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What Drives the Erasure of Protected Areas? Evidence from across the Brazilian Amazon

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ABSTRACT

Protected areas (PAs) are a widely used strategy for conserving forests and ecosystem services. When PAs succeed in deterring economic activities that degrade forests, the impacts include more forest yet less economic gain. These economic opportunity costs of conservation lead actors with economic interests to resist new PAs, driving their sites away from profitable market centers and towards areas featuring lower opportunity costs. Further, after PAs are created, economic actors may want PA downgrading, downsizing, and degazettement (collectively PADDD). We examine reductions in PAs' spatial extent – downsizings (partial erasures) and degazettements (complete erasures) − that presumably reduce protection. Using data for the entire Brazilian Amazon from PADDDtracker.org, our empirical analyses explore whether size reductions from 2006 to 2015 resulted from bargaining between development and conservation. We find that the risks of PA size reductions are raised by: lower travel costs (as implied by distances to roads and cities), which affect economic gains and enforcement; greater PA size, which affects enforcement; and more prior internal deforestation, which lowers the impacts of size reductions. These dynamics of protection offer insights on the potentially conflicting factors that lead to PA size reductions, with implications for policymaking to enhance PA effectiveness and permanence.

1. Introduction

Establishing national parks and other types of protected areas (PAs) is the most extensively employed tool to conserve biodiversity (Deguignet et al., 2014; Naughton-Treves et al., 2005; Watson et al., 2014). Over 23% of lands are classified as in PAs within Latin America, with a particularly high concentration in Brazil, which is responsible for over half of those lands (UNEP-WCMC, 2020).

Restrictions inherent in protection generate conflicts over land use between advocates for biodiversity conservation and advocates for economic development (hereafter, 'conservation' versus 'development') (Deguignet et al., 2014; Naughton-Treves et al., 2005; Watson et al., 2014). PAs can deter development (Albers, 2010; Naughton-Treves et al., 2005; Nicolle and Leroy, 2017). In turn, lobbying against PAs by those actors who prefer development often leads PAs to be located where the economic opportunity costs of protection are lower (Baldi et al., 2017; Joppa and Pfaff, 2009; Pfaff and Robalino, 2012). Locating PAs where the forces of economic development are lower weakens PAs' forest impacts: even a fully forested PA may not indicate much conservation impact, or any, when that PA is quite far from market centers (Anderson et al., 2016; Kere et al., 2017; Pfaff et al., 2017; Robalino et al., 2017). Even without PAs, little of the forests in such locations would be lost, since many economic activities are not profitable there (Abman, 2018; Ferraro et al., 2013; Nolte et al., 2013; Pfaff et al., 2015a, Pfaff et al., 2015b; Pfaff et al., 2014). Yet while the impacts of some PAs can be as low as zero, studies which control for biases in PAs' locations conclude that, on average, PA networks deter some human activities and, thereby, lower deforestation on average (Andam et al., 2008; Joppa and Pfaff, 2011; Jusys, 2018; Pfaff et al., 2009; Robalino et al., 2017; Sims, 2014). Thus, PAs may indeed generate conflicts because of associated land-use restrictions.

Lobbying against PAs may continue even after PA establishment. Yet at that point, such lobbying would be for PA downgrading, downsizing and degazettement (PADDD) (Qin et al., 2019), i.e., legal changes in PA status or PA size (Mascia and Pailler, 2011). Following Mascia

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and Pailler (2011): downgrading is "a decrease in legal restrictions on the number, magnitude, or extent of human activities within a PA"; downsizing is "a decrease in size of a PA as a result of excision of land or sea area through a legal boundary change"; and degazettement is "a loss of legal protection for an entire PA". The common proximate causes of PADDD are economic activities related to industrial scale resource extraction and development and, to a lesser degree, local land pressures and land claims (Golden Kroner et al., 2019).

PADDD events affect the forest impacts from establishing PAs (Pack et al., 2016). Forrest et al. (2015) find that PADDD raised the carbon emissions from deforestation within a number of tropical countries (specifically the Democratic Republic of Congo, Malaysia and Peru), while Golden Kroner et al. (2016) document increased habitat fragmentation in the US. Thus, the occurrence of PADDD events clearly affects optimal public decisions on PA establishment, type, location, and enforcement. Designing more robust PA networks going forward − with strategies to enhance durability and permanence, as well as impacts, while pursuing realistic targets − clearly depends on PADDD and PA networks' risk factors.

Yet the implications of PA dynamics for future PA decisions depend on how PADDD happens, exactly, as well as where PADDD events occur. Forest impacts of PADDD depend on a PA's prior effectiveness: if a PA was well-enforced, then PADDD could unleash deforestation; but if a PA was not enforced, so that economic activities occurred inside the PA (Tesfaw et al. (2018) find that the frequency of PADDD rises with the deforestation in a PA), then PADDD may not affect forest loss. As a result, conservation advocates might contest PADDD less where it matters less, i.e., the PAs which already were ineffective.

Looking at factors in PADDD frequencies, Symes et al. (2016) find that PAs' sizes raise degazettement, over 44 countries and 110 years. Tesfaw et al. (2018) find that more deforestation inside PAs within the Amazonian state of Rondônia increases their risk of degazettements or downsizings in 2010 and 2014.

As in many countries, Brazilian agencies' objectives and policies vary across space, over time, and also across agencies. For instance, the desire to placate rural development interests can politically internalize economic interests that lobby against the creation of PAs or, after PA establishment, for PADDD events (Bernard, 2014; de Marques and Peres, 2015). Changes in agencies' objectives are likely to be linked to the economy, federal budgets, and elections. In Brazil, from 1980 to 2000 there was a considerable public effort to extend the government's PA networks. Over time, though, nearly 7% has been lost from their SNUC (*Sistema Nacional de Unidaded de Conservação*) or PA system (Supplementary Materials in Golden-Kroner et al., 2019). Proposals for PADDD events – especially size reductions − rose greatly after 2000, putting at risk 10% of the PA estate (Supplementary Materials - Golden-Kroner et al., 2019), given public support for economic gains (Bernard, 2014; Soares-Filho et al., 2014). By 2012, this public orientation resulted in a new forest code, which made development projects easier (Soares-Filho et al., 2014), as illustrated by a doubling of prices for forest lands between 2010 and 2012 (Miranda et al., 2019). For optimal conservation strategies, it is important to understand that such shifts in priorities are one part of the typical political dynamics regarding economic development (Carvalho et al., 2019).

To better inform future decisions by extending understanding of PADDD, we assess how conservation-development conflicts over PAs have contributed to PADDD events across the entire Brazilian Amazon. We focus on PA size reductions, or erasures, because both downsizings and degazettements presumably imply an effective reduction in the constraints upon economic development activities.[1] We do not study here the downgrading of PAs. While it can be the case that permitting a greater set of activities in PAs – as in downgrading from a strict PA to a multiple-use PA − negatively affects forests and biodiversity (Mascia et al., 2014), such a shift might primarily reduce conflict by reconciling PA management with traditional local land use. A downgrade, then, might not erode but instead increase effective protection (Naughton-Treves and Holland, 2019), in particular via increased buy-in by locals (roughly half of the downgradings were for "rural settlement" or "subsistence"). Global average (Nelson and Chomitz, 2011) and Brazilian Amazon (Pfaff et al., 2014) results both have demonstrated that multiple-use PAs sometimes have had greater forest impacts than strict PAs. Changing PA type is thus not the same as eliminating protection.

We contribute both theoretically and empirically to scientific understanding of the dynamics of PA size reductions. We formalize the conceptual framework in Tesfaw et al. (2018), then add one critical issue, deforestation inside of imperfectly enforced PAs, which greatly influences impacts of size reductions. After describing some benefits and costs assumed to be central within conflicting agencies' objectives, we consider how interactions between agencies, over PA reductions, might play out across landscapes: spatial gradients in benefits and costs affect where each agency is most for, or against, size reductions. Both benefits and costs feature travel costs − which links conceptual discussions to our empirical work.

Empirically, we analyze PADDDtracker.org Data Release Version 1.1 (Conservation International and World Wildlife Fund, 2017) concerning PA size reductions observed for the entire Brazilian Amazon. We use measures of land and PA characteristics that seem relevant for agencies' decisions, stressing the economic opportunity costs of PAs. We examine a binary indicator for 'reduced in size' (degazetted or downsized), with a logistic probability model, to study the determinants of size reductions from 2006 to 2015. As the weight placed upon conservation likely varies across the states in the Brazilian Amazon (Abman, 2018; Ferraro et al., 2013; Pfaff et al., 2015a, 2015b), we use state dummies to control for fixed but unobserved state differences, which might influence these decisions about PA size reductions.

We find that PA size reductions are affected by factors in PA enforcement and opportunity costs, such as travel costs (measured by distance) that significantly affect private landusers' profits as well as public enforcement. In particular, we find that PA size reductions are more common nearer to cities. This result suggests bargaining power for development: with the deforestation inside PAs being more common far from cities, size reductions nearer to cities bring more environmental concern but also more economic development gains. We also find that, as in Symes et al. (2016), the risk of reductions increases with PA size, which raises enforcement costs. Lastly, greater prior deforestation inside PA boundaries, which indicates a lack of PA effectiveness, increases the risk of size reduction. That result suggests bargaining power for conservation because size reductions bring less environmental concern when there has been more internal deforestation. Thus, development and conservation agencies both seem to have influences on where size reductions occur. That conclusion, from our full set of results, suggests agency bargaining rather than any process dictated solely by economic development. Further, the nature of these results is consistent across PA subsets defined by sub-region, PA type, and level of government − implying that bargaining is a sensible framing of decisions about past PA size reductions across the Brazilian Amazon. Our results offer a new empirical perspective on what to expect from PA creation and PA enforcement.

Below, Section 2 presents a model with two agencies, focused on economic development and ecosystem conservation, respectively, each with spatial gradients in their views about PA size reductions. Section 3 presents the data and our empirical strategy, Section 4 our results, and Section 5 additional discussion.

[1] This is harder to assert if a PA degazettement is followed by relabeling as Indigenous Lands, which can deter activities. In our dataset, though, none of the downsizings or degazettements were followed by a designation as Indigenous Lands.

2. Agency Perspectives on PA Size Reductions

2.1. Agency Benefits/Costs from Reducing Enforced PAs

In formalizing and extending ideas in Tesfaw et al. (2018), we consider existing PAs. PA sites and their costs are not issues, as PAs already are established. We consider the net benefit or net cost of continuing protection versus allowing a PA size reduction to occur. Thus, the choices to be made concern which PAs are left untouched versus which are reduced. Formalizing this, for every PA *i* the choice is to reduce size ($R_i = 1$) or not to reduce ($R_i = 0$). Reductions (R_i) refer to degazettement or downsizing; either of those events lowers PA size. We have an environment agency (*E*) as well as a development agency (*D*). Given these interests, agency bargaining determines whether any given PA suffers a size reduction (R_i).

One key issue is profitability. When a PA has its intended positive impacts − improving environmental outcomes by deterring some economic activity – some private profits necessarily are foregone.[2] A PA's opportunity cost (OC $= \sigma_i$) can equal the entire potential profit, if the PA does not allow any activities, but more generally a PA's OC (σ_i) is the fraction of profit foregone due to the PA (noting that multiple-use PAs allow some activities and thus profits). OC varies with land characteristics that affect profits, i.e., characteristics that raise profits raise economic loss (σ_i) from protection. For any given conservation gain from a given PA, for a higher economic OC (σ_i) that PA offers lower total social welfare, on net.

2.1.1. Development Agency (D)

For agencies with development objectives, PAs are constraints whose costs rise with OC (σ_i). Thus, the development agency *D*'s economic gain from a PA size reduction is $\delta\sigma_i$. Its (simple) preferences are:

$$B^{D}(R_i) = \delta\ \sigma_i\ R_i$$

$$R_i^{*} = 1\ ?\ \sigma_i \geq 0$$

The development agency D would like all of the PAs with positive OCs to be reduced in size, with even stronger preferences for reducing those PAs that have higher opportunity costs (σ_i). When considering social welfare or agency bargaining, we can overlay these views with the environmental agency's views.

To consider whole landscapes, presuming dependable determinants of σ_i we assume a profit function $\pi_i = (P^Q - T_i)^*Q_i - (P^K + T_i)^*K_i$, with market prices ($P^{Q,\ K}$) for goods (Q) and capital inputs (K), plus travel costs (T_i) to any PA_i. High prices (P^Q) for goods such as soy, gold or energy, as well as high yields (Q, affected by rainfall and topography) affect profits. Below, however, our exposition will be focused upon travel costs (T) that are a factor in not only the PAs' opportunity costs but also their enforcement costs.

2.1.2. Environment Agency (E)

PA environmental gains are achieved by deterring activities that, without protection, would have led to environmental loss. Profitable activities imply high OCs of PAs, and environmental gains from PAs, while low economic profits imply low OCs of PAs but also less consequential impacts from protection. Specifically, PA environmental benefits can be computed as the environmental value for any area (V) multiplied by the probability that area is developed without protection. That baseline risk of damage (or d_i^D) rises with profits or opportunity cost (σ_i), so environmental agency *E*'s benefit B^E from avoiding a size reduction ($R_i = 0$) rises with OC (σ_i), just as did the development agency D's benefit from having a PA size reduction ($B^D(R_i = 1)$). *E* prefers no reductions if PAs deter environmental loss (any positive σ_i):

$$B^{E}(R_i) = V\ d_i^{D}(\sigma_i)\ (1 - R_i)$$

$$R_i^{*} = 0\ ?\ \sigma_i \geq 0.$$

2.1.3. Tradeoffs

Higher OC raises both environmental loss for *E* and economic gain for D from a size reduction ($R_i = 1$). Thus, a higher OC does not make a PA look better, socially speaking, or worse. As in Pfaff and Sanchez-Azofeifa (2004), for optimal size reductions (R_i) a society might search for PAs where *D* and *E* views are less correlated: e.g., $R_i = 1$ where economic gains from a size reduction are high but environmental losses are low; or $R_i = 0$ where economic gains from a size reduction are low but environmental losses are high. One basis for making such choices may be factors independent of the economic OC, e.g., values of species (V).

We note that *E* might make socially efficient decisions about R_i if faced with the OC σ_i. Analogously, development agency *D* might make socially efficient R_i decisions if forced to trade off with PA gains. The former occurs if *E* pays to conserve on private land – e.g., payment for ecosystem services (PES) − or buys land for PAs or expends limited political capital to counter lobbies for economic development. For instance, if required to surrender a PA, *E* might keep the PAs where valued species (high *V*) thrive.

2.2. Allowing for Illegal Environmental Damage in PAs

2.2.1. Probabilities of Illegal Activities

The discussion above presumes that, once any PA is established, no illegal damages occur inside of its boundaries (no deforestation occurs or we can redefine this as the legally permitted amount occurring[3]). Put another way, all of the PAs considered above have perfect and costless enforcement: if financial or political capital is spent to establish a PA, and to maintain it, then all the lands inside are fully protected.

Yet, in fact, enforcement varies. Given enforcement effort, higher profits raise the benefits from illegal activities: low travel costs (T), e.g., raise profits from illegality. Yet low T also improves enforcement (Ferraro et al., 2011; Sims, 2014). Thus, T's level does not predict the probability of illegality: near to a city, low T raises private profits from illegality, yet also facilitates public monitoring and enforcement. If enforcement is perfect, then there are no illegal environmental damages inside any established PA. At the other extreme, if for any PA there is no enforcement, then there is no effective protection,[4] i.e., de facto degazettement. In that case, environmental impacts from official size reductions would be zero: whatever development would occur without any protection is the same as what occurs with official PAs.

Beyond those cases, it is theoretically unclear how effects of T play out across a landscape. We do not take a stand, theoretically, but consider two importantly distinct cases: illegal use rises or falls with T. We describe agency preferences in each case, then compare to the data on observed PA size reductions. Thus, our empirical conclusions will include inference about which of these two effects of T dominates.

2.2.2. Implications for Agencies from Illegal Activities

To represent those gradients across a landscape, consider a non-zero probability of illegal damage (d_i^I), i.e., the protection inside a PA is imperfect. We consider two cases − both linear in T, for simplicity − that yield importantly distinct possibilities: either illegal damage (d_i^I

[2] Tourism can generate meaningful economic gains based on protection, as illustrated in Naidoo et al. (2019). However, while this possibility should be considered if tourism is common, it is not common for PAs in the Brazilian Amazon.

[3] Within our empirics, we cannot distinguish between illegal deforestation and internal from permitted economic activities.

[4] "Paper parks" − in a sense of no enforcement resources − have avoided deforestation under certain condition by shaping capital intensive behavior, yet that is due to future prospects of stronger enforcement (see, e.g., Blackman et al., 2015).

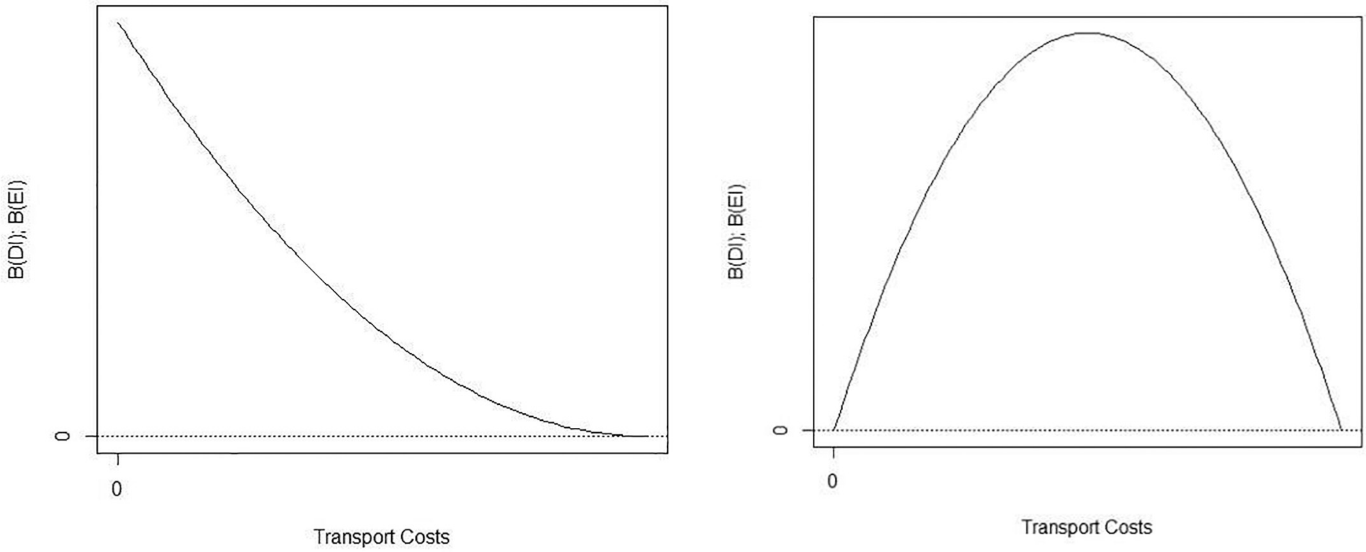

Fig. 1. (1A & 1B) D benefits (B^{DI}) & E costs (B^{EI}) of PA size reductions given internal damages.

(T)) rises with T, i.e., is lower near cities due to strong monitoring; or illegal damage (d_i^I) falls with T, i.e., is relatively high near to cities, since economic pressure is higher there. Each scenario has implications for each agency. If extraction occurs even with protection, D's gains from a PA size reduction are lower. From above, in the extreme $B^{DI} = 0$: when protection is fully unenforced, there is no difference between size-reduced and original-size PAs. E's benefits from keeping the original PA are also lower if there is illegal damage ($B^{EI} < B^E$). Thus, B^D and B^E are lower by the fraction of PA environmental value that has already been lost (1 - d_i^I).

$$B^{DI}(R_i) = B^D(R_i)\,(1 - d^I{}_i) = \delta\,\sigma_i\,R_i\,(1 - d^I{}_i)$$

$$B^{EI}(R_i) = B^E(R_i)\,(1 - d^I{}_i) = V\,d_i{}^D(\sigma_i)\,(1 - R_i)\,(1 - d^I{}_i)$$

It is still true that if either agency were to dictate R_i, given positive profits and baseline environmental loss, D would choose reduction while E would choose protection. Yet without illegality, all else equal, nearer to cities is where D most wants to have PA size reductions. Going beyond Tesfaw et al. (2018), we can hypothesize different patterns over space in terms of benefits and costs from PA size reductions.

Fig. 1A and B show relevant possibilities for when T has a clear net effect on illegal damages in protected areas. Fig. 1A presumes that illegal PA damages are less likely near cities (low T) because the higher effective monitoring nearer to cities outweighs the higher pressure there. Profits from clearing and thus baseline development fall with T. Thus, the probability of illegal activity is rising with T. As we move to the right in Fig. 1A, the illegal damages rise. Thus, both the benefits for D and the losses for E from reductions ($R_i = 1$) fall more steeply to the right, and they always fall with T. For this case − as for perfect costless enforcement − agency D pushes reduction near cities, where E most fights them. Thus, if size reductions occur near cities, that suggests bargaining power for development agencies (D).

Views on size reductions differ given the assumptions underlying Fig. 1B, where illegal damages are expected to occur more near cities because the higher profits there win out over the ease of monitoring (again, we cannot be sure a priori which net effect T will have, but we will examine internal damages).[5] Near cities, E effectively already loses most PA value and, thus, loses less from reduction. There is also low value for E from keeping PAs very far from cities, where pressure is low. At intermediate distances where profit and economic pressure remain high despite illegal deforestation, we see more gain for the development agency D from a size reduction ($R_i = 1$), as well as more loss for conservation agency E. A big difference in Fig. 1B is that E would focus less on contesting PA size reductions near to cities.[6]

The above extension of our simple discussion of bargaining considered *expected* illegality within PAs, given the benefits of illegal activities inside PAs as well as costs of public monitoring and enforcement. However, as time passes, agencies also observe the *actual* illegal activities within the PAs and, thereby, can update perspectives on each PA. Thus, actual PA illegalities should also affect PA size reductions.

3. Data & Empirical Strategy

3.1. Data

3.1.1. Scope & Observational Units

The Brazilian Amazon comprises nine states (Roraima, Amazonas, Acre, Rondônia, Amapá, Pará, Mato Grosso, Tocantins and the western part of Maranhão) covering over 5 million km2. In 2010, over one third of this enormous region was under some form of protective zoning, namely Conservation Units (CUs) and varied territories of traditional occupation (Indigeneous Land and Quilombola Territories) (Veríssimo et al., 2011). CUs are managed by the federal, state, or municipal governments and they can be classified according to their degrees of permitted intervention (strict conservation or sustainable use). Sustainable-use PAs may allow for economic activities and thereby limited legal deforestation. We do not have sufficient information about legal deforestation, hence we examine all internal deforestation.

Our observational units are CUs (here PAs). We do not consider the fates of unprotected, unzoned land or traditional occupation. For PA boundaries, we use the World Database on Protected Areas (WDPA) from the IUCN (IUCN and UNEP-WCMC, 2016) − dropping both

[5] Consider $B^D(T)$ = Profit(T) = $B^E(T)$ = Baseline Environmental Damage $d_i^D(T)$ = 10-T, for T = 0–10. Illegal damage $d_i^I(T)$ = .1 T, rising with T, or (1 - .1 T), falling with T. Thus, (1 - $d_i^I(T)$) is either (1 - .1 T) or .1 T, falling or rising. If (1 - $d_i^I(T)$) = (1 - .1 T), in fact, then $B^{DI}(T) = B^{EI}(T) = 10–2\,T + .1\,T^2$, in which the gains or losses of $R_i = 1$ will always fall with T. If (1 - $d_i^I(T)$) = .1 T, though, $B^{DI}(T) = B^{EI}(T) = T - .1\,T^2$, implying that the gains or losses of of $R_i = 1$ rise then fall in T.

[6] If profits are very flat, in T, while illegal damage is more likely near cities given higher pressure with low travel costs, costs to E of $R_i = 1$ could rise with T. Since we do not think that profits are very flat in T, however, we ignore this case.

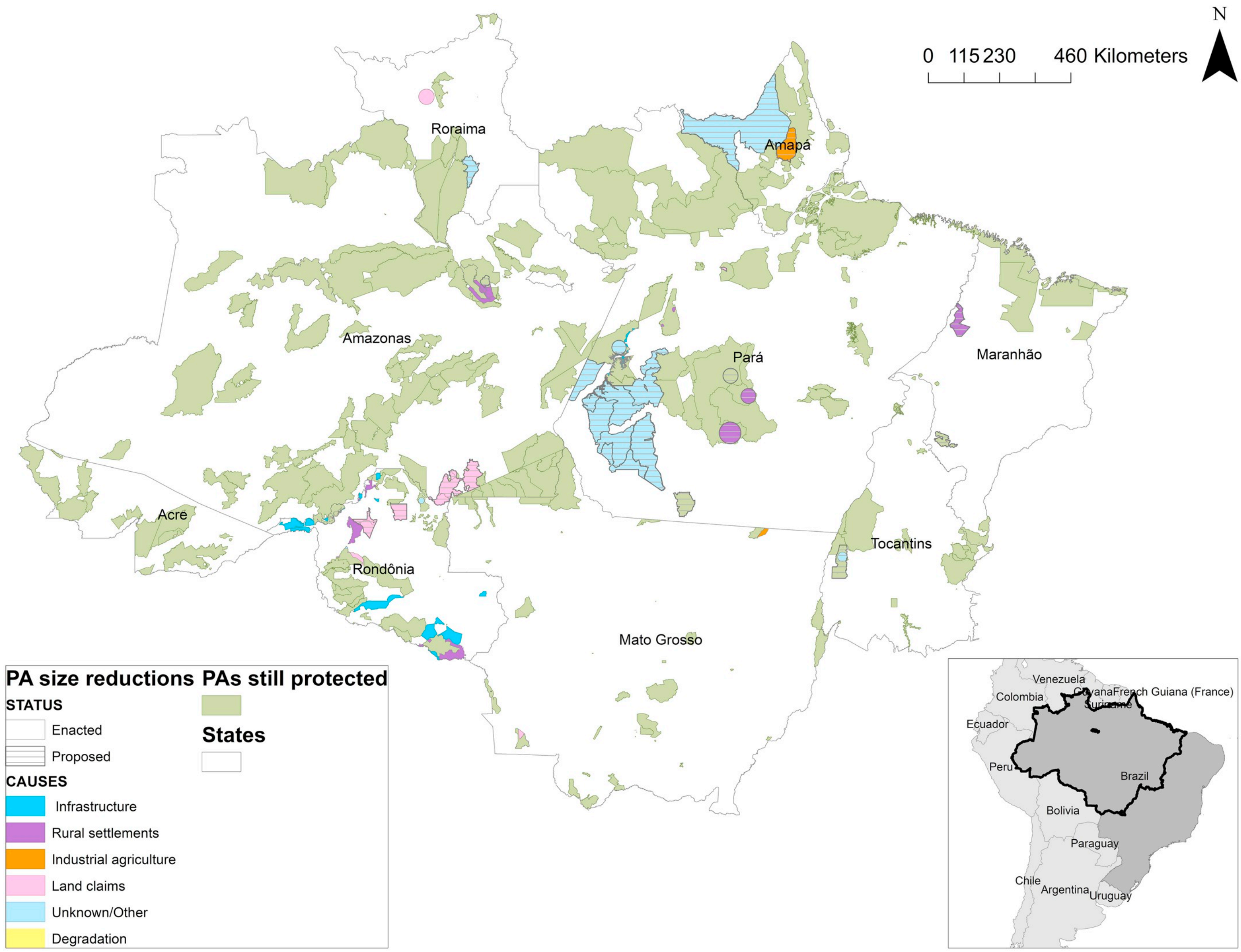

Fig. 2. Listed Proximate Causes of Braziliam Amazon PA Degazettements & Downsizings.

Indigenous Lands and Quilombila Territories that do not suffer official PADDD – which also has a location and IUCN category per PA.

3.1.2. Dependent Variable (PADDD)

We use the spatially explicit PADDDtracker.org Data Release Version 1.1 (Conservation International and World Wildlife Fund, 2017), which provides a location and description for each for size reduction of a state-managed PA. Events are classified by type (downgrading, downsizing, degazettement), status (enacted versus proposed), and listed primary cause (hydropower, other infrastructure, rural settlement, broad policy changes, and other causes (Fig. 2)). Other facts include the year of PADDD enactment.

These two databases have been compared so that, for each point in time, each PA is indicated as either having the same boundaries as at the start of our period or, instead, having undergone a size reduction (i.e., downsizing, eliminating part of a PA, or degazettement, eliminating a PA entirely). A few PAs have been downgraded to have lowered status (e.g., strict PA to extractive reserve). We exclude all downgrade events from our analysis since, again, we cannot be sure that such a change in status lowered effective protection. Yet where PA size is reduced, we are confident effective protection does not rise.

A dummy variable indicates the PAs that suffered either degazettement or downsizing. Degazettement is more severe, and the downsizings vary in size, however the set of size reductions is limited enough that we combined them. For a PA, we examine whether any size reduction occurred from 2006 to 2015.

From 1998 to 2009, 77 PAs in the Brazilian Amazon experienced PADDD events (Pack et al., 2016). Most were degazettements (30) and downsizings (44) that, in total, reduced 'the PA estate' by over 20% (Veríssimo et al., 2011). Most PADDD events were enacted, i.e., passed into law (48), but 29 remained proposals (Pack et al., 2016). PA creation follows a clear process of civil discussions and technical studies, while size reductions are proposed and enacted federally with less consultation (Bernard, 2014; Pack et al., 2016; Veríssimo et al., 2011). Our analyses consider enacted and proposed events, as we are interested in intentions to remove protection (at least partially, i.e., including the downsizings). Most of these reductions in PA sizes were from 2006 to 2015 (30 degazettements and 21 downsizings).

3.1.3. Independent Variables

We collected data for other independent variables during the 2000–2005 time period. These dates avoid endogeneity by depicting the landscapes before size reductions. We obtain measures for the independent variables for all intact PAs (281 observations) and size reductions from 2006 to 2015 (51 observations).

We use 2000–2005 average municipal Gross Domestic Product (GDP), both its level and growth (IBGE, 2017), as proxies for economic activity. We measure development pressure using access to markets, agriculture profitability and population (Tesfaw et al., 2018): distance to nearest urbanized area in 2005; distance to nearest road in 2006

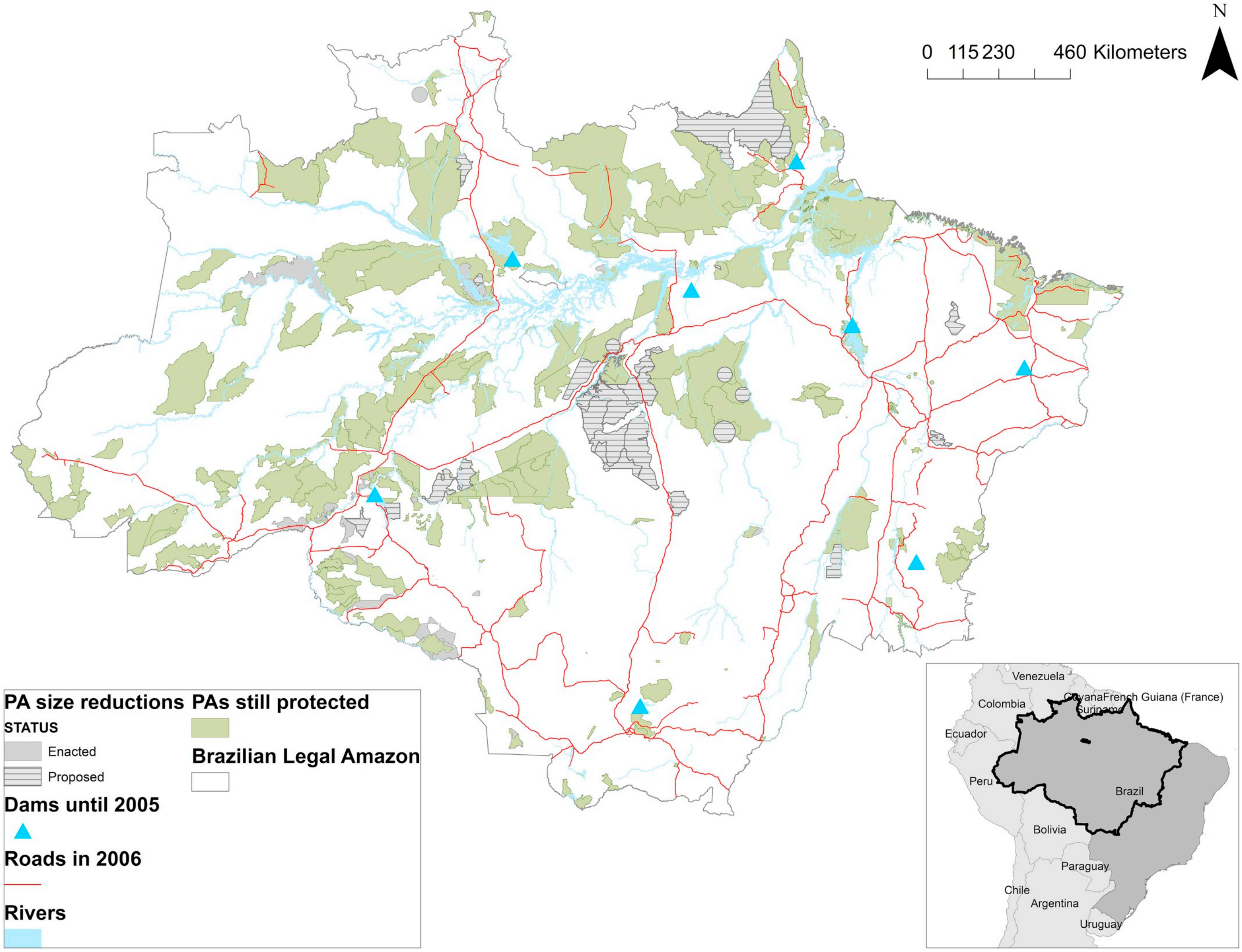

Fig. 3. Roads, rivers and dams in the Brazilian Amazon.

Table 1A
Descriptive statistics.

	PAs still fully protected			PAs reduced in size		
	Mean	Min	Max	Mean	Min	Max
Distance to nearest road	85	0.1	400	56	2.4	274
Distance to nearest river	46.4	0	306	43.3	0	270
Distance to nearest dam	344	36	1065	282	6.8	644
Distance to nearest city	271	0	846	261	0.1	721
Average GDP (10,000 reals)	86,701	328	2,000,000	158,040	1660	2,000,000
Average population density	165	0	8815	63	0	3033
Average slope	1.68	0.15	8.19	2.05	0.43	6.93
Average rainfall	2080	954	3218	2086	1273	2990
Total prior deforestation	19	0	22,357	115	0	832
PA Size	3669	0.01	48,267	7115	0.5	38,870
Perimeter-to-Area Ratio	1.78	0.03	74	0.18	0	1.34
High Endemism (< 21) [a]	20.85			3.91		
Low Endemism (1–5)	37.81			37.25		
Medium Endemism (6–20)	26.15			39.21		
No Endemism (0)	15.19			19.61		
IUCN Category Ia	11.53			7.84		
IUCN Category II	20.28			27.45		
IUCN Category III	1.75			–		
IUCN Category IV	8.39			–		
IUCN Category V	14.33			7.84		
IUCN Category VI	43.71			56.86		
Observations	286			51		

[a] For Endemism and IUCN Category, we report the frequency.

Table 1B
Variables' sources & descriptions.

Name	Date	Units	Sources	Treatment
GDP	2000 to 2005	1000 reals, current prices	Vector format from the IBGE at the level of the municipality (IBGE, 2017).	Average from 2000 to 2005.
Distance to the nearest road	2006	km	Vector format from the Brazilian Departamento Nacional de Infraestrutura de Transportes (DNIT, 2017).	Geodesic distance of the centroid to the nearest roads with ArcGIS 10.4.
Distance to the nearest city	2005	km	Urbanized spots of more than 100,000 inhabitants in vector format from the IBGE (IBGE, 2017).	Geodesic distance of the centroid to the nearest city with ArcGIS 10.4.
Slope		250 m*250 m	Gridded elevation data from the Shuttle Radar Topography Mission (SRTM) (Jarvis et al., 2008).	Computed in degree from the horizontal with ArcGIS 10.4.
Rainfall	2000 to 2005	mm/year 5 km*5 km	Gridded data from the Climate Hazards Group InfraRed Precipitation with Station (CHIRPS) (Funk et al., 2015).	Average from 2000 to 2005.
Distance to the nearest river		km	Lake, pond and rivers, permanent and navigable in vector format from the IBGE (IBGE, 2017).	Geodesic distance of the centroid to the nearest river with ArcGIS 10.4.
Population density	2005	1 km*1 km	Gridded data from The Gridded Population of the World (GPW) version 4 from the 2006 Global Rural-Urban Mapping Project (GRUMP) of the Center for International Earth Science Information Network (CIESIN, 2015).	Average from 2000 and 2005.
Total deforestation	2001 to 2005	squared km	Vector format from the PRODES System of the Instituto Nacional de Pesquisa Espacial (INPE, 2017).	Total from 2001 to 2005.
Distance to the nearest dam	1975 to 2005	km	Dams > 0,1 km3 in points format from the Global Reservoir and Dam (GRanD) database of the Department of Geography of Mc Gill University in Montreal (Lehner et al., 2011).	Geodesic distance of the centroid of each PA to the nearest dam with ArcGIS 10.4.
PA size		squared km	WDPA (IUCN and UNEP-WCMC, 2016)	For PAs reduced in size, we use the size of the PA before the event.
Perimeter-to-Area Ratio		ratio	WDPA (IUCN and UNEP-WCMC, 2016)	For PAs reduced in size, we use the perimeter of the PA before the event.
Number of endemic species	Before 2006	No: 0 species; Low: 1 to 5; Medium: 6–20; High: 21–47	Vector format from the WWF WildFinder database of species distributions (Olson et al., 2001; World Wildlife Fund (WWF), 2006).	
IUCN category		See IUCN for all details but lower is stricter.	WDPA (IUCN and UNEP-WCMC, 2016) and PADDDtracker (Conservation International and World Wildlife Fund, 2017)	
Administrative boundaries			Vector format from data on Global Adminsitrative Area (GADM, 2012)	

Table 2A
Internal deforestation within PAs – area (sq.km).

Total internal deforestation 2003–2005 [OLS]	(1)	(2)	(3)	(4)
ln(Distance to the nearest road in 2006)	4.840	5.067	5.556	3.495
	(0.62)	(0.66)	(0.72)	(0.51)
ln(Distance to the nearest river)	7.085	6.672	9.446	8.912
	(3.05)***	(2.98)***	(2.49)**	(2.55)**
ln(Distance to the nearest dam in 2005)	−0.001	−2.837	10.663	−0.764
	(0.00)	(0.59)	(1.16)	(0.13)
ln(Distance to the nearest city in 2005)	9.727	7.197	0.173	
	(2.12)**	(1.82)*	(0.05)	
Distance to the nearest city in 2005				0.074
				(1.97)*
(Distance to the nearest city in 2005)^2				0.001
				(1.98)**
ln(Average GDP from 2000 to 2002)	10.272	7.351	2.339	3.690
	(2.62)***	(2.18)**	(0.87)	(1.39)
ln(Average population density in 2000)	0.822	6.008	5.233	5.767
	(0.28)	(1.51)	(1.79)*	(2.24)**
ln(Average slopes)	8.103	9.693	4.562	0.753
	(0.80)	(0.93)	(0.59)	(0.13)
ln(Average rainfalls from 2000 to 2002)	2.066	−11.861	−12.748	−9.543
	(0.24)	(1.24)	(0.97)	(0.73)
ln(PA size)		6.516	9.042	8.445
		(3.52)***	(3.89)***	(3.92)***
low endemism (1–5)[a][b]			(1.95)*	(1.93)*
			6.417	6.417
medium endemism (6–20)			(0.78)	(0.78)
			12.183	12.183
no endemism (0)			(1.03)	(1.03)
			−0.778	−0.778
IUCN cat. IV[b]			32.083	30.052
			(1.86)*	(1.74)*
Acre[c]			−3.221	−2.610
			−68.055	−58.485
Amapá			(2.32)**	(2.53)**
			−48.858	−41.063
Amazonas			(2.56)**	(2.53)**
			−54.947	−50.223
Maranhão			(3.23)***	(3.44)***
			−41.436	−37.436
Mato Grosso			(1.93)*	(1.97)**
			−40.206	−39.797
Pará			(2.50)**	(2.62)***
			−13.950	−12.006
Roraima			(0.95)	(0.88)
			−70.582	−48.599
Tocantins			(2.97)***	(2.96)***
			−48.673	−45.269
Constant	−212.385	−93.004	−75.138	−50.355
	(1.67)*	(0.85)	(0.60)	(0.39)
R2	0.05	0.07	0.14	0.14
N	355	355	354	355

* $p < .1$; ** $p < .05$; *** $p < .01$.

[a] The effects of these categorizations of the number of endemic species are compared to high endemism (> 21).

[b] IUCN categories are compared to Category Ia (and all were included but are not shown here as insignificant).

[c] States are compared to Rondônia, which is omitted as we can estimate only N-1 relative effects of the states.

Table 2B
Internal deforestation within PAs – fraction of PA area.

Ln (Total internal deforesation 2003–2005 / PS size) [OLS]	(1)	(2)	(3)	(4)
ln(Distance to the nearest road in 2006)	0.001	0.007	0.005	0.006
	(0.02)	(0.17)	(0.12)	(0.14)
ln(Distance to the nearest river)	0.051	0.040	0.031	0.038
	(1.77)*	(1.46)	(1.17)	(1.39)
ln(Distance to the nearest dam in 2005)	−0.100	−0.018	−0.015	−0.039
	(1.47)	(0.27)	(0.20)	(0.55)
ln(Distance to the nearest urban area in 2005)	0.017	0.041		
	(0.34)	(0.83)		
Distance to the nearest urban area in 2005			−0.001	0.000
			(2.03)**	(1.39)
Squared distance to the nearest urban area in 2005			0.000	
			(2.77)***	
ln(Average GDP from 2000 to 2002)	0.056	−0.001	−0.029	−0.007
	(1.98)**	(0.04)	(1.02)	(0.26)
ln(Average population density in 2000)	−0.017	0.012	−0.009	0.002
	(0.36)	(0.29)	(0.26)	(0.06)
ln(Average slopes)	0.198	0.255	0.257	0.254
	(2.31)**	(3.12)***	(3.17)***	(3.15)***
ln(Average rainfalls from 2000 to 2002)	0.291	0.524	0.386	0.515
	(2.00)**	(2.81)***	(2.04)**	(2.75)***
ln(PA size)	−0.075	−0.092	−0.092	−0.093
	(3.41)***	(3.71)***	(3.69)***	(3.74)***
low endemism (1–5)[a]		0.289	0.246	0.277
		(2.18)**	(1.80)*	(2.04)**
medium endemism (6–20)		0.221	0.279	0.227
		(1.41)	(1.75)*	(1.44)
no endemism (0)		0.447	0.416	0.439
		(2.68)***	(2.45)**	(2.60)***
IUCN cat. II[b]		−0.192	−0.282	−0.210
		(1.16)	(1.54)	(1.26)
IUCN cat. III		−0.677	−0.711	−0.689
		(3.73)***	(4.05)***	(3.81)***
IUCN cat. IV		−0.249	−0.274	−0.256
		(2.46)**	(2.71)***	(2.51)**
IUCN cat. V		0.257	0.246	0.251
		(2.25)**	(2.23)**	(2.21)**
IUCN cat. VI		0.078	0.092	0.079
		(0.85)	(1.00)	(0.88)
Acre[c]		−0.178	−0.098	−0.155
		(0.61)	(0.33)	(0.52)
Amapá		−1.240	−1.160	−1.239
		(6.65)***	(6.03)***	(6.57)***
Amazonas		−0.939	−0.857	−0.935
		(6.60)***	(5.93)***	(6.53)***
Maranhão		−0.152	−0.073	−0.149
		(0.53)	(0.25)	(0.52)
Mato Grosso		−0.611	−0.544	−0.615
		(3.67)***	(3.18)***	(3.69)***
Pará		−0.625	−0.527	−0.636
		(3.97)***	(3.29)***	(4.16)***
Roraima		−0.836	−0.723	−0.803
		(5.04)***	(4.17)***	(4.67)***
Tocantins		−0.695	−0.621	−0.687
		(3.56)***	(3.00)***	(3.48)***
Constant	−1.789	−3.110	−1.444	−2.711
	(1.58)	(2.08)**	(0.93)	(1.78)*
R2	0.11	0.40	0.41	0.40
N	355	354	354	354

* $p < .1$; ** $p < .05$; *** $p < .01$.

[a] The effects of these categorizations of the number of endemic species are compared to high endemism (> 21).

[b] IUCN categories are compared to Category Ia.

[c] States are compared to Rondônia, which is omitted as we can estimate only N-1 relative effects of the states.

(DNIT, 2017), as seen within Fig. 3 (Barber et al., 2014; Bax and Francesconi, 2018; Laurance et al., 2014); average rainfall from 2000 to 2005 (Funk et al., 2015), per the suitability of land for agriculture (Kirby et al., 2006; Sombroek, 2001; Tesfaw et al., 2018); and market sizes, as measured using an average density of population during 2000 to 2005 (CIESIN, 2015).

Characteristics that could raise the economic returns from infrastructure, including hydropower, include average slope (Jarvis et al., 2008) and proximity to rivers (IBGE, 2017). Being nearer to rivers and on higher slopes (see Fig. 3) is more suitable for implementing a hydroelectric dam (Finer and Jenkins, 2012; McClain and Naiman, 2008). We want to account for this type of infrastructure investment, in particular, since hydropower development has for some time been a leading objective for infrastructure investments within the Brazilian Amazon (Araújo et al., 2012; Fearnside, 2014; World Wildlife Fund (WWF), 2006).

Internal forest loss from 2001 to 2005 (INPE, 2017) indicates a lack of enforcement within a given PA. We also use the number of terrestrial endemic species (World Wildlife Fund (WWF), 2006) to indicate conservation priorities (Tesfaw et al., 2018) and in addition include the proximity to dams (Olson et al., 2001) to indicate potential habitat fragmentation (Fearnside, 2014; Finer and Jenkins, 2012; McClain and Naiman, 2008).

PA management costs, per unit area, can fall (or rise) with PA size depending on (dis-) economies of scale (Bruner et al., 2004). We use perimeter-to-area ratio (World Wildlife Fund (WWF), 2017) as a proxy. That ratio is lower if a protected unit is larger. It also can measure habitat or PA fragmentation (Albers, 2010; Sims, 2014). We sometimes directly include the PA's size itself (Robinson et al., 2011) as a variable in the analyses, recalling that it already has been found to affect the likelihood of having a PADDD event (Symes et al., 2016). Lastly, the IUCN PA categories (IUCN and UNEP-WCMC, 2016) indicate a PA's management objectives and thus also the level of cost faced (Bruner et al., 2004; IUCN and UNEP-WCMC, 2016; Symes et al., 2016).

All the covariates were transformed in Geographic Coordinate System “South American Datum 1969” and projected into “UTM Zone 18S (meters)” using ArcGIS 10.4.1. The raster and vector covariates have not been treated similarly, though. A grid of 1.8 × 1.8 km was used to sample the raster dataset (slopes, population density and rainfalls). We extract means, for each cell, allowing us to describe our smallest degazetted or downsized unit. Only averages and weighted averages (by proportion of the unit) have been included in the final estimations. The covariates (GDP, endemic species, deforestation) have been intersected with protected units to compute (weighted) averages for the cells. Geodesic distances to the nearest road, dam and river have been computed in kilometers from the centroid of each PA. A complete description of the source and statistical treatment of these covariates is available in Table 1B.

3.2. Empirical Strategy

Our objective is to reveal factors in the probability of a PA being reduced in size from 2006 to 2015. In a bargaining model, that decision should reflect net benefits or costs for agencies D and E (B^{DI} and B^{EI}). Thus, we want to consider all of the factors above in the net impacts of decisions about size reductions. We will represent as $U^*(R_i)$ the effective ‘joint objective function’ that arises from agency bargaining.

$$U * (R_i) = \beta X_i + \varepsilon_i$$

with X_i the covariates that affect agencies' benefits, β their associated parameters, and ε the error term. As our dependent variable $U^*(R_i)$ is latent, we can consider a dummy variable R_i taking the value one if a decision to reduce PA size was made and the value zero otherwise, i.e., a binary indicator of U(R). Thus, our regression estimates the probability of PAs' size being reduced using that binary variable:

$$\text{Probability } (R_i = 1) = F(\beta X_i) \text{ or, expanding upon the factors } X_i \text{ in net impacts of } R_i$$

$$\text{Probability } (R_i = 1) = \alpha o_i + \beta d_i + \omega V_i + \rho C_i + \eta_i + \varepsilon_i$$

Assuming the cumulative distributive function of residuals to be logistic − as a default model to start − we use a logistic probability model estimated using maximum likelihood, where: o_i are characteristics of land that affect the economic return from activities that yield PA size reductions (de Marques and Peres, 2015; Mascia et al., 2014; Pack et al., 2016; Tesfaw et al., 2018); d_i refers to illegal damages, measured by deforestation inside the PAs before any size reduction; while V_i are species values and C_i references PAs enforcement costs that affect the net benefits of keeping any PA as it is (i.e., $R_i = 0$) versus reducing its size (Abman, 2018; Joppa and Pfaff, 2011; Nolte et al., 2013).

We believe that the two agencies' relative bargaining power, concerning decisions about PA reductions, is likely to be influenced by some fixed characteristics that vary across Amazon states (Abman, 2018; Joppa and Pfaff, 2011; Nolte et al., 2013), which vary in environmental and development objectives (Pfaff et al., 2015a; Tesfaw et al., 2018). For example, environmental regulatory history in Rondônia is consistent with numerous PADDD events over time, reflecting distinct local perceptions of net benefits from PAs (Sauquet et al., 2014). We account for this political heterogeneity by including state dummies.

Table 2C
Internal deforestation within PAs – binary independent variable.

Total internal deforestation 2003–2005 [Logit]	(1)	(2)	(3)	(4)
ln(Distance to the nearest road in 2006)	0.907	0.843	0.842	0.850
	(−0.47)	(−0.77)	(−0.78)	(−0.75)
ln(Distance to the nearest river)	1.055	0.924	0.934	0.930
	(0.41)	(−0.40)	(−0.35)	(−0.38)
ln(Distance to the nearest dam in 2005)	0.911	0.777	0.717	0.805
	(−0.31)	(−0.56)	(−0.66)	(−0.46)
ln(Distance to the nearest city in 2005)	1.461	1.590		
	(2.08)**	(2.07)**		
Distance to the nearest city in 2005			1.008	1.002
			(1.56)	(1.34)
(Distance to the nearest city in 2005)^2			1.000	
			(−1.16)	
ln(Average GDP from 2000 to 2002)	1.293	1.040	0.998	0.948
	(1.98)**	(0.25)	(−0.01)	(−0.40)
ln(Average population density in 2000)	0.986	1.015	0.944	0.926
	(−0.11)	(0.08)	(−0.35)	(−0.46)
ln(Average slopes)	1.501	1.562	1.672	1.643
	(1.01)	(0.88)	(1.03)	(1.03)
ln(Average rainfalls from 2000 to 2002)	136.043	52.660	71.985	45.273
	(5.57)***	(3.14)***	(3.32)***	(2.91)***
ln(PA size)	1.452	1.294	1.311	1.302
	(5.37)***	(2.21)**	(2.35)**	(2.33)**
low endemism (1–5)[a]		1.256	1.172	1.087
		(0.40)	(0.26)	(0.14)
medium endemism (6–20)		1.280	1.042	1.196
		(0.31)	(0.05)	(0.22)
no endemism (0)		2.750	2.493	2.135
		(1.09)	(0.98)	(0.86)
IUCN cat. IV[b]		0.096	0.106	0.084
		(−1.92)*	(−1.91)*	(−1.98)**
Acre[c]		12.558	12.469	13.130
		(1.15)	(1.06)	(1.05)
Amapá		0.085	0.071	0.090
		(−1.91)*	(−2.06)**	(−1.92)*
Amazonas		0.483	0.409	0.485
		(−0.77)	(−0.95)	(−0.78)
Maranhão		0.415	0.317	0.366
		(−0.84)	(−1.14)	(−0.98)
Mato Grosso		0.377	0.313	0.375
		(−1.19)	(−1.44)	(−1.17)
Pará		1.423	1.130	1.371
		(0.33)	(0.12)	(0.30)
Roraima		1.168	1.128	1.242
		(0.10)	(0.08)	(0.14)
Tocantins		0.070	0.059	0.071
		(−2.60)***	(−2.86)***	(−2.61)***
Pseudo R2	0.31	0.44	0.44	0.44
N	355	354	354	354

* $p < .1$; ** $p < .05$; *** $p < .01$.

[a] The effects of these categorizations of the number of endemic species are compared to high endemism (> 21).

[b] IUCN categories are compared to Category Ia (and all were included but are not shown here as insignificant).

[c] States are compared to Rondônia, which is omitted as we can estimate only N-1 relative effects of the states.

Table 3
Risks of PA size reductions.

[Logit [a]]	(1)	(2)	(3)	(4)	(5)
Distance to the nearest road in 2006	0.991	0.989	0.981	0.989	0.980
	(−2.21)**	(−2.58)**	(−2.23)**	(−2.18)**	(−2.26)**
(Distance to the nearest road in 2006)^2			1.000		1.000
			(1.09)		(1.45)
Distance to the nearest river	0.999	1.000	1.001	1.002	1.003
	(−0.13)	(−0.02)	(0.14)	(0.23)	(0.45)
Distance to the nearest dam in 2005	1.000	1.001	1.001	1.002	1.002
	(0.26)	(0.50)	(0.50)	(0.23)	(1.04)
Distance to the nearest urban area 2005	0.999	0.991	0.992	0.992	0.993
	(−0.50)	(−2.32)**	(−2.16)**	(−1.97)**	(−1.75)*
(Distance to nearest urban area 2005)^2		1.000	1.000	1.000	1.000
		(1.98)**	(1.82)*	(1.46)	(1.20)
ln(Average GDP from 2000 to 2005 + 1)	1.206	1.005	1.006	1.069	1.071
	(1.05)	(0.03)	(0.03)	(0.38)	(0.39)
Average population density2000–2005	1.000	1.000	1.000		
	(0.23)	(0.31)	(0.28)		
(Average popden 2000–2005) in buffer				0.999	0.999
				(−0.63)	(−0.65)
ln(Total deforestation 2000–2005)	1.380	1.265	1.256	1.378	1.366
	(2.45)**	(1.96)*	(1.89)*	(2.71)***	(2.59)***
ln(Size of the PA)	1.437	1.514	1.527		
	(2.80)***	(3.42)***	(3.47)***		
Perimeter-to-area ratio				0.190	0.186
				(−2.55)**	(−2.59)***
Low endemism (1–5)[b]	0.326				
	(−0.82)				
Medium endemism (6–20)	0.301				
	(−1.00)				
No endemism (0)	0.628				
	(−0.37)				
IUCN cat II[c]	4.796				
	(1.61)				
IUCN cat V	1.790				
	(0.50)				
IUCN cat VI	2.537				
	(1.18)				
Amapá[d]	1.131	1.109	1.030	1.395	1.200
	(0.09)	(0.10)	(0.03)	(0.27)	(0.14)
Amazonas	0.080	0.127	0.120	0.137	0.128
	(−2.39)**	(−2.82)***	(−2.79)***	(−2.53)***	(−2.51)**
Maranhão	0.252	0.451	0.397	1.120	0.947
	(−1.11)	(−1.01)	(−1.17)	(0.09)	(−0.04)
Mato Grosso	0.025	0.050	0.046	0.063	0.056
	(−2.76)***	(−2.65)***	(−2.69)***	(−2.50)**	(−2.58)***
Pará	0.493	0.630	0.605	1.173	1.146
	(−0.68)	(−0.67)	(−0.72)	(0.22)	(0.19)
Roraima	0.207	0.212	0.213	0.236	0.244
	(−1.14)	(−1.45)	(−1.44)	(−1.16)	(−1.13)
Pseudo R2	0.31	0.30	0.30	0.32	0.32
MacFadden's ajusted R2	0.14	0.16	0.16	0.18	0.18
AIC	233.34	226.29	227.59	211.41	212.29
Number of observations	292	292	292	284	284

* $p < .1$; ** $p < .05$; *** $p < .01$.

[a] Results are robust to a changing from logit to probit or ordinary least square. In regressions (2) to (5), removing the number of endemic species and IUCN categories has no impact and allows us to gain degrees of freedom.

[b] The effects of these categorizations of the number of endemic species are compared to high endemism (> 21).

[c] IUCN categories are compared to category Ia (some IUCN, slope and rainfall are insignificant and dropped).

[d] States are compared to Rondônia, which is omitted as we can estimate only N-1 relative effects of the states.

4. Results

4.1. Descriptive Statistics

Table 1A offers summary statistics for our covariates for sets of PAs − broken down into still protected (1st large meta-column) versus experienced size reductions from 2006 to 2015 (2nd large meta-column) − while Table 1B extends the information concerning the sources for and descriptions of those variables. We see differences in land characteristics between these groups, with significant *t*-tests on the inequality of means and Pearson's pairwise correlations. On average, size-reduced PAs were: in areas with higher 2000–2005 GDP; closer to 2006 roads; and, consistent with those features, also more deforested from 2001 to 2005 (Table 1A). Size-reduced PAs were also larger and endowed with fewer endemic species.

4.2. Deforestation inside PAs

Tables 2A, 2B and 2C consider deforestation inside of the boundaries of PAs before any size reduction, a central factor within our theory. Table 2A considers the area deforested regardless of PA size, while Table 2B considers area deforested as a share of total PA area. For the latter, we expect different effects of size and distance as political economy tends to push larger PAs to more distant sites. Finally, as

Table 4A
Risks of PA size reductions within the higher pressure 'Arc' region.

[Logit [a]]	'Arc of deforestation'	
Distance to the nearest road in 2006	0.986	0.986
	(−2.37)**	(−2.86)***
Distance to the nearest river	0.988	0.988
	(−2.80)***	(−8.77)***
Distance to the nearest dam in 2005	1.003	1.003
	(1.65)*	(1.94)*
Distance to the nearest urbanized area in 2005	0.990	0.989
	(−1.87)*	(−3.46)***
Squared Distance to the nearest urbanized area in 2005	1.000	1.000
	(1.65)*	(2.76)***
ln(Average GDP from 2000 to 2005 + 1)	1.097	1.027
	(0.73)	(0.17)
Average population density from 2000 to 2005		1.000
		(−0.05)
Average population density from 2000 to 2005 in the buffer zone	1.000	
	(−0.29)	
Average slopes	1.144	1.126
	(0.97)	(1.28)
Average rainfalls from 2000 to 2005	1.000	0.999
	(−1.00)	(−1.12)
ln(Total deforestation from 2000 to 2005 + 1)	1.573	1.538
	(3.22)***	(1.81)*
Ln(PA size)		1.342
		(1.16)
Perimeter to area ratio	0.198	
	(−1.84)*	
Pseudo R2	0.28	0.25
MacFadden's ajusted R2	0.16	0.13
AIC	161.06	170.16
Number of observations	180	183

* $p < .1$; ** $p < .05$; *** $p < .01$.
[a] Results without state dummies and with clustered standard errors given insufficient observations.

Table 4B
Risks of PA size reductions for different types of protected areas.

[Logit [a]]	Mixed use PAs		Strict PAs	
Distance to the nearest road in 2006	0.981	0.981	0.996	0.996
	(−4.41)***	(−5.19)***	(−2.74)***	(−3.93)***
Distance to the nearest river	0.991	0.992	1.007	1.006
	(−1.40)	(−1.61)	(0.92)	(0.99)
Distance to the nearest dam in 2005	1.001	1.001	0.993	0.993
	(0.59)	(0.68)	(−2.39)**	(−2.40)**
Distance to nearest urbanized area 2005	0.992	0.991	1.001	1.000
	(−2.32)**	(−2.30)**	(0.24)	(0.08)
(Distance to nearest urbanized area 2005)^2	1.000	1.000	1.000	1.000
	(2.18)**	(2.50)**	(0.23)	(0.45)
ln(Average GDP from 2000 to 2005)	1.051	1.033	1.359	1.277
	(0.29)	(0.15)	(2.44)**	(4.20)***
Average population density in 2000–2005		0.991		1.000
		(−2.22)**		(0.24)
(Average popden 2000–2005) in buffer zone	0.994		1.000	
	(−2.18)**		(2.30)**	
Average slopes	1.494	1.463	1.036	1.008
	(2.87)***	(3.49)***	(0.16)	(0.07)
Average rainfall from 2000 to 2005	1.000	1.000	1.001	1.001
	(−0.07)	(−0.01)	(0.71)	(1.00)
ln(Total deforestation from 2000 to 2005 + 1)	1.557	1.417	1.785	1.852
	(4.38)***	(2.15)**	(6.28)***	(5.13)***
Ln(PA size)		1.295		1.235
		(1.76)*		(0.93)
Perimeter to area ratio	0.302		0.173	
	(−2.21)**		(−1.38)	
Pseudo R2	0.30	0.28	0.30	0.27
MacFadden's ajusted R2	0.16	0.15	0.05	0.03
AIC	144.09	158.41	93.61	96.02
Number of observations	211	219	112	113

* $p < .1$; ** $p < .05$; *** $p < .01$.
[a] Results without state dummies and with clustered standard errors given insufficient observations.

Table 2B suggests that vulnerable forest areas near boundaries of large PAs do not scale linearly with PA area, Table 2C considers the odds of having any internal deforestation at all. This offers a robustness check that is 'scale free' − in a sense that neither total area deforested nor the PA share deforested can be, as each of those might naturally vary with PA size (the former positively while the latter likely negatively).

Tables 2 confirm that states differ, with states other than Rondônia having less deforestation inside PAs. Also, only IUCN category V (multiple-use PAs) correlates with higher deforestation rates within PAs. PAs in areas with lower species endemism experienced more deforestation. PA size matters too – yet, as expected, area deforested does not scale linearly with PA size: larger PAs may be invaded up to some distance from their edges, such that share deforested falls with PA size. Finally, while using state effects can obscure part of such effects, Tables 2A and 2C indicate that deforestation occurs farther from cities.[7]

These results suggests that transport costs raise costs of effective management − and by even more than they lower economic profits and pressures. This supports the presumptions underlying Fig. 1A in our theory: if agency E enforces best near cities, agency D would especially want PA size reductions there. Environmental agency E would most contest PA size reductions there, given higher environmental loss from eliminating restrictions on activities there, as that is where enforcement blocked human pressure (up until any size reductions occurred). Consequently, if PA size reductions occur more near cities, that result would suggest that development actors have greater bargaining power than environmental actors.

4.3. Drivers of PA Size Reductions

Table 3 presents the results of a logit model for PA size reductions from 2006 to 2015. As broad effects,[8] state dummies are significant (relative to Rondônia, the omitted state). This result is consistent with the numerous PADDD events in Rondônia in 2010 and 2014 considered in Tesfaw et al. (2018).[9] As noted, the bargaining power of environment and development agencies can differ significantly across states. Here, being in Amazonas lowers the chance of a size reduction by roughly 10%, relative to Rondônia.

Table 3 shows that higher distances to the nearest road and city are associated with lower likelihoods of PA size reductions (non-linear versions add nothing robust). From our theory, if size reductions are more common with lower travel costs (T), we infer more bargaining power for development agencies because an environment agency E would rather see any PA size reductions in remote areas, where they

[7] We did not find a consistent difference in urban distance between strict PAs and multiple-use PAs with some legal internal deforestation. Yet IUCN category V, which allows internal deforestation linked to traditional management practices, tended to be closer to cities, for instance relative to IUCN category Ia, which is strict about not permitting any human disturbances. Thus, we find internal deforestation farther from cities despite the legal internal deforestation in multiple-use being closer.

[8] For instance, without the state dummies in Table 3, the coefficients for the influence of average GDP are highly significant.

[9] Some states (Acre and Tocantins) do not have any degazettement events after 2005. We replaced them by clustered standard errors at the level of the state in the Robustness Checks (section 4.4), allowing residuals to be correlated within states without losing the observations. We have 9 clusters, not enough to guarantee consistent estimates of standard errors (Cameron and Miller, 2015), yet we cannot rely a on non-parametric bootstrap (Esarey and Menger, 2018) because we don't have enough variations within each cluster.

Table 4C
Risks of PA size reductions for different levels of government agency.

[Logit [a]]	State agencies		Federal agencies	
Distance to the nearest road in 2006	0.986 (−3.72)***	0.990 (−2.05)**	0.990 (−3.13)***	0.988 (−4.73)***
Distance to the nearest river	1.008 (1.04)	1.005 (0.65)	0.994 (−0.49)	0.995 (−0.65)
Distance to the nearest dam in 2005	0.997 (−1.02)	0.998 (−0.90)	0.998 (−1.06)	0.997 (−1.28)
Distance to the nearest urbanized area in 2005	0.992 (−2.76)***	0.994 (−1.85)*	1.002 (0.32)	0.999 (−0.11)
(Distance to the nearest urbanized area in 2005)^2	1.000 (5.58)***	1.000 (3.26)***	1.000 (−0.28)	1.000 (0.14)
ln(Average GDP from 2000 to 2005 + 1)	1.336 (1.41)	1.405 (1.26)	0.868 (−0.60)	0.902 (−0.48)
Average population density 2000–2005		0.999 (−0.63)		0.537 (−2.76)***
(Average popden 2000–2005) in the buffer zone	0.999 (−0.96)		0.827 (−2.97)***	
Average slopes	0.567 (−4.45)***	0.599 (−6.33)***	1.845 (3.27)***	1.940 (4.56)***
Average rainfalls from 2000 to 2005	0.999 (−0.71)	0.999 (−0.66)	1.000 (0.19)	1.000 (−0.37)
ln(Total deforestation from 2000 to 2005 + 1)	1.613 (4.20)***	1.681 (2.64)***	1.657 (4.42)***	1.501 (2.73)***
Ln(PA size)		1.020 (0.08)		2.761 (2.95)***
Perimeter to area ratio	0.564 (−2.22)**		0.023 (−2.96)***	
Pseudo R2	0.36	0.30	0.31	0.37
MacFadden's ajusted R2	0.16	0.11	0.14	0.21
AIC	101.86	111.76	124.61	118.42
Number of observations	178	181	145	150

* $p < .1$; ** $p < .05$; *** $p < .01$.
[a] Results without state dummies and with clustered standard errors given insufficient observations.

add less damage. As that is not what we see, we infer some bargaining power for development agencies.

We also consistently find larger PAs are more likely to be downsized or degazetted − be that using the area itself or a perimeter-to-area ratio (negatively correlated with size, as the perimeter rises linearly with the radius while the area rises with the square of the radius). Small PAs with high perimeter-to-area ratios are less likely to experience size reductions, consistent with lower management costs for smaller PAs because they are not spread out across vast landscapes (Albers, 2010; Bruner et al., 2004).[10]

Finally, higher internal 2001–2005 deforestation raises the likelihood of PA size reduction, extending across the Amazon the result in Tesfaw et al. (2018) for Rondônia. This suggests environmental power, as losses from size reductions fall with prior internal deforestation. In contrast, development gains from size reduction fall with internal deforestation. Thus, agency D would push less for such size reductions.

Controlling for states, the number of endemic species has no effect. We do not find consistent impacts for average slopes, distance to river, average population density, or average rainfall. The insignificance of average rainfall may reflect a lack of interactions with more detailed soil quality data (Bax and Francesconi, 2018; Sombroek, 2001) or the averaging of different marginal effects as crops gain but then lose as rainfall rises (Bax and Francesconi, 2018; Chomitz and Thomas, 2003; Kirby et al., 2006).

[10] It may also be more common for large PAs that internal outcomes vary considerably across the sub-regions within the PA, and that this is relevant for PADDD events (see results for Rondônia indicating such a possibility within Tesfaw et al., 2018).

4.4. Drivers of PA Size Reductions – Robustness Checks

We reassess the drivers of PA size reduction using a number subsets of interest: 'arc of deforestation' PAs only (Table 4A); different PA types, i.e., strict versus mixed use per IUCN categories (Table 4B); and different levels of PA governance, i.e., state versus federal (Table 4C).[11] In our sample, most of the PAs that have been degazetted are located in the arc of deforestation, where they are more likely to face high economic pressures (Pfaff et al., 2015a, 2015b; Pfaff et al., 2014). However, the PA type and the level of PA governance are evenly distributed across the PAs without and with size reductions (Pack et al., 2016).

PA types differ substantially. Multiple-use PAs may be closer to pressure – which may imply higher impacts, even with higher internal clearing (Ferraro et al., 2013; Jusys, 2018; Nolte et al., 2013). Levels of PA governance also differ significantly. For example, it is broadly hypothesized that federal actors place more importance on environmental gains than do state or local public actors (Herrera et al., 2019). For instance, PAs implemented by states may be expected to lack enforcement or be farther from threat.

Across subsets, Tables 4A to 4C confirm lower distances, greater size, and higher internal deforestation raise the likelihood of size reductions. In the arc of deforestation (Table 4A), where profits are higher, lower distances to the nearest road, river, and urban area, where profits are highest, are associated with more size reductions – suggesting development bargaining power. Also, across Tables 4A to 4C, greater internal deforestation is associated with more PA size reductions, reflecting environmental preferences.

5. Discussion

Consistent with previous research (Symes et al., 2016; Tesfaw et al., 2018), our analysis shows larger PAs and PAs with greater prior internal deforestation are more likely to experience PA size reductions – even when looking across states for the entire Brazilian Amazon and controlling for sites' differences. We also take a further step to explore how travel costs influence the outcomes of bargaining between development and environmental agencies over the locations of PA size reductions. To start, we show that high travel costs are associated with greater internal deforestation, suggesting that travel costs raise the costs of effective PA management even more than they lower economic profits and pressures (this cost tradeoff can vary over national settings, as highlighted in Ferraro et al. (2011) and Sims (2010)).

Pairing that finding with our theory, we infer that a development agency would especially want PA size reductions nearer to cities because of lower transportation costs and higher profits from size reductions, while an environmental agency would most contest those PA size reductions because they would cause higher additional environmental losses. We also confirm empirically for the entire Brazilian Amazon that in fact the likelihood of PA size reductions is higher near cities (consistent with Symes et al. (2016) and Tesfaw et al. (2018)). In light of our prior finding, and our theory, that suggests bargaining power in the hands of development forces, since they would prefer those locations for any PA size reductions.

In contrast, our empirical results on greater PA size and more prior internal deforestation suggest some environmental influence. Size is clearly relevant for enforcement costs (Albers, 2010; Symes et al., 2016) while more prior internal deforestation lowers the environmental costs of PA size reductions. Thus, our results indicate bargaining among actors, rather than a process dictated solely by development interests.

[11] Results are presented without state dummies and with clustered standard errors because of the lack of sufficient observations to identify these effects, once we split the data by subset. However, results for deforestation and development objectives are consistent with inclusion of state dummies (significant for Amazonas, Mato Grosso and Maranhao, compared to Rôndonia).

This is consistant with the study of PADDD in Rondônia (Tesfaw et al., 2018), which found upgrades of remaining PA areas with less deforestation, in parellel with reductions for the more deforested parts.

Many possible future directions for research could build upon this work and sparse existing literature. To start, it would be helpful to replicate such studies of risks of size reductions for other geographies. We also need to examine ecological and social impacts of PADDD, controlling for confounding factors, extending a small literature (including Forrest et al., 2015; Golden Kroner et al., 2016; Pack et al., 2016; Tesfaw et al., 2018) by building on our understanding of PADDD risks (Golden-Kroner et al., 2019).

Impermanence could be studied for other interventions in the conservation portfolio (Qin et al., 2019), with both qualitative and quantitative extensions concerning the political bargaining among agencies to understand the factors shaping those outcomes. For instance, it would be useful to test this framework for area-based conservation measures such as indigenous lands. There, enforcement is partly carried out by the residents within the conserved area. Hence, it may be less affected by distances and travel costs. Looking over time, we could study periods before proposed PADDD events are enacted (Tesfaw et al., 2018). Spatially, there may be interactions across events, given all the key actors (Sauquet et al., 2014).

These results have important policy implications since a pattern of PA erasures affects the calculus for optimal investments in PA creation, siting, enforcement, and management. For instance, as larger PAs are more likely to be reduced in size (Symes et al., 2016), attention to limiting the cumulative impacts of downsizings is important. In addition, our results for prior internal deforestation and size reductions suggests that enhancing PA performance through enforcement and capacity building may also enhance the durability and permanence of PAs, because ineffectiveness can lead to erasure (Tesfaw et al., 2018). Lastly, given the impermanence of PAs, we need to consider the portfolio of all conservation strategies for their complementary roles in conserving ecosystems − including in jointly confronting implications of development pressures for the durability of each of these conservation initiatives (Qin et al., 2019).

As bargaining between development and environment agencies over conservation is likely to continue, this research broadly informs policy debates concerning conservation-development tradeoffs (Ferreira et al., 2014; Mascia et al., 2014). Given ongoing ambitions for hydroelectric dams and mining (Anderson et al., 2019; Araújo et al., 2012), as seen in Rondônia (Ferreira et al., 2014; Tesfaw et al., 2018) and in PADDDtracker.org Data Release Version 2.0, other PAs have lost protection over time (Conservation International and World Wildlife Fund, 2017). Within Brazil in particular, shifts in federal orientation and thus policy (see, e.g., Aamodt, 2018; Carvalho et al., 2019; Ferrante and Fearnside, 2019; Viola and Franchini, 2014) make PAs and other types of conservation areas like indigenous lands additionally vulnerable to development pressures. Global actors eager to support conservation and local economic development, i.e., a full suite of sustainable development goals, must consider how PADDD events play out over time and space in order to reduce tradeoffs and to enhance PAs' effectiveness and permanence.

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None.

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