Conserving What's Left: The Political Economy of Protected Area Location

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Abstract

Protected areas are one of the best policy solutions to address the twin crises of climate change and biodiversity loss. Despite the importance of protected areas (PAs), we know little about how they are affected by political factors. This paper offers a political economy analysis of the protection of ecologically important areas. I argue that misallocation of PAs occurs because the decision to protect land is inherently a political one. Domestic interest group contestation leads governments to place PAs in areas with low economic opportunity cost as opposed to diverse ecosystems and biodiversity hotspots. I examine the role of domestic and international political institutions in facilitating better environmental outcomes through shifting this distributional conflict. In order to explore these dynamics, I construct an original time-series geospatial dataset on the coverage of 846 ecoregions worldwide from 1992-2020, which to my knowledge is the first of its kind. I find that when green factors, such as membership in international environmental regimes and the participation of environmental NGOs, are stronger, distributional bargaining shifts in favor of conservation. However, when extractive interests are more important in a particular place, conservation is less prioritized. These findings contribute to our understanding of politics at the nexus of the environment and the economy, land use, and the role of international actors in domestic decision making.

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Introduction

The world is in the midst of a biodiversity crisis. The current rate of extinction is believed to be around 1000 times the "background rate" that would be expected under normal environmental conditions (Pimm et al. 2014). The collapse in biodiversity around the world not only places non-human species in peril, but endangers humans as well by harming agricultural and fishery output (Allen-Wardell et al. 1998, Worm et al. 2006), vastly increasing instability in food chains and ecosystem resilience (Ehrlich and Ehrlich 2013), and destroying natural carbon sinks (Naeem et al. 2009, Sala 2020, Jung et al. 2021). Scholars have called the crisis civilization-threatening (Ehrlich and Ehrlich 2013), while even security analysts have begun to warn of the instability that can stem from this ecological decline: the 2021 Annual Threat Assessment produced by the U.S. intelligence community states that "the degradation of…biodiversity resources almost certainly will threaten infrastructure, health, water, food, and security…" (p18). Signaling the depth of the crisis, the recent Conference of Parties for the Convention on Biological Diversity (CBD) led to a new agreement to combat biodiversity loss that many analysts argue has more actionable targets than the Paris Climate Agreement (Einhorn 2022).

Protected land areas (PAs) are one of the most effective tools for preserving biodiversity (Geldmann et al. 2019, Gray et al. 2016).[1] The growth of protected area networks globally is hailed as one of the greatest successes of environmentalism (IUCN 2010). Since 1990, the amount of land area covered by PAs has nearly quadrupled, from 4% in 1990 to almost 17% today (Reid and Lovejoy 2021). However, significant concerns remain about whether PAs are effectively located, even as coverage has nominally expanded.[2] PAs are often placed in remote, economically low-value areas rather than where they are needed for conservation of delicate ecosystems, carbon-sequestration hotspots, or highly biodiverse places (Joppa and Pfaff 2009, 2011, Venter et al. 2014, 2018). As a consequence, even though PAs are effective at conserving biodiversity within

[1] PAs are formally defined as a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values (Dudley 2008).

[2] A common way of measuring the effectiveness of a PA network, which will be used in this paper, is representativeness — the extent to which a network covers a representative sample of the different ecosystems and biomes in a country.

their boundaries, they have failed to meaningfully slow rates of biodiversity loss globally both in terms of species and population (Barnosky et al. 2011).

What explains why PAs are sometimes misallocated and sometimes more representative? Scholars have largely focused on exclusively economic factors, which are important, but I argue that the decision to designate a PA is inherently political, and it is interest group contestation that has often resulted in governments designating PAs in places with low economic opportunity cost, rather than broadly covering different ecosystems and biodiversity hotspots. In this paper, I present a political economy model of PA location decisions in which government decisionmakers make choices based on the preferences of interest groups and the relative importance of those groups to the government. Protection represents an investment in a public good, whereas extraction is mostly a private good benefiting the extractive firms. In this distributive conflict, I predict that when extractive groups, which prefer less protection in order to maintain access to rents, have more influence, there will be less broad protection across biomes and ecoregions. When green groups, such as environmental NGOs or international organizations (IOs), have more access and power, more targeted PA networks will emerge.

To test these theoretical expectations, I create a new time-series dataset of the coverage of ecoregions by PAs over time. For all countries from 1992-2020, I measure what proportion of each ecoregion within that country is covered by a PA. Ecoregions are defined as "relatively large units of land containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land-use change" (Olson et. al 2001, p.993), and are commonly used to measure PA networks' effective location (e.g. Dinerstein et al. 2017). This data collection effort represents, to my knowledge, the first global dataset of ecoregion protection by countries over time. By using this subnational data rather measuring country-level total protection, I measure the *quality* of a PA network as opposed to simply its size. I supplement this data with political-economy variables at sub-national and country levels to probe the implications of the distributive model I propose, and undertake several strategies to address endogeneity concerns around my independent variables. The findings support my theoretical arguments. Greater participation in the international environmental regime and greater presence of environmental NGOs are associated with broader coverage of ecoregions, but higher extractive interest in particular ecoregions is associated with less protection.

This paper makes three contributions. First, I contribute to a growing body of research on environmental politics. While this subfield has become more mainstream in political science recently, the majority of the work is focused on climate change (Keohane 2015). This is understandable given the magnitude of that issue, but other environmental crises interplay with climate in addition to deserving attention in their own right (Green and Hale 2017). A more direct focus on the politics of biodiversity and land use is overdue in political science, and this paper focuses squarely on this critical issue. Despite the political nature of PAs, relatively little political science work has focused on them. Existing approaches have focused primarily on economic incentives, focused on one or two countries (Alger 2023, Mangonnet et al. 2022), or used coarse measures of PAs within broader arguments (Kashwan 2017, Hawkins et al. n.d., Neumayer 2002a, 2002b). My findings build on this work by offering a political-economic explanation, and I capture important differences among PA networks by introducing the coverage of ecoregions as an important outcome.

My findings also speak to the current debate about models of global environmental politics. Traditionally, scholars have seen environmental challenges as a global collective action problem (Keohane and Oppenheimer 2016; Terhalle and Depledge 2013; Underdal 2017). More recently, scholars have argued that domestic distributive conflicts are more important than international free-riding concerns (Aklin and Mildenberger 2020; Colgan, Green, and Hale 2020; Genovese 2019; Kennard 2020). I synthesize these approaches by proposing a distributive conflict, but one that cannot be understood without considering the role of international politics in that conflict. Indeed, my findings suggest that international actors have influence in the domestic distributional conflict, and they may be more powerful than some domestic actors in the case of PAs, in line with other work (e.g. Bernauer et al. 2010). This paper is also one of the first attempts to model a broad, environmental distributional conflict empirically.

Third, I contribute to the literature on the influence of international actors and agreements on domestic politics. Debates continue as to the degree to which nonbinding international agreements can affect domestic policy outcomes (Simmons 2009, Fang and Stone 2012, Goes and Chapman n.d., Kaiser and Meyer 2019, Dai 2005, Hafner-Burton and Tsutsui 2005). My results show that, when part of a broader international environmental regime, even small environmental agreements can move states over time toward more "cooperative" outcomes. In my framework, these agreements can go beyond institutionalizing aligned interests (Keohane 1984), since they push outcomes beyond the explicit aims of the international organization and agreement in many cases. I also contribute to literature on transnational nonstate actors, showing that the presence of NGOs can make a significant impact on domestic distributional outcomes (Dörfler and Heinzel 2022, Shibaike 2022).

The Importance of Protected Areas and Representativeness

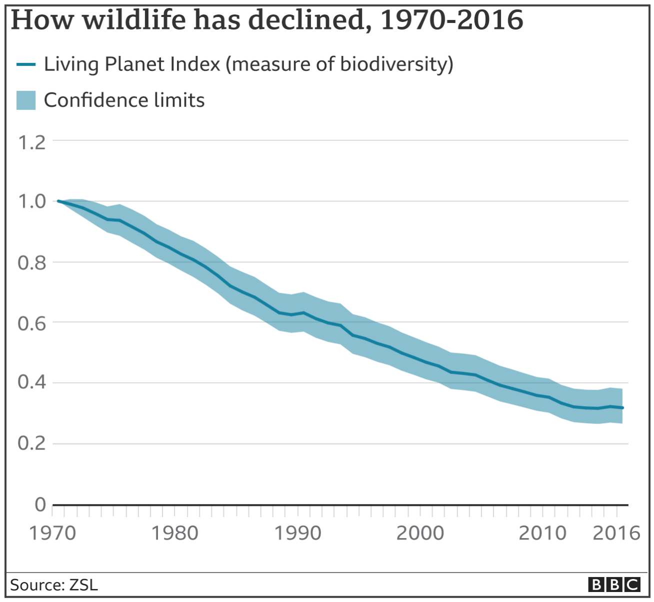
The biodiversity crisis is on par with the climate crisis in urgency, and each exacerbates the other. Figure 1 below demonstrates the precipitous decline in non-human or human-domesticated life on earth in recent decades.[3] Protecting biodiversity is critical for preserving the ecological health of the planet and ensuring its continued habitability for humans. There are also strong normative arguments for conserving biodiversity for its own sake (Sala 2020, Ephraim 2022).

PAs help protect biodiversity and ecosystems. They are locations set aside for preservation in order to protect biodiversity, ecology, or occasionally cultural value. The National Parks system in the United States is a well-known example of PAs, but they can also be on smaller or even larger scales. The level of protection can vary depending on the specific designation and the country that creates the PA, but traditional economic development, such as clearing forests for agriculture, tends to be strictly limited or prohibited within PAs. The restriction of economic exploitation is one of

[3] The Living Planet Index is "a measure of the state of the world's biological diversity based on population trends of vertebrate species from terrestrial, freshwater and marine habitats" (WWF 2022).

the primary ways that PAs can help preserve ecosystems and biodiversity. They prevent agricultural monocrops from replacing plant life, and preserve the habitats of animal life. PAs also protect against poaching and wildlife trafficking, in comparison to non-protected land.

Figure 1. Biodiversity Loss Since 1970 (1970 = 1.0)



PAs do, of course, vary in their effectiveness. Illegal logging and wildlife poaching in PAs are major concerns (Burgess et al 2012, Haass 2020). Additionally, the size of PAs matters, since larger PAs allow for deeper intermingling of species and prevent the isolation of small groups of species. This isolation can lead to their die-off even within a PA, known as habitat fragmentation (Wilson 1984). PAs also vary in the stringency of their protection. For example, many allow resource extraction such as mining and oil drilling within them, which disrupts habitats. These sorts of PAs are less effective at protecting ecology and biodiversity than more strict PAs. Empirically, it is common to include all PAs in analysis (e.g. Venter et al. 2018), although some do restrict to only those in International Union for the Conservation of Nature (IUCN) Categories I-IV which prohibit almost all economic activity. This paper includes all categories in the empirical analysis.

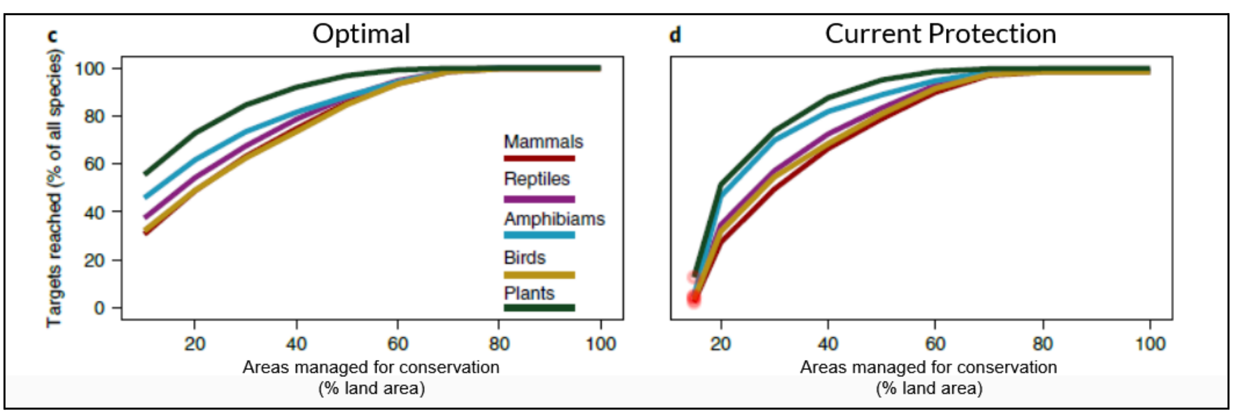
Beyond concerns about effectiveness for conservation, PAs also have a complicated and at times violent history. Governments have often displaced indigenous people when created PAs, disregarding sustainable traditional knowledge in favor of "fortress conservation" (Dowie 2006, Eisenfeld 2015, Gibson 1999). While keeping this dark history in mind, for the purposes of this paper I posit that PAs are environmentally beneficial in the sense that they help to conserve biodiversity where they are located.

Overall, then, PAs have been shown to benefit biodiversity (Geldman et al 2019, Gray et al 2016). They improve the ecological health of ecosystems within them, compared to areas directly outside. They are not perfect, and areas with the strictest protections perform better than those with some economic activity allowed (Sala and Giakoumi 2018), but any PA is vastly superior to allowing open exploitation, and is one of the best tools available for slowing the biodiversity crisis.

The importance of PAs, despite their relatively low profile compared to other environmental issues like carbon reduction, has not gone wholly unnoticed: the UN recently agreed to a framework increasing global commitments to protecting 30% of the world by 2030.[4] Similarly, Sustainable Development Goal 15 is directly related to area-based conservation. Many major donors, including states, are looking to PAs as a way to slow environmental degradation globally and prevent carbon emissions via deforestation, particularly in heavily publicized places like the Amazon.

Given the extent of the overlapping climate and biodiversity crises, the important role that PAs can play in ameliorating them, and the renewed focus on them by the international community, designating effective PAs is crucial. As discussed in the introduction, the current global PA network is misallocated by most environmental standards, a failure that is commonly explained through economic opportunity costs. The figure below demonstrates that "optimal"

Figure 2. Difference in Optimal and Current Protection by PAs



[4]The 15th Conference of Parties of the CBD took place in December 2022, where this framework was finalized.

allocation of a PA network the same size as the one that currently exists would result in reaching targeted coverage (based on international goals) of 30-55% of species, compared to 5-15% in the current network.[5] "Optimal" in the figure is based solely on biodiversity, but the same figure based on optimizing for biodiversity, carbon sequestration, and water retention would look similar.

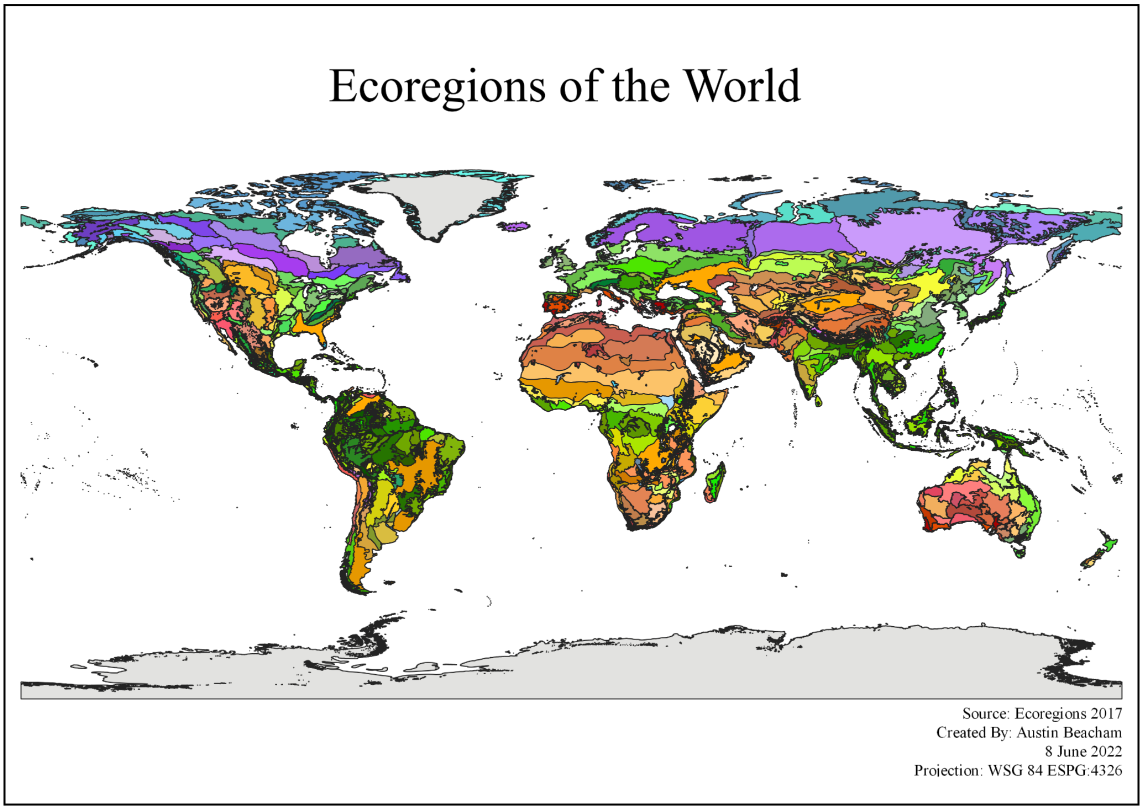
The sort of intensive species and carbon mapping used to create figure 2 has only become available in recent years. Before then, the concept of representativeness was how environmental analysts tried to account for proper allocation. At first, environmentalists advocated for coverage of each biome within a country. The fourteen biomes of earth are simply areas of the planet that can be classified according to the plants and animals that live within them.[6] However, there is significant variation within biomes that meant that smaller-scale ecosystem mapping was needed. A commonly used approach now is the ecoregion, defined above. There are 846 distinct ecoregions according to the latest data, shown in figure 3, which perfectly nest within biomes (Dinerstein et al. 2017). An example is the Western shortgrass prairie in the central United States, which differs from Northern shortgrass prairie in climate and species habitation in a way that makes the two ecologically distinct, despite both being part of the temperate grasslands, savannas, and shrublands biome.

While biomes are a more commonly understood way of dividing the natural world, I follow the majority of the scientific community and most new policy goals by focusing on a more granular measurement — in this case, ecoregions. While the creators of PAs may not have had this specific division in mind, protecting a diverse array of environments has been a key goal since the early days of conservation (Adams 2004). This method is easily understood (since ecoregions nest within biomes), rigorously mapped, and able to be compared across countries that may have differing levels of species diversity density. For the purposes of this paper, therefore, I introduce the concept of representativeness as the extent to which a PA network covers a variety of ecoregions. This is one measure of the quality of a PA network. While protecting biodiversity hotspots is also critically

[5] Adapted from Jung et al. (2021). The red dots on the right represent the current network's coverage. Looking at where the lines begin on the left chart versus the right shows the difference in protection that would emerge from a more targeted global PA network the same size as the one that exists today.
[6] See Appendix for map.

important, protecting some of each type of natural environment on earth is another worthy goal, since even low-biodiversity density places perform critical ecosystem services, are home to unique and inherently valuable flora and fauna, and their health potentially impacts upon other ecosystems both near and far. Representativeness is an explicit measure of PA network quality.

Figure 3: Global Distribution of Ecoregions[7]



The Basic Ecological and Political Problem

Protecting diversity of life on earth is critical, and decisionmakers have had tools to do so for generations. However, biodiversity loss has been part of human development for millennia. This is because development for most of the last 10,000 years, and particularly starting in the industrial revolution, has been driven by converting land from diversity-rich to diversity-poor (Swanson 1994). For example, a field full of many different kinds of native grasses, along with the animal life whose natural habitat is located within this particular mix, can be cleared in order for a monocrop

[7] Due to the large number of ecoregions, figure 4 does not have a scalar and intends simply to reflex the diversity of different ecoregions.

of wheat to be planted. According to Swanson, the fundamental issue is that the diverse resources that were located in that place are not valued as highly as the low-diversity alternative, such as a farm or a mine. When this is the case, as it has been for most of history, land conversion occurs, with accompanying biodiversity and species loss. Additionally, biodiversity itself can be the resource that is more highly valued in its exploited rather than preserved state. Many local populations in Africa chastened against colonial-era wildlife management policies because bush meat was a large source of income and food for rural communities (Gibson 1999).

In this framework, PAs restrict this method of development where biodiversity is replaced with more "productive" alternative uses of space. They restrict the conversion process of space and allow for the preservation or recovery of the natural diversity present there. When governments are involved in the creation of a PA, they make a conscious decision to restrict conversion in exchange for the more diffuse benefits derived from intact ecosystems such as biodiversity conservation, flood resistance, increased productivity in adjoining areas, tourism, or international prestige (Mangonnet et al. 2022, Sala 2020). This creates winners and losers and divides preferences on PAs between those who prefer the shorter term, conversion-derived benefits, and those who place more value on the diffuse, long-term benefits.

Beyond this basic setup, there are three principal reasons PAs should be seen as political. First and most importantly, most PAs are established by or at minimum recognized by governments. The International Union for the Conservation of Nature (IUCN), a mixed-membership IO that is arguably the leading authority on biodiversity conservation, identifies four types of PA governance: governance by government, shared governance, private governance, and governance by indigenous people and local communities (Borrini-Feyerabend et al. 2013). Three of these four type require recognition and cooperation on the part of national governments. Within the government, PAs are usually established by executive agencies or legislatures, and management is often then delegated to subnational administrations or municipal governments. For example, in Western Europe many PAs are established by regional governments with the approval of national governments, but in Eastern Europe PAs are still normally centralized and administered by national agencies (Borrini-Feyerabend et al. 2013, p31). PAs normally do not require formal legislative approval in terms of passing a bill and having it signed by the executive. However, the fact that government is largely responsible for establishing these PAs means that they are subject to the same influences on policymaking as other government decisions.[8]

Second, PAs create significant distributional consequences both for the immediate surrounding area and for economic interests that may want to exploit the protected natural resources for short-term economic benefits. Recent work has begun to examine the domestic political economy of PAs, and describes the situation helpfully (Mangonnet et al. 2022, p6-7):

> Protected areas generate winners and losers (e.g., Fernández Milmanda and Garay 2019). On the one hand, they can mitigate climate change, improve national reputation, appease environmental interest groups, and secure forest peoples' livelihoods... On the other hand, they are costly to the local primary sector because they prevent the extraction of natural resources. The problem is one of public goods provision (Olson 1965; Samuelson 1954): locally costly actions generate broader social benefits at the national, and even global, level.

Third, they represent at minimum a symbolic commitment to conserving natural ecosystems that in theory precludes political exploitation of those resources. This can have ramifications for politicians that create PAs or have them designated in their jurisdiction by an outside party. Locals who would prefer to exploit may push back and punish a politician electorally, or pro-protection locals and other groups may reward them with support. International donors and NGOs are often very invested in PA policy and can rally for or against politicians that create or destroy PAs. Even more directly, Sanford (2021) discusses how forests and other valuable ecosystems can be used as a political tool in winning votes during close elections. By sanctioning a PA, a politician is consciously preventing themself from accessing that tool in the future, at least legally.[9] Political actors are aware of these possibilities and take them into account when making PA decisions. For all

[8] For the purposes of this paper, I focus mostly on national governments when I refer to government decisionmakers, since these actors often have the final say even if authority is formally delegated to subnational bodies.

[9] Degazetting, or unprotecting, does occur, but it is rare, and would presumably engender some reputational cost on the part of the politician, especially if it were the same politician under which the PA was originally created.

of these reasons, the decision to create a PA is clearly political. It can engender faraway support while simultaneously causing local resistance, depending on the characteristics of the location.

The Political Economy of Protected Area Location

Given this setup, I now turn to addressing why the current PA network varies in representativeness. I argue that the fundamental conversion tendency described above often wins out over desires to conserve representative ecosystems, but that this can vary depending on interest group contestation and institutions.

Who Resists PA Location on Representative Lands?

I argue that organized, extractive interest groups, which are the dominant force undertaking the global conversion process of land in the last fifty years, push against PA creation in the first place, and especially PA location in representative areas.[10] By extractive interest groups, I refer to organized economic groups such as firms and industries that are interested in land use for resource extraction such as mining, logging, and most agricultural practices. These economic elites prefer fewer PAs overall, and that PAs be placed in areas with lower economic opportunity cost, rather than places that will best protect representative biodiversity. They prioritize the short-term economic windfalls that can be gained from this sort of placement, eschewing the long-term benefits that biodiversity can provide (Sanford 2021). Organized agricultural interests, such as large landowners or multinational agricultural corporations with investments in a country, would clearly prefer this type of placement. While numerous small farmers may have a difficult time coordinating (De Gorter and Swinnen 2002, Park and Jensen 2007), the actors I refer to here are fewer in number but greater in power, due to economic strength and advantage in overcoming the collective action problem (Olson 1965). Consolidation of agricultural industry is becoming more prevalent throughout the world (Ceddia 2019, Skillman n.d.).

[10] For a detailed discussion of this group and their role in PA policymaking, see Alger 2023.

Beyond organized agricultural interests, resource extractive industries would also prefer that PAs not be placed in areas that would complicate their business. Examples of this sort of industry would be oil extraction, mining, or logging. Even if some PAs still allow resource extraction, the process generally requires more bureaucratic hurdles if it is within a PA than outside of it. Therefore, I assume that overall, organized, economic interest groups prefer fewer PAs overall, and less representative coverage when such coverage would coincide with high economic opportunity cost areas. This overlap is common: the most biodiverse places are often those with the most economic opportunity, because they contain potentially arable farmland, vast tracts of forest, or other valuable ecosystems. Density of life coincides with high resource potential more than enough for my arguments about economic groups to hold.

These groups have been successful during the expansion of the global PA network: while PAs have been created for reasons discussed below, which these groups do not want, they have varied widely in representativeness, because such representativeness conflicts with continued economic extraction and land conversion. Extractive interests are resource-powerful, have persuasive economic arguments for national governments, and are more consolidated and therefore better organized than groups that desire a more representative PA network. They have therefore been able to achieve their second-best outcome: PA expansion, but broadly in low economic opportunity cost places. They have less interest in low-economic opportunity areas, and so they will not push as hard against PAs created there. I expect the power of these extractive groups to manifest most heavily in ecoregions with significant potential for economic opportunity, which would concentrate protection in low-opportunity cost areas. This leads to my first hypothesis:

H1: The more interest that extractive groups have in land in a country, the less representative that country's PAs will be, all else equal.

Who Advocates for Representative Protection?

Given how deeply-rooted the conversion process is and the power of extractive groups, what forces can alleviate these pressures and allow for more representative PA networks? I focus on two factors: the presence of environmental NGOs, and the embeddedness of a country in the international environmental regime that has emerged particularly since the end of the Cold War. Each of these factors can empower other green groups and influence outcomes directly. I start more broadly with international actors, before focusing on domestic groups.

The International Environmental Regime

Many national governments have experienced international pressure to expand their PA networks in a representative way for decades. Beginning in the 1970s, PAs became one of the most widespread policy responses to environmental problems. Creating PAs became accepted as "what states do," in the vein of the World Society approach (Meyer et al. 1997). This helps to explain the vast proliferation of PAs across states starting in this time period, although growth was initially quite slow. International agreements centered on the environment and specifically around biodiversity protection, beginning with the Conventional on Biological Diversity in 1992, started including targets for the percentage of a county's land area that is covered by a PA. Such goals give a rallying point for interested parties, which in the aggregate can shift interests in the country. These agreements often also include more specific targets about ecosystem representativeness in the PA network.

Additionally, it may not even take direct participation in agreements with specific targets to create this pressure. States are socialized by the international community to view certain policies as mainstream and beneficial (Pevehouse 2002, Checkel 2001) — this likely happens across IOs with environmental interests and international environmental agreements (IEAs), even if they are not specifically addressing PA policy. In essence, association with the international regime complex around these issues, which states may join for related or unrelated reasons to PA and biodiversity policy itself, may increase a state's propensity to implement thess policies. Foreign governments may also take part in this socialization, diffusion, and pressure process: they may want more protection globally, or they may be trying to externalize policy change rather than make it within their own country (Hafner-Burton, Schneider, Pevehouse n.d.).

International actors can also become involved beyond membership. They can provide information, for example through scientific research on the state of biodiversity. For example, the Millenium Ecosystem Assessment was published by a collection of IOs and TNGOs, and involved input from over one thousand scientific researchers around the world. The report was widely covered in the news and helped shift public opinion in favor of biodiversity conservation, as well as empowering pro-biodiversity environmental groups. Information provision by international actors can have an especially pronounced effect in concert with domestic experts, where they can mutually reinforce conclusions and policy recommendations (Fang and Stone 2012).

The international community, particularly IOs, can also coordinate with domestic and transnational NGOs that are already embedded in the distributive conflict at the domestic and local level. They provide strategic advice, share best practices, and coordinate diffuse groups around single campaigns. The embeddedness of NGOs in transnational networks coordinated by IOs has been shown to be important in empowering them with strategies and resources that make them more effective in advocacy, lobbying, and public opinion campaigns (Abbott et al. 2015, Dörfler and Heinzel 2022). This channel of influence is similar to past work on IOs empowering civil society (e.g. Dai 2005, Darst 2001, DeSombre 2000, Green and Hadden 2021, Hafner-Burton and Tsutsui 2005, Hafner-Burton et al. 2016), but involves even more direct cooperation than some of these works discuss, particularly because the transnational policy community on biodiversity is relatively small compared to other issue areas. This channel of activity fits well with the "IOs as orchestrators" literature, but I add to it by providing broader empirical testing and analyzing an understudied policy space (Abbott and Snidal 2010, Abbott et al. 2015).

Through these avenues, the international environmental regime and its NGO partners can both empower domestic pro-biodiversity domestic interests and directly shift the government's understanding of the relative utility of pro-PA policy compared to the interests of extractive groups. Indeed, some scholars claim that these international influences may even preclude domestic ones, which my empirical findings somewhat bear out (Bernauer et al. 2010). Thus, I put forth my third hypothesis:

H2: Increased state involvement in the international environmental regime complex is associated with more representative PAs, all else equal.

Domestic Green Groups

By green groups generally, I am referring to organized, engaged citizens that lobby for pro-biodiversity policies, NGOs such as the National Audubon Society in the U.S., and engaged scientific groups and individuals such as the Union of Concerned Scientists or Enric Sala, a famous marine biologist who has shifted his career towards advocacy in addition to research. The relative abundance of actors within this group will vary by country based on economic development, political system, and educational attainment levels, but they all have similar goals even if they may have different strategies to achieve them.

These actors strongly prefer coverage of a broad range of ecosystems with PAs. Their preferences in this area are intuitive because they are explicitly organized around or focused on it. They can be seen as part of a transnational activist network focused on species and ecosystem conservation (Keck and Sikkink 1998). The scientific community within this group can additionally be conceived of as an epistemic community (Haas 1992, Raustiala 1997). These groups are also important for the role that international actors may play, because their presence is part of what allows the international activity and influence to be credible to the national government (Fang and Stone 2012). These groups are organized around environmental issues, sometimes well-resourced through international connections and funding, and deeply embedded in the distributional conflict. If the fundamental conflict is green groups against extractive groups in deciding government policy, the activity of domestic green groups should clearly matter.

Additionally, the transnational connections of many of these groups, particularly NGOs, means that they are likely connected to international resources such as contacts and funding that may allow them to be even more effective, particularly in developing countries where resources for civil society organizations are scarce at the purely domestic level. Green groups represent an international actor that operates distinctly from, but also occasionally within, formal intergovernmental fora. More green groups being active in a country would mean more pressure from the green side of the distributional conflict and, therefore, more coverage of ecoregions. As discussed above, they are also most likely to coordinate with IOs that may be exerting influence in the country (Abbott et al. 2015, Dorfler and Heinzel 2022). This leads to my second hypothesis:

H3: Increased strength of domestic green groups in a country is associated with more representative PAs, all else equal.

Research Design

Units of Analysis

Because of the nature of the distributive conflict, variation at both the ecoregion level and national level are important both theoretically and empirically. For this reason, I employ a hierarchical model, as seen below. The unit of analysis of the dependent variable is the country-ecoregion-year. Independent variables vary at both the ecoregion level and country-year level.

Dependent Variable

To test the empirical implications of this argument, I analyze the **proportion of each country-ecoregion** (that is, all ecoregions within each country, splitting ecoregions by national boundaries where appropriate) that is **covered by a PA** each year between 1992 and 2020.[11] As stated, the unit of analysis for the dependent variable is country-ecoregion-year. The variable can change over time as each state's PA network expands. The aggregate effect of each ecoregion being covered is representativeness, the policy outcome of interest in this theoretical argument. The more likely that any one ecoregion has significant PA coverage, the more representative the PA network will be overall. As discussed, this is a measure of PA network quality as opposed to simply quantity or size. Measuring coverage at the ecoregion level means that a country covering all of one ecoregion but none of the rest would not give it a high "country-level" representativeness rating, which

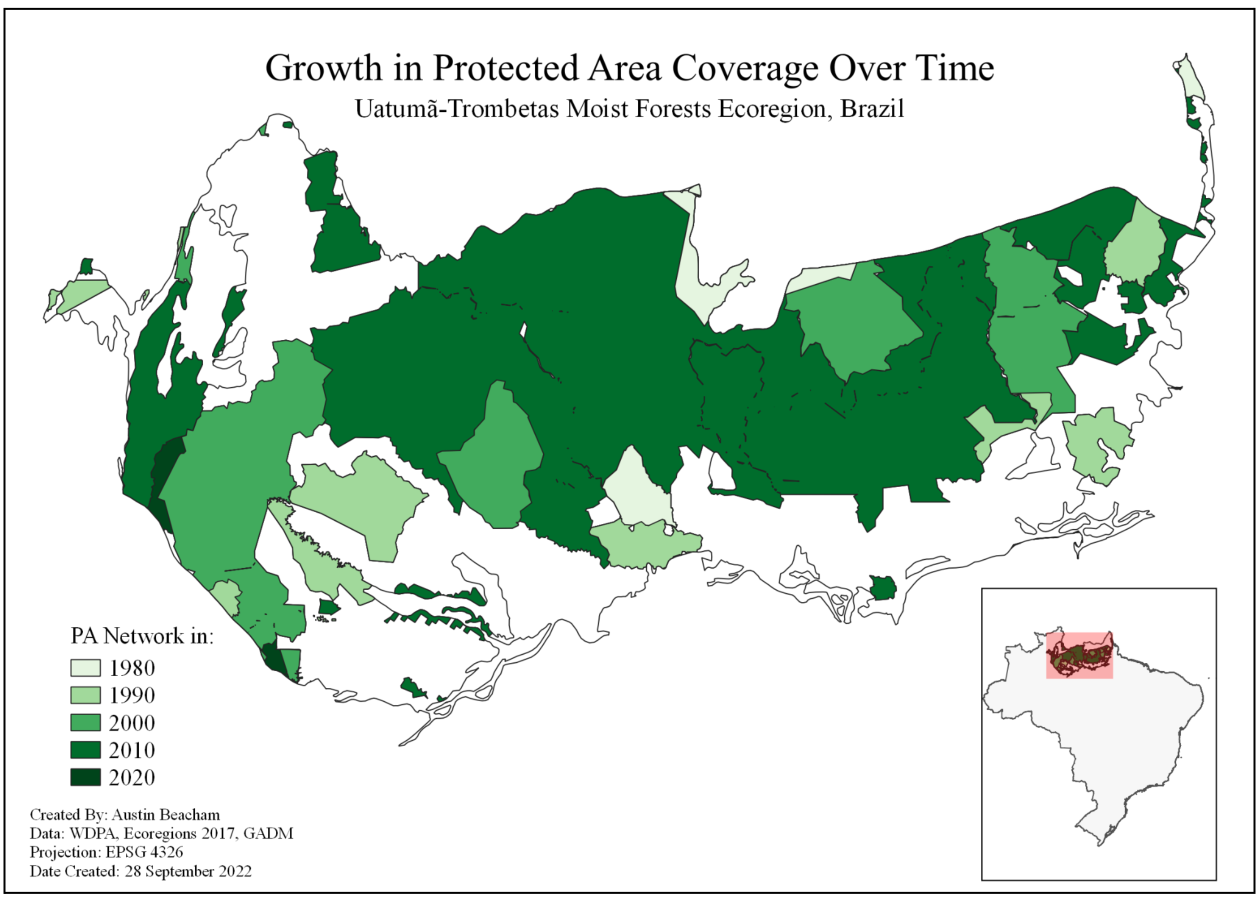
[11] Appendix with details on spatial data processing in progress.

aggregating this measure to the country level by averaging would do. An example of how this variable can change is shown in figure 5 below.

Data on ecoregions is sourced from Ecoregions 2017 (Dinerstein et al. 2017). This dataset is slightly modified, updated version of a commonly-used dataset from 2001 (Olson et al. 2001). Since conservationists and ecologists have been using a similar approach for years, and since protecting a representative sample of life on earth has been a priority even before the signing of the CBD in 1992, it is reasonable to assess coverage of ecosystems using this new map. While the creators of PAs may not have had access to these maps when designating PAs, they still had a similar goal in mind at least in theory and based on international goals.

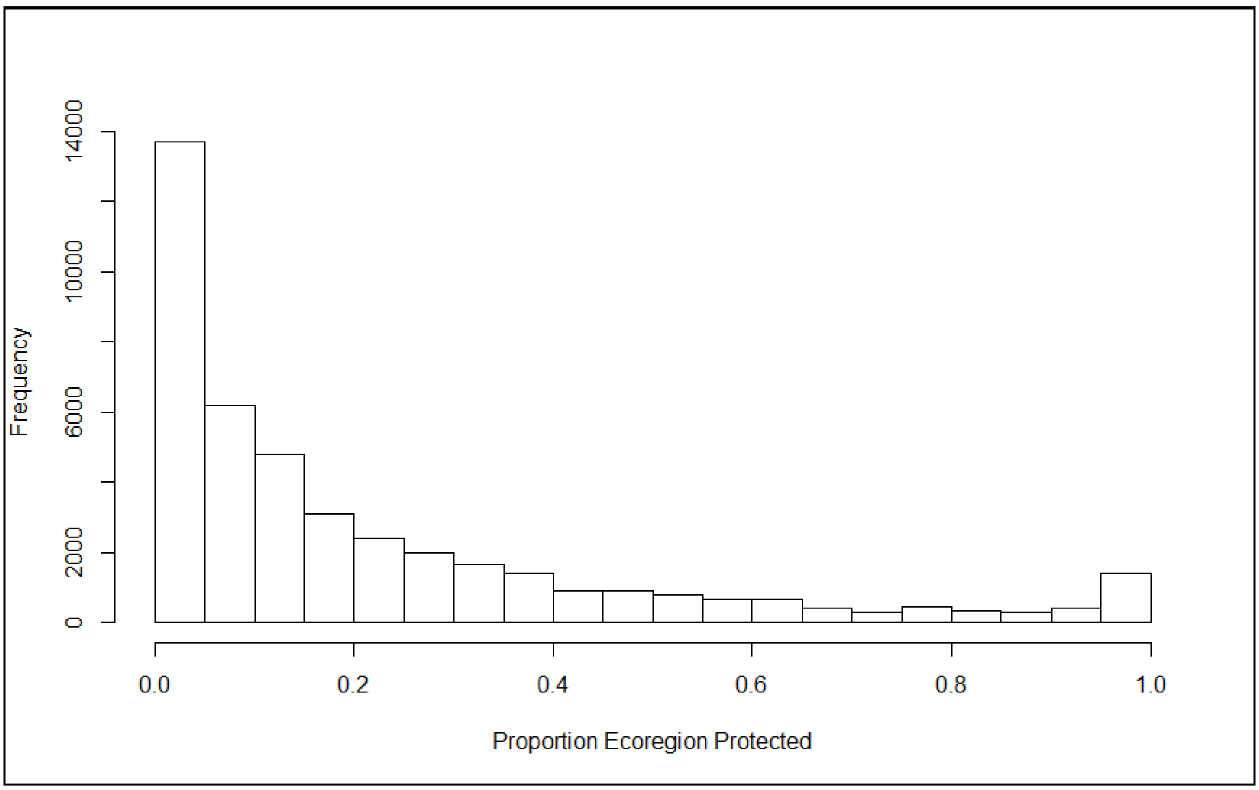
Spatial data on PAs come from the World Database on Protected Areas (WDPA), a comprehensive database from the UN Environmental Programme (UNEP) and the International IUCN. The database is updated monthly, but for my purposes I limit it to PAs designated by the

Figure 5



end of 2020 to have a consistent time period. One potential issue with this is that if a PA is degazetted, or abolished, it is removed from the WDPA data entirely, rather than having a date created and date degazetted field. However, the WDPA data is still by far the most comprehensive and representative dataset we have for PAs, and degazetting is a relatively rare occurrence. To analyze country level effects spatially, I use Database of Global Administrative Areas (GADM) at level 0 (national boundaries). To my knowledge the dataset created and used here is the first that can measure how ecoregion coverage has changed within countries, across the entire world, and over several decades. The distribution of the dependent variable is shown below in Figure 6. Very little protection is common, with frequency decreasing steadily until a jump at complete protection.[12]

Figure 6: Histogram of Dependent Variable (Country-Ecoregion-Year Protection)



[12] This is most common with small country-ecoregions, which is accounted for empirically by controlling for ecoregion size.

Independent Variables

To measure the impact of extractive interests (H1), I proxy for economic potential with **land suitability** for staple crops, derived from Ramankutty et al. (2002). This variable is similar to others used in the environmental literature as a proxy for economic cost), but is more commonly used in the social sciences. Agricultural expansion is "associated with 90% of observed cases of habitat conversion" (Venter et al. 2018, p.129). More specifically, I take the median land suitability of each country-ecoregion, derived using geospatial techniques. This variable does not change over time, but represents the fixed extractive potential of each ecoregion. My results are robust to other measure of geographically-specific economic opportunity cost.

The international environmental regime (H2) is measured using the total number of **international environmental agreements (IEAs)** in force for a particular state. This measure is used elsewhere in the literature as a proxy for participation in global environmental governance,[13] and is appropriate for the same purposes here as well. These variables are also at the country-year unit of analysis. For simplicity I use the same year as the ecoregion, but results are robust to lagging this variable various years.

To test the influence of domestic green groups (H3), I draw on data from Bernauer et al. (2013) and use a count of **IUCN-affiliated NGOs** present in each country. While many NGOs of this type are transnational, even transnational NGOs usually have dedicated office space and national employees in the countries in which they operate, and they act as domestic interest groups in the distributive conflict. Bernauer et al. acknowledge that IUCN-affiliated NGOs are more internationalized than purely domestic NGOs, but they are compellingly used to represent domestic interests in their work, and to my knowledge this data represents the best systematic collection of green NGO presence in terms of temporal and country coverage.

Control Variables

I include **total natural resource rents** at the country-year level as an alternative measure for extractive interests. I believe that the underlying agricultural potential for land will be more

[13] Data from the International Environmental Agreements Database (IEADB). See Mitchell et al. (2020) for a review of uses and Andonova et al. (2017) for an example of similar use.

important than overall natural resource rents, which can be spatially concentrated in areas like oil reserves — broad, land-based extractive interests will be powerful in the distributive conflict regardless of their overall importance to the economy. However, to account for changes over time and other types of extraction, I include this variable. I control for **GDP per capita** to account for the fact that plentiful economic resources may make effective PA creation easier. In robustness checks I included the square of GDP per capita to account for the environmental Kuznets curve hypothesis (Grossman and Krueger 1991), but results remain unchanged. I also control for the **total percent of a country's land area that is covered by each ecoregion**, since a larger ecoregion is inherently more difficult to protect.

Lastly, I control for **Participatory democracy**, from V-Dem, to measure democratic institutions that allow for the participation of civil society in public life (H2). This measure is different from the typical polyarchy index in V-Dem used by many political scientists, but better represents the aspects of democracy that might be important in the distributive conflict that I present above. There is a long debate in the literature on whether or not democracy leads to better environmental outcomes (Midlarsky 1998, Battig and Bernauer 2009, Sanford 2021, Buitenzorgy and Mol 2011). The answer seems to depend on the particular environmental issue under investigation, measurement strategy, and time period. Following selectorate theory (Bueno de Mesquita et al. 2003), democracy can lessen the power of the extractive interests discussed above by expanding the selectorate to include more groups (i.e. green groups).

However, other groups gain access in democracies that either counteract or overpower these green groups. Workers in brown industries, agricultural workers, and individuals who rely on more basic natural resource exploitation such as hunters and loggers would likely prefer less protection, because it would threaten their livelihood. Additionally, there is some evidence of the "not in my backyard" phenomenon in the case of PAs, where those most directly affected by the creation of a PA do not want it near them. Citizens would possibly feel this way even more strongly in biodiversity than climate policy, since land use has such a direct and obvious effect on their lives. Therefore, my theoretical expectations about changes in democracy within a country are unclear, and I leave it as a control variable. I investigate the relationship between PA outcomes and democracy in other work.

Model Specification

Since the dependent variable is a proportion that varies between 0 and 1, I estimate fractional logistic regressions in my main models. I include country and year fixed effects, and clustered standard errors at the country level.[14] The results are robust to including a time-trend rather than year fixed effects. To address concerns over incidental parameters in these models, conventional OLS models of the main results are included in the Appendix, with equivalent substantive results. The general expression is as follows:

$$Y_{ecy} = \gamma_{00} + \gamma_{01} LS_{e} + \gamma_{02} IEA_{cy} + \gamma_{03} NGO_{cy} + \gamma_{04} Controls_{cy}$$ [15]

Because I am using country fixed effects, I am analyzing where or changes within a country, rather than *across* countries, matter. This will be important when discussing my findings. Country fixed effects are appropriate here because my theory is based on domestic distributive conflicts that domestic and international factors influence, so I am interested in changes within countries over time, rather than comparing across countries. The Hausman test indicates a significant difference between models with and without fixed effects. As a final note, independent variables are standardized with a mean of 0 and standard deviation of 1 in the tables below so that magnitude is comparable across covariates.

Findings

The results in table 1 below provide fairly strong evidence for my hypotheses. I test each hypothesis in a bivariate model (Models 1-4) and individually with controls (Models 5-7), finding consistent significance. In the models that include all elements of the distributive conflict discussed here (Models 4 and 8), I include all three independent variables of interest together, both with and without controls. Throughout these specifications, I find support for my argument.

[14] To account for spatial autocorrelation across ecoregions, I also cluster robust standard errors at the ecoregion-level in robustness checks (currently not in appendix).

[15] Notation for multilevel model comes from Snijders and Bosker (2011).

In the "main model," Model 9, the land suitability variable is significant ($p < .01$) and in the expected direction: a increase in land suitability is associated with decrease in coverage by PAs. This is in line with the political economy argument I described above. Supporting Hypothesis 2, IEAs are significant ($p < .01$) and in the expected direction. A increase in IEAs in force is associated with an increase in protection, The coefficient for NGOs (*H3*) is also significant ($p < .01$), and provides evidence that stronger domestic green groups are associated with higher quality PA network location. Democracy is not significant, which is in line with the mixed impact of democracy that I conjecture. Ecoregion size is significant ($p < .001$) and in the expected direction — a larger ecoregion tends to be less protected, even as the distributive conflict changes over time. Other control variables do not achieve significance. It is notable that natural resource dependency, measured through natural resource rents, is not associated with less protection as one might expect. This points to the overwhelming influence of agricultural interest in land conversion that is accounted for with the land suitability variable.

Because the coefficients of logistic regression models are difficult to interpret substantively, I also include predicted value plots for my main independent variables in Model 8.[16] The plots show that a standard deviation increase in land suitability from the mean decreases predicted protection by approximately 1.5% of ecoregion area, while a standard deviation increase from the mean of IEAs and NGOs increases predicted protection by approximately 1.5% and 2%, respectively. These are substantively meaningful changes, considering the mean of country-ecoregion-year protection is 21.4%. These sorts of changes in protection represent be a 7-9% change from the mean, which would represent substantially more or less investment in biodiversity conservation.

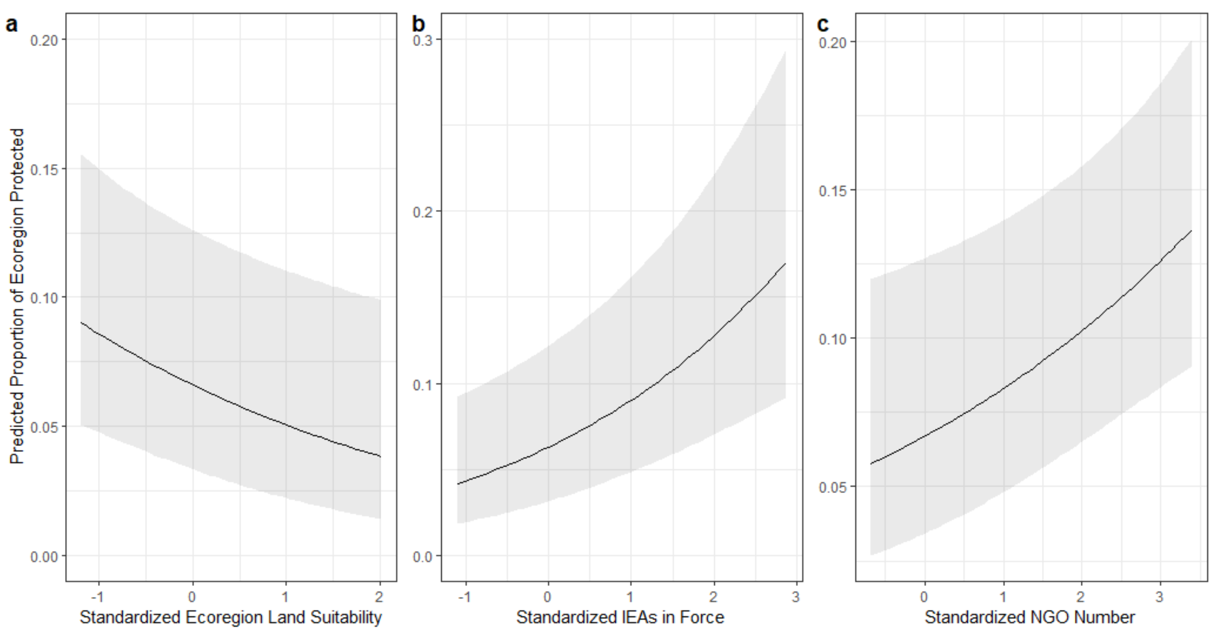
[16] These plots were made using the marginaleffects package in R (https://vincentarelbundock.github.io/marginaleffects/articles/marginaleffects.html). The package holds all other variables at their mean when creating plotting predictions.

Table 1: Testing Hypotheses 1-3

	Dependent variable:							
	% Ecoregion Covered							
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Ecoregion Land Suitability Median	−0.248**			−0.240**	−0.289***			−0.284***
	(0.098)			(0.113)	(0.096)			(0.110)
IEAs in Force		0.346***		0.379***		0.365***		0.390***
		(0.085)		(0.099)		(0.088)		(0.102)
NGO Number			0.216***	0.228**			0.222***	0.233***
			(0.081)	(0.092)			(0.080)	(0.090)
Natural Resource Rents (% GDP)					0.038	−0.006	0.007	−0.001
					(0.025)	(0.033)	(0.040)	(0.038)
Log GDPPC					0.068	0.107	0.038	−0.034
					(0.079)	(0.103)	(0.071)	(0.067)
Ecoregion Size (% Country Area)					−0.611***	−0.584***	−0.617***	−0.657***
					(0.070)	(0.066)	(0.082)	(0.085)
V-Dem Participatory Democracy					0.026	0.045	0.016	0.035
					(0.084)	(0.076)	(0.106)	(0.081)
Constant	−3.539***	−2.914***	−3.971***	−3.597***	−4.035***	−3.361***	−4.525***	−3.719***
	(0.077)	(0.159)	(0.057)	(0.167)	(0.431)	(0.550)	(0.351)	(0.384)
Observations	39,730	40,601	20,664	19,298	37,095	38,710	20,139	18,876
Log Likelihood	−13,105.350	−13,969.690	−6,313.428	−5,588.126	−11,452.480	−12,641.990	−5,873.122	−5,190.406
Akaike Inf. Crit.	26,586.710	28,365.390	12,974.860	11,498.250	23,276.950	25,679.980	12,080.250	10,698.810

Note: *p<0.1; **p<0.05; ***p<0.01

Figure 7: Predicted Values for Ecoregion Protection, Varying Main Independent Variables



Robustness

I perform several robustness checks to build confidence in my main empirical results. First, as mentioned above, I include linear and quadratic time trends instead of year fixed effects. Results are unchanged.[17] Second, I use OLS regression instead of fractional logit regression.[18] There is evidence that fractional logit regressions with fixed effects in panel data may produced biased estimates because of the incidental parameters problem. Results are robust when using OLS, with similar substantive effects to those discussed above. Third, as discussed in the variable descriptions, I included several alternative measures of key variables.[19] I use potential agricultural rents (Naidoo and Iwamura 2007), at the ecoregion level, instead of land suitability. While land suitability is a more common measure, it does not include pastureland and this is one of main drivers of land conversion, especially in the developing world. This alternative measure does, and with it results still confirm the three hypotheses. I also use squared GDPPC, to account for an environment Kuznets curve, as well as more fine-grain measures of extractive groups via oil rents, agricultural rents, and lagged natural resource rents. The main results remain robust to all of these alternative specifications, except that *H3* on domestic green groups loses significance when agricultural rents are used.

Next, I include two other variables that may be related to ecoregion protection.[20] First, I add tourism as percent of GDP to control for the growing role of ecotourism in promoting PA creation and location on valuable ecosystems. Second, I include the Environmental Performance Index's adjusted emissions growth rate for carbon dioxide, as similar measures to which have been used as a proxy for overall environmental performance elsewhere in the literature (Bättig et al. 2008, Bättig and Bernauer 2009, Von Stein 2008). By controlling for this, I attempt to reduce concerns about endogeneity in my measure of IEAs — that is, that countries with better PA policy are also more likely to join many IEAs, rather than the IEAs leading to better PA policy.

[17] Table A.1
[18] Table A.2
[19] Table A.3
[20] Table A.4

Controlling for overall environmental policy, which this construct is explicitly intended to measure according to the EPI codebook, should help with this concern.[21]

These variables are of obvious interest, but I do not include them in my main results because they are endogenous to the dependent variable. Tourism, especially eco-tourism, is affected by how much protection of ecoregions has occurred in the past: people are more likely to visit countries with vast protected areas full of intact nature, even though it is also true that more tourism likely increases the chance of future protection. Emissions trends are also affected by protection, since many healthy, natural ecosystems provide significant carbon sinks that reduce aggregate emissions. It is unclear from the codebook if the EPI's indicator takes this into account. Therefore, while emissions trends do capture environmental policy, ecoregion protection potentially also does. These controls are therefore not suitable to include in my main model, but since they are of theoretical interest, I include them as robustness.

Results are relatively robust. The main change is that the coefficient for domestic green groups (*H3*) lowers in significance to the 10% level. Tourism is not significant in any model, while the emissions trend variable is consistently significant at the 10% level as well.

Endogeneity

Despite inclusion of the EPI emissions trend variable, concerns may still remain over IEAs representing the influence of international community, rather than representing a broader commitment by the state to good environmental policy. Perhaps a state signs up to more IEAs for the same reason that it creates more representative PAs: it simply has pro-environment preferences. However, this is may not be the case. I employ an instrumental variable approach to assuage endogeneity concerns. I instrument for a country's IEAs by a inverse-distance weighted average of the lagged IEA memberships of all other countries. This measure uses a similar logic to the instruments in Acemoglu et al. (2017) and Lang and Tavares (2018) for democracy and globalization, respectively. The number of IEAs to which a country is party is affected by its neighbors' IEA memberships, since they are often joining the same agreement or are participating

[21] I perform further checks in the next section to reduce endogeneity concerns.

in regional efforts, but those neighbors' IEA memberships do not directly affect ecoregion coverage, except through the country's own IEA membership.

Results using the instrumental variable are substantively similar to the non-instrumented models above.[22] The measure of integration into the international environmental regime is still positive and significant with similarly large effect size. The measure of domestic green groups drops to 10% significance, but other results remain unchanged. The *F*-statistic for the Wald test IV regression is 60.44, indicating a strong instrument. Additionally, while there are theoretical concerns that the non-instrumented IEA measure may be endogenous, the Wu-Hausman test is insignificant. This set of endogeneity checks does not eliminate the possibility that IEAs and PAs both represent the same underlying state preference for positive environmental outcomes, but they build some confidence in my results and construct validity.

Conclusion

Protected areas represent an investment in the preservation of ecosystems and the valuable services they can provide. They are critically important to slowing the biodiversity crisis and protecting remaining invaluable carbon sinks. In this paper, I have presented a first step to understanding the political economy of PA location dynamics. I argued that domestic green groups and international environmental regimes can help encourage protection, but that extractive, economic interests are still very powerful This paper provides the foundation for future work that can delve into the mechanisms that cause the patterns that I describe, and introduces an ecologically invaluable, politically-driven dynamic to a broader political science audience.

The paper contributes to literature comparing international and domestic influences on political outcomes. I demonstrate that international political influences may as more associated with PA outcomes as domestic ones, which would be consequential. This finding demonstrates the importance of including international forces in distributive conflict models of environmental

[22] Table A.5

politics, which are the focus of a promising new strand of literature (Aklin and Mildenberger 2020; Colgan et al. 2020; Genovese 2019; Kennard 2020).

In pointing out this dynamic, I also contribute to work that analyzes the influence (or lack thereof) of nonbinding international cooperation. IEAs are a diverse set of agreements in terms of stringency and formality. Despite these limitations, this paper shows that they are associated with more environmentally friendly outcomes. Debates continue as to whether or not international cooperation can shift domestic outcomes or if it is cheap talk that represents inherent preferences. I provide some evidence that even the kind of cooperation and interaction discussed in this paper can matter.

Lastly, this paper shines a light on understudied spatial dynamics in environmental politics, and political science more broadly. Looking at national or local indicators is the most common approach in the field, but even fine-grain non-spatial data can miss important dynamics such as underlying geographic and land characteristics that shape distributive conflict. More specifically, this paper demonstrates that the spatial demands of biodiversity conservation can have both synergy and conflict with climate change mitigation. Carbon sequestration is a major service that ecosystems can provide, but renewable energy sources such as wind and solar also require land use. PAs already cover land totaling to the size of South America — understanding why they are located where they are and what politics undergird those decisions will be important for future, sustainable land use planning (Schmidt-Traub 2021).

I highlight three main ways that future work can build on this paper. First, the democracy finding can be explored beyond the planned work discussed above. Work on deforestation has looked at the role of elections and district changes (Burgess et al. 2012, Sanford 2021) — perhaps similar dynamics play out in PA designation. Second, future work could examine the particular activities of international actors that I posit matter, testing whether they have a substantive effect at a more micro, causal level. Third, this paper does not discuss the role that the broader public may have on PA placement. While some individuals may want protection near them because of the ecological value or ecotourism opportunities, others may experience NIMBYism and not want PAs near them, precisely because they prohibit legal exploitation of resources. Future work could examine the conditions under which publics prefer protection or do not, which can help flesh out the political economy argument laid out here.

This discussion should make clear that this paper offers many pathways for future study. It highlights that both domestic and international factors still matter for domestic environmental outcomes, and shows that strong economic incentives cannot be ignored when it comes to political areas as sensitive as land use. As environmental crises continue to mount, understanding the dynamics that have led to underinvestment in environmental protection is critical to moving toward a more sustainable future.

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Appendix

Figure A.1: Biome Map

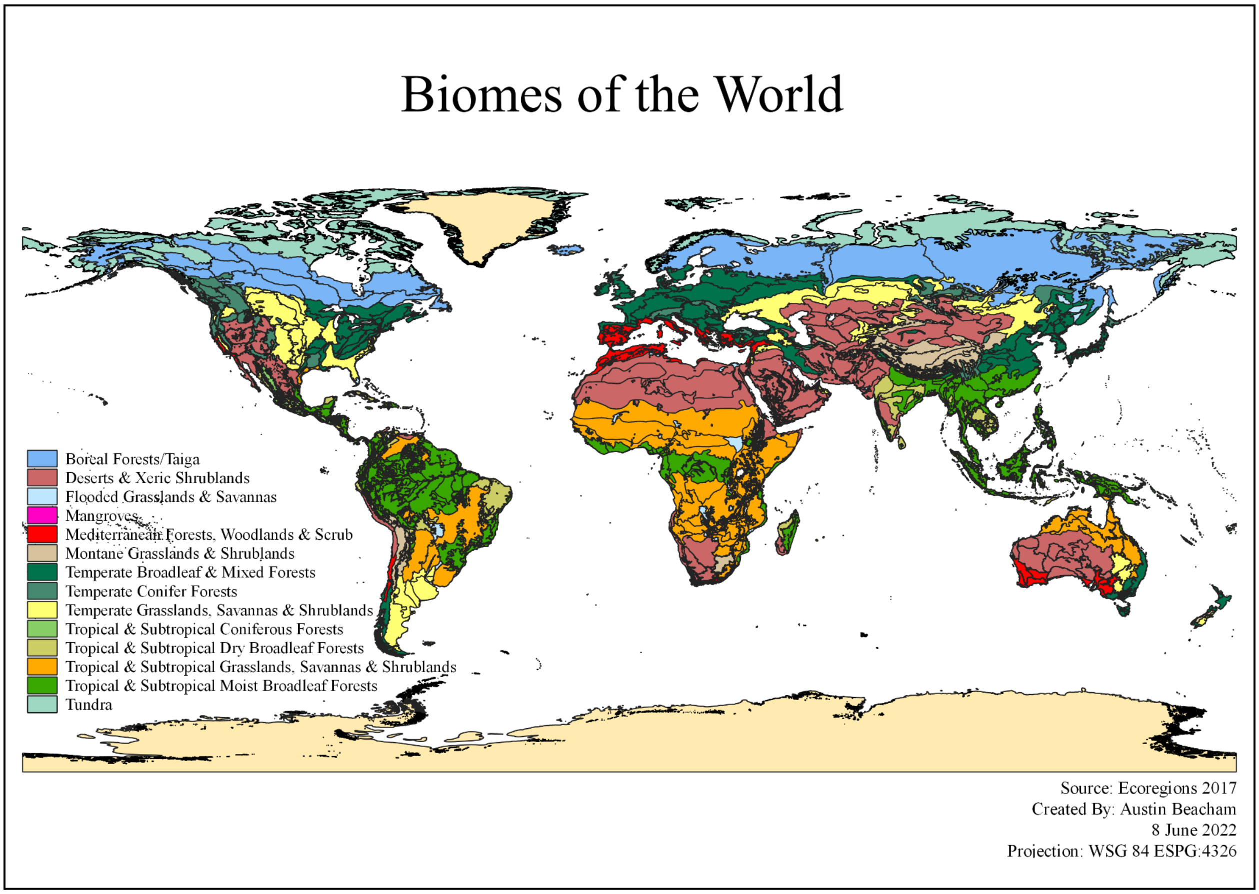

Table A.1: Linear and Quadratic Time Trends

	Dependent variable:	
	% Ecoregion Covered	
	(1)	(2)
Ecoregion Land Suitability Median	−0.284***	−0.284***
	(0.110)	(0.110)
IEAs in Force	0.382***	0.382***
	(0.102)	(0.102)
NGO Number	0.228***	0.228***
	(0.081)	(0.081)
Natural Resource Rents (% GDP)	−0.004	−0.004
	(0.033)	(0.033)
Log GDPPC	−0.052	−0.052
	(0.049)	(0.049)
Ecoregion Size (% Country Area)	−0.657***	−0.657***
	(0.085)	(0.085)
V-Dem Participatory Democracy	0.036	0.036
	(0.081)	(0.081)
Time Trend	0.013	
	(0.009)	
Quadratic Time Trend		0.00000
		(0.00000)
Constant	−28.848	−16.189*
	(18.222)	(9.113)
Country Fixed Effects	Yes	Yes
Year Fixed Effects	No	No
Observations	18,876	18,876
Log Likelihood	−5,190.761	−5,190.763
Akaike Inf. Crit.	10,671.520	10,671.520
Note:	*p<0.1; **p<0.05; ***p<0.01	

Table A.2: Main results using OLS

	Dependent variable:
	% Ecoregion Covered
Ecoregion Land Suitability Median	−0.034**
	(0.014)
IEAs in Force	0.056***
	(0.015)
NGO Number	0.024**
	(0.012)
Natural Resource Rents (% GDP)	−0.004
	(0.004)
Log GDPPC	−0.015
	(0.009)
Ecoregion Size (% Country Area)	−0.073***
	(0.009)
V-Dem Participatory Democracy	0.003
	(0.009)
Country Fixed Effects	Yes
Year Fixed Effects	Yes
Observations	18,876
R^2	0.331
Adjusted R^2	0.325
Residual Std. Error	0.188 (df = 18717)
Note:	*p<0.1; **p<0.05; ***p<0.01

Table A.3: Regression results using alternative measures of land suitability, GDPPC, and resource rents

	Dependent variable:				
	% Ecoregion Covered				
	(1)	(2)	(3)	(4)	(5)
Ecoregion Potential Agricultural Rents	−0.254**				
	(0.103)				
Ecoregion Land Suitability Median		−0.284***	−0.284***	−0.256***	−0.285***
		(0.110)	(0.110)	(0.092)	(0.110)
IEAs in Force	0.364***	0.390***	0.391***	0.387***	0.392***
	(0.101)	(0.105)	(0.102)	(0.106)	(0.103)
NGO Number	0.247***	0.232***	0.232***	0.124	0.233***
	(0.084)	(0.085)	(0.088)	(0.102)	(0.089)
Natural Resource Rents (% GDP)	0.003	−0.001			
	(0.038)	(0.038)			
Oil Rents (% GDP)			0.025		
			(0.062)		
Agricultural Rents (% GDP)				0.042	
				(0.048)	
Lagged Natural Resource Rents (% GDP)					−0.009
					(0.027)
Log GDPPC	−0.009	−0.029	−0.034	−0.036	−0.031
	(0.065)	(0.276)	(0.064)	(0.065)	(0.065)
Squared Log GDPPC		−0.0003			
		(0.017)			
Ecoregion Size (% Country Area)	−0.607***	−0.657***	−0.657***	−0.649***	−0.660***
	(0.074)	(0.085)	(0.085)	(0.083)	(0.086)
V-Dem Participatory Democracy	0.025	0.034	0.039	0.013	0.035
	(0.083)	(0.082)	(0.083)	(0.083)	(0.081)
Constant	−3.877***	−3.735***	−3.695***	−3.902***	−3.757***
	(0.370)	(1.027)	(0.374)	(0.438)	(0.390)
Country Fixed Effects	Yes	Yes	Yes	Yes	Yes
Year Fixed Effects	Yes	Yes	Yes	Yes	Yes
Observations	19,524	18,876	18,878	17,939	18,764
Log Likelihood	−5,446.320	−5,190.401	−5,190.898	−4,948.954	−5,181.746
Akaike Inf. Crit.	11,224.640	10,700.800	10,699.800	10,215.910	10,679.490
Note:					*p<0.1; **p<0.05; ***p<0.01

Table A.4: Including Tourism and Emissions Trends

	Dependent variable:		
	% Ecoregion Covered		
	(1)	(2)	(3)
Ecoregion Land Suitability Median	−0.282***	−0.283***	−0.282***
	(0.107)	(0.107)	(0.107)
IEAs in Force	0.403***	0.409***	0.418***
	(0.099)	(0.094)	(0.096)
NGO Number	0.157*	0.145*	0.151*
	(0.084)	(0.083)	(0.085)
Natural Resource Rents (% GDP)	0.006	0.010	0.019
	(0.040)	(0.035)	(0.038)
Log GDPPC	−0.047	−0.035	−0.020
	(0.082)	(0.071)	(0.079)
Ecoregion Size (% Country Area)	−0.674***	−0.661***	−0.674***
	(0.085)	(0.083)	(0.085)
V-Dem Participatory Democracy	0.022	0.025	0.028
	(0.058)	(0.063)	(0.060)
Tourism (% GDP)	0.059		0.060
	(0.055)		(0.055)
EPI Emissions		0.043*	0.043*
		(0.024)	(0.024)
Constant	−1.138*	−3.625***	−1.278**
	(0.604)	(0.385)	(0.593)
Country Fixed Effects	Yes	Yes	Yes
Year Fixed Effects	Yes	Yes	Yes
Observations	15,317	15,577	15,317
Log Likelihood	−4,406.619	−4,433.397	−4,406.387
Akaike Inf. Crit.	9,115.239	9,180.794	9,116.773
Note:			*p<0.1; **p<0.05; ***p<0.01

Table A.5: OLS with Instrument Variable for IEAs.[23]

	Dependent variable:
	% Ecoregion Covered
Ecoregion Land Suitability Median	−0.038***
	(0.013)
IEAs in Force (Instrumented)	0.065***
	(0.023)
NGO Number	0.024*
	(0.012)
Natural Resource Rents (% GDP)	−0.004
	(0.004)
Log GDPPC	−0.015
	(0.009)
Ecoregion Size (% Country Area)	−0.071***
	(0.009)
V-Dem Participatory Democracy	0.003
	(0.009)
Constant	0.112*
	(0.068)
Country Fixed Effects	Yes
Year Fixed Effects	Yes
Observations	18,568
R^2	0.338
Adjusted R^2	0.332
Residual Std. Error	0.186 (df = 18412)
Note:	*p<0.1; **p<0.05; ***p<0.01

[23] I have not had success running a IV GLM regression, but since my main results are robust to OLS, I employ OLS here.