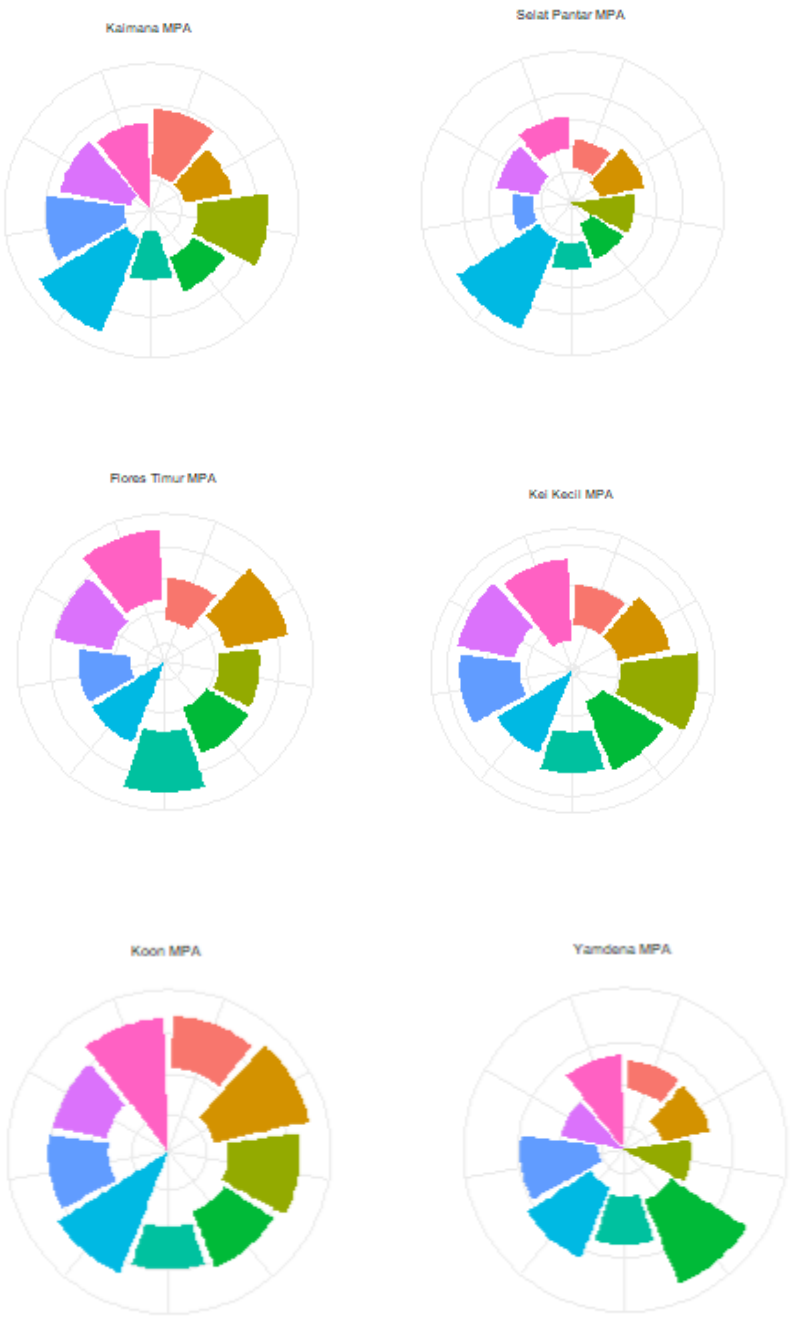


Figure 12: Governance Indicator Distribution by MPA

As shown, there are clear differences in the patterns of indicators across MPAs. The Misool Selatan Timur MPA, for example, has much higher levels of accountability for

monitors, while the Selat Pantar MPA has higher indicators of nested governance, both relative to the overall averages in the study area. Others, such as the Kei Kecil MPA, have settlements which are more uniform across indicators.

In general, a few specific patterns in MPAs stand out, as shown in Figure 13. First are four MPAs which generally have higher than average governance indicators across all metrics: Teluk Mayabit, Delat Pantar, Selat Danpier, and Misool Selatan Timur. On the other side, however, two MPAs have low levels of alignment with governance principles across all principles, Teluk Cenderiwasih and Koon, while Kei Kecil has low alignment, with the exception of "Clearly Defined Boundaries" and "Participation in Decision Making." In both directions, the fact that within a specific geographic area, there is uniformity in the directionality of indicators provides support for the hypothesis that there is a uniform measure of good governance, albeit with heterogeneity. The remainder of MPAs have more heterogeneity, indicating that different principles are measuring different components of governance.

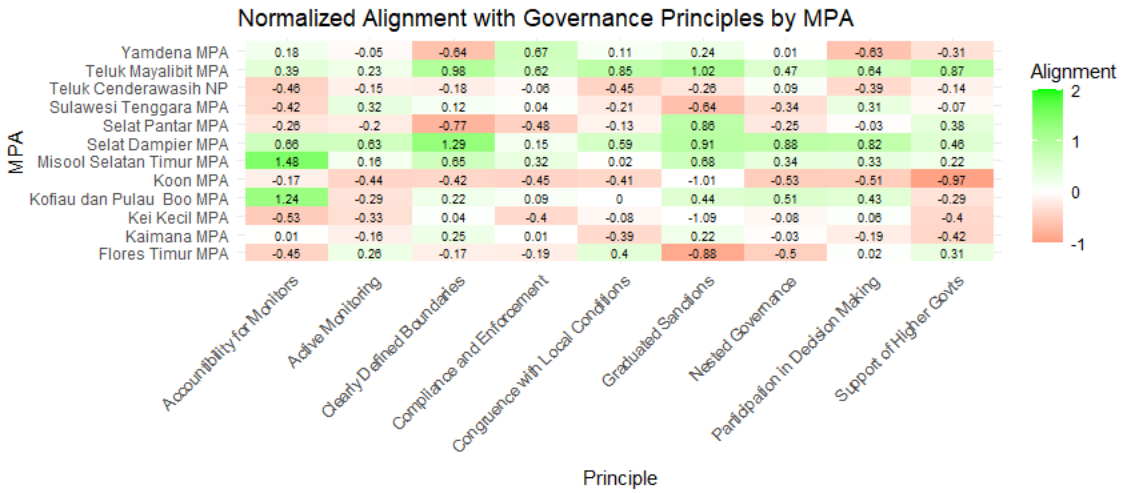


Figure 13: Normalized Alignment with Governance Principles by MPA

3.12 Cluster analysis

Beyond the indicators themselves, a central part of the research question is the extent to which there are common governance types, where indicator values for each principle is similar across settlements, even in different MPAs. E.g., we may observe certain indicators being more likely to be observed in tandem, and others less so. The aggregation by MPA (in Figure 12) is purely geographical, whereas here I take all the settlements as independent, and ask what patterns may arise when we look at governance using the framework used to operationalize design principles. As such, I conduct a form of cluster analysis, Latent Profile Analysis (LPA), to identify which settlements fall in different governance categories. I then examine the resulting patterns between indicators of each principle in what determines governance, aiming to find commonalities across the range of governance indicators (Bauer et al. 2022). Due to the process of normalization and equal weighting, each governance principle has equal weight in the LPA, and each component is equally weighted in its respective principle, regardless of the underlying distribution.

The goal in this analysis is to identify different types of governance that are observed across all the settlements in each of the MPAs. The LPA process assumes that there are some categories of settlements which observe common trends in governance practices, attributes of which are observed in each of the governance indicators (Bauer et al. 2022). This is a form of cluster analysis, aiming to find common links across a range of variables. I do this inductively, not assuming any relationship across the indicators, but rather that each of the nine are independent. I take the cleaned and aggregated governance indicators from each settlement, as described in section (i). There are various missing data points; the average settlement is missing 4 of the 24 indicators. I first choose the "baseline" year from each of

the settlements, the earliest year. In order to complete the data and as is common practice in this type of cluster analysis (Bauer et al. 2022), I then impute missing values, using a k-Nearest neighbor algorithm in the biokNN package to assign values from the most similar settlement (Cubillos et al. 2022). This assumes that the NAs are all Missing At Random, and that links between the clusters themselves are Missing Completely At Random. The imputation processes uses existing data to make predictions about what we are likely to observe in the missing data, making the most of the information we have available. While this may be subject to bias if all missing data is of a particular governance type, the fact that the average settlement is only missing 4 of 24 governance indicators is likely to limit the extent of potential bias. As in the aggregation process, I make the assumption that for the ordinal data (e.g., 1-5 scales), the distance between the levels is homogeneous, such as that the difference between a 1 and 2 on the scale is the same as a difference between a 4 and a 5, a necessary assumption given the structure of the questions (See robustness checks and question text in appendices). I then run a series of cluster models to group the settlements. I take the model with the minimum Bayes Information Criterion.

The initial cluster analysis settles around 4 different governance types, using the "VVE" model type. "VVE" allows for different sized and shaped clusters, with a common elliptical covariance structure, and as such is the most flexible form of the model, based on our lack of assumptions about the structure of the data (Bauer et al. 2022). Below in Figure 14 I show the characteristics of each of the clusters, summarized by the mean values in each governance type for each of Ostrom's indicators.

Each of the clusters has clearly distinct characteristics, as shown in Figure 14. Cluster 1 is the "good governance" type: clearly higher normalized values of the indicators are

observed across all indicators. The fourth is the inverse, with uniformly less good values of the indicators across all metrics. The second and third are more heterogeneous, with good governance in some distinct areas. The second is generally similar in pattern to the fourth, but with a distinct difference in support of higher governance. The third exhibits higher scores in graduated sanctions and compliance and enforcement, with middling scores across the others of Ostrom's indicators. Of the 230 settlements, roughly one third (74) fall in the first "good governance" cluster. Nearly half (108) fall into either the "poor governance" fourth cluster or "poor governance except for higher government support" second cluster, while the remaining 20% fall into the "heterogeneous" third cluster. Colors indicate the scale of values, with a reference value of 0 such that green values are greater than average, yellow are close to the average, and red values are lower than average.

Clusters	Clearly Defined Boundaries	Participation in Decision Making	Active Monitoring	Compliance and Enforcement	Accountibility for Monitors	Congruence with Local Conditions	Support of Higher Govts	Nested Governance	Graduated Sanctions
Good Governance	0.58	0.94	0.51	0.42	0.58	0.53	0.23	0.64	0.52
Poor Governance, High Support	-0.39	-0.45	-0.37	-0.38	-0.40	-0.20	0.45	-0.30	-0.80
Heterogeneous	-0.02	-0.42	0.01	0.21	-0.05	-0.22	0.38	-0.22	0.92
Poor Governance	-0.38	-0.47	-0.32	-0.38	-0.34	-0.38	-1.56	-0.41	-0.72

Figure 14: Governance Type Mean Characteristics

3.13 Patterns in governance types among MPAs

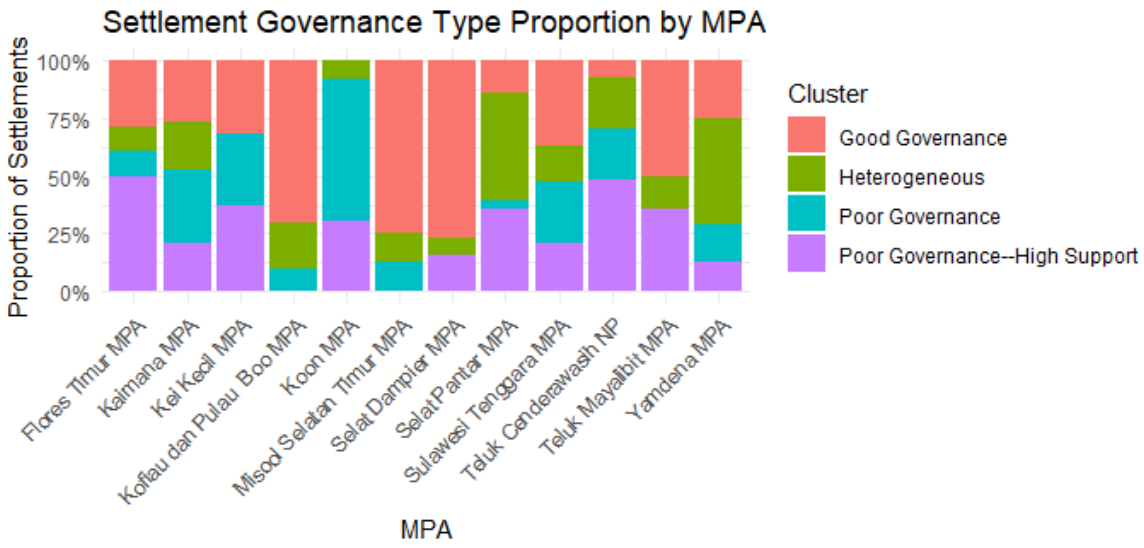


Figure 15: Governance Type Distribution by MPA

In examining the distribution of clusters across MPAs in Figure 15, no clear pattern is apparent for the majority of MPAs. However, three of the MPAs, Kofiau dan Pulau Boo, Misool Selatan Timur, and Selat Dampier, all have a much higher proportion of settlements in the "good governance" cluster. These MPAs span both the Birds Head and Sunda Banda Seascapes, and a range of sizes (See Tables 4 and 5). There may be some geographic or political characteristics of these MPAs which have affected the governance metrics, highlighting the need for further investigation. On the other side, while most MPAs have a heterogeneous distribution across the other three governance types, two stand out as having a higher proportion in the two "poor governance" clusters: Flores Timur MPA and Teluk Cenderawasih, each of which has fewer in the heterogeneous and "good governance" clusters. Lastly, Koon MPA stands out as having a particularly bimodal distribution of governance types, with nearly all settlements in the MPA falling in the good or poor governance type categories.

3.14 Conclusions

First, the report provides one approach for operationalizing Ostrom's design principles using survey data. The approach is robust to a number of potential changes and ambiguities, illustrating that even with the inherent challenges of survey questions with a number of missing responses and the multifaceted nature of the design principles themselves, the data available can demonstrate one way to measure alignment with those principles of good governance, and aggregate to a measure of good governance in a community, which is robust across a variety of assumptions in the aggregation process (See Appendix B). The high degree of congruence we observe across questions within the indicators, as well as between indicators within settlements indicates that the questions do provide some measure of a cohesive whole for each principle, i.e., that they are measuring something similar. Nevertheless, the approach does have its downsides in that we do not measure all design principles completely (Conflict Resolution, for example, was the sole principle left out entirely), being limited by existing questions present in the survey data.

We next look at variation in governance itself across the MPAs. The specific MPAs in which we observe a greater degree of alignment with good governance across the board provide evidence that there are some characteristics of those MPAs that are leading to better aligned governance practices, spanning different settlements in those MPAs, with the reverse being true for MPAs with comparatively and consistently poor governance. In those MPAs in which we observe significant heterogeneity across all indicators, further research is necessary to determine the causes of that variation, but no clear patterns are observed.

Lastly, the results of the cluster analysis illustrate that there are indeed patterns of relatively good and poor governance, with clear groups arising which exhibit uniformly high

or low levels of alignment with Ostrom's principles. It is important to emphasize that all metrics are normalized, meaning that we are measuring alignment with good governance practices in a relative rather than absolute sense. As such, it may be the case that from the survey questions governance indicators are well aligned or poorly aligned with good governance practices on the whole, but that question is outside the scope of this study. Importantly, the clusters we find span many MPAs, indicating that there is a significant amount of variation within MPAs between settlements, moderating the effectiveness of that protection. At the same time, the fact that some MPAs do see greater concentrations of the good or poor governance clusters demonstrates that there are indeed geographic patterns in which good governance is likely to be observed, but that the MPAs themselves are not the sole determining factor. In the remaining clusters, the difference in the indicator of support for higher governments stands out, indicating that particular indicator may represent a measure distinct from the cohesive concept of good governance across other indicators, which makes sense due to the presence of outside agencies which may affect other governance practices. This too, highlights the need for further research on the underlying drivers of this qualitative difference in that specific indicator, as well as the relationships between the variation we identify as a moderator of PA effectiveness.

4. A Long-run Evaluation of the Protected Area Network in Brazil's Mangrove Forests

4.1 Introduction

As the most widely used conservation tool worldwide, protected areas (PAs) are of great interest for policymakers and local populations alike. Existing evaluations of PA effectiveness leave many open questions and mixed results, highlighting the need for

context-specific research (Geldmann et al. 2018). Further, recent updates to land cover datasets with a much greater level of detail (Diniz et al. 2019, MapBiomas 2024) provide an opportunity to reevaluate their effectiveness, as do innovations in difference-in-differences estimation techniques (de Chaisemartin and d'Haultfoeuille 2023).

In particular, evaluating PA impacts over a long time span in a particular ecosystem provides a greater opportunity to investigate their overall effects, which may not be present over a short time period. The PA network in the mangrove forests of Brazil offer such an opportunity. Mangrove forests are a critical biome located along tropical coastlines around the world, occupying the estuarine space between marine and terrestrial landscapes, resulting in high levels of biodiversity. However, mangrove forest loss is an ecological crisis, contributing to biodiversity loss and exacerbating climate change (IUCN 2017). With high levels of carbon sequestration (Siikamäki et al. 2012), 10 times more carbon per acre than terrestrial forests (IUCN 2017), mangrove protection is a major conservation priority (Polidoro et al. 2010). Further, mangroves face disproportionate risk on two fronts (Luther and Greenberg 2009): they are home to significantly higher than average levels of species found nowhere else, as well as some of the greatest levels of human-driven land use pressure of any biome.

As home to the second largest mangrove area in the world (Hamilton and Casey 2016), the effectiveness of Brazil's network of coastal PAs is important for the future of global mangrove health. Further, the majority (75%) of mangrove forest loss in contexts outside Brazil has been found to occur outside of PAs, highlighting the need to understand the extent to which PAs are able to provide additional protection within their boundaries and to offset losses outside the PA system (López-Angarita 2018).

The network of PAs in Brazil is among the world's largest, having been created piecemeal over the past century (ICMBio 2023). Since 1985, when data are available, there is an increase in mangrove forest cover within PAs, shown in the data section below. The central question of this paper is whether this increase is directly attributable to the increase in PAs. I use remotely sensed high resolution (30m) annual dataset on mangrove forest cover (generated in Diniz et al. 2019, as part of MapBiomas 2024) and a difference-in-differences estimation technique with staggered implementation (Gardner et al. 2024) to estimate the causal impact of Brazil's mangrove PAs during 1988 to 2020 on mangrove forest cover. I investigate variation in estimated effects by PA governance regime (federal, state, and municipal), timing of PA creation, and PA category (wildlife reserve or recreation permitted).

Much recent work on PA effectiveness exists elsewhere in Brazil, including some causal inference studies (Goncalves-Souza et al. 2021, Pfaff et al. 2015, Herrera et al. 2019). Research suggests that PA impacts in Brazil differ by governance regime (N. Alves-Pinto et al. 2022), but existing studies that focus on mangroves specifically do not take a causal inference approach (Almeida et al. 2016). The literature on the effects of governance on PA effects is wide-ranging, but open questions remain on the effect of more stringent governance and greater capacity (e.g., Gill et al. 2017, Muñoz Brenes et al. 2017). Meanwhile, the few causal inference studies of mangroves around the world do not examine such a long time period of impacts at high temporal resolution, instead aggregating to larger land areas and/or two time periods (Miteva et al. 2015, López-Angarita et al. 2018, Liao et al. 2019, He et al. 2023). Higher temporal resolution has the potential to illustrate the specific recovery path for reforestation in particular, while long study periods are often necessary to observe the regrowth practitioners aim for (Brancalion et al. 2019).

Thus, estimating the time-specific long-term impacts of PAs on mangrove forests using modern causal inference procedures remains an open question, and of significant importance for conservation. Examining PAs in the mangrove forests of Brazil over the past four decades, I aim to understand how their creation has affected the recovery and extent of Brazil's mangrove forests, and in particular, how their observed effects have varied, spatially, temporally, and by type of PA.

4.2 Methods

4.2.1 Estimation strategy

My general identification strategy is a difference-in-differences (DiD) design, with staggered implementation over the study period (1985-2021). I classify one-kilometer grid cells across the study area in each year as either treated (protected) or untreated and observe changes in mangrove forest cover within each grid cell as an outcome, examining the data by reforestation events (positive changes) and deforestation events (negative changes). I am interested in estimating treatment effects by year before or after PA creation. These year-by-year effects would be particularly relevant if there were to be an effect of reforestation which accumulated over time, or a time-varying effect of avoided deforestation.

I follow the model specified in Gardner et al. (2024) for staggered treatment implementation, chosen for its flexibility and similarity to longstanding two-way fixed effects (TWFE) estimators for DiD estimation procedures. This approach is in the category of new DiD estimators which avoid bias associated with group-time negative weights, by not relying on homogeneous treatment effects and instead estimating cohort-time effects (de Chaisemartin and D'Haultfoeuille 2023). The basic outcome $y_{i,g,t}$ is mangrove forest cover in grid cell *i*, PA cohort *g*, and year *t*. For example, a grid cell that contains a PA created in

2002 will be in cohort *g* for all years *t*. In all equations, each grid cell is weighted by its mangrove potential, as calculated in Data below. I further define the estimated treatment effect $\tau_{g,t}$ for each cohort *g* in year *t* relative to treatment, μ_i as grid cell-specific effects, η_t as time-specific effects, treatment status $P_{g,t}$ as indicating whether a point is protected in each cohort g and year t, and $e_{i,g,t}$ as the error term for each data point i, cohort g, and year t. I thus aim to estimate $y_{i,g,t}$ as follows:

$$y_{i,g,t} = \mu_i + \eta_t + \tau_{g,t} P_{g,t} + e_{i,g,t}$$

The parameters in $\tau_{g,t}$ represent the group-time average treatment effects. To yield overall event-time effects, we aggregate effects across cohorts and years (Gardner et al. 2024). The central modification from a traditional TWFE model is the ability of treatment effects to vary by time *t* and the treatment cohort *g*, here representing PA cohort, which allows us to estimate different effects for different yearly cohorts of protection (Butts and Gardner 2022). The treatment effects are then aggregated into a weighted average for overall cohort treatment effects, reported in the results section below.

Estimating this model directly results in traditional problems associated with TWFE estimators, namely the possibility of negative weights on certain subgroups (Callaway and Sant'Anna 2021). Thus, Gardner outlines a two-step procedure to estimate each cohort-time treatment effect, which I follow here, using the did2s package in R (Butts and Gardner 2022). The first step is to estimate a first stage model using a subsample of both untreated and non-yet-treated observations, with the following specification:

$$y_{i,g,t} = \mu_g + \eta_t + e_{i,g,t}$$

The goal is to use this first stage to measure the specific effects of both time and group. For each observed outcome, the procedure uses the estimated group and time effects to construct an adjusted outcome as follows, where the hat represents the effect estimated in the sample using the model above, and the tilde represents the adjusted value.

$$\tilde{y}_{i,g,t} = y_{i,g,t} - \widehat{\mu_g} - \hat{\eta}_t$$

The second stage is then to empirically regress adjusted outcomes on treatment status in the full sample, estimating treatment effects independent of cohort and time effects. Where each outcome *y* is as defined above, the second stage of the model is thus as follows, where *X* is a vector of *n* covariates, specified below:

$$\tilde{y}_{i,g,t} = \tau_{g,t} P_{i,g,t} + \beta_n X_{i,t} + \tilde{e}_{i,g,t}$$

I then re-estimate the models for aggregate outcomes across four-year intervals, to aid in interpretation of results and to allow for the construction of a more balanced sample. Averages are taken in the intervals 1989-1992, 1993-1996…2016-2020, corresponding with Brazilian federal government election cycles (Koleros et al. 2024). Standard errors used are clustered by grid cell, with a comparison given in the results with clustering by PA.

4.2.1 Hypotheses

Given the varied literature on the topic, initial hypotheses are mixed. With the aims of this particular PA network, I would expect to see an overall positive effect of PAs on forest cover (Goncalves-Souza et al. 2021). I investigate different effects across two categories: management level (federal, state, or local), and strictness of protection. I would expect to see different effects across each of these, in line with other papers on evaluating PA effects on mangroves (Miteva et al. 2015, Almeida et al. 2016). However, heterogeneities

across governance level may be different in the Brazilian federal system, in line with PA evaluations in the Brazilian Amazon (Pfaff et al. 2015). I do not have a unidirectional hypothesis as to the effects of federal PAs, given ambiguities identified in the literature which find variation across protection categories elsewhere in Brazil (Alves-Pinto et al. 2021). Other work which finds sustainable use PAs having a stronger effect, highlights this ambiguity (Pfaff et al. 2014), but sustainable use PAs are not included in this study. Further, siting considerations that cause PAs to be placed in areas that are likely to undergo reforestation or reduced deforestation regardless of PA creation may moderate this effect of governance level, indicating that there may be different effects broadly with our quasi-experimental approach when compared to the overall change over time (Joppa and Pfaff 2010, Ribas et al. 2020).

4.2 Data

The outcome variables I am working with are defined by mangrove forest cover. MapBiomas has a series of annual forest cover maps at 30-meter resolution spanning 1985-2021 for Brazil. Each is a raster covering all of Brazil. The raster layers are categorized into 45 land use codes, spanning anthropogenic and natural land use and cover categories. In the wetlands category, one land use code is mangrove forest, a layer generated by the machine learning algorithm processing of satellite data in Diniz et al. (2019).

From the land cover data, I define two sets of polygons, delineating the treated and untreated areas. To be included in either, an area must have the potential to host a mangrove forest. As such, I take the union of any raster cell which has mangrove forest cover in any time period in the data, and mangrove potential polygons published in Worthington (2019). The resulting union covers the extent of possible mangrove habitat along Brazil's coast. I

then remove areas with PAs established prior to 1985 to generate the study area. While the lack of satellite data prior to 1985 does limit the potential scope of the study, recently created PA areas are more likely to be relevant to present-day questions of additionality, highlighting the significance of the effects identified.

To measure the outcome variable, I reclassify the raster layer into two categories for each year, with mangrove forest cover as 1, and all other land types being 0. I thus have a measure for each pixel (at 30m resolution) at each point in time, defining the variable $Y_{i,t}$ as being equal to 1 if mangrove forest cover is observed in year t, and 0 if it is not. I take the extent of presently existing PAs from the data portal of ICMBio, Brazil's federal conservation agency (ICMBio 2023). Each PA has an associated shapefile, including its year of initiation (when the protection goes into effect), level of governance (federal, state, or municipal), and level of protection, within IUCN-standardized tiers. All mangrove PAs included are at least as strict as IUCN category III, which corresponds to national parks in both the US and Brazil, allowing for recreation but no harvesting of resources. The dataset also includes PAs in IUCN categories I and II, which are more strict, allowing no recreation or limited recreation respectively. I supplement these data with PADDDtracker, a database of downgraded, degazetted, or downsized PAs. One downgraded PA was removed from the sample to focus on measuring effects of presently existing PAs (Conservation International and World Wildlife Fund 2024). This results in a variable identifying protection for each grid cell i in time t, defining the variable $P_{i,t}$ as equal to 1 if it is under protection in year t and 0 if it is not.

The treated area is then defined as the intersection of all PAs created in the study period and the study area, while the untreated area is all remaining study area. Within both

areas, I create a grid of 1km square cells, which cover all possible mangrove area. I aggregate to 1km resolution, to remain within the bounds of computing capability. In contrast to a sampling-based approach, such as the random points method used in He et al. 2023, this approach enables me to retain all available information for each PA, which becomes important given the small sample size in each cohort-year. Within each grid cell and for each year, I sum the area of mangroves present. For each grid cell overall, I also calculate the sum of the extent of protected area within that grid cell and the extent of mangrove potential. The extent of protected area is used as a threshold to classify a grid cell as treated or not: over 50% coverage for a grid cell results in the grid cell being classified as protected. While it is possible that the protected area within a grid cell may not intersect with the mangrove potential area completely, it is rare that a PA does not contain some mangroves, particularly so given the relatively small size of grid cells. Since PAs are generally much larger than grid cells, the vast majority (>95%) of cells have either 0% or 100% PA coverage. The mangrove potential in each grid cell, meanwhile, is used to weight each grid cell in each regression specification.

An example area of coastline is shown in Figure 16, with the grid cells used shown as magenta squares, and treated area (PAs) shown in green.

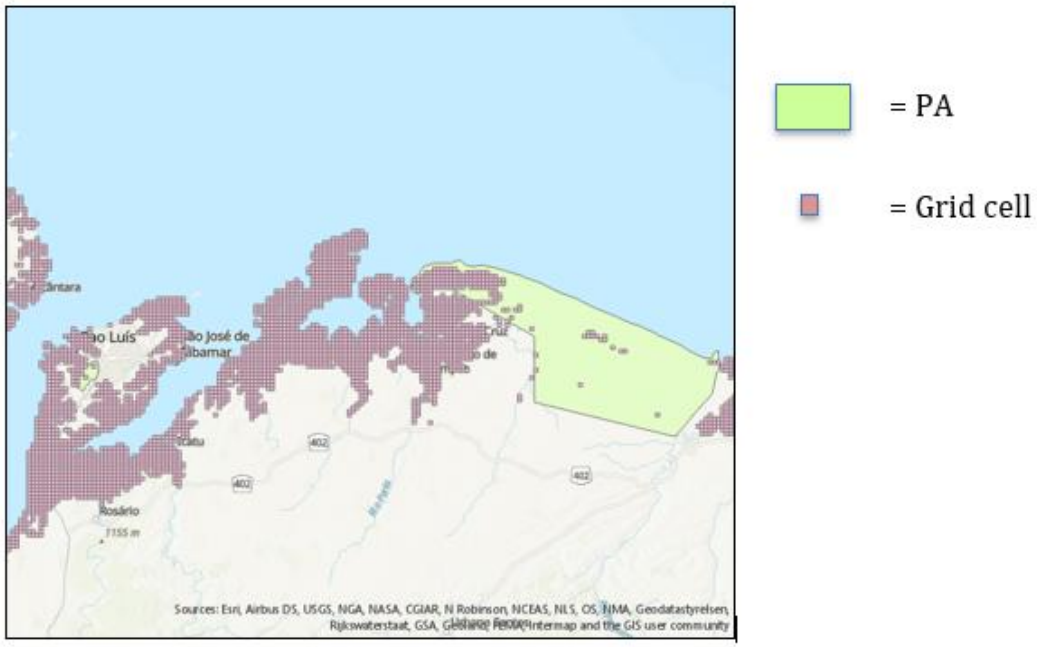


Figure 16: Grid Cell Polygons

After taking the sample of treated and control areas, I am left with ~1.25M observations in total, 37,840 grid cells for each year, for the overall outcome variable of forest cover.

4.2.1 Trends

I first examine the trend of mangroves in the treated group, shown in Figure 17, in contrast to overall changes in Figure 18. I trim the data to years 1988 to 2020, due to measurement error in years 1987 and 2021. In treated areas, mangroves undergo a noticeable recovery over the study period from the late 1980s until approximately 2008, before undergoing a slight decline. It is an open question, however, the extent to which this increase is in part due to the increase in protected area coverage. In contrast, the trend in overall mangrove forest cover in Figure 18 shows a slower and more steady increase over the study period: mangroves outside of PAs did not undergo the same degree of recovery. Because a

strong majority of mangrove area (95.3%) is untreated, the trend for the untreated group is nearly identical to the overall trend in Figure 18.

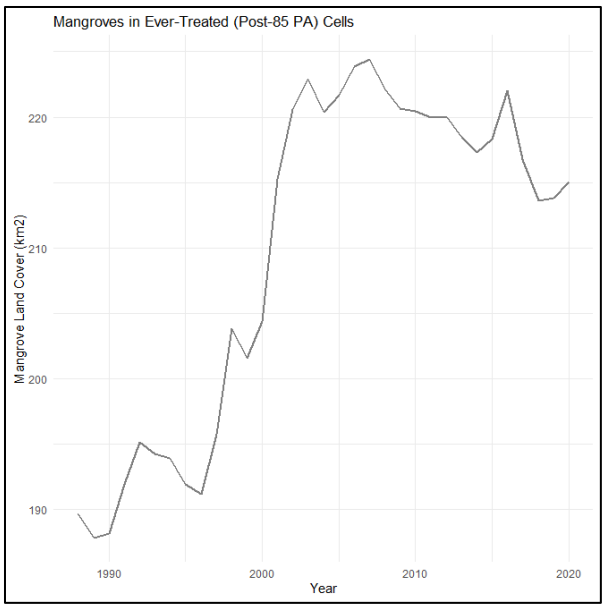


Figure 17: Mangrove Forest Cover Trend: Treated Area

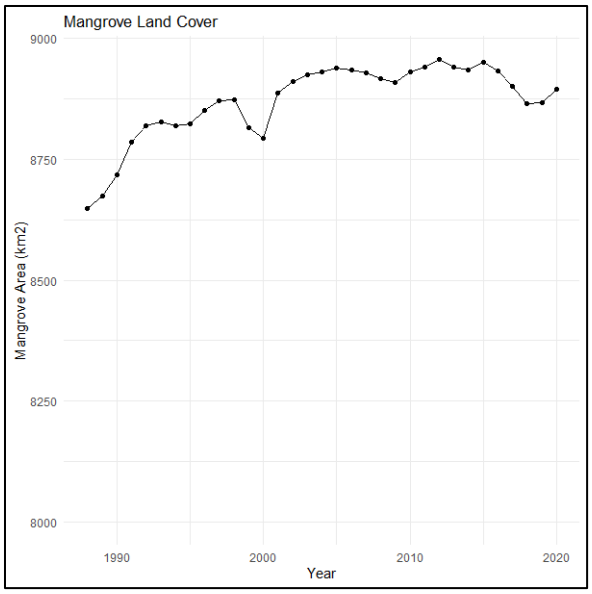


Figure 18: Mangrove Forest Cover Trend: Entire Mangrove Area

The central question of this paper is whether the observed increase in Figure 17 is can be causally attributed to increasing protected area coverage. I show in Figure 19 the growth in protected area extent over the study period. PA coverage increases substantially, with all but three years (1995, 1999, and 2019) seeing the creation of at least one PA.

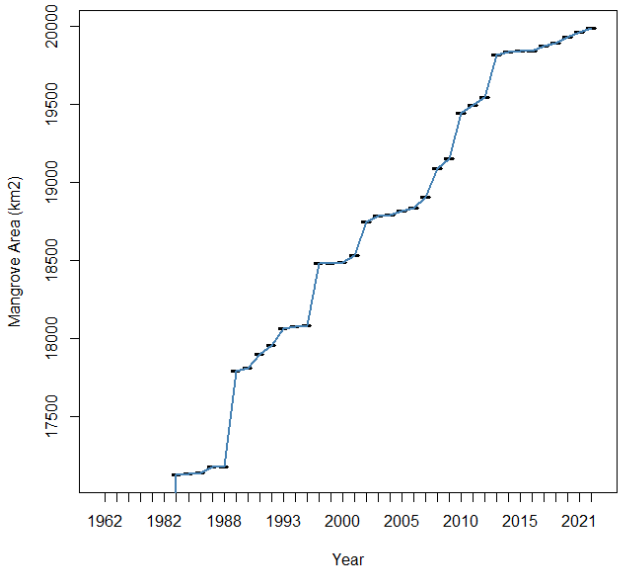


Figure 19: Protected Area Trend: Entire Mangrove Area

I also examine the changes observed by deforestation and reforestation events. While there is significant fluctuation in each year (~3% on average) relative to the net change, the length of the study period and increased change in treated areas shown in Figure 20 means that the small average change produces the overall trend above from 1988-2020. In each grid cell, I find the net mangrove change compared to the prior year, and sum these grid cell-level net changes to the entire sample. Thus, in years where the green line exceeds the red line in Figure 20, the net mangrove forest cover change in the full sample is positive, and vice versa.

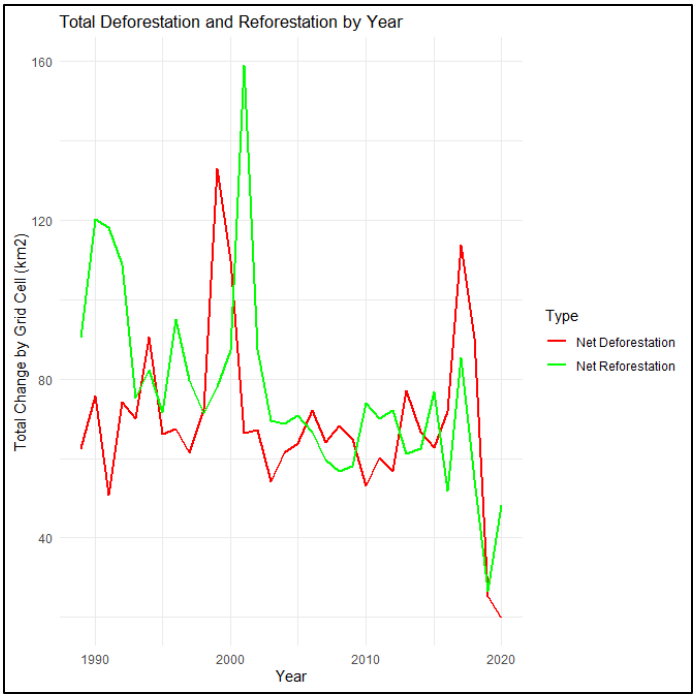


Figure 20: Annual Mangrove Forest Cover Change

4.2.2 Covariates

For each grid cell in the sample, I develop covariates that may affect mangrove regrowth. I summarize the covariates in Table 7, along with the hypothesized mechanism for each. Covariates are taken as historical data where available, as seen in Table 4.

Table 7: Summary of Covariates

Covariate	Data source	Mechanism
GDP per capita (1km)	Global Gridded GDP 1990-2015, in constant 2011 PPP USD (Kummu et al. 2018)	Economic activity generating extractive activities and urbanization (Barbier and Cox 2003)

Population density (1km grid cells)	Gridded Population of the World v4 2000-2020 (CIESIN 2018)	Urbanization infringing on mangrove forest (Barbier and Cox 2003)
Travel time to a city (minutes, 1km resolution)	Weiss and Nelson (2018)	Connections to major markets and economic payoffs for harvest (Barbier and Cox 2003)
Distance to previously existing PA	ICMBio (2023)	Possibility of spillovers from previously existing (prior to study) PAs (Miteva et al. 2015)

GDP per capita is taken at a yearly level from 1990 to 2015, with years in the study period before 1990 and after 2015 being taken as the nearest year. Population density is available at 5 year intervals from 2000-2020 and is taken as 2000 values for years before 2000, 2020 values for years after 2020, and interpolated for years between 2000-2020. The distance to a city and latitude are taken as time-invariant, while the distance to a PA changes when a new PA is created nearby. Time-invariant covariates fill an analogous role to grid cell-level fixed effects, with the grid cell-specific effects being a linear combination of the covariates used.

4.2.3 Matching

In the initial sample, neither trends nor covariates are comparable between the treated and untreated groups. As such, I employ pre-matching to construct a balanced sample, with trends in each after matching shown in Figure 21. In line with recent papers showing that matching on pretreatment levels for DiD can improve balance (Rico-Straffon et al. 2023), I include covariates, pretreatment levels, and pretreatment trends as matching variables. Each is then used as a covariate in a standard covariate matching procedure, with Mahalanobis distances (Austin 2009), as covariates can affect both treatment and outcomes. I first include all covariates used above, with the exception of distance to a PA, which is less likely to be balanced due to abrupt changes and shifts to 0 after treatment. Since pretrends diverge significantly from 1991-1998, I also match directly on mangrove forest cover in those years. Pretreatment mean and slope are also included for the same reason, accounting for trends and outcomes. prior to treatment. I show in Table 8 the balance of covariates after matching. Recommended diagnostics show all covariates are within acceptable limits of .10 standardized mean differences, and variance ratios (Austin 2009, Zhang et al. 2021).

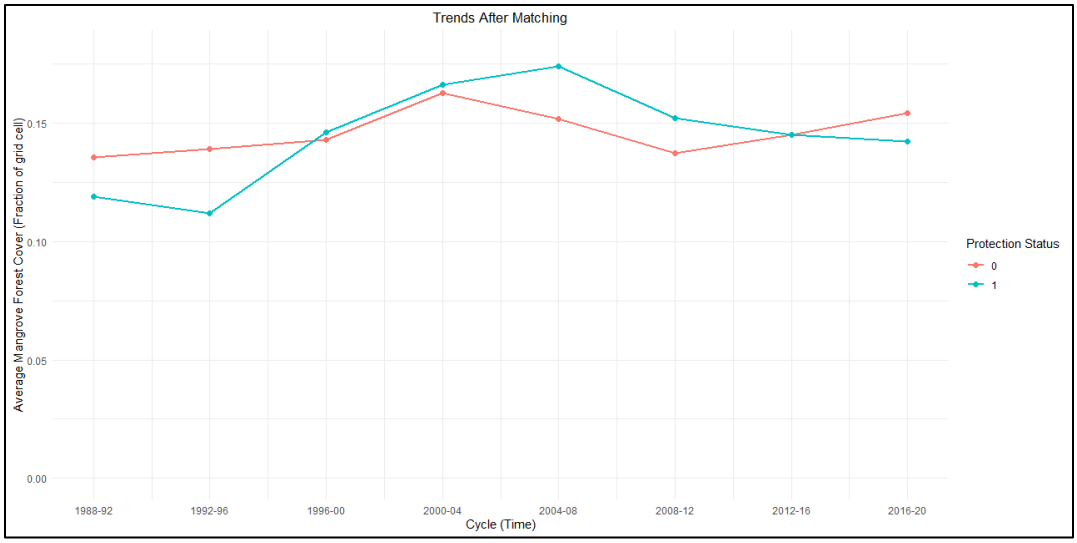


Figure 21: Mangroves by Ever-Treated Status After Four-Year Aggregated Matching

Table 8: Matched Year-by-Year Sample Characteristics

Balance Summary After Matching

	Covariate Balance			
	Means Treated	**Means Control**	**Std. Mean Diff.**	**Var. Ratio**
Pre-treatment Mangroves	1.485940e-01	1.500077e-01	-0.0096350	0.9589814
Pre-treatment Slope	7.027000e-04	7.145000e-04	-0.0023734	1.1429782
Pop. density (/km2)	4.496948e+02	4.341878e+02	0.0086802	1.0892557
Per Capita GDP (2010 USD)	1.371239e+04	1.362969e+04	0.0113724	1.0463301
Distance to City (min)	7.860686e+01	7.318576e+01	0.0652090	1.0394409
Distance to PA	3.481749e+04	3.875699e+04	-0.0611924	1.0159994

Despite the matching procedure used, there still exists divergence in the treated and untreated groups in certain years, creating the possibility of calendar-specific effects which would bias the results. In order to create a more balanced sample and reduce year-to-year noise in outcomes, I aggregate to four-year intervals, corresponding to federal planning

cycles. I first match directly on solely average mangrove cover and slope, so as to minimize the possibility of year-to-year differences, and after observing a slight divergence in the second cycle (1992-95), include mangrove levels in that cycle directly as a matching variable. This greatly improves the parallel trends across all cycles, as seen in Figure 21. Matching is done separately with the same procedure across all subgroups: overall, federal, state and municipal, strict, and less strict. Because covariates are not included in the matching procedure, they are left out of DiD estimation procedures for four-year aggregation as well.

4.3 Results

4.3.1 Specifications

We first estimate the staggered DiD model (Gardner et al. 2024) with no covariates, aggregating to event-time effects relative to treatment, as shown in Figure 22a. These results indicate the average change in the likelihood of mangrove forest cover in an area where mangroves are possible, in the year relative to PA creation, aggregated across the entire study area and time period. Because of the limitations of the study period, years further from 0 have fewer grid cells in their sample, thus causing increases in the standard error. For example, a grid cell which contains a PA created in 2015 will only have 5 years of data post-PA creation. Error bars thus widen moving further from zero in the below results, which is a feature of the data structure, not inherent variability in the data.

I next include the full set of covariates, as shown in Figure 22b, accounting for the differences across sites and time in geographic and socioeconomic characteristics. This allows for pre-trends to be constructed conditional on covariates, accounting for differences after matching. I compare these two specifications to examine the effect of measured characteristics on estimated impacts of protection.

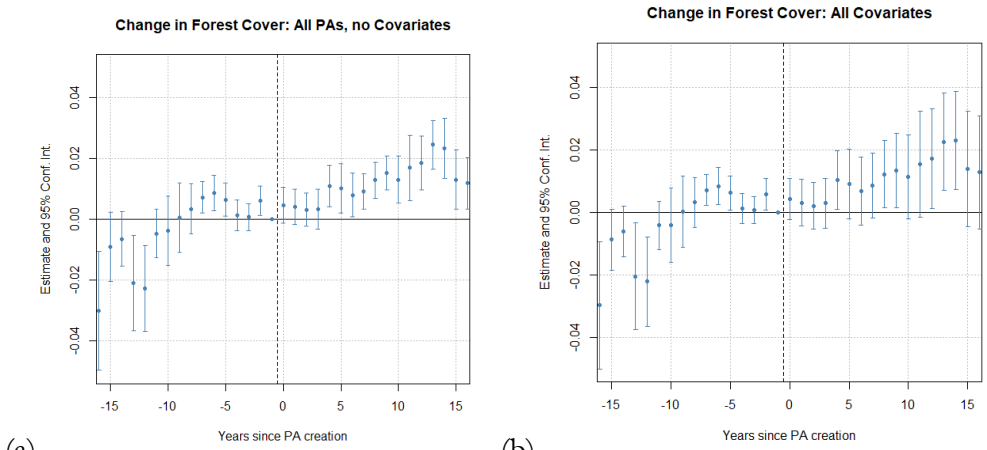


Figure 22: Annual Effects (a) Without Covariates (b) With Covariates

In Figure 22, I cluster standard errors (SEs) at the grid cell level. Clustering at the PA level would be possible given the process of treatment assignment (PA zonation), but would certainly an overestimate of standard error sizes, given that the number of clusters in the sample is a significant fraction of the number of clusters in the population (Abadie 2023). As such, I compare the outcomes when clustering at the PA level and the grid cell level, and proceed with clustering at the grid cell level, given the minimal difference between the two, shown in Figure 23 with no covariates. All following results have standard errors clustered at the grid cell level.

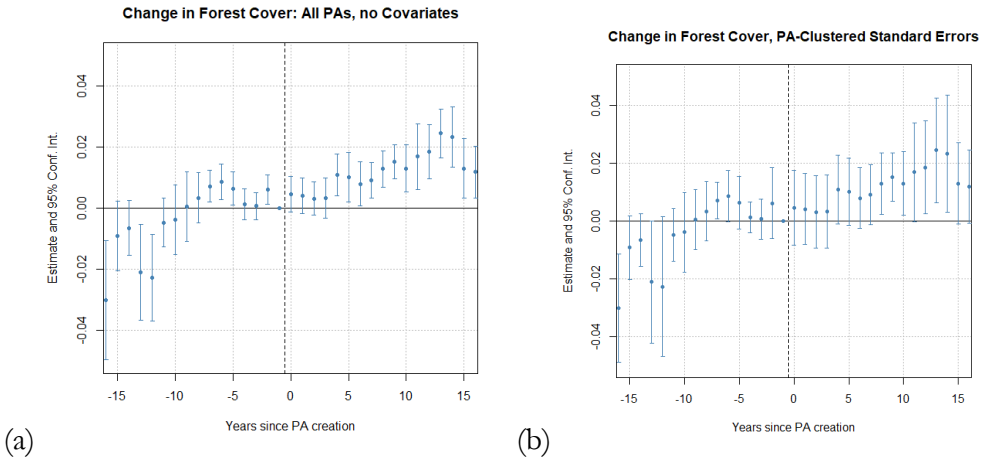


Figure 23: Annual Effects (a) PA-Clustered SEs (b) Grid Cell-Clustered SEs

Next, I compare results across two heterogeneities: federal PAs as compared to state and municipal PAs, and strict PAs (those which do not allow recreation, IUCN Category I) compared to those in which recreation is allowed, IUCN Categories II and III. For all these analyses, full covariates are included, to account for possible siting considerations in these subgroups.

I also examine year-aggregated effects, to look for patterns over time or specific years driving the overall effects. Because the sample size in each year is small, most have a wide confidence interval, while those that are more precisely estimated contain a larger amount of grid cells which gain protection in those years. I also examine cohort-specific

year-by-year effects in Figure 24, estimated using Callaway and Sant'Anna (2021).

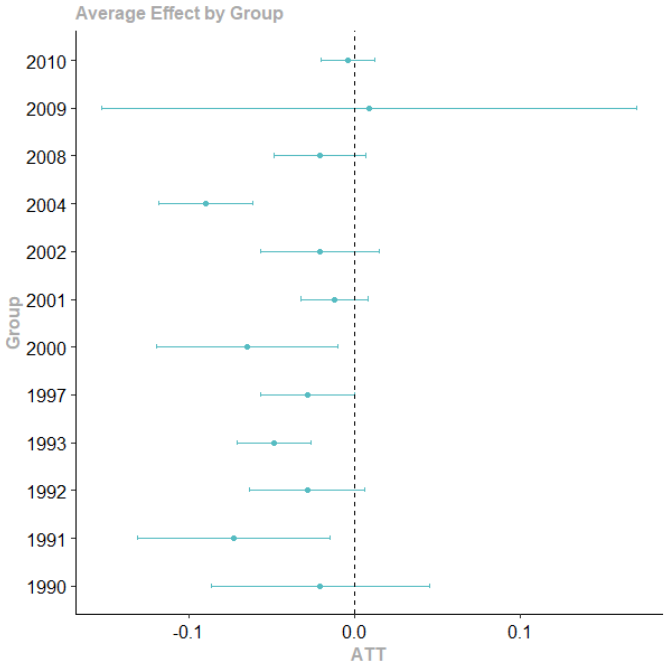


Figure 24: Year-by-Year Effects Aggregated by PA Creation Year

I complete the same overall and subgroup analyses of federal as compared to state and municipal PAs and strict as compared to less strict PAs for four-year aggregated outcomes, with results reported in Figure 25, Figure 26, and Figure 27. These outcomes have grid cell-clustered standard errors and do not include covariates.

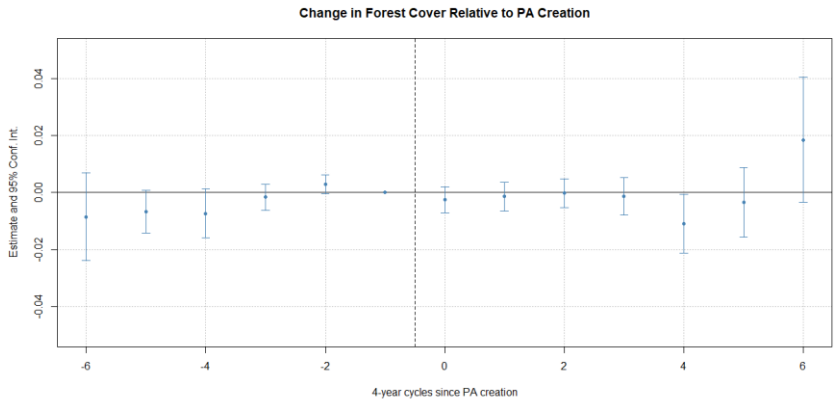


Figure 25: Overall Four-Year Aggregated Effects

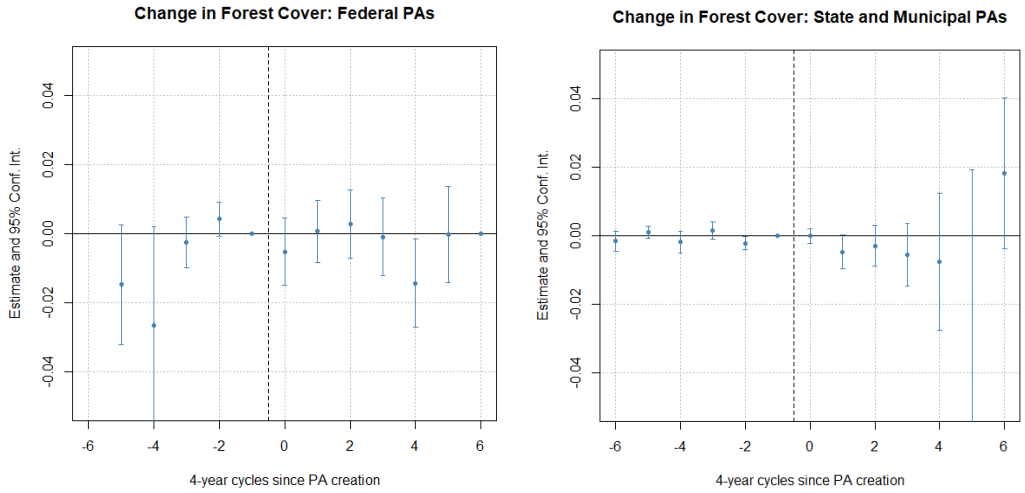


Figure 26: Aggregated Four-Year Effects in (a) Federal PAs (b) State and Municipal PAs

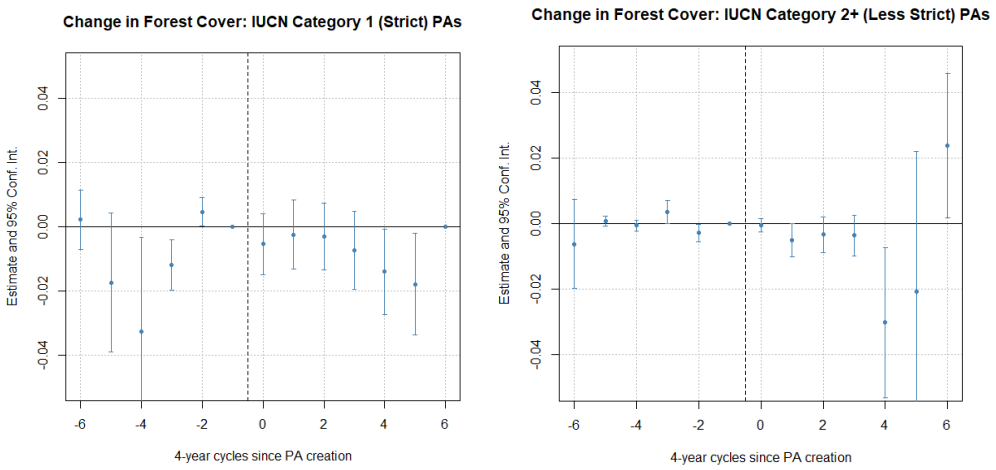


Figure 27: Aggregated Four-Year Effects in (a) Strict (b) Recreation-Permitted PAs

4.3.2 Year-by-year findings

The unconditional event-time estimates show an overall positive effect compared to the time period prior to implementation, as shown in Figure 22a. However, both specifications, shown together in Figures 22a and 22b, exhibit a decrease in forest cover approximately 15 years prior to PA creation, of approximately 2 percentage points. The likelihood of mangrove forest cover then steadily increases, before reaching an approximately 1-2 percentage point increase, which is maintained until approximately 10 years after PA creation, at which point there is a partial return to the baseline. As discussed in the conclusion, these pretrend effects may be anticipatory or due to selection effects (Malani and Reif 2015). While results where anticipation predominate do occur (Coglianese et al. 2016), even if there is not clear evidence for the pretrend being solely due to anticipation, such violations in parallel trends can be informative, and do not negate post-treatment observed effects when the pre-treatment effects are informative of post-treatment trends, as is the case with a change in deforestation pressure (Rambachan and Roth 2021).

However, these differences are not maintained when aggregated to the four-year cycles, indicating that there is not necessarily an effect in the aggregate.

Differences between the specifications with and without covariates appear to be highly time-specific, but small in the aggregate, following the same general pattern of a decrease followed by a steady increase. This may be indicative that there are particular PAs which are affected by siting considerations, but that in sum, the sample of unprotected areas and not yet protected areas is able to serve as an equally valid comparison group under this methodology. I do not observe clear patterns by year, as shown in Figure 24.

4.3.3 Four-year aggregate findings

Given that we cannot determine the cause of the year-by-year changes in mangrove cover, the aggregated results provide a better opportunity to determine whether we observe an overall causal effect. As seen in Figure 25, we do not observe an effect of protection in any aggregated cycle following PA creation. The same is true in two subcategories: federal and state PAs, as well as strict and less strict PAs, as seen in Figures 26 and 27. While variation increases further from PA creation, both prior and following, the margin of error also increases to the extent that no consistent effect is observed.

4.4 Conclusions and implications

It is difficult to provide certainty on the exact mechanism behind the observed outcomes and lack of overall effect, given the large time span and geographic area. These are aggregate estimates, and as such should be interpreted as the average long-run effects of PAs in a particular ecosystem and over the timespan, rather than telling a specific story about how the implementation of a particular PA affects its forests. Indeed, the two specific heterogeneities that I do examine suggest differing effects and mechanisms, determined by

the characteristics of that management level or set of rules. Were a study to be designed to estimate the impact of a single PA in a specific time period, the approach would no doubt be different. However, the overall patterns present and the heterogeneities in those results do suggest some underlying explanations.

First, the presence of anticipation suggests that some actors could be aware that protection may happen in the future, and are adjusting behavior accordingly, perhaps by increasing extractive activity. However, the pretreatment recovery following this change indicates that the forest is able to regrow before the PA is implemented, or that deforestation activity decreases. It is possible that the regrowth then progresses after PA creation, since the increase in forest cover continues. Given the limitations of this methodology, it is unclear whether true anticipation is the underlying mechanism, selection effects, or some other factor. The actors are more likely to engage in both changes in behavior, deforestation before implementation and the following recovery, when the effect of protection is binding, under federal or strict PAs. The fact that in some categories, the effect diminishes after a long time period indicates that the PA may be unable to change behavior over a long time period, and perhaps at a certain point, extractive activities resume.

The central interpretive question of the observed year-by-year trends is whether they are due to true foreknowledge of the PA, selection effects, or some other factor. While some illustrative case studies show that land use plans for a PA can indeed provide indication of the possibility of the creation of a PA over 15 years in advance, such plans also directly take into account observed changes in forest cover (Nery et al. 2024), suggesting the possibility of PA sites being selected based on existing trends. Given the wide variety of plans for each

individual PA, it is difficult to make broad conclusions about the predominant mechanism over the entire sample size.

Overall, the findings do not find a consistent positive effect of protected areas on the regrowth and protection of mangrove forests. This finding is not out of step with recent meta-analyses of quasi-experimental evaluations of protected area impacts, which find mixed results (Geldmann et al. 2018, Burivalova et al. 2019) and "persistent shortfalls" in PA effectiveness (Maxwell 2020). Where we observe some changes in year-by-year trends, changes are small relative to the size of mangrove forests, suggesting that other underlying trends are at play in driving the overall mangrove forest change present over the study period. Particularly in Brazil, a country that has observed more positive changes in mangrove forest cover than many others during this time span (Diniz et al. 2019), it appears that while aggregate patterns exist, many other factors may predominate in producing the overall avoided deforestation of mangroves, relative to elsewhere in the world.

Further, the larger observed effects of federal and strict PAs, with a more prominent version of the same underlying trend in both cases, underscore the importance of understanding how specific PAs are able to change behavior. The need for further research is highlighted on two fronts: first, for research which aims to uncover a specific mechanism behind PA effects, and second, for examining heterogeneity across PA impacts by individual PA, instead of solely in the aggregate. Nevertheless, the observed pattern and variation which holds widely across a common ecosystem, as well as the absence of an overall effect adds to the body of literature on PA impacts in a novel setting of major ecological and socioeconomic importance. Given the importance of Brazil's PA system in its mangrove

protection plan and its importance for climate goals, the lack of a clear long-run impact heightens the need to understand the true effects of its PAs on mangrove recovery.

5. Conclusion

Given the disparate methodologies used in each chapter, overarching empirical conclusions are elusive. In each essay, we arrive at separate conclusions about how restoration programs can be planned and funded, how marine protected governance works in practice, and on the aggregate impact of protection status implied by a particular quasi-experimental methodology. Given the nature of conservation practice, the inability to draw deeper theoretical insights surrounding actions towards supporting ecological flourishing is in part due to the inherent limitations of the methods used.

While the first essay provides evidence for the ability of future information to affect restoration planning decisions, normative statements about either restoration research or restoration practice are not directly implied. There are significant empirical realities which are omitted from the analysis, to the extent that it is difficult to argue that the estimates are close to a "true" restoration priority. For example, cultural values are left out, yet are a significant motivator for restoration (Wehi and Lord 2017), as is relational value. Ethical justifications for restoration which fall outside of utilitarian value frameworks are similarly outside the scope of the analysis, yet remain relevant to the restoration decision (Wainaina et al. 2023).

Even in the context of the methods used, there are limits to the conclusions reached. The monetization process of restoration benefits is based on a sole meta-analysis (Brouwer et al. 2022) which suffers from the long-recognized inherent limitations of contingent valuation studies, including incommensurability and pluralistic value frameworks (Beckerman and Pasek 1997). Further, large-scale restoration programs under current governance structures often engender displacement, a primary concern not studied (Hernandez and Vogt 2020).

Nevertheless, the conceptual advance on the utility of projections in informing restoration priorities does imply specific directions for further research. Examining the implications of uncertainty directly would be well served, beyond the point estimate projections used here. Empirically, an approach that directly observes changing restoration decisions in response to changing ecological or economic conditions would also be a valuable application in further research.

In the second chapter, two main conceptual advances emerge, in addition to contextual insights. The first is a framework for operationalizing Ostrom's principles. Second is a methodology for identifying governance types within a PA using cluster analysis. With limited research studies that look at variation in governance practice within an MPA, the use of this dataset across a wide geographic area to identify that variation has important consequences for practice, implying that even within a single protected area, there exists variation in governance that could affect the effects of that PA.

Three context specific conclusions also emerge. First, the existence of clusters of settlements which are primarily aligned with good governance and poor governance practices provides support for the concept of a unitary standard of good governance. Second, while some MPAs have relatively poor or good governance, the fact that alignment varies far more within MPAs than between them highlights the need for further research of the variation which we observe. Lastly, the emergence of "Support of Higher Governments" as a distinct indicator in this dataset points to the role it plays in determining other management practices in the context of Ostrom's principles, as well as in the governance practices of this particular region.

These findings provide important insight into how protected area governance works in its present form. Nevertheless, common pool resource governance theory has inherent limitations as an area of study in addressing real world problems. While Ostrom's framework is grounded in observations of small-scale research users, it is ignorant of theories of power which drive the decisions of those resource users and thereby shape global resource use patterns (Hardt and Negri 2017). As such, it is unable to imagine alternatives to current systems of resource distribution and control, a major limitation given the changes in political systems necessary to reverse the decline of nature (Díaz et al. 2019).

The null result of the third chapter is a novel result in the context of the impact evaluation literature on mangrove PAs, yet unsurprising given shortfalls identified in PA capacity worldwide (Maxwell 2020). Nevertheless, it is not itself evidence of a minimal impact of protected areas, in this context or globally: it is possible that impacts could be identified by other methods. It is also certainly not evidence against the effectiveness or value of protected areas in practice, given the numerous motivations for PA creation. Establishing causation in coupled-human natural systems presents inherent challenges in identifying mechanisms (Ferraro et al. 2019), challenges perhaps exacerbated by the aggregate nature of the study. There is also significant variation in the impacts across all of the PAs studied, indicating that PA-specific factors play a major role in determining effects, beyond the two heterogeneities of management level and strictness which I study. These factors could be geographic, political, or ecological, but are not apparent from the approach used here.

Finally, returning to the introductory theme of conservation science as being guided by practice, I summarize the most relevant facts to emerge for policy implementation as follows:

1. Restoration priorities are affected by projected impacts of climate change.
2. Settlements within a network of MPAs in Eastern Indonesia show varying degrees of alignment with Ostrom's design principles of good governance.
3. DiD identification methods do not find significant aggregate causal effects of the PAs created since 1985 on mangrove forest cover in Brazil.

Appendix A: Focus group survey questions

Survey Question	Text
FG-11	How are the rules and boundaries of [name of MPA/name of usual fishing grounds] made clear to individuals who use marine resources within [name of MPA/name of usual fishing grounds]?
FG-14	Who actively participated in or is actively participating in making the decision to establish [name of MPA]?
FG-15	Who actively participated in or is actively participating in deciding upon the boundaries of [MPA name]?
FG-16	Who actively participated in or is actively participating in designing the organization that manages [name of MPA/name of usual fishing ground]?
FG-17	Who actively participated in or is actively participating in making rules governing [name of MPA/name of usual fishing ground]?
FG-18	How frequently does each of the following groups monitor the condition of marine resources in [name of MPA/name of usual fishing ground]?

FG-19	How frequently does each of the following groups monitor the well-being of people who depend upon marine resources in [name of MPA/name of usual fishing ground]?
FG-20	How frequently does each of the following groups monitor the compliance of resource users with the rules governing marine resource use in [name of MPA/name of usual fishing ground]?
FG-21	What penalties exist to encourage compliance with the rules governing marine resource use in [name of MPA/name of usual fishing ground]?
FG 22	How frequently does each of the following groups hand out penalties when people break the rules governing use of marine resources in [name of MPA/name of usual fishing ground]?

Appendix B: Key Informant Interview survey questions

Survey Question	Text
KII-2	Can you please tell me… a. What types of formal and informal management zones exist within the [name of MPA/name of usual fishing grounds]? b. How many examples are there of each type of zone within the [name of MPA/name of usual fishing grounds]? c. For each type of zone, is there a formal or informal organization that helps to manage ONLY this type of management zone? d. For each organization listed in 2c, to what extent does this zone-specific organization coordinate its activities with an organization that manages all of [name of MPA/name of usual fishing grounds]? Do they never coordinate, rarely coordinate, sometimes coordinate, usually coordinate, or always coordinate?
KII-6	I would now like to ask you about the rules governing the behavior of each of the following important groups of individuals in [name of MPA/name of usual fishing ground]. In particular, are there special rules governing the behavior of these each user groups inside the [name of MPA/name of usual fishing grounds]? For example, are there rules that apply to these groups and nobody else?

KII-7	Do the rules use of governing marine resources in [name of MPA/name of usual fishing grounds] change based on changes in ecological conditions in and around [name of MPA/name of usual fishing grounds] over time?
KII-8	Do the rules governing marine resources use in [name of MPA/name of usual fishing grounds] change based on changes in social conditions in and around [name of MPA/name of usual fishing grounds] over time?
KII-14	First, for each of the following groups, to what extent does the national government oppose or support the rights of users to develop their own rules governing resources within [name of MPA/name of usual fishing grounds]?
KII-18	If caught breaking rules in [name of MPA/name of usual fishing grounds], how often do rule-breakers receive one of these penalties?
KII-19	What factors influence the choice of penalty?
KII-20	What incentives are provided to encourage compliance with the rules governing marine resource use?
KII-21	I'm curious who "monitors the monitors" in [name of MPA/name of usual fishing grounds]. What happens if the people

	responsible for monitoring and enforcement don't do their job properly? Are there penalties?
KII-22	What happens if the monitors do not effectively monitor social conditions? Are there…
KII-23	What happens if the monitors do not actively monitor the compliance of individuals who use marine resources in [name of MPA/name of usual fishing grounds]? For the monitors, are there…

Appendix C: Limitations and robustness checks of design principles

Governance Principle	Limitations	Sensitivity tests	Results
Clearly Defined Boundaries	This approach assumes they are equal and contextually appropriate, while in reality, all may not apply to all circumstances. Some particular means of communication involve active outreach, while others are more passive, such as signage.	I recalculate the indicator with only active means of outreach: written, audio, or visual notice, door to door, town meetings, etc.	Negligible change in indicator distribution. Thus, all means of outreach are included, since all provide some additional information.
Participation in Decision Making	The equal weighting of each is arbitrary, and the inclusion of the second metric is tenuous. One challenge in interpreting that question is that the actors being involved	I recalculate the indicator with only the first metric.	Sizeable shift to the left in the entire distribution, and overall compression of the distribution. Given Ostrom's principle,

	could include the national government and foreign NGOs.		decisions about the MPA implementation are relevant to the structure of rules, and thus are still included, but with the understanding that distinct aspects of decision making are included.
Active Monitoring	Many different indicators, corresponding to different components of the Accountable Monitoring governance principle	I check how well this metric is congruent with others using Cronbach's alpha.	Combined Accountable Monitoring indicator had low level of overall congruence (<.1), hence split into 3 separate indicators. Active monitoring component has

			much higher congruence, with a Cronbach's alpha of .718.
Compliance and enforcement	There is a question as to whether these types of penalties are aligned and likely to be observed together.	I check correlation between the indicators.	Correlation is low, .109. Thus, these types of penalties appear to be unrelated, but both are important aspects of enforcement as it relates to Ostrom's principle.
Accountability for Monitors (Sanctioning)	The indicators may not be measuring similar components of a single metric.	I check congruence among these metrics to see how well they align.	Overall congruence among the 4 indicators for complete cases is .960, indicating a high level of a alignment among the indicators.

Congruence with Local Conditions	The third question could be a metric of congruence, but also may indicate a power imbalance, with each group doing what it wants.	I run congruence tests to check for correlation between the variables (Cronbach's alpha) with and without the third indicator.	Overall congruence among the 3 indicators is moderately high (.657), and higher among the first two (.864). I include all 3 indicators, since the third likely measures heterogeneity among groups in a way not captured by the first two.
Support of Higher Governments	None	N/A	N/A
Nested Governance	There are concerns about situations where there is only a single zone, and	Since I lack alternative information in those cases, I include both	The metric is calculated using both criteria, requiring the presence of both

	the irrelevance of the question in those cases.	criteria in a single metric.	multiple zones and different organizations in each zone (See Appendix A).
Graduated Sanctions	This metric assumes that each of the sanctions are relevant in that particular settlement.	Since the second question doesn't have the same tiers, I test the indicator with solely the first metric.	Negligible change in overall indicator and governance categorization, so I include both, to maximize available information.

Appendix D: Nested governance indicator calculation

Multiple levels of governance

Multiple zones	Yes	No
Yes	1	0
No	0	0

Appendix E: Variation across MPAs for governance principles

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