Conservation-development trade-offs in the Brazilian Amazon : the case of protected areas downgrading, downsizing and degazettemen

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The case of Protected Areas Downgrading, Downsizing and Degazettement

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Résumé de la thèse

En Amazonie Brésilienne, les aires protégées induisent un arbitrage entre conservation de la forêt et développement économique.

Un tiers des forêts tropicales humides – et plus spécifiquement - 60% de la forêt tropicale Amazonienne se trouve au Brésil (The World Bank, 2020; West et al., 2019). Ce pays détient un rôle majeur dans l'atténuation du changement climatique, et fournit également un très grand nombre de services écosystémiques tel que le maintien de la biodiversité (Fearnside, 2008). Pourtant, L'Amazonie Brésilienne Légale, sur laquelle nous nous concentrons dans cette thèse, a été définie juridiquement dans les années 1960 afin de désenclaver le territoire et de promouvoir son développement économique (IBGE, 2017). Depuis cette période, la déforestation a fortement augmenté, entraînant des niveaux élevés d'émissions de gaz à effet de serre, et une perte conséquente de biodiversité (Fearnside, 2008; IPBES, 2019). Les activités de déforestation sont principalement liées au secteur de l'utilisation des terres et des changements d'affectation des terres. L'économie brésilienne se caractérise en effet par une volonté d'exploitation de ses forêts pour la production de bois, l'élevage et l'agriculture. De plus, l'insécurité du régime foncier entraîne une spéculation renforçant les activités de déforestation pour l'accaparement des terres.

Plusieurs efforts témoignent néanmoins d'un engagement du Brésil pour la réduction de la déforestation et de la dégradation des forêts. En 2015, lors de la 11ème Conférence des Parties ayant donné lieu à l'accord de Paris, Le Brésil s'est engagé à réduire ses émissions de gaz à effet de serre de 43% par rapport au niveau de 2005 en 2030. Le Brésil est aussi membre de la Convention sur la Diversité Biologique créé lors du sommet de la Terre à Rio de Janeiro depuis 1992 et s'est donc engagé « à la conservation de la diversité biologique » à son « utilisation durable » et « au partage équitable des avantages découlant des ressources génétiques et des savoirs traditionnels associés » (MMA, 2017). A ce titre, le Brésil s'est par exemple engagé à ce qu'en 2020, « 17 % des zones terrestres [...], y compris les zones qui sont particulièrement importantes pour la diversité biologique et les services fournis par les écosystèmes » soient « conservées au moyen de réseaux écologiquement représentatifs et bien reliés d'aires protégées gérées efficacement et équitablement [...] » lors du plan stratégique 2011-2020 (Convention on Biological Diversity, 2020).

Le Brésil, et plus particulièrement l'Amazonie Brésilienne, fait en plus face à de nombreux défis en termes de lutte contre la pauvreté : accès aux infrastructures d'éducation et de santé, à l'emploi, au logement, à l'énergie, etc. (Federative Republic of Brazil, 2015; The World Bank, 2020). Il semble donc primordial que la lutte contre le changement climatique n'entrave pas le développement économique. Depuis 2015, le Brésil fait ainsi partie du programme de développement durable à l'horizon 2030 des Nations Unis dont l'objectif est d'éradiquer la pauvreté et de construire un monde plus soutenable (United Nations, 2015).

L'expansion et l'harmonisation du réseau d'AP, qui s'est fortement accélérée à partir des années 2000, conjointement à un renforcement des dispositifs de détection de la déforestation illégale, et à un contexte macroéconomique favorable, aurait contribué à une nette réduction de la déforestation à partir de 2004. Pour que les AP aient un impact dans la diminution de la déforestation, elles doivent réduire la déforestation par rapport à un scénario de référence. Deux conditions doivent être respectées: i) les AP doivent être localisées sur des terres autrement profitables pour le développement d'activités économiques et, ii) les AP doivent disposer de frontières suffisamment respectées. Or, il a été démontré que pour minimiser les conflits d'utilisation des terres, celles-ci étaient en moyenne situées sur des terres à faible potentiel de développement économique (Joppa and Pfaff, 2009; Nelson and Chomitz, 2011), générant un biais de localisation. En effet, lorsqu'elles empêchent l'utilisation des terres pour d'autres activités économiques productives, les AP génèrent un coût d'opportunité élevé (Pfaff et al., 2015a, 2014, 2009; Sims, 2010), ce qui augmente le prix d'acquisition des terres (Ando et al., 1998; Börner et al., 2020; Pfaff and Sanchez-Azofeifa, 2004) et le risque de déforestation interne illégale (Blackman and Villalobos, 2019; IPBES, 2019; Thieme et al., 2020).

Les travaux empiriques qui tiennent compte de ce biais de localisation montrent que les AP sont en moyenne beaucoup moins efficaces que prévu (Herrera et al., 2019; Kere et al., 2017; Pfaff et al., 2015b, 2015a, 2014) puisque les terres sur lesquelles elles sont localisées n'auraient dans tous les cas pas été défrichées. Ainsi, certains travaux montrent que lorsque les aires sont strictement protégées et localisées au plus près des pressions économiques, elles permettent de diminuer la déforestation (Jusys, 2018; Pfaff et al., 2015b), mais cette efficacité est souvent réduite quand leurs frontières ne sont pas suffisamment respectées (Pfaff et al., 2014). Lorsqu'elles sont efficaces, les AP entravent toutefois le développement économique local: il a ainsi été démontré que la pauvreté est plus importante près des AP réduisant la déforestation en Amazonie Péruvienne et en Bolivie par exemple (Hanauer and Canavire-Bacarreza, 2015; Miranda et al., 2019) ou encore qu'elles diminuent le taux de croissance du secteur industriel en Amazonie Brésilienne (Kauano et al., 2020).

Question de recherche : les événements PADDD comme résultat d'un arbitrage entre objectifs de conservation et objectifs de développement économique ?

Face au constat que les AP manquent d'efficacité et entravent le développement économique, une accélération des évènements de déclassement, de réduction et de suppression des AP, désignés plus généralement sous l'acronyme « PADDD », a été observée à partir du milieu des années 2000. Le déclassement représente une diminution des restrictions légales sur le nombre, l'ampleur ou l'étendue des activités humaines dans une AP. La réduction est une diminution de la taille de l'AP à travers une excision de zones maritimes ou terrestres grâce à une modification légale de frontières. Enfin, la suppression représente une perte légale de protection sur la totalité de l'AP (Mascia and Pailler, 2011).

Cette accélération reflète un manque d'engagement du Brésil pour les objectifs de conservation de la forêt puisque ces phénomènes ont principalement pour cause le développement et l'extraction de ressources à l'échelle industrielle et les revendications territoriales. Dès 2012, les taux de déforestation cessent de diminuer en Amazonie Brésilienne, et en 2014, 242 000 km^2 de surfaces protégées étaient déjà concernées par ces types d'évènements.

Dans cette thèse, j'examine si, dans quelle mesure, et selon quelles caractéristiques des terres et des AP, les événements PADDD proviennent d'arbitrages entre objectifs de conservation de l'environnement et objectifs de développement économique (chapitre 1). J'évalue ensuite leurs impacts sur la déforestation (chapitre 2) et sur le développement économique local (chapitre 3) en fonction de l'hétérogénéité des terres et des AP. Plus spécifiquement, dans le premier chapitre, nous examinons théoriquement les facteurs entrainant les agences de développement et de conservation de l'environnement à négocier pour ou contre des réductions de taille d'AP. Nous vérifions ensuite empiriquement dans quelle mesure les caractéristiques des terres et des AP entrant en compte dans le processus de négociation influencent les réductions de taille d'AP de 2006 à 2015, qu'elles aient étaient été promulguées, ou simplement proposées. Dans le second chapitre, nous évaluons l'impact des réductions de taille d'AP ayant été promulguées de 2009 à 2012. Nous utilisons des techniques d'appariement au niveau du pixel et distinguons les terres selon leur profitabilité et les AP selon leur efficacité antérieure. Dans le troisième chapitre, nous évaluons l'impact des évènements PADDD ayant été promulgués à partir de 2001 sur la distribution des revenus des ménages et sur les inégalités en 2010. Nous utilisons la méthode des doubles différences sur des données appariées à une échelle de 100km^2 en distinguant les terres selon leur profitabilité et les évènements PADDD selon leur influence sur la protection restante.

Chapitre 1 : quels sont les facteurs à l'origine des réductions de taille d'AP ?

Alors que plusieurs travaux apportent un éclairage descriptif concernant l'accélération des événements PADDD lors des dernières décennies, la littérature évaluant théoriquement et empiriquement ses facteurs d'influence reste très limitée. Nous nous basons principalement sur les travaux réalisés par Tesfaw et al. (2018) sur l'État du Rondônia : il est plus probable que les événements PADDD soient promulgués lorsque ceux-ci permettent des bénéfices économiques élevés alors que les coûts environnementaux sont faibles. Ce résultat est plus incertain et dépend des pouvoirs de négociation de chaque acteur lorsque les coûts environnementaux sont élevés ou que les gains économiques sont faibles. Nous formalisons ce cadre conceptuel en décrivant les bénéfices retirés par les agences de développement lors de la réduction d'AP et par les agences environnementales lorsqu'elles évitent ces réductions.

Si les frontières de l'AP sont parfaitement respectées, les bénéfices économiques et environnementaux de chaque agence sont respectivement égaux aux bénéfices perdus liés à la mise en place d'AP et à la déforestation que les AP ont permis d'éviter. Ils augmentent donc avec le coût d'opportunité de la conservation, que nous approchons ici grâce à une mesure de distances aux routes. Il est difficile de rencontrer des situations socialement efficaces dans lesquelles les bénéfices de chaque agence seraient dé-corrélés. Toutefois, si nous considérons que les AP sont illégalement défrichées, les bénéfices économiques et environnementaux sont réduits d'un montant égal à la perte de ces terres, que ce soit pour des activités productives ou pour des activités de conservation. Cette hypothèse, plus proche de la réalité, nous permet d'étudier la façon dont l'hétérogénéité des terres et des caractéristiques des AP influencent les bénéfices économiques et environnementaux de chaque agence.

Près des routes, où le coût d'opportunité de la conservation est le plus élevé, la déforestation interne illégale peut être faible si, malgré des pressions économiques importantes, les frontières sont parfaitement défendues grâce à de faibles coûts de transport aux villes. Les bénéfices économiques et environnementaux seraient alors élevés et diminueraient à mesure que l'on s'éloigne des routes. Observer des réductions de tailles près des routes permettrait de démontrer que les agences de développement ont un pouvoir de négociation plus important que les agences environnementales.

La déforestation interne illégale peut également être importante près des routes si, malgré les faibles coûts de transport aux villes, les pressions économiques sont trop importantes et ne permettent pas une défense parfaite des frontières des AP. Dans ce cas, les bénéfices économiques et environnementaux seraient les plus élevés à des distances intermédiaires aux routes, où la défense des frontières est facilitée par de plus faibles pressions économiques. Des réductions de taille près des routes seraient ici moins contestées par les agences environnementales.

Empiriquement, nous observons dans un premier temps que les AP étaient moins défrichées de 2004 à 2005 lorsqu'elles étaient situées près des routes, confirmant ainsi que les frontières étaient mieux défendues, malgré des pressions économiques plus importantes, grâce aux faibles coûts de transport aux villes. Dans un second temps, nous utilisons la version 1.1 de la base de données PADDDtracker.org (Conservation International and World Wildlife Fund, 2017) afin d'évaluer les caractéristiques des terres et des AP qui influencent la probabilité que des réductions de taille d'AP soient proposées ou promulguées de 2006 à 2015. Nous trouvons que les AP étaient plus souvent réduites près des routes, confirmant l'importance du pouvoir de négociation des agences de développement. En outre, la déforestation interne, la taille, ainsi que des catégories de gestion plus strictes augmentent les risques que les AP soient réduites. En plus de confirmer les résultats antérieurs (Symes et al., 2016; Tesfaw et al., 2018), ces résultats impliquent que les agences environnementales ont un certain pouvoir de négociation: elles contestent moins les réductions de taille d'AP lorsque celles-ci deviennent inefficaces et plus coûteuses.

Chapitre 2: quels sont les impacts des réductions de taille d'AP sur la déforestation ?

Dans le premier chapitre, nous avons démontré que les réductions de taille d'AP se situent principalement sur des terres où les pressions économiques sont fortes, générant un biais de localisation. On pourrait donc s'attendre à ce que la déforestation augmente suite aux réductions de taille d'AP. Or, les agences environnementales ont également un pouvoir de décision, puisque les AP sont plus souvent réduites lorsqu'elles ont déjà été défrichées et lorsqu'elles sont coûteuses à gérer. Ce résultat impliquerait que les réductions de taille n'influencent pas la déforestation. Pack et al. (2016) et Tesfaw et al. (2018) trouvent en effet que les évènements PADDD n'ont en moyenne pas augmenté la déforestation au Brésil de 2000 à 2011 et dans l'Etat du Rondônia en 2010, car ces AP étaient déjà défrichées et inefficaces. Dans ce deuxième chapitre, nous menons une évaluation d'impact quasi-expérimentale afin de tenir compte des biais de sélection en fonction de l'hétérogénéité des terres et des AP.

A partir d'un modèle de Von Thünen, démontrant que les AP réduisent la déforestation quand elles réduisent les rentes agricoles, nous supposons que les réductions de taille d'AP augmentent la déforestation quand elles limitaient auparavant la déforestation. Ces conditions sont satisfaites: près des centres urbains car les frontières sont moins perméables aux pressions économiques, et à des distances intermédiaires, car les pressions économiques existantes sont contraintes par les capacités de défenses des frontières restantes. Nous supposons que les réductions de taille d'AP n'influencent pas la déforestation si elles ne limitaient pas auparavant la déforestation : loin des centres urbains où la rente agricole est nulle et, près des centres urbains si les frontières de l'AP n'étaient pas suffisamment défendues.

Empiriquement, nous évaluons l'impact des réductions de taille d'AP ayant été promulguées de 2009 à 2012 sur la déforestation de 2010 à 2015 à l'échelle de pixels de $90m^2$. Afin de tenir compte de l'efficacité antérieure des AP, nous évaluons tout d'abord leur impact lors de leur mise en place avant 2000, sur la déforestation de 2001 à 2008 en fonction de leur distance aux routes et de l'État dans lequel elles se situent. Nous obtenons des effets moyens de traitement sur les traités en appariant les pixels traités à des pixels non traités aux caractéristiques similaires grâce à des scores de propensions, puis à des distances de mahalanobis.

Nous trouvons tout d'abord que les réductions de taille d'AP n'ont pas influencé la déforestation entre 2010 et 2015 lorsque les pressions économiques étaient faibles: dans les Etats situés en dehors de l'arc de la déforestation et loin des routes. Conformément à notre cadre théorique, ces AP ne limitaient pas la déforestation de 2001 à 2008 car les terres n'étaient pas profitables. Les réductions d'AP n'ont pas non plus influencé la déforestation dans les zones où les pressions économiques étaient les plus fortes car elles étaient déjà entièrement défrichées de 2001 à 2008. Même si la défense des frontières était plus simple, il semble que la volonté politique de défense de ces AP était aussi plus faible. Leur réduction a toutefois contribué à accélérer la déforestation à des distances intermédiaires aux routes: ces AP étaient défrichées de 2001 à 2008, mais probablement dans une moindre mesure.

Chapitre 3: quels sont les impacts des évènements PADDD sur la distribution des revenus des ménages et sur les inégalités ?

Quand elles réduisent la déforestation, les AP empêchent le développement d'autres activités économiques. Ainsi, nous avons constaté dans le premier chapitre que les réductions de taille d'AP se situent en moyenne près des villes, car les opportunités économiques sont meilleures, même si les dommages environnementaux peuvent être élevés. Cependant, dans le deuxième chapitre, nous avons constaté que même si la déforestation augmente à la suite des réductions de taille d'AP, celles-ci étaient déjà défrichées, au moins en partie. Si les performances économiques étaient déjà élevées, les réductions de taille d'AP pourraient alors ne faire aucune différence sur le développement économique local par rapport à la situation initiale.

Il existe de nombreux types d'évènements PADDD et ceux-ci peuvent être promulgués pour des raisons allant du développement d'activités extractives à grande échelle à l'autorisation d'activités touristiques de plus faible intensité. En outre, les AP peuvent améliorer le développement économique local lorsqu'elles permettent un accès aux ressources forestières aux populations locales qui en sont dépendantes, entraînent le développement d'infrastructures ou encore, le développement d'activités touristiques (Wunder, 2014 ; Sims, 2010 ; Duchelle, 2014). De ce fait, nous utilisons les données de Masson (2020) qui mesurent la distribution des revenus des ménages sur une grille de cellules de 100km² en 2000 et en 2010. Ces données nous permettent de mesurer les inégalités et d'évaluer quelle partie de la population a bénéficié ou non des évènements PADDD ayant été promulgués entre 2001 et 2010.

Nous utilisons la méthode des doubles différences sur des données pré-appariés pour obtenir des effets moyens de traitement sur les traités qui tiennent compte des biais de sélection liés à l'existence de variables influençant le développement économique et l'occurrence d'évènements PADDD. De plus, nous utilisons une distance aux routes afin d'identifier les niveaux de pression économique sur les terres. Enfin, nous distinguons les cellules dans lesquelles la protection a été réduite du fait de réductions et de suppressions d'AP et les cellules dans lesquelles la protection a augmenté du fait de simples déclassements d'AP ou de la combinaison de réductions, suppressions, déclassement et mise en place de nouvelles AP.

En moyenne, bien que la proportion de ménages appartenant à la classe moyenne soit déjà élevée, les évènements PADDD ont contribué à accélérer leur augmentation par rapport aux cellules entièrement protégées jusqu'en 2010.

Dans les cellules où la taille de la protection a diminué, c'est l'augmentation de la classe moyenne-supérieure qui s'est accélérée près des routes, bien que celle-ci était déjà importante par rapport aux cellules non traitées. Ce résultat peut provenir d'un mécanisme distributif inter-classes si les ménages se sont enrichis, ou d'un mécanisme distributif intra-classes si les ménages de la classe moyenne supérieure ont immigré dans les cellules traitées. La réduction de la protection aurait permis un meilleur accès à de nouvelles opportunités économiques liées au développement de barrages et d'autoroutes ou une augmentation des rentes agricoles, facilitée par de meilleurs moyens de communication.

Dans les cellules où la taille de la protection a augmenté, les inégalités se sont réduites loin des routes. La classe moyenne-inférieure, qui était déjà plus importante dans les cellules traitées, a vu son expansion s'accélérer, alors que celle des ménages sans revenus, qui étaient moins importante, a vu son expansion ralentir. Un mécanisme distributif inter-classes pourrait provenir de l'enrichissement des ménages les plus pauvres si les AP ont permis le développement d'activités touristiques ou un accès aux ressources forestières. Un mécanisme distributif intra-classes pourrait toutefois être lié à une émigration des ménages sans revenu vers des cellules non traitées si les AP n'étaient pas mieux gérées et que de nouvelles activités économiques ont entravé leur accès aux ressources forestières. Des ménages de la classe moyenne-inférieure auraient aussi pu immigrer plus facilement vers ces cellules traitées pour accéder à ces nouvelles activités économiques.

Conclusion, Implications et Travaux futurs

Comme en témoignent ses nombreux engagements internationaux, l'Amazonie Brésilienne fait face à des enjeux de conservation de l'environnement et de lutte contre la pauvreté. Malgré un réseau d'AP de plus en plus étendu, celui-ci est considéré comme étant peu efficace pour réduire la déforestation. En effet, les conflits entre développement économique et conservation des terres ont entraîné un biais de localisation des AP vers des zones où les pressions économiques sont faibles. Lorsqu'elles sont situées dans des zones à plus fortes pressions économiques, celles-ci manquent souvent d'application effective et souffrent de déforestation illégale interne. Quand les AP parviennent enfin à réduire la déforestation, elles entravent le développement économique local. Ainsi, des mouvements de protestation des AP ont émergé et se sont accélérés à partir du milieu des années 2000.

Dans cette thèse, j'analyse la relation existante entre les événements PADDD et les arbitrages entre objectifs de conservation de l'environnement et objectifs de développement économique en termes de décision (chapitre 1) et en termes d'impact sur la déforestation (chapitre 2) et sur le développement économique (chapitre 3) en fonction de l'hétérogénéité des terres et des caractéristiques des AP.

Dans le chapitre 1, nous avons démontré que les agences de développement ont un pouvoir de négociation important concernant les réductions de taille d'AP. En effet, ces décisions ont plus souvent été proposées et promulguées de 2006 à 2015 près des villes, alors qu'elles pouvaient entraîner d'importants dommages environnementaux. Dans le chapitre 3, nous avons observé en effet que les réductions de taille d'AP ayant été promulguées de 2001 à 2010 ont contribué à augmenter la taille de la classe moyenne supérieure près des routes, alors que celle-ci était déjà conséquente. Que ce résultat soit lié à un enrichissement des ménages des classes inférieures ou à une immigration des ménages de la classe moyenne-supérieur, il résulte probablement d'une amélioration des infrastructures, de la rente agricole et des opportunités d'emploi non agricoles pour les ménages, facilitée par de faibles coûts de transport aux villes. Loin des routes, nous n'avons pas trouvé d'effet de la réduction de taille d'AP. Au contraire, les inégalités ont diminué à la suite d'évènements PADDD si la taille de la protection restait constante ou augmentait.

Dans le chapitre 1, nous avons également démontré que les agences environnementales ont un certain pouvoir de décision, puisque les réductions de taille d'AP étaient plus souvent proposées et promulguées de 2006 à 2015 lorsqu'elles étaient déjà défrichées et coûteuses à gérer. Dans le chapitre 2, nous avons trouvé que les réductions de taille d'AP ayant été promulguées de 2009 à 2012 ont augmenté la déforestation de 2010 à 2015, bien qu'elles étaient déjà défrichées en partie, dans les états de l'arc de la déforestation, et à des distances intermédiaires aux routes. Les pressions économiques existantes étaient en effet élevées comparé aux faibles capacités de défense des frontières des AP. Nous n'avons pas trouvé d'effet des réductions de taille d'AP loin des routes, où les pressions économiques étaient trop faibles ou trop près des routes car les AP étaient entièrement défrichées.

Cette thèse a de nombreuses implications. Dans un premier temps, elle met en lumière la nécessité de tenir compte de la dynamique du réseau d'AP dans les évaluations d'impact des AP, en fonction des caractéristiques des terres et des AP. Des efforts supplémentaires pour compléter les initiatives mondiales de documentation des événements PADDD, comme PADDDtracker.org, sont donc nécessaires pour réaliser des analyses d'impact complètes et aider à la prise de décision (Cook et al., 2017 ; Forrest et al., 2015 ; Golden Kroner et al., 2019 ; Pack et al., 2016 ; Qin et al., 2019).

Dans un contexte où les AP restent sous-financées, cette thèse peut aider, dans un second temps, à orienter les investissements nécessaires à la défense des AP là où ils sont susceptibles d'être les plus efficaces; et de mieux choisir les sites et types de gestion optimaux pour la mise en place de nouvelles AP. Par exemple, alors que nous trouvons que les réductions d'AP n'influencent ni la déforestation, ni la distribution des revenus dans les zones à faible pression économique, cette pression peut augmenter, notamment en raison du développement des routes. Il semble donc plus pertinent de renforcer les AP loin des pressions économiques en améliorant l'inclusion des populations locales. Plus près des pressions économiques, les réductions de taille ne semblent pas augmenter la déforestation et permettent une augmentation de la taille de la classe moyenne supérieure. Pourtant, les frontières de ces AP semblent moins coûteuses à défendre grâce aux faibles coûts de transport aux villes. Il conviendrait donc d'investir pour renforcer ces AP et ainsi éviter les évènements PADDD (Bernard et al., 2014 ; Forrest et al., 2015 ; Pack et al., 2016 ; Symes et al., 2016 ; Tesfaw et al., 2018). C'est à des distances intermédiaires aux routes que les évènements PADDD contribuent le plus à la déforestation. Il semble donc pertinent de renforcer les AP, en tirant éventuellement parti du budget dégagé par la réduction des aires protégées défaillantes (Fuller et al., 2010 ; Kareiva, 2010 ; Mascia et al., 2014). Pour renforcer leur efficacité, et légitimité, une attention particulière doit être portée au type d'AP mises en place (strictes ou durables) ainsi qu'à leur niveau de gouvernance (Etat ou gouvernement fédéral). Enfin, pour renforcer le réseau d'AP afin d'éviter les évènements PADDD ou s'il n'est pas possible d'éviter les évènements PADDD, il est également possible d'utiliser d'autres types d'instruments de conservation comme les Paiements pour Services Environnementaux (Mascia et al., 2014 ; Mascia and Pailler, 2011). Ceux-ci peuvent légitimer la défense des frontières des AP (Sims and Alix-Garcia, 2017) et se sont révélés efficaces dans la lutte contre la pauvreté (Alix-Garcia et al., 2018).

Plusieurs questions restent toutefois à traiter pour renforcer les implications politiques et permettre des prises de décision concrètes. Tout d'abord, les travaux portant sur les déclassements d'AP, sur les différences entre réductions et suppressions d'AP ou encore sur leurs causes, doivent être renforcés. En effet, il n'est pas certain que ces différents types d'évènements réduisent la protection de la même manière, ce qui peut générer des différences dans la façon dont les décisions sont prises ainsi que dans leurs impacts (Cook et al., 2017; Naughton-Treves and Holland, 2019 ; Mascia et al., 2014 ; Tesfaw et al., 2018). Il conviendrait également de répliquer les analyses de risque et d'impact des événements PADDD à d'autres pays et à d'autres types d'instruments de conservation (Qin et al., 2019; Symes et al., 2016 ; Forrest et al., 2015 ; Golden Kroner et al., 2016 ; Tesfaw et al., 2018). De plus, la nature dynamique des négociations entre les agences, des objectifs fédéraux ainsi que l'évolution de la pression économique dans le temps doivent également être pris en compte, en utilisant des données plus précises et d'autres méthodes empiriques (Pack et al., 2016 ; Tesfaw et al., 2018 ; Mascia et al., 2014). Pour finir, il est important que les recherches futures élargissent l'étude des impacts des évènements PADDD pour tenir compte de la déforestation à long-terme (Tesfaw et al., 2018), d'autres types de services écosystémiques (Cook et al., 2017 ; Tesfaw et al., 2018), et pour mieux évaluer le bien-être des populations locales.

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General Introduction

1 Protected Areas as Conservation Tools in the Brazilian Amazon

1.1 State of the Brazilian Legal Amazon within Brazil

Brazil is the 5th largest country in the world and encompasses several terrestrial ecoregions, of which the Amazon (MMA, 2017). The Brazilian Amazon contains 60% of the total amount of Amazon rainforest (West et al., 2019) and one third of world tropical moist forests (The World Bank, 2020). The Amazon rainforest acts as a large carbon reservoir, allows the maintenance of water cycling and contains among the highest biodiversity levels of the planet (IPBES, 2019; MMA, 2017). Thus, it provides a large extent of ecosystem services, of which climate change mitigation participates, as the release of carbon could lead to a wide increase in global average temperatures (Fearnside, 2008). The Amazon rainforest extends over nine states: Roraima, Amazonas, Acre, Rondônia, Pará, Amapá, and one part of Tocantins, Mato Grosso and Maranhão, which all forms the Brazilian Legal amazon, defined by law in an attempt to develop the region in the 1960's (IBGE, 2017).

The Brazilian economy has been based on the use of its forests through the collection of non-timber forest products, logging for timber, cattle pasture and agricultural production (Fearnside, 2008), especially since the 1990's when the country become a large supplier of beef and soybeans (Arima et al., 2014). As a result, Brazilian Amazon deforestation and forest degradation has been increasing (Figure 1), along with greenhouse gas emissions, and consequent biodiversity losses (Fearnside, 2008; IPBES, 2019). The largest contributor is the land-use and land cover change sector (Stabile et al., 2020; West et al., 2019) but infrastructure development (e.g. roads, dams) (Arima et al., 2014; Fearnside, 2008) has also been a large contributor, in an attempt to answer the need of an increasing population density. Another factor accelerating deforestation is land speculation linked to insecure land tenure, and the presence of illegal economic activities (e.g. drug traffic) (Fearnside, 2008).

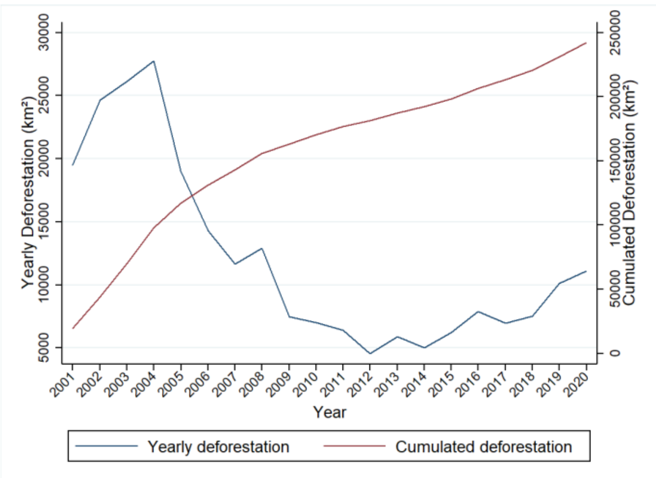


Figure 0-1 - Evolution of deforestation in square kilometers from 2000 to 2020
Source: INPE (2019)

Brazil committed to reduce its greenhouse gas emissions by 37% below the 2005 levels during the 11th COP of the United Nation Framework Convention on Climate Change (UNFCCC), held in 2015 in Paris, and to reduce them by 43 below 2005 levels in 2030. Brazil's intended Nationally Determined Contribution (NDC) is carried out under the National Policy on Climate Change. Objectives related to land use change and forests are the following (Federative Republic of Brazil, 2015):

- "strengthening policies and measures with a view to achieve, in the Brazilian Amazonia, zero illegal deforestation by 2030 and compensating for greenhouse gas emissions from legal suppression of vegetation by 2030";
- "restoring and reforesting 12 million hectares of forests by 2030, for multiple purposes";
- "enhancing sustainable native forest management systems, through georeferencing and tracking systems applicable to native forest management, with a view to curbing illegal and unsustainable practices".

Brazil also seeks to improve water security and the conservation and sustainable use of biodiversity (Federative Republic of Brazil, 2015). Indeed, from 1992, Brazil has been a member of the Convention on Biological Diversity (CBD), which objectives are "the conservation and sustainable use of biodiversity, and the fair and equitable sharing of benefits arising out of the use of genetic resources and of associated traditional knowledge." (MMA, 2017).

However, Brazil, and more specifically the Brazilian Legal Amazon, faces multiple challenges in terms of poverty eradication, education, health, infrastructure, and employment among others (Federative Republic of Brazil, 2015; The World Bank, 2020). Protecting the forest and reducing greenhouse gas emissions could thus prevent economic development (Federative Republic of Brazil, 2015; IPBES, 2019). Yet, the country will likely bear most costs from climate change (IPBES, 2019), since, for example, environmental degradation can threaten the provision of ecosystem services such as timber and non timber forest products, especially for nature-dependent local population or indigenous people (Wunder et al., 2014), and can lower future land productivity (Celentano et al., 2012).

Therefore, Brazil must also adapt to climate change by building conservation tools that do not hinder development opportunities (Börner et al., 2020). The resilience of population, infrastructures and production systems is thus a core concept of their NDC, which underlines the principle of common but differentiated responsibilities in climate change (Federative Republic of Brazil, 2015). As a result, reducing emissions from deforestation and forest degradation (REDD+), that aims to reward developing countries when they achieve reduction emissions as compared to a business as usual scenario (UNFCCC, 2020), is also integrated in Brazil NDC (Gallo and Albrecht, 2019). Since 2015, Brazil is also part of the 2030 Agenda for Sustainable Development that aims to eradicate poverty and build a sustainable world (United Nations, 2015).

1.2 Protected Areas to Safeguard Biodiversity and Reduce Greenhouse Gas Emissions

Protected Areas (PAs) are defined as "clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values." (IUCN WCPA, 2020). Since the 1980 first world park congress, PAs have been a key component of the Brazilian strategy to safeguard forest and maintain the production of their ecosystem services (MMA, 2017; Veríssimo et al., 2011). PAs extent have been increasing especially from the 2000's (Veríssimo et al., 2011), following Brazilian membership in the CBD. Under the CBD strategic plan 2002-2010, countries recognize that the high increase in tropical deforestation poses severe threats to biodiversity and commit themselves to create a comprehensive and representative system of PAs. The first phase of the Amazon Region PA program (ARPA) was designed in 2003 in order to invest in PA creation and management and to regularize land tenure.

The National System of Protected Areas (SNUC) was thus created to harmonize existing PAs as Conservation Units (CUs) according to management categories defined by the International Union for the Conservation of Nature (IUCN)[1] (Naughton-Treves et al., 2005; OECD, 2015; Veríssimo et al., 2011). CUs are designed at federal, state and municipal levels to conserve biodiversity. While in strict PAs, the use and consumption of natural resources are not allowed, sustainable use PAs permit human settlements and the use of natural resources sustainably (MMA, 2017; OECD, 2015), since their expansion might come at the expense of local communities (Naughton-Treves et al., 2005). Other types of PAs are composed of territories of traditional occupation (i.e. Quilombola territories) and indigenous lands, which safeguard the "social organization, uses, languages, beliefs and traditions of these peoples and communities" (MMA, 2017).

In 2010, during the 10th Conference of the Parties, Brazil then committed to the 2011-2020 strategic plan for biodiversity of the CBD. Under the 11th target, countries committed that "In 2020, at least 17 percent of terrestrial and inland water, [...], especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas [...]." (Convention on Biological Diversity, 2020). The national Brazilian biodiversity strategy and action plan initiated more ambitious projects specifying that at least 30% of the Amazon should be conserved under the SNUC in 2020.

[1] IUCN management categories are defined according to PAs management objectives. From category I to III, no human or minimum human intervention are allowed and PAs are considered as strictly managed. Category I "Strict Nature Reserve" and "Wilderness Area" are managed for science or wilderness protection. Category II "National Park", for ecosystem protection and recreation and category III "Natural monument" for specific natural features. From category IV to VI, human interventions are allowed and PAs are considered to be sustainably managed. Category VI "Habitat/Species management" aims to conservation through management intervention, category V "Protected Landscape/Seascape" are managed for landscape/seascape conservation and recreation, and category VI "Managed Resource PA" for the sustainable use of natural resources (Dudley and Phillips, 2006; UNEP-WCMC and IUCN, 2016)

PAs could further play an active role to meet the goals defined under the Paris agreement and REDD+ framework (Forrest et al., 2015; Mascia and Pailler, 2011; UNEP-WCMC et al., 2020; Watson et al., 2014) as they protect forests and relevant ecosystems with potentially high level of carbon storage. International pressures to ensure that PAs contribute to supporting local livelihoods have also been growing since the 1980's (Naughton-Treves et al., 2005). PAs can indeed contribute to the 2030 sustainable development goals if they are equitably and effectively managed, e.g. by enhancing food security, improving water security, reducing inequalities and by offering development opportunities through tourism (UNEP-WCMC et al., 2020). In that sense, target 14 of the 2011-2020 strategic plan for biodiversity also indicates that the benefits related to the conservation of biodiversity should be distributed to all (MMA, 2017).

2 Challenges Associated with PA Designation and Maintenance

2.1 Conflicts over Land-Use and Location Bias

In 2017, 17.5 % of Brazilian continental areas were protected under the SNUC law. In the Brazilian Amazon, this number was 27% in 2015 (MMA, 2017; OECD, 2015). In 2019, progress toward the biodiversity Aichi target were achieved in terms of surface but PAs remained poorly managed and lacked ecological representativeness (IPBES, 2019; UNEP-WCMC et al., 2020).

When PAs are implemented to conserve forests, they prevent other productive use of land such as agriculture or logging, which are characterized by rents for private landowners (Pfaff et al., 2015a, 2014, 2009; Sims, 2010). Setting aside land that would have otherwise generated private profits thus have an economic opportunity cost. This relation is often studied using a von Thünen model (Angelsen, 2007; Thünen V., 1875) where lands are allocated to the uses producing the highest rents. Rents are determined by several factors such as the price of products sold in the nearest market, transport costs to the city and other inputs costs that depend on available technologies and lands agro-ecological conditions. If we consider a monocentric city surrounded by homogenous lands, the rent is simply the difference between the price of the product and transport costs, which increase with the distance to the city. Consequently, intensive economic activities such as agriculture occur nearest to the city, followed by extensive agriculture or managed forests for logging. As protected forests are often considered to have no or low value for private land rents, they are located far away from the city center. As a result, designing PAs near to cities or on lands that are suitable for the development of economic activities could constrain local economic development (Robalino, 2007; Sims, 2010).

Designing PAs on high economically valuable lands might in addition be challenging when comparing expected costs and benefits of conservation. First, lands are more expensive for conservation agencies when they are located where the opportunity cost is high (Ando et al., 1998; Börner et al., 2020; Pfaff and Sanchez-Azofeifa, 2004). Second, high opportunity costs generate high threats on forests, with potentially more conservation benefits, but also more risks of losing them (Pfaff and Sanchez-Azofeifa, 2004). Globally, PAs in high-pressure locations would have a higher likelihood of being invaded with capital-intensive land uses (Blackman and Villalobos, 2019; IPBES, 2019; Thieme et al., 2020). Indeed, if a high opportunity cost land is protected, the rent associated with e.g. illegal logging might exceed the costs related to illegally invading the PA (e.g. due to a fine) (Robinson et al., 2010; Sims, 2014). Since enforcement is not perfect within PAs (Albers, 2010), it might be more optimal for a conservation agency, when comparing expected costs and conservation benefits, to buy cheaper land but with less conservation value (Ando et al., 1998) or that are less threatened (Pfaff and Sanchez-Azofeifa, 2004).

Areas that are less profitable for the development of economic activities have a lower economic opportunity cost, which make them less expensive to conserve but results in a "location bias" of PAs. Globally, PAs have been found to be located on average at higher elevation, on steeper slopes and at greater distance from roads and cities (Joppa and Pfaff, 2009; Nelson and Chomitz, 2011). This location bias has been confirmed for the Brazilian Amazon (Anderson et al., 2016; Herrera et al., 2019; Kere et al., 2017; Nepstad et al., 2006; Nolte et al., 2013; Pfaff et al., 2015a, 2015b, 2014), where a lack of effective governance (Reydon et al., 2020) and budget allocated for PA management (Veríssimo et al., 2011) have been reported. Besides, the PA network would on average be biased toward sustainable use areas, as they do not formally exclude human activities (Peres, 2011). Strictly managed PAs would rather be located in lower opportunity costs areas, since fewer efforts are thus needed to enforce and manage them (Nolte et al., 2013). This location bias can vary according to time as deforestation activities increase (Nepstad et al., 2006). For example, strict PAs were designed nearer to threats when they were designed after 2000 (Nolte et al., 2013) and in Acre, PAs created before 2000 were located nearest to forest edges than PAs designed in 2004-2008 (Pfaff et al., 2014).

2.2 Have Protected Areas Been Effective?

2.2.1 Methodological Challenges

PAs must still extend in the Brazilian Amazon to meet the goals of the national biodiversity strategy and progresses in terms of management still need to be achieved (Veríssimo et al., 2011) to meet several CBD targets (UNEP-WCMC et al., 2020; Visconti et al., 2019). Under strong budget constraints and increasing economic pressures within and around PAs (Heino et al., 2015; Soares-Filho et al., 2010; Watson et al., 2014), it is necessary they have an impact by avoiding deforestation. In other words, PAs must reduce deforestation as compared to a baseline scenario where they would not have existed (Joppa and Pfaff, 2009; Pfaff and Sanchez-Azofeifa, 2004). Besides, since progresses toward achieving conservation might undermine sustainable development goals (IPBES, 2019; Naughton-Treves et al., 2005), PAs must also contribute to enhance local livelihood (Naughton-Treves et al., 2005; Watson et al., 2014).

Conceptually, a parcel of land is cleared if the rent a landowner can make is positive (Pfaff, 1999), which is the case of lands with suitable conditions for the development of economic activities. If a PA is implemented and reduces the rent so that clearing becomes unprofitable, the PA will have avoided deforestation as compared to that baseline scenario (Ferraro and Hanauer, 2014; Joppa and Pfaff, 2010). Hence, when PAs avoid deforestation, they prevent other economic activities, which result in a trade-off between conservation and economic development objectives (Ferraro and Hanauer, 2011). PAs impacts, on both deforestation and development, could however be limited for two reasons (Anderson et al., 2016; Nepstad et al., 2006; Nolte et al., 2013):

- If the rent would not have been positive in the absence of the PA, i.e., in land were economic activities are not profitable and where PAs are on average located;
- If the rent would have been positive in the absence of the PA, but the PA is not effectively enforced and faces some internal deforestation.

These counterfactual situations are not observed, which makes it empirically challenging to measure PAs impacts (Ferraro and Hanauer, 2014). Indeed, correlations between treatment and outcome are often observed due to the presence of confounders driving both PAs and deforestation location. Researchers have estimated those counterfactual outcomes on a global scale and for developing region and countries (see e.g. Andam et al. (2008); Blackman et al. (2015); Bowker et al. (2017); Ferraro et al. (2013) or Pfaff et al. (2017) for deforestation impact analysis). Matching strategies or difference-in-differences estimations are often used to compare PAs to similar unprotected areas along key characteristics driving both conservation, deforestation and economic development outcomes (Ferraro and Hanauer, 2014).

2.2.2 Summary of Main Empirical Results

Regarding PAs impacts on deforestation, researchers find that simply comparing deforestation in PAs versus in unprotected areas lead to an overestimation of avoided deforestation due to location bias (Anderson et al., 2016; Jusys, 2018; Kere et al., 2017; Pfaff et al., 2015a, 2015b). Yet, despite being low, PAs impacts remain negative on average in the Brazilian Amazon (i.e. PAs reduce deforestation) (Nolte et al., 2013; Soares-Filho et al., 2010).

Indeed, PAs implemented nearer to economic pressures and sufficiently enforced do avoid deforestation. This was the case for strict PAs in Pará (Jusys, 2016), and in the Brazilian Amazon (Jusys, 2018; Pfaff et al., 2015b), even when levels of economic pressures are rather low (Anderson et al., 2016; Nolte et al., 2013). PAs managed at the federal level would also be more effective as they would benefit from a large amount of resources, better management and would be supported by broader environmental policy (Herrera et al., 2019). Indigenous Lands seemed to be more effective too when they had better management conditions (Kere et al., 2017; Soares-Filho et al., 2010), but they did not block as much pressures, due to lack of political power (Herrera et al., 2019). High economic pressures may induce lower PAs impacts if enforcement fails. For example, even though sustainable use PAs were found to block some threats in Acre (Pfaff et al., 2014), they did not longer avoided deforestation from 2009 to 2014 in the Brazilian Amazon, as they were highly deforested (Jusys, 2018), suggesting lack of enforcement. PAs located farther from economic pressures always have low or no impact on avoided deforestation since lands are not profitable (Herrera et al., 2019; Kere et al., 2017; Pfaff et al., 2015b, 2015a, 2014).

Regarding PAs impacts on economic development, research that both evaluate deforestation and economic development outcomes following PAs implementation do not concern the Brazilian Amazon but rather Costa Rica, Bolivia and the Peruvian Amazon (Ferraro and

Hanauer, 2011; Hanauer and Canavire-Bacarreza, 2015; Miranda et al., 2019). As expected, while no effect was found far from economic pressures, PAs exacerbated poverty when they reduced the rent from agricultural activities. However, on a global scale, PAs have been found to positively influence human well being when they provided tourism opportunities and allowed access to forest resources (Naidoo et al., 2019). In the Brazilian Amazon, Kauano et al. (2020) found no impact of PAs from 2004 to 2014, except on the growth rate of the industry sector, which was reduced.

3 Protected Area Downgrading Downsizing and Degazettement

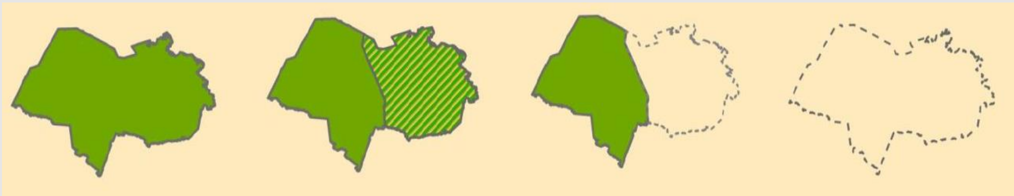
3.1 Definition, Description and Extent in the Brazilian Amazon

PAs lack of effectiveness (Jusys, 2018) and their expected detrimental effect on economic development (Kauano et al., 2020), have eventually poses the question of PAs permanence (Golden Kroner et al., 2019; Naughton-Treves and Holland, 2019; Qin et al., 2019), beyond their implementation and even beyond their impact. Widespread PA downgrading downsizing and degazettement (PADDD) events have occurred on a worldwide basis and affected more than 500,000 square kilometers of PAs since the beginning of the twentieth century (Mascia et al., 2014; Mascia and Pailler, 2011), mostly across developing regions such as Africa, Asia and Latin America, and in areas of high biodiversity value and greenhouse gas stocks.

In Latin America, Brazil includes the highest number of PADDD events (Mascia et al., 2014) – 93 events were recorded from 1981 to 2012 (Bernard et al., 2014) –, despite Brazilian commitments to the CBD and the Paris agreement. PADDD started as of 1970 but increased a lot from the mid 2000's (Figure 1) (Golden Kroner et al., 2019; Naughton-Treves and Holland, 2019; Qin et al., 2019). Yet, the literature related to PADDD in Brazil remains scarce. While in the beginning of the 2000's, PADDD events were mostly related to reclassification of PAs under the SNUC law (Bernard et al., 2014), from the mid-2000's, PADDD events have been related to increasing development pressures and lack of effective PA management, combined to a rather weak influence of environmental agencies (Bernard et al., 2014; Marques and Peres, 2015). Increase in presidential decrees, without technical studies and consultation of civil society (Bernard et al., 2014), and the fact that the executive branch did not veto proposals would reflect a new orientation of executive branches toward development objectives (Bernard et al., 2014; Marques and Peres, 2015).

PADDDtracker, WWF and Conservation International

Downgrading is defined as “a decrease in legal restrictions on the number, magnitude, or extent of human activities within a PA (i.e., legal authorization for increased human use)”; downsizing as a” decrease in size of a PA as a result of excision of land or sea area through a legal boundary change”; and degazettement as “a loss of legal protection for an entire PA” (Mascia and Pailler, 2011).



Original PA downgrade downsize degazettement

Source: Mascia et al. (2014)

The reasons behind PADDD mostly include industrial-scale resource extraction and development (forestry, mining, oil and gas, industrial agriculture, industrialization, infrastructure), local land pressures and land claims (rural settlement, land claims, subsistence, degradation, shifting sovereignty, refugee accommodation), as well as conservation planning, i.e., revision of PA systems. (Mascia et al., 2014).

PADDD proposals take the form of bills in the chamber of deputies or in the senate, at state or federal level, often following regional developers and local communities’ proposals. To be valid, they must be enacted, i.e., pass both congress chambers and then be sanctioned by the president at the federal or state level (Marques and Peres, 2015). Decisions can also be unilaterally taken by executive decrees, which does not require congress approval (Pack et al., 2016).

These events have been recorded since 2009 by WWF and Conservation International through PADDDtracker.org (Conservation International and World Wildlife Fund, 2017a; Mascia et al., 2012) using a review of the available scientific and grey literature (Mascia et al., 2014). PADDDtracker.org spatially records validated data about the frontier and year of proposed events, which are under consideration by legal authorities, as well as enacted events, which have been passed into law. Their cause, as well as the size of affected PAs are also reported. The base layer map is the 2012 version of World Database in Protected Areas (WDPA). As a result, these data also contain the name, type and the year of creation of PAs. The version 1.1 that we use throughout this thesis has been released in 2016 and records events until 2015 only. A new version have been released in 2016 and accounts for events until 2017 (Conservation International and World Wildlife Fund, 2017a).

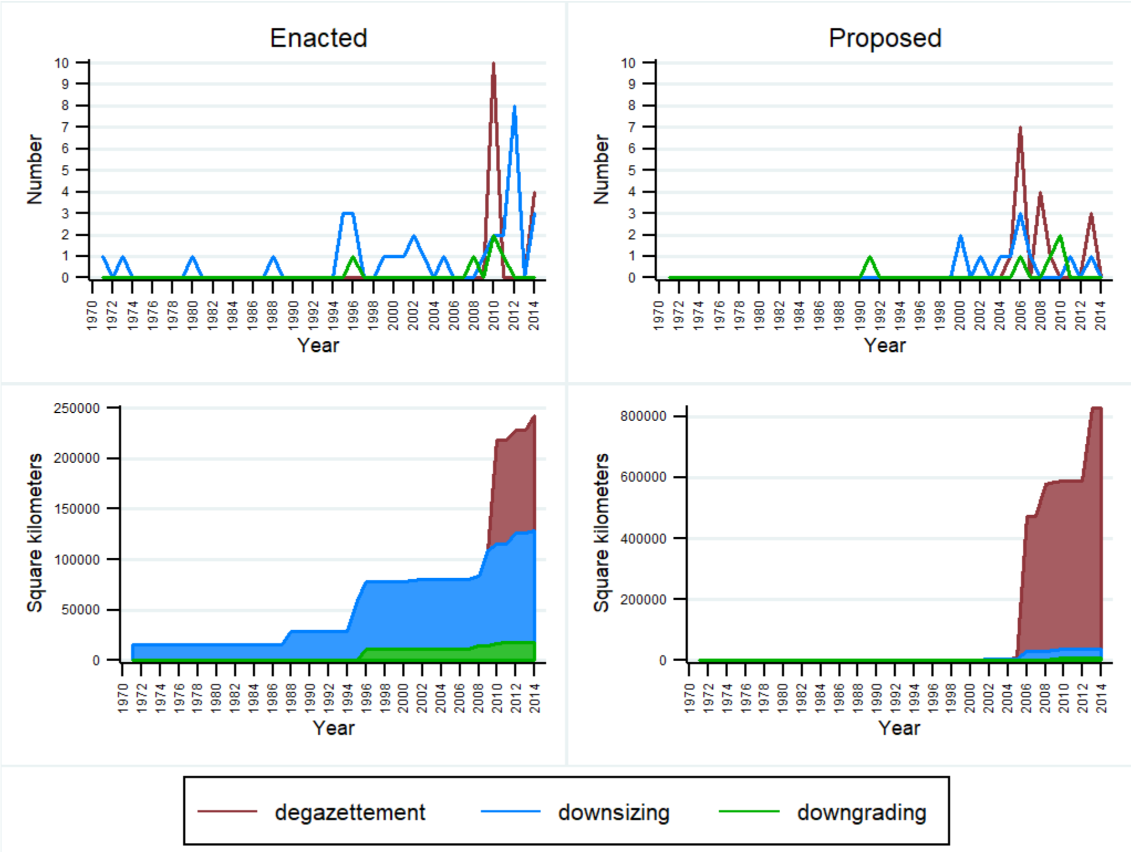


Figure 0-2 Number and extent of PADDD in the Brazilian Amazon
Source: author calculation (Conservation International and World Wildlife Fund, 2017b)

Using PADDDtracker Data Release version 1.1. (Conservation International and World Wildlife Fund, 2017b), as well as existing descriptive research, helps us having a first sight at the extent of the phenomenon. In 2014, approximately 242,000 square kilometers of PAs in the Brazilian amazon were already undermined through PADDD, and ~828,000 supplementary square kilometers were under risk of being downgraded, downsized and degazetted (Figure 1 and 2). PADDD, whether enacted or proposed, especially involved degazettements and downsizings, both in number and extent (Figure 1 and 2). 14 and 16 degazettements, as well as 33 and 11 downsizings were respectively enacted and proposed while it was the case for only 3 and 5 downgradings from 1970 to 2014. Enacted downsizings were more frequent than any other type of PADDD events before 2000 and represented most lost PA surface, while after 2010, enacted degazettement are as numerous and represent as much PA lost surface. Proposals for PADDD, especially for degazettements, severely increased after 2000. Enacted PADDD occurred more at the state level, by congress decisions as their members would be more vulnerable to political lobbying from agribusiness, construction or energy sectors at local levels (Bernard et al., 2014). They were not uniformly distributed across states (Figure 2). In Acre, Amapá, Roraima, Maranhão, Tocantins and Mato Grosso, which are, either small states, or states with few PAs (Figure 2), none, or a maximum of 3 PADDD per states were proposed or enacted. Conversely, 31 events were enacted in Rondônia, 14 in south-east Amazonas and 8 in Pará. These states are largely covered with PAs, and are located in the active deforestation frontier. PADDD also occurred more in strict than in sustainable use PAs even though proposals and enactments are evenly distributed across each PA type (Pack et al., 2016). While, there would be some cases of PADDD in indigenous territories, they have not been formally recorded yet (Marques and Peres, 2015). 3 enacted downsizings have been compensated with increased protection elsewhere in 2002 and 2011 in Rondônia and in 2012 in Amazonas.

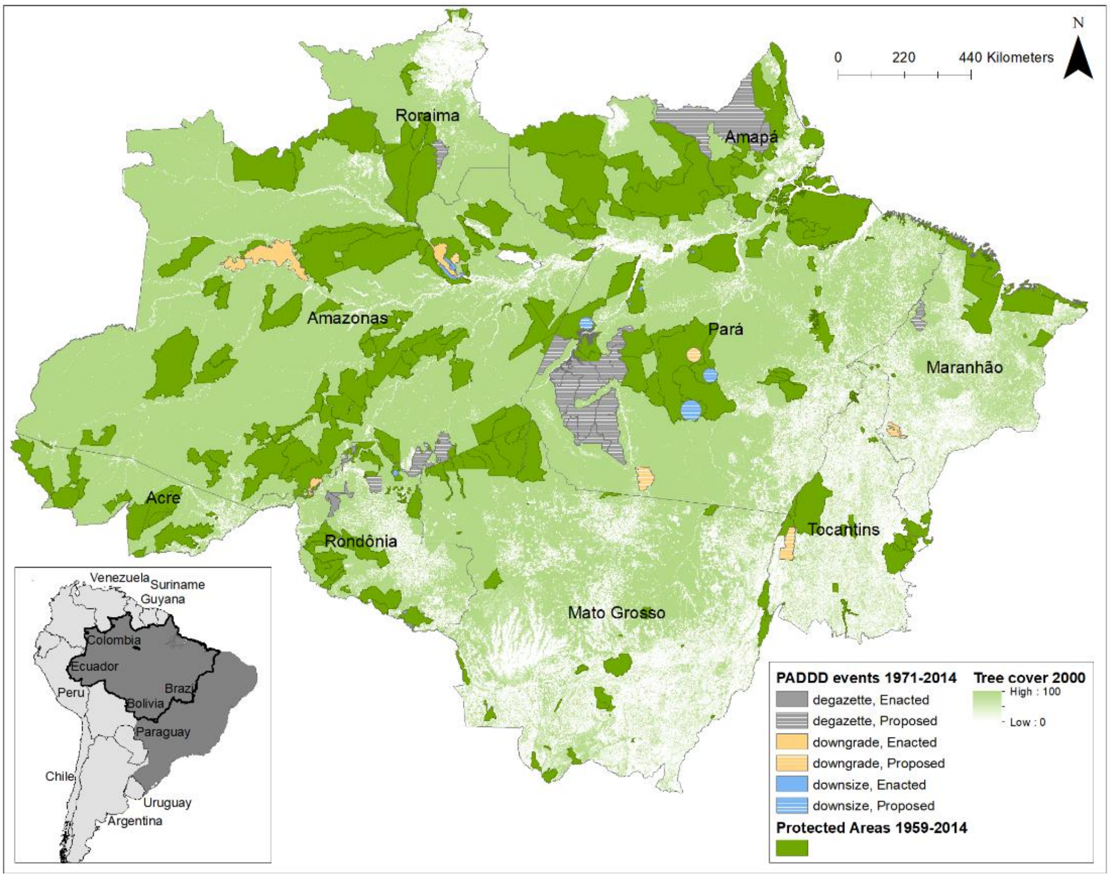


Figure 0-3 Location of PADDD in the Brazilian Amazon
Source: author calculation (Conservation International and World Wildlife Fund, 2017b; IUCN and UNEP-WCMC, 2016)

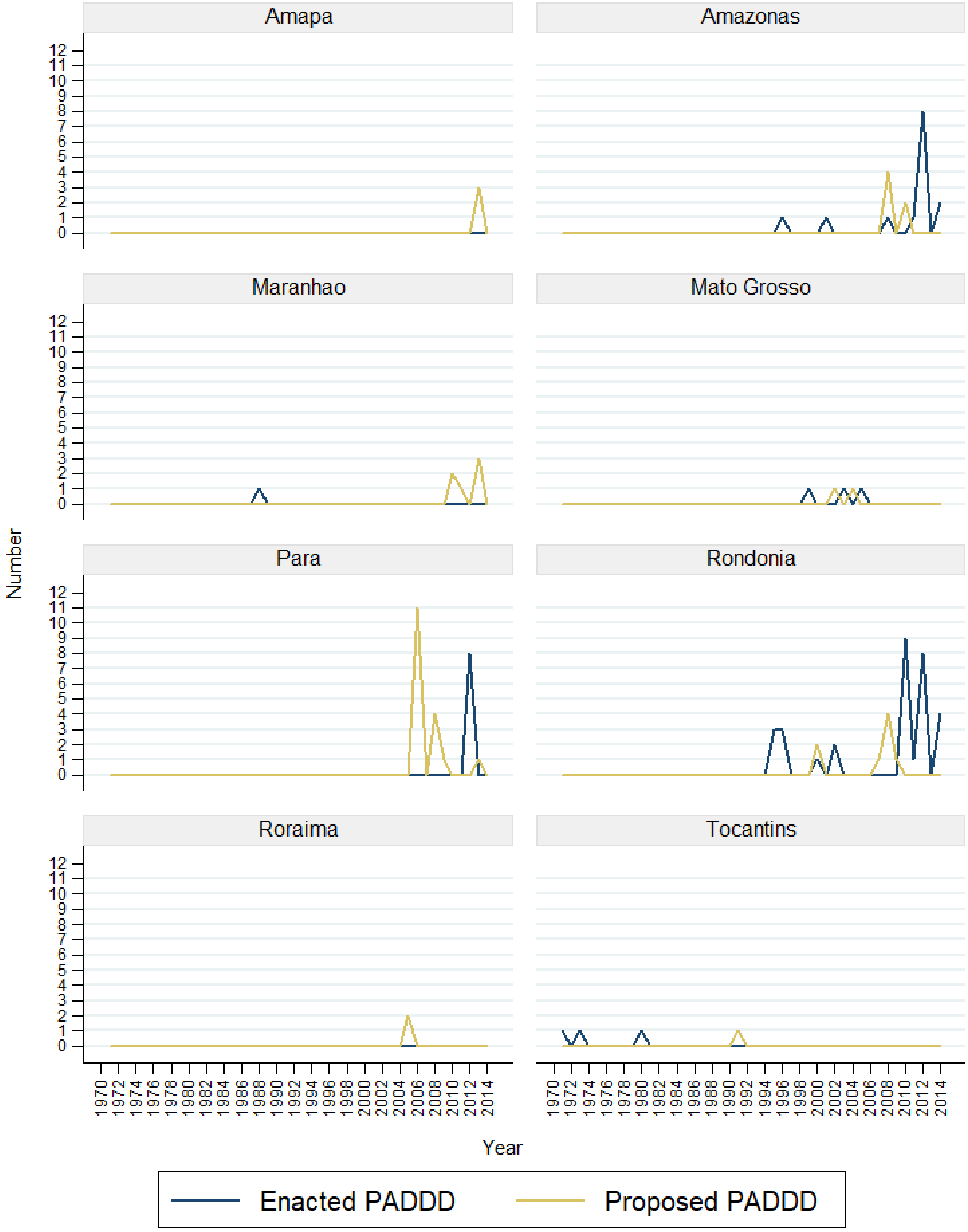


Figure 0-4 Temporal variation of proposed and enacted PADDD within states
Source: author calculation (Conservation International and World Wildlife Fund, 2017b)

3.2 PADDD as an outcome of the conservation-development trade-off in the Brazilian Amazon?

PADDD have recently increased from 2015 onwards as can be observed in PADDDtracker data release version 2.0 (Conservation International and World Wildlife Fund, 2017a) along with a surge in deforestation rates (INPE, 2019). PADDD are likely to intensify in the future, especially now that Bolsonaro's administration has put continuous efforts in dismantling environmental laws (Escobar, 2019; Fearnside, 2016; Ferrante and Fearnside, 2019). Besides, the covid-19 pandemic might increase economic pressures around PAs due to lack of financial and human resources for their management and enforcement, and may deepen regulatory rollbacks against conservation policies, despite their importance in preventing future pandemics (Hockings et al., 2020).

The objective of this thesis is to examine if, where, and to what extent PADDD events interact with conservation-development trade-offs in the Brazilian Amazon, both in terms of decisions (chapter 1) and impacts on deforestation (chapter 2) and local economic development (chapter 3).

Besides the conservation of biodiversity, PAs could represent additional and permanent emissions reductions if they were additional in reducing deforestation and deforestation increased following PADDD, which has strong implications for the Paris Agreement and REDD+ (Forrest et al., 2015; Mascia and Pailler, 2011). In addition, PADDD could influence Brazilian progress toward several targets of the 2030 agenda for sustainable development, depending on PAs contributions to economic development. The question of where and why PADDD events occur, as well as their impacts is thus of primary importance to design relevant conservation instruments that fit with international commitments.

Since location biases are likely to influence PADDD events' additional effect on deforestation and economic development, the first aim of this thesis is to highlight the factors driving PADDD and the characteristics of their location in the Brazilian Amazon. Precisely acknowledging their drivers will lead us, in a second chapter, to better assess PADDD impacts on deforestation according to lands and PAs heterogeneities. Finally, as PAs have been criticized due to the land restriction they impose, it seems important to understand whether PADDD events had their intended effect on economic development, according to landscape and PA heterogeneities. This will be the focus of our third chapter.

Recognizing where and how conservation and development outcomes have been affected by PADDD events could help to allocate scarce resources and to guide investments to consolidate the protected area network where it would be most effective (Fuller et al., 2010; Symes et al., 2016). For example, if some PAs were to be downgraded, downsized or degazetted due to environmental inefficiencies and because they induce poor economic outcomes, new areas (Fuller et al., 2010), or new conservation solutions that better integrate both objectives, could replace them (Golden Kroner et al., 2019; Qin et al., 2019; Tesfaw et al., 2018).

3.2.1 Chapter 1: What drives the Erasure of PAs? Evidence from the Brazilian Amazon

We assess PADDD events drivers in the Brazilian Amazon by examining how interactions between development and conservation agencies may induce PA size reductions (downsizing and degazettement) according to PAs and lands characteristics. This article has been published in Ecological Economics and in the BETA working paper series.

First, we use Tesfaw et al. (2018) framework, conceptualize it, and go further. We spell out how PA size reductions affect development and environmental agencies when PAs are enforced, i.e., they entirely block economic pressures. Expected economic benefits of PA size reductions equal foregone profits related to PAs designations. Environmental benefits of avoiding PA size reductions equals land clearings that PAs helped deterring. These are actually conservation opportunity costs, which we represent with distances to nearest cities. Socially efficient decisions might be reached when economic gains from PA size reductions are high while environmental benefits from avoiding them are low, resulting in PA size reductions, or when economic gains are low and environmental benefits high, resulting in PAs maintenance. To allow for spatial variations, we assume PAs faced internal deforestation activities, which reduces both development and environmental agencies benefits from PA size reductions. PAs internal deforestation can be either high or low near cities depending on the effects of transport costs on enforcement and illegal deforestation activities. We obtain several spatial predictions on agencies' relative bargaining power had PA size reductions occurred near cities according to both assumptions.

Second, in order to test these predictions, we study drivers of proposed and enacted PADDD from 2006 to 2015 over the entire Brazilian amazon. While we confirm that illegal activities within PAs rather occur far from cities, suggesting PAs are better enforced near cities in spite of high economic pressures, we also find PA size reductions were more likely to occur near to these cities. Since PA size reductions could have induced strong environmental damages, this suggests that development agencies had more bargaining power. However, environmental agencies had some power too as, consistently with Tesfaw et al. (2018), PA size reductions were more likely when PAs were already illegally invaded or when they featured high management costs, as demonstrated by their size and their IUCN categories.

3.2.2 Chapter 2: Does the Selective Erasure of Protected Areas Raise Deforestation in the Brazilian Amazon?

In this chapter, we assess whether PA size reductions had an additional impact on deforestation in the Brazilian Amazon. Since both development and environmental agencies have varying bargaining power in the landscape, it is actually challenging to predict whether PA size reductions actually increased deforestation. As a result, we take into account location bias according lands and PAs heterogeneities.

First, we follow the Von Thünen model and assume that PAs have an additional impact on deforestation where they reduced the rent from agricultural activities. PAs may discourage agricultural activities either near cities, if they are enforced, or at intermediate distance from cities, if diminishing economic pressures are outweighed by remaining PAs enforcement. We predict that PA size reductions will induce additional deforestation activities, where PAs have actually avoided them.

Second, we estimate the impact of PA size reductions enacted between 2009 and 2012 on deforestation occurring from 2010 to 2015, as compared to constant PAs using matching techniques at the pixel level. To assess whether reduced PAs previously blocked deforestation, we estimate their impact when they were protected before 2000, on deforestation occurring from 2001 to 2008, as compared to similar unprotected lands. We estimate their impact according to road distance, which stands for distances to cities, and according to states, which features varied contexts of economic pressures. We confirm the existence of strong location biases for PAs designations and size reductions. Then, we find PA size reductions had no additional impact, both before and after being enacted, in low economic pressures areas. Conversely, in high economic pressures areas, PAs selected for size reductions were already deforested, consistent with Pack et al. (2016) and Tesfaw et al. (2018) leading to no additional impact. However, enacting size reductions seems to have reinforced deforestation as compared to similar protected lands when they were invaded to some extent, at intermediate distanced from roads.

3.2.3 Chapter 3: Who benefits from PADDD in the Brazilian amazon?

In this third chapter, our aim is to provide the first empirical assessment of PADDD events' impacts on socioeconomic outcomes in the Brazilian amazon. This question was indeed neglected in the literature while allowing land to be used productively is one of the primary objectives of PADDD.

Even though PAs may enhance local livelihood when they attract tourism activities, foster the development of infrastructure or by acting as a safety net for forest dependent communities, they are generally expected to prevent economic development when they avoid deforestation. PADDD events are located in profitable areas and seem to have increased deforestation despite their illegal invasions. It is thus challenging to evaluate whether PADDD actually enhanced economic development. In addition, they were enacted for various reasons, from allowing tourism activities, to implementing large-scale industrial agriculture.

First, we distinguish between enacted PADDD from 2001 to 2010 that contributed to reduce PAs size and PADDD where protection increased, due to downgrading or a combination of downgradings, size reductions and replacement by new PAs designations. Second, we use a difference-in-differences estimations on pre-matched samples, according to road distances to account for location bias according to lands characteristics. Third, we work on the distribution of revenues and on inequalities to appreciate whether some part of the population have benefited from these events more than other parts. We use data provided by Masson (2020) that offer the distribution of households' incomes at a fine scale over the Brazilian Amazon.

When PAs size were reduced, near roads, the growth of the higher-middle income class accelerated. New economic opportunities and enhanced access to markets might have enriched lower-middle income class households, or proximity of existing markets might have spurred in-migrations of higher-middle income class households. Far from roads, when PAs size increased, inequalities were reduced by limiting the growth of households without revenue, and fostering that of lower-middle income class households. Better management of PAs and new economic activities induced by the development of tourism within PAs might have led poorest households to access richer classes or, loss of secure access to forest resources might also have led to out-migration of poorest households. New activities and infrastructure might have spurred in-migrations of lower-middle income class households.

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Chapter 1

What Drives the Erasure of Protected Areas?

Evidence from the Brazilian Amazon

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Abstract (199 words)

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.

Keywords

land use, forest, protected area, conservation; PADDD; Amazon; Brazil

Publications

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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, 2015b, 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 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

[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 downsizing or degazettements were followed by a designation as Indigenous Lands.

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.

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 = σ_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:

$$B^D(R_i) = \delta \, \sigma_i \, R_i$$

$$R_i^* = 1 \; \forall \, \sigma_i \geq 0$$

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] 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.

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):

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

$$R_i^* = 0 \;\forall\, \sigma_i \geq 0$$

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

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

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 ($d_i^I(T)$) 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 ($1 - d_i^I$).

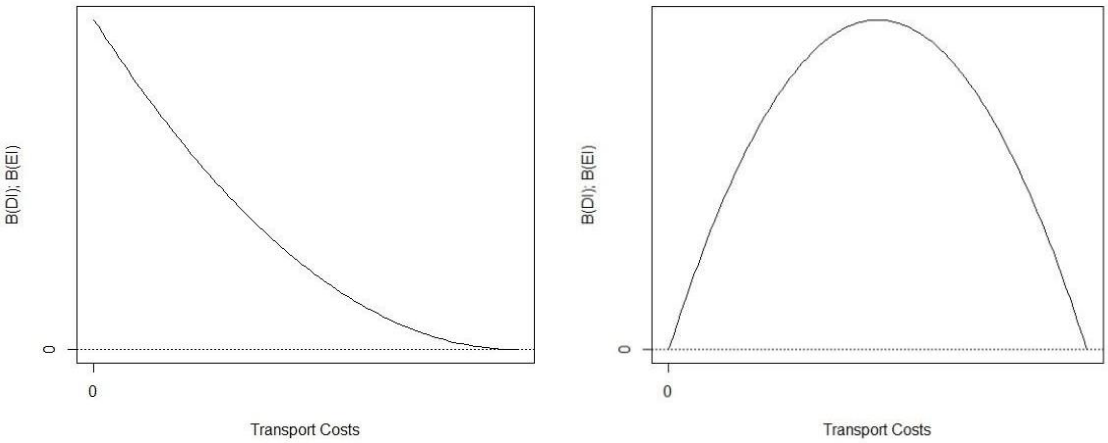
$$\mathrm{B}^{\mathrm{DI}}(R_i) = \mathrm{B}^{\mathrm{D}}(R_i)\,(1 - d^I_i) = \delta\,\sigma_i\,R_i\,(1 - d^I_i)$$

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

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.

[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).

<u>Figure 1 (1A and 1B)</u> D benefits (BDI) & E costs (BEI) of PAs size reductions given internal damages



Figures 1A and 1B show relevant possibilities for when T has a clear net effect on illegal damages in protected areas. Figure 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 Figure 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 Figure 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 Figure 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

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

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 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 (Figure 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.

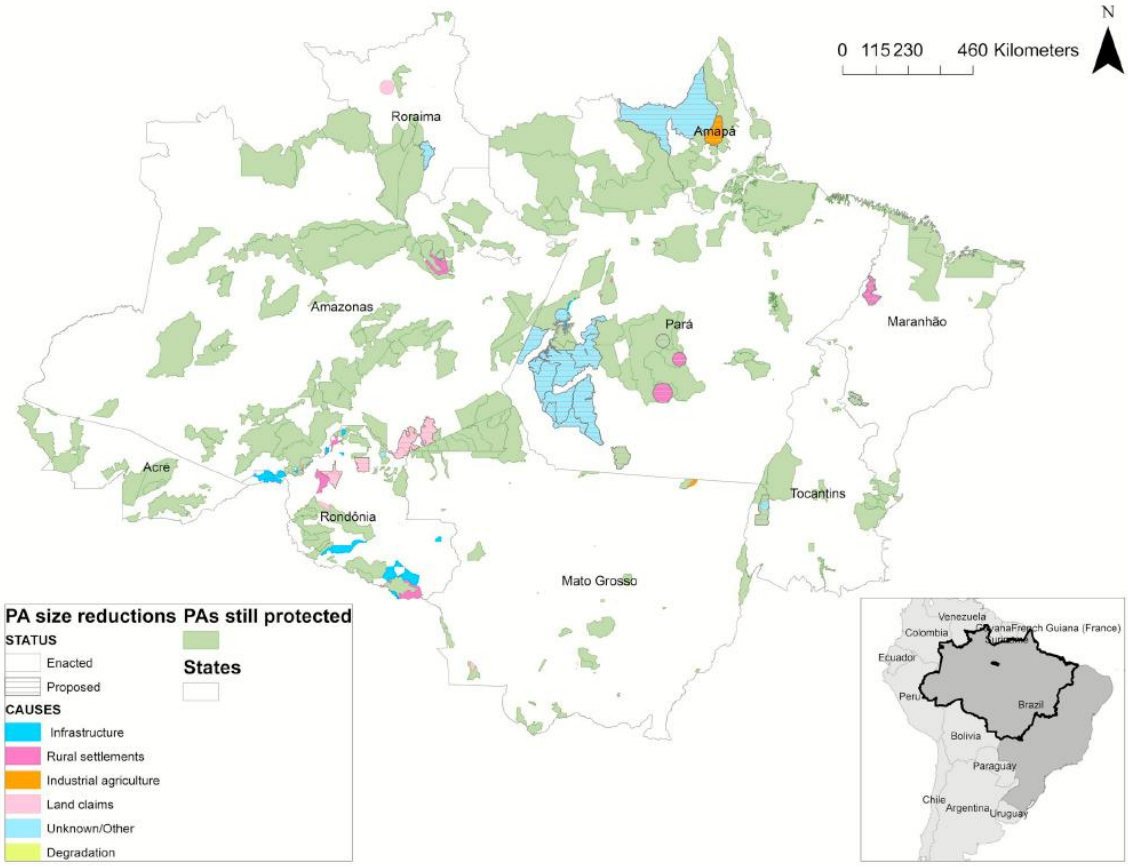

Figure 1-2 Listed Proximate Causes of Brazilian Amazon PA Degazettements & Downsizings

Source: author calculation (Conservation International and World Wildlife Fund, 2017b; IUCN and UNEP-WCMC, 2016)

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 (DNIT, 2017), as seen within Figure 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 Figure 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), 2017).

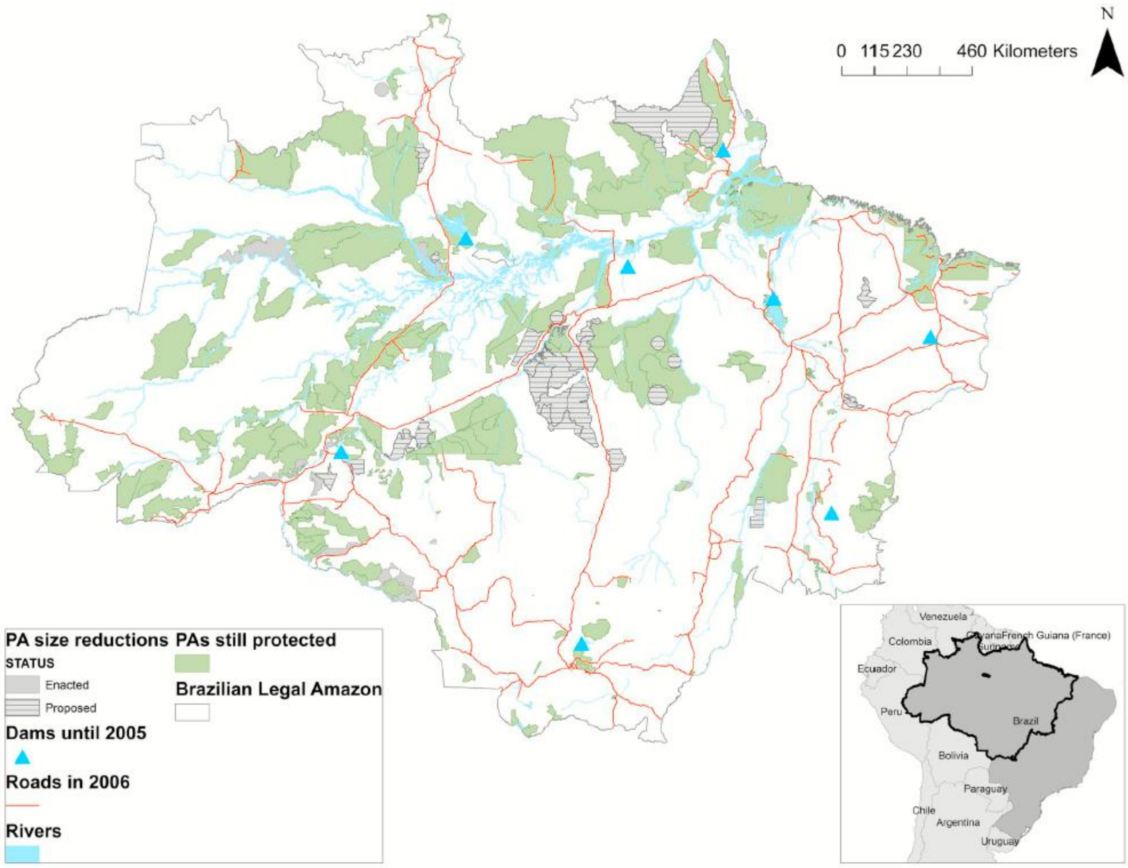

Figure 1-3 Roads, rivers and dams in the Brazilian Amazon

Source: author calculation (Conservation International and World Wildlife Fund, 2017b; IUCN and UNEP-WCMC, 2016)

Table 1-1A 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 (10000 reals)	86,701	328	2000000	158,040	1660	2000000
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	22357	115	0	832
PA Size	3669	0.01	48267	7115	0.5	38870
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 1-1B 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		250m*250m	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 5km*5km	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	1km*1km	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 > 0,1km3 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)	
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; 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, 2017a)	
Administrative boundaries			Vector format from data on Global Adminsitrative Area (GADM 2012)	

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

* $p<0.1$; ** $p<0.05$; *** $p<0.01$

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

[b] IUCN categories are compared to Ctegory 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 1-2B Internal Deforestation Within PAs – Fraction of PA Area

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

* $p<0.1$; ** $p<0.05$; *** $p<0.01$

[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.

Table 1-2C 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 c		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

[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.

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 x 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 :

Probability (R_i=1) = F(βX_i) or, expanding upon the factors X_i 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).

Table 1-3 Risks of PAs size reductions

[Logit [a]]	(1)	(2)	(3)	(4)	(5)
Distance to the nearest road in 2006	0.991 (-2.21)**	0.989 (-2.58)**	0.981 (-2.23)**	0.989 (-2.18)**	0.980 (-2.26)**
(Distance to the nearest road in 2006)^2			1.000 (1.09)		1.000 (1.45)
Distance to the nearest river	0.999 (-0.13)	1.000 (-0.02)	1.001 (0.14)	1.002 (0.23)	1.003 (0.45)
Distance to the nearest dam in 2005	1.000 (0.26)	1.001 (0.50)	1.001 (0.50)	1.002 (0.23)	1.002 (1.04)
Distance to the nearest urban area 2005	0.999 (-0.50)	0.991 (-2.32)**	0.992 (-2.16)**	0.992 (-1.97)**	0.993 (-1.75)*
(Distance to nearest urban area 2005)^2		1.000 (1.98)**	1.000 (1.82)*	1.000 (1.46)	1.000 (1.20)
ln(Average GDP from 2000 to 2005+1)	1.206 (1.05)	1.005 (0.03)	1.006 (0.03)	1.069 (0.38)	1.071 (0.39)
Average population density 2000-2005	1.000 (0.23)	1.000 (0.31)	1.000 (0.28)		
(Average popden 2000-2005) in buffer				0.999 (-0.63)	0.999 (-0.65)
ln(Total deforestation 2000-2005)	1.380 (2.45)**	1.265 (1.96)*	1.256 (1.89)*	1.378 (2.71)***	1.366 (2.59)***
ln(Size of the PA)	1.437 (2.80)***	1.514 (3.42)***	1.527 (3.47)***		
Perimeter-to-area ratio				0.190 (-2.55)**	0.186 (-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 (0.09)	1.109 (0.10)	1.030 (0.03)	1.395 (0.27)	1.200 (0.14)
Amazonas	0.080 (-2.39)**	0.127 (-2.82)***	0.120 (-2.79)***	0.137 (-2.53)***	0.128 (-2.51)**
Maranhão	0.252 (-1.11)	0.451 (-1.01)	0.397 (-1.17)	1.120 (0.09)	0.947 (-0.04)
Mato Grosso	0.025 (-2.76)***	0.050 (-2.65)***	0.046 (-2.69)***	0.063 (-2.50)**	0.056 (-2.58)***
Pará	0.493 (-0.68)	0.630 (-0.67)	0.605 (-0.72)	1.173 (0.22)	1.146 (0.19)
Roraima	0.207 (-1.14)	0.212 (-1.45)	0.213 (-1.44)	0.236 (-1.16)	0.244 (-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

* $p<0.1$; ** $p<0.05$; *** $p<0.01$

[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.

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.

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 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]

[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

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 Figure 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 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

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 Appendix, 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.

[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).

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).

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.

[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).

Table 1-4A Risks of PAs size reductions within the Higher Pressure 'Arc' Region

[Logit [a]]	'Arc of Deforestation'	
Distance to the nearest road in 2006	0.986 (-2.37)**	0.986 (-2.86)***
Distance to the nearest river	0.988 (-2.80)***	0.988 (-8.77)***
Distance to the nearest dam in 2005	1.003 (1.65)*	1.003 (1.94)*
Distance to the nearest urbanized area in 2005	0.990 (-1.87)*	0.989 (-3.46)***
Squared Distance to the nearest urbanized area in 2005	1.000 (1.65)*	1.000 (2.76)***
ln(Average GDP from 2000 to 2005+1)	1.097 (0.73)	1.027 (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 (0.97)	1.126 (1.28)
Average rainfalls from 2000 to 2005	1.000 (-1.00)	0.999 (-1.12)
ln(Total deforestation from 2000 to 2005+1)	1.573 (3.22)***	1.538 (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<0.1; ** p<0.05; *** p<0.01

[a] Results without state dummies and with clustered standard errors given insufficient observations.

Table 1-4B Risks of PAs size reductions for Different Types of Protected Areas

[Logit [a]]	Mixed Use PAs		Strict PAs	
Distance to the nearest road in 2006	0.981 (-4.41)***	0.981 (-5.19)***	0.996 (-2.74)***	0.996 (-3.93)***
Distance to the nearest river	0.991 (-1.40)	0.992 (-1.61)	1.007 (0.92)	1.006 (0.99)
Distance to the nearest dam in 2005	1.001 (0.59)	1.001 (0.68)	0.993 (-2.39)**	0.993 (-2.40)**
Distance to nearest urbanized area 2005	0.992 (-2.32)**	0.991 (-2.30)**	1.001 (0.24)	1.000 (0.08)
(Distance to nearest urbanized area 2005)^2	1.000 (2.18)**	1.000 (2.50)**	1.000 (0.23)	1.000 (0.45)
ln(Average GDP from 2000 to 2005)	1.051 (0.29)	1.033 (0.15)	1.359 (2.44)**	1.277 (4.20)***
Average population density in 2000-2005		0.991 (-2.22)**		1.000 (0.24)
Average population density from 2000 to 2005 in the buffer zone	0.994 (-2.18)**		1.000 (2.30)**	
Average slopes	1.494 (2.87)***	1.463 (3.49)***	1.036 (0.16)	1.008 (0.07)
Average rainfall from 2000 to 2005	1.000 (-0.07)	1.000 (-0.01)	1.001 (0.71)	1.001 (1.00)
ln(Total deforestation from 2000 to 2005+1)	1.557 (4.38)***	1.417 (2.15)**	1.785 (6.28)***	1.852 (5.13)***
Ln(PA size)		1.295 (1.76)*		1.235 (0.93)
Perimeter to area ratio	0.302 (-2.21)**		0.173 (-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<0.1; ** p<0.05; *** p<0.01

[a] Results without state dummies and with clustered standard errors given insufficient observations.

Table 1-4C Risks of PAs 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 population density from 2000 to 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

* $p<0.1$; ** $p<0.05$; *** $p<0.01$

[a] Results without state dummies and with clustered standard errors given insufficient observations.

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. This is consistant with the study of PADDD in Rondonia (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 Rondonia (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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Chapter 2

Does the Selective Erasure of Protected Areas Raise Deforestation in the Brazilian Amazon?

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Abstract (179 words)

Protected areas (PAs) have long been the leading conservation tool for deterring deforestation, yet resistance by land users leads PAs to be established in remote sites, lowering their impacts. After protection is established, ongoing resistance includes deforesting illegally inside a PA or advocating for its downgrading, downsizing, or degazettement ('PADDD') to reduce its status (downgrading) or size (downsizing/degazettement, i.e., partial/full erasure). For the Brazilian Amazon, we estimate impacts of 2009-2012 PA erasures on 2010-2015 post-erasure forest loss. We match to similar controls, on average and for many subsets, in light of significant temporal and spatial variation in the risks of both deforestation and PADDD. For the Brazilian Amazon, erasures of PAs tend to be relatively near to economic pressures, where deforestation is likely. Conceptually, erasures will cause deforestation if erased PAs had faced and blocked pressures. Ours results support that. Erasures that faced no pressure had no impact. Facing pressures, erasure's impact depends on protection's prior impact: if erased PAs had blocked the pressures, erasure raised loss; yet when erased PAs had not blocked pressures, erasures had no impacts.

Keywords:

protected areas, PADDD, forest conservation, Brazil, Amazon, impact evaluation

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1 Introduction

Brazil's Amazon region contains half of the world's tropical rainforest and is a biodiversity hotspot (Campos-Silva et al., 2015; Gallo and Albrecht, 2019). However, Brazil is the 7th-largest GhG-emissions contributor, due mostly to the conversion of Brazilian Amazon rainforest to pasture and agricultural lands (Azevedo-Ramos and Moutinho, 2018). Facing those pressures, protected areas (PAs) have been a leading conservation strategy in the Brazilian Amazon since the 1980s (Campos-Silva et al., 2015; Nogueira et al., 2018; Veríssimo et al., 2011). The national PA system expanded, including with international commitments at the World Parks Congress in Bali, 1992 United Nations conference on environment and development, and Conventions on Biological Diversity in 2004 and 2010 (Visconti et al., 2019). In the 2016 Paris agreements, Brazil committed to lower GhG emissions to 43% below 2005 levels by 2030, mainly through further reductions in deforestation (Gallo and Albrecht, 2019). The PA network now covers over 30% of Brazilian territory (UNEP-WCMC, 2020) and over 50% of the Brazilian Amazon (Campos-Silva et al., 2015) and on average has reduced deforestation (e.g., Pfaff et al., 2015).

Yet in Brazil, as elsewhere, PAs' impacts upon deforestation are constrained by PAs' locations. Outside high-pressure areas such as the 'Arc of Deforestation', few land uses are profitable so forest often remains standing without any protection (Herrera et al., 2019; Pfaff et al., 2015). Thus, the PAs outside of higher-pressure areas often do not reduce deforestation significantly. PAs impacts also are limited by illegal deforestation inside of PAs, due to imperfect enforcement (Amin et al., 2019; Carranza et al., 2014; Jusys, 2018, 2016; Kere et al., 2017; Nolte et al., 2013).

After a suite of conservation policies helped to reduce Amazon deforestation by over 70% from 2004 to 2011 (Azevedo-Ramos and Moutinho, 2018), deforestation then rose from 2012 to 2019 (INPE, 2019). The initial fall in deforestation followed from increases in enforcement and areas protected. The recent rise in deforestation in turn followed decreases in enforcement, even 'a license to deforest', including in PAs (Carvalho et al., 2019; Ferrante and Fearnside, 2019). There are also decreases in areas protected, with protection being legally challenged by permitting additional activities inside PAs, partially erasing PAs to decrease their sizes, or fully erasing some PAs (Mascia and Pailler, 2011). Those events – i.e., PA downgrading, downsizing, and degazettement (collectively known as 'PADDD') – have accommodated industrial-scale resource extraction and development and, to a lesser degree, local land pressures as well as land claims (Golden Kroner et al., 2019; Naughton-Treves and Holland, 2019; Qin et al., 2019).

After little PADDD events in the Brazilian Amazon from 1970 to 2000, its pace rose, especially during 2008-2015 when 44,000 square kilometers of PAs were lost to reductions in PAs' sizes, i.e., the partial or complete erasure of PAs (Campos-Silva et al., 2015; Golden Kroner et al., 2019). This acceleration reflected a lack of political and technical support for conservation objectives within Brazil, including as expressed through the scarcity of funds and human resources allocated for the management of PAs (Bernard et al., 2014; Campos-Silva et al., 2015; Ferreira et al., 2014).

Given that acceleration, and similar actions around the world, we want to better understand PADDD impacts to inform optimal conservation strategies. Conceptually, development agents propose PADDD events for PAs if they would gain from deforesting in those PAs. Given contrasting objectives, conservation agents bargain against PADDD, as a function of gains from those PAs (Keles et al., 2020; Tesfaw et al., 2018). Many outcomes are possible when such agents bargain. Empirically, in a quite limited literature, it has been found that accessibility of a PA to markets, PA size, and the rate of internal deforestation all increase the likelihood, or risk, of a PA being reduced in size (Keles et al., 2020; Pack et al., 2016; Symes et al., 2016; Tesfaw et al., 2018).

Those results for the risks of PADDD – specifically of size reduction (Keles et al. (2020) for the entire Brazilian Amazon) – make it hard to predict whether selective erasures of PAs will raise deforestation. If a PA's size was reduced due to internal deforestation – e.g., profitable internal deforestation has occurred – then a size reduction may have no impact. This is a relevant case, given that prior internal deforestation has increased risks of size reductions (Keles et al., 2020; Tesfaw et al., 2018). If instead the context for a PA size reduction is a remote location with low pressure, again a size reduction may have no impact: in such locations, even without PAs there may be no profits from deforestation. However, the fact that size reductions are more frequent near markets suggests that at least some of the size-reduced PAs did face economic pressures. If a PA has at least somewhat blocked pressures, then a size reduction may raise deforestation.

The few empirical results on PADDD's forest impacts are mixed. Golden Kroner et al. (2016) report that 150 years of legal changes within the Yosemite National Parks in the United States and infrastructure to accommodate rural settlement and resource extraction increased habitat fragmentation. Moreover, higher deforestation rates and greenhouse gas emissions followed from PADDD in Peru and Peninsular Malaysia (Forrest et al., 2015). Yet for Rondônia State in the Brazilian Amazon, Tesfaw et al. (2018) found no average short-term forest impact. That is consistent with size reductions occurring where significant internal deforestation has already occurred (which can result from bargaining if 'failing' PAs are more likely to be allowed to be reduced). Considering the entire Brazilian Amazon, Pack et al. (2016) use difference-in-differences to separate PADDD's influence upon 2002-2011 deforestation from those of fixed other factors. They also find no impact, i.e., erased and still-protected area parts had the same deforestation.

Yet results can differ over both space and time for geographic, economic, and political reasons. Considering remote geographies, investments in infrastructure would not always immediately raise deforestation (Pfaff et al., 2018; Tesfaw et al., 2018) yet over time activities tend to build. Similarly, responses after PADDD could be slow. Another temporal issue is that change in the views of government shift expectations and investments (per recent political shifts see, among others, Carvalho et al., 2019; Escobar, 2019; Ferrante and Fearnside, 2019). As noted above, while after 2004 rates of Brazilian Amazon deforestation fell for some years, due to new policy, they have been rising since 2014. Reconsidering in detail the impacts of size reductions in PAs seems worthwhile, at the very least to allow for spatial and temporal variation in their contexts.

To consider all of these details concerning impacts on deforestation of reductions in PAs sizes, we evaluate the impacts of enacted 2009-12 PA size reductions (downsizing/degazettement) on 2010-2015 post-reduction forest losses. Like Pack et al. (2016), we study all of the Brazilian Amazon. We extend their results with more recent events and recent deforestation. Critically, we also distinguish multiple subsets of PAs across which the impact of reductions might differ.

Our contributions are the following. First, Section 2 offers a conceptual model of where PA size reduction is more likely to raise deforestation. Second, in light of that model, we focus not on average impact, for an enormous and incredibly diverse region, but instead impacts by context. To reduce bias, for each context we employ matching at pixel level over both space and time. We use observed characteristics of forest lands that influence the likelihood of deforestation and the chance of a PA size reduction, as well as pre-PADDD outcomes, all varying by PA subsets, to estimate pre-reduction (2001-2008) forest impacts of all PAs. That is a critical summary of PAs' effectiveness across our contexts which vary in expected deforestation pressures. We proxy for differences in economic pressure using large regions, states, and road distances. Our estimates of 2001-2008 pre-reduction forest impacts confirm that those contexts do differ.

On average for the PA size reductions that occurred in the Brazilian Amazon, we find that PAs selected for erasures in 2009-2012 (i.e., "to-be-reduced" PAs) actually raised 2001-2008 rates of deforestation relative to areas with no protection. That shows a lack of effective protection, which differed from the majority of PAs that remained constant in size. Consistent with our theory concerning partial PA enforcement, PA size reductions also increased the post-erasure 2010-2015 forest-cover loss, i.e., loss was higher in reduced areas versus constant-sized PAs. Overall, PAs reduced in size had internal forest loss then, after reduction, losses went further.

However, for the PAs outside of the 'arc of deforestation', given low pressure unsurprisingly erased PAs neither had impacts during 2001 to 2008 nor raised deforestation due to erasures. Further, even among reduced PAs that had faced clearing pressure, outcomes were not equal and the variations in impacts of erasures were in line with the variations in prior PA impacts. For example, when relatively higher pressures were not blocked by PAs during 2001-2008, after erasure forest loss continued not to be blocked, thus erasure had had little if any impact. In between those two pressure extremes (very low and very high), reduced PAs had impacts before they were erased and thereby, as predicted, their erasures allowed forest losses to rise.

The rest of the manuscript is organized as follows. Section 2 offers relevant background and a simple framework for considering where PA erasures are more likely to increase forest loss. Section 3 presents our empirical strategy, Section 4 provides our estimates for impacts of PAs and PA erasures – including, critically, for distinct subsets of PAs – then Section 5 discusses.

2 Historical Background & Conceptual Model

2.1 Deforestation & Forest Protection in the Brazilian Amazon

Brazilian Amazon deforestation rose during the 1960s as a military dictatorship opened up this forest region (Hargrave and Kis-Katos, 2013; Souza-Rodrigues, 2019). To support settlement and economic activities, roads were built and incentives given, all interacting with insecure land tenure and the consequent land grabbing and illegal logging (Araujo et al., 2009). When the economy stabilized during the 1990s, deforestation was also driven by rising demand for exports, as the country became a major supplier of beef and soybeans (Arima et al., 2014).

Following international interactions and domestic public concerns (Naughton-Treves et al., 2005; Veríssimo et al., 2011), the Brazilian Institute of the Environment and Renewable Natural Resources (IBAMA) was created to enforce environmental laws (Arima et al., 2014). PAs were established (Figure 1) to conserve forest habitat for species and the required share of forest ('legal reserve') on private land rose from 50% to 80% (Arima et al., 2014; Souza-Rodrigues, 2019). Yet enforcement was poor (Naughton-Treves et al., 2005; Veríssimo et al., 2011) and deforestation rose until a peak in 2004, when 26,800 km2 of land were cleared (Figure 2).

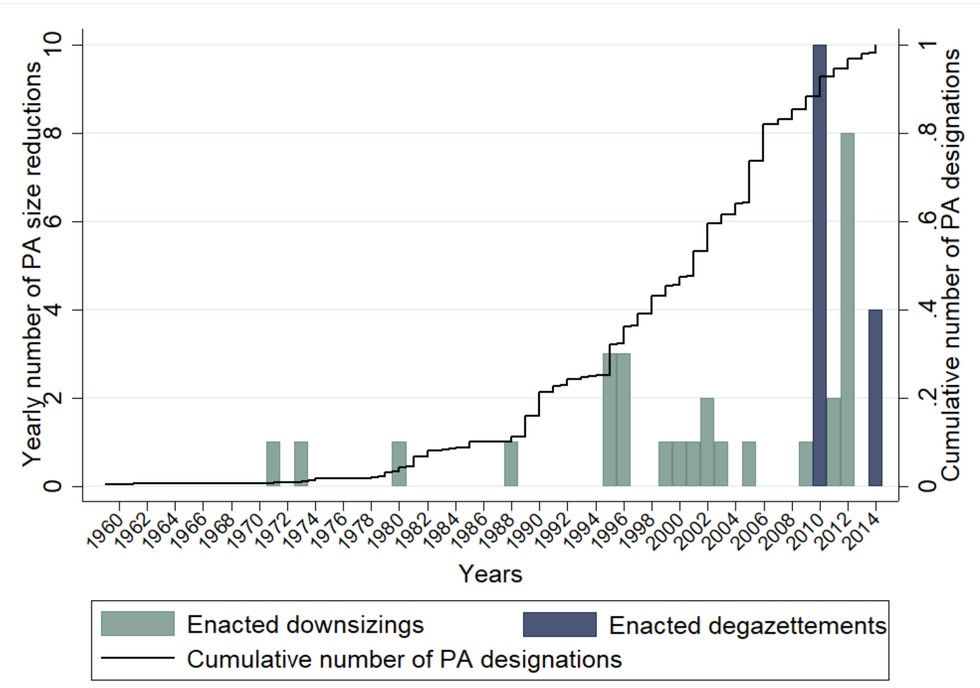

Figure 2-1 Timing of PA designations (we use whether pre-2001) & PA size reductions.
Source: author's calculation (WWF,2017a; IUCN and UNEP-WCMC, 2017 with creation date confirmed in Brazilian data).

Figure 2-2 Deforestation in the Brazilian Amazon
Source: author's calculation from INPE (2019)

Given multiple federal policy responses, as well as the 2008 economic crisis (Arima et al., 2014; Assunção et al., 2015; Soares-Filho et al., 2014), deforestation fell sharply from 2004 to 2011, when it was 5,800 square kilometers. In 2002, for instance, the 'The Amazon PA Program' (ARPA) was initiated in order to extend the PA network as well as to improve PA management. To enforce environmental laws, the Real-Time System for Detection of Deforestation (DETER), using satellites, was implemented by the National Institute for Space Research (INPE) in the 1st phase of the Action Plan to Prevent and Control Deforestation in the Amazon (PPCDAm-I) (Arima et al., 2014; Souza-Rodrigues, 2019; Veríssimo et al., 2011). PPCDAm-II (2009-2011) added federal measures such as: more frequent inspections and applications of sanctions by IBAMA; a list of priority municipalities subject to stricter enforcement (Arima et al., 2014; Assunção et al., 2015; Souza-Rodrigues, 2019); new punishment instruments (embargoes, seizures); and conditioning of rural credit on environmental compliance (Gibbs et al., 2016).

By 2012, however, the trend of falling deforestation had ceased. Then, from 2014, deforestation again started to rise, as a result of political changes which weakened environmental laws (Campos-Silva et al., 2015; Fearnside, 2016; Rochedo et al., 2018). For example, a 2012 revision of the Forest Code provided amnesties to landowners whose legal forest reserves had been cleared before 2008 (Soares-Filho et al., 2014). In addition, environmental requirements were lowered, further investments in infrastructure (highways and dams) were promoted, and the PA network was undermined (Fearnside, 2016; Naughton-Treves and Holland, 2019). PAs had been reduced in size as early as 1970, yet the phenomenon accelerated, mostly to accommodate infrastructure projects, settlements, and expansions of agriculture (Golden Kroner et al., 2019; Mascia and Pailler, 2011; Pack et al., 2016). Specifically, size reductions were enacted within the Brazilian Amazon during recent decades for 40 PAs, covering an area of 157,377 square kilometers, with 25 PA erasures concentrated between 2009 and 2012 covering 42,113 square kilometers (Figure 1). The recent election of and actions by President Jair Bolsonaro also have sent clear signals per the prioritization of economic growth over conservation. Thus, incentives to clear the forest have been increased. Forest fires, which have doubled compared to last year, are in part a consequence of such deforestation (Casarões and Flemes, 2019; Escobar, 2019).

2.2 Impacts of Protection & PA size reductions

2.2.1 Deforestation Baselines & PA Impacts By Location

Frontier deforestation following agricultural expansion is often thought to follow patterns that have been considered since von Thünen (Angelsen, 2010, 2007; Sims, 2014). A risk-neutral agent faces a choice to clear forest on unprotected parcel i and earns rents Y_i that equal p, the price in the nearest market, times parcel yield, which is a function of land quality $f(Q_i)$. From those revenues, one must subtract the transport costs to move outputs to market T_i. Assuming per-unit T_i is linear, costs T_i*d_i rise with the distance of the parcel to the nearest market. Thus, Y_i falls with d_i and, in turn, the forest clearing falls with d_i, ending at the $\bar{d}_i$ for which $Y_i = 0$.

$$Y_i = pf(Q_i) - T_i\, d_i \quad \rightarrow \quad \bar{d}_i = \frac{pf(Q_i)}{T_i} \qquad (1)$$

Within a PA, fines for deforestation F add to all other costs, reducing the rents from agriculture, so $Y_i^{PA} < Y_i$ given some positive chance of being caught illegally deforesting ($\pi_i^{PA} > 0$). Yet transport costs hinder PA enforcement as well. For the Brazilian Amazon, illegal deforestation in PAs is higher far from urban centers (Keles et al., 2020). Thus, we assume that enforcement near cities, where agencies are located, could be sufficient to prevent any illegal deforestation nearer to cities, yet in any case its effectiveness falls as the distance to city rises. We would then expect that PAs might not lower deforestation in two important cases: either beyond $\bar{d}_i$, where already profit is negative without any PAs; or when enforcement was effectively zero, due to transport costs. PAs at low to intermediate distances may discourage deforestation (Figure 3):

$$Y_i^{PA} = pf(Q_i^{PA}) - T_i^{PA}(d_i^{PA}) - \pi_i^{PA}F \quad \rightarrow \quad \bar{d}_i^{PA} = \frac{p\, f(Q_i^{PA}) - \pi_i^{PA}F}{T_i^{PA}} \qquad (2)$$

Proposition 1: PAs reduces forest clearing (i.e., deforestation would occur without any PA but not given a PA and enforcement) near to urban centers if enforcement is strong and a bit further out from cities, where enforcement is weaker yet profit is lower than near cities.

2.2.2 Locations & Impacts of PA size reductions

Development agents will advocate (and when necessary bargain) to downsize or degazette PAs when the profit from clearing forest would be high (Keles et al., 2020; Tesfaw et al., 2018). When a full PA or a part of a PA is erased, i.e., its size is reduced, the fines disappear for clearing those forests, making the rents from agriculture after size reductions Y_i^{RS} = Yi. Of course, this raises deforestation only for the same range of distances for which the PA was saving forest:

close enough to cities that deforestation occur without PAs (below $\bar{d}_i$); and also close enough that enforcement, which falls with city distance, generates expected fines which reduce rents. Those are the conditions under which reducing a PA should raise deforestation (see Figure 3).

Proposition 2: Reducing PAs size raises deforestation if PAs have impact (see Proposition 1).

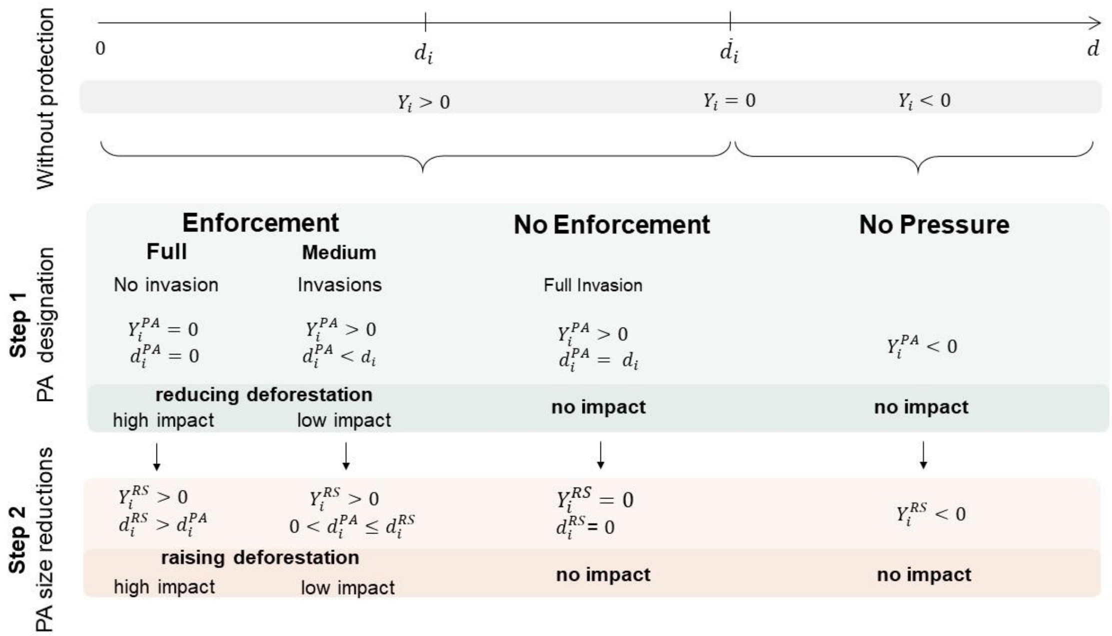

Figure 2-3 Locations & Impacts of PAs Designations and Size Reductions

3 Empirical Approach

3.1 Impact Estimation: matching both before & after erasures of protected areas

We estimate deforestation impacts of PA size reductions. As is summarized above (Figure 3), we do not expect impact above a threshold distance at which rents are not positive. Below that, impacts depend on enforcement. Not knowing all these distance ranges in practice, we simply estimate PAs' forest impacts before size reduction. Only for PAs that had lowered deforestation before they were reduced in size would we predict a rise in deforestation after that reduction. We estimate impacts of PAs before size reductions (step 1) and of PA size reductions (step 2).

If a PA lowered deforestation before its size was reduced, there should be a difference between the observed deforestation with protection and an estimate of the counterfactual deforestation that would have occurred had the PA never been created (Ferraro and Hanauer, 2014; Velly and Dutilly, 2016). Thus, we compare each PA to similar unprotected forest. If a size reduction raised deforestation, there should be a difference between the deforestation after the reduction and an estimate of the counterfactual deforestation that would have occurred had the PA been retained. Thus, we compare with deforestation within appropriately selected protected forests.

As our counterfactuals, we could use observed outcomes for treated parcels when not treated, e.g., a protected forest parcel before its PA was created. Yet deforestation rates vary over time independent of whether any PA was created, due to external 'shocks' (e.g., a fall in agricultural prices (Assunção et al., 2015)). We could instead use average outcomes for untreated parcels to compare with treated parcels. Yet baseline deforestation would have to be the same for those groups for a valid comparison (Ferraro and Hanauer, 2014; Velly and Dutilly, 2016). In contrast to that requirement, within the Brazilian Amazon PAs have tended to be located farther from pressure than unprotected forests (Pfaff et al. 2015). Also, protected forests that lose PA status have been in relatively accessible sites (Keles et al., 2020; Pack et al., 2016; Tesfaw et al., 2018).

Thus, we use matching to find comparison groups that are similar as, for both protection itself and PA size reductions, the treated and untreated parcels are not expected to be fully similar. For each treatment – both protection and a reduction of protection – we will find the untreated forest parcels that are most observably similar to whatever forest parcels were treated and we will use both fixed characteristics relevant for deforestation plus the pre-treatment outcomes. To estimate impacts of protection before reduction (step 1), we use unprotected forests most similar to each protected parcel. To estimate the impacts of PA size reductions (step 2), though, we use the still-protected forest parcels that are most similar to the parcels that lost protection.

3.2 Data

3.2.1 Units of Observation

We randomly drew 1,028,230 pixels from across the entire Brazilian Legal Amazon. To address the possibility of spatial autocorrelation, we enforced a 1km minimum distance between draws (Avelino et al., 2016; Pfaff et al., 2009; Velly and Dutilly, 2016). We also drop some observations due to the possibility of local PA 'leakage' which affects land use nearby. Specifically, we exclude from the controls a 20km buffer zone around each PA (Joppa and Pfaff, 2011; Nolte et al., 2013).

3.2.2 Variables

We use forest loss at a 30x30m resolution from the new version of the Global Forest Change (Hansen et al., 2013). These data indicate tree-cover density (10% to 75%) in 2000, for trees over 5 meters in height, and whether a pixel was cleared each year from 2000 to 2015 (the distribution of forest loss can be seen below in Figure 4, juxtaposed with PAs' size reductions). We apply the label 'forest' if the tree-cover density is at least 30% at the start of any period.[1] It

[1] This threshold corresponds to the definition of tropical forest within the United Nations Framework Convention on Climate Change (UNFCCC): any area of at least 0.5 ha with 10 to 30% tree cover density (Chazdon et al., 2016). This is also the official definition of tropical forest used in the CBD (Convention on Biological Diversity, 2019).

is important to note that the Global Forest Change data do not indicate a difference between natural and secondary planted forests (Sexton et al., 2016; Tropek et al., 2014). Also, Hansen et al. (2013) have not to date applied and provided data from the same methodology for tree-cover gains, making it impossible to compute net forest-cover loss across any given region.

Our deforestation outcome is a binary variable that indicates whether the forest cover has been lost during either of two periods: between 2001 and 2008, for estimating the pre-reduction PA impacts on deforestation; and between 2010 and 2015, for estimating the impacts of the PA size reductions. For the former estimation, we use locations forested in 2001 and forest cover is considered to be lost wherever it has fallen to zero between 2002 and 2008. For the latter estimation, we use locations forested in 2010. Forest cover is seen as lost if, post-reduction, the forest indicator falls to zero during 2011-2015 (PA size reductions are between 2009 and 2012).

We use the PA data from the World Database on Protected Areas (WDPA) (IUCN and UNEP-WCMC, 2016), which is a spatially explicit database that describes the PAs' locations and their boundaries. To avoid misinterpretation, we only use those PAs that could have been recorded in the PADDD data as reduced in size, i.e., all of the 'units of conservation' included within the National System of Protected Areas (Sistema Nacional de Unidades Conservação - SNUC) for Brazil. Thus, we have dropped both Indigenous Lands and Quilombola Territories.

These conservation units are all classified according to the extent of the activities permitted inside. When a location appears in multiple PAs, we assigned it to the strictest classification among those PAs. In our sample, we find 58,944 protected pixel locations before 2000, while 460,256 pixel locations were never protected. In estimating the 2001-2008 impacts of PAs on forest, the treated group of observations is all the locations inside PAs established before 2000, while the pool of potential controls is all of the locations that were never protected by 2008.

For considering PADDD, and more specifically all PA size reductions, we use PADDDtracker.org Data Release Version 1.1 (Conservation International and World Wildlife Fund, 2017). That is a spatially explicit database with descriptions of PA size reductions, their locations, and their changes in boundaries. To study the deforestation impacts of these PA size reductions, for all of the pixel locations protected by 2008 we distinguish all of the PAs that were reduced in size between 2009 and 2012 from the rest of the PAs, which remained protected through 2015 (i.e., constant-sized PAs, see Figure 4). We find 2,653 of the former pixels and 145,170 of the latter.

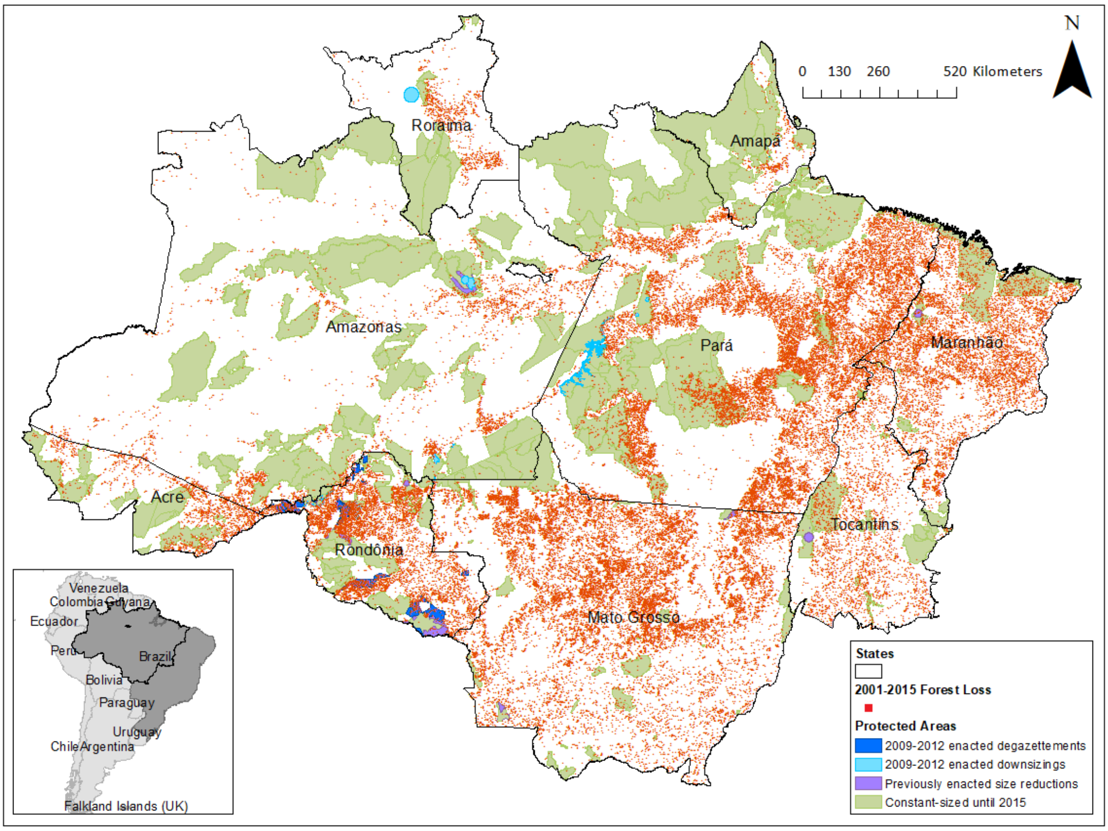

Figure 2-4 PAs, PA size reductions and deforestation in the Brazilian Amazon
Source: author calculation (Conservation International and World Wildlife Fund, 2017b; Hansen et al., 2013; IUCN and UNEP-WCMC, 2016)

As noted above, we would expect that both the initial locations proposed and chosen for the establishment of PAs and the proposal and implementation of PA size reductions may well be explained by bargaining between parties with different interests, in light of rents that can be earned from using cleared forest lands and, thus, are lost with the enforcement of protection. Conservation's opportunity cost is affected by biophysical and socioeconomic characteristics of lands which influence agricultural suitability. We include slope and elevation from the Shuttle Radar Topography Mission (SRTM) (Jarvis et al., 2008). We also obtain 1995-2015 rainfall levels in millimetres per year from version 2.0 of Climate Hazard Group InfraRed Precipitation with Station Data (CHIRPS) (Funk et al., 2015). We also use an indicator of soil quality from the Global Agro-Ecological Zone (FAO and IIASA, 2020), which equals one if the land is suitable for high-input rain fed farming but zero otherwise. Rents of course also depend upon market access. We use the road network as a proxy: in 1996, from the Center for International Earth Science Information Network (CIESIN, 2015); and in 2006 from the Brazilian Departamento Nacional de Infraestrutura de Transportes (DNIT, 2017). In addition, we use the network of navigable rivers, from the Environmental Systems Research Institute (ESRI, 2019a) as well as a set of 'major cities' from the Environmental Systems Research Institute (ESRI, 2019b).

PAs' sizes may influence average enforcement costs (higher for larger sizes). So too may IUCN categories (which affect what is allowed), as collected from WDPA (IUCN and UNEP-WCMC, 2016). We use the number of endemic species in 2006, from WWF WildFinder data on species distributions (Olson et al., 2001; WWF, 2006), as one proxy for environmental values of PAs.

3.3 Methods

3.3.1 Nearest-neighbour matching

Forest-cover loss for treated pixels is compared with the losses for similar untreated pixels. If matching for similarity on key observable covariates eliminates all of the relevant differences, then the latter outcomes differ from the former outcomes solely due to impacts of treatment.

We match treated pixels with the single most similar untreated pixels, defining similarity using the shortest Mahalanobis distance for covariates expected to influence treatment probability and forest loss (Caliendo and Kopeinig, 2008; Ferraro and Hanauer, 2014; Velly and Dutilly, 2016) such as biophysical and socioeconomic characteristics we believe affect the opportunity cost of conservation as well as the forest outcome (Andam et al., 2008; Carranza et al., 2014; Nolte et al., 2013). To avoid endogeneity, those covariates are fixed or measured pre-treatment. For estimating PAs' impacts, we use slope, elevation, rainfall in 1995 and soil quality as well as access to markets measured by distances to 1996 roads, rivers, and major cities. For estimating the impacts of PA size reductions, we can also use PAs' characteristics. For both size-reduced and constant-sized PAs, the size of a PA before erasure, its IUCN category before erasure, and its number of endemic species are proxies for enforcement costs and environmental values.

Mahalanobis distances take into account interactions among covariates in selecting controls (Stuart, 2010) and yield better balance in our sample. To lower variance, we can match treated observations to multiple similar untreated observations, though this raises dissimilarities and, thus, bias (Caliendo and Kopeinig, 2008). Given the trade-off, we did matching several times with different parameters to maximize matched observations while minimizing standardized bias[2] (King et al., 2011). For impacts both of protection and of reduction, we use the two most similar untreated observations and drop treated observations for which no matches are found in a caliper of 1 standard deviation of each matching covariate (Caliendo and Kopeinig, 2008).

3.3.2 Match Quality

Concerning whether similarity was achieved, we use standardized mean differences as well as tests of distributions to assess differences between the treated and controls. After matching, no significant differences in means should remain between treated and matched untreated controls. Any gains from the matching depend on these differences being significantly reduced (Caliendo and Kopeinig, 2008; Ferraro and Hanauer, 2014; Velly and Dutilly, 2016).

Hidden biases may remain, however, if important confounders – that influence treatment and outcome – are unobserved. In light of this possibility, our standard errors for the Mahalanobis distance matching use the variance approximation developed by Abadie and Imbens (2006).

[2] We try different parameters – e.g., 1 to 3 neighbors – dropping observations for which matches were not found for calipers of 0.10, 0.25, and 0.5 covariate standard deviations. We report the number of matched observations as well as the mean standardized bias. We tried to maximize common support and minimize standardized bias.

As matching on Mahalanobis distances can face challenges with many confounding variables, and if variables are discontinuous, for robustness we also used propensity-score matching, which assesses similarity using treatment probabilities based on probit models using the same covariates (Caliendo and Kopeinig, 2008; Ferraro and Hanauer, 2014; Velly and Dutilly, 2016). We use the nearest neighbour and a caliper of 0.01 standard deviation of estimated propensity scores, without replacement. We follow Rosenbaum (2002) to check how sensitive the latter results are to remaining hidden biases, if they exist (the test does not prove whether they exist).

3.3.3 Adding Temporal Analyses

While we do not have pre-treatment outcomes before protection, we do observe all of the PAs before any size reductions. Thus, we can make use of their outcomes while still protected as a way to control for differences across PAs in testing for impacts of size reductions based on shifts in forest loss over time. We are starting to make use of this temporal perspective in two ways.

First, we can now visually inspect deforestation trends for each of the comparison subgroups. When testing the effects of PA size reductions, that means comparing PAs that will be reduced with PAs that remain constant in size – specifically, during the years that all are still protected. That reveals significant selection for size reduction, on average, i.e., that the to-be-reduced PAs had significantly higher deforestation, while still protected, than did the constant-sized PAs. This motivates our matching of both relevant characteristics and the pre-treatment outcomes. Second, as a form of differences in differences, we run panel models to test reductions' impacts. To date, these figures and specifications have confirmed our low and highest pressures results.

3.4 PA Subsets: varying in expected impacts

PAs facing different expected economic pressures are defined using distance to the nearest road for each pixel. Roads are important factors in market access, since they lower transport costs. They strongly affect deforestation rates (Angelsen and Kaimowitz, 1999; Barber et al., 2014; Cropper et al., 2001; Laurance et al., 2001; Pfaff, 1999), especially if prior development is not yet very high (Pfaff et al., 2018). Thus, often roads are used to distinguish between lands that have higher versus lower opportunity costs of conservation (Jusys, 2018; Pfaff et al., 2015, 2014). We also use regions and states, which feature distinct economic and political contexts.[3]

To divide subsets, we calculate the lowest Euclidian distance to a road for each deforested pixel then sum deforestation within each 1km from the nearest road to get a 'distance-accumulated deforestation' curve (Figure 5) (Barber et al., 2014; Jusys, 2018). We define below 46km as the 'more accessible' locations, as half of deforestation occurs in that range. To more continuously represent Figure 3's cases, we created smaller road-distance subsets using ranges containing

[3] The regions are 'outside the arc of deforestation' (Amazonas, Roraima, Amapá), with low economic pressures, versus 'inside the arc of deforestation' (namely Rondônia and Pará), where economic pressures are much higher. While there are other Amazonian states both inside and outside the arc, they have not featured in size reductions.

20% of total deforestation: 0-10km; 10-30km; 30-68km; 68-143km; and above 144km. Yet as no size reductions were observed within the fifth subset, we merged that with the fourth one.

This methodology was implemented using forest-cover losses during 2001 to 2008, from the Global Forest Change database (Hansen et al., 2013). We then use the same subsets to predict impacts of 2009-2012 PA size reductions using 2001-2008 pre-reduction impacts of the PAs.[4]

4 Results

4.1 Descriptive Statistics

Table 2 frames our analyses by presenting regressions that link all our observed covariates to deforestation for the periods we consider. Measured in 1996, for its influence on 2001-2008 deforestation, or 2006, for 2010-2015 deforestation, road distance discourages economic activities, reducing pressures and clearing. The same relationships hold for distance to cities, for slope, and for rainfall that is excessive (i.e., more rainfall than aids agricultural production).

Within PAs, as in Table 2's three right columns, we can also examine influences of PA characteristics. Interestingly, having additional endemic species seems to be associated with lower levels of deforestation, perhaps suggesting additional environmental enforcement effort. Also concerning enforcement, a stricter IUCN category is associated with lower deforestation.

For either outside or inside PAs, deforestation rates clearly vary a great deal among the states. States are distinct political regulatory regimes and feature quite distinct economic contexts. For instance, they vary in average distances to important national markets and in their access to markets via roads or via major rivers. States' effects are strong and consistent in Table 2.

4.1.1 Protection Subsets vs. Unprotected (in terms of 2001-2008 deforestation)

Among all locations that were protected, ~ 93% of the pixels were within PAs that remained constant in size until at least 2015, while ~ 7% were reduced in size between 2009 and 2012. Only 0.4% of tree cover was lost during 2001-2008, on average, inside constant-sized PAs. Tree-cover loss was far higher (8%), though, within the to-be-reduced PAs. Indeed, forest-loss rates in the PAs later selected for size reductions were above unprotected forests (Table 3A).

As to why that might be the case, the constant-sized PAs are, on average, farther from roads than are the never-protected lands (as in Joppa and Pfaff (2009)). Quite the opposite is true of to-be-reduced PAs (and p-values confirm that these differences are significant at the 1% level). Thus, some to-be-reduced PAs faced more pressures than unprotected due to urban proximity (Keles et al., 2020; Pack et al., 2016; Tesfaw et al., 2018). Matching approaches help to address such observable differences (Table 3A) between PA groups and unprotected lands. This helps to avoid spurious inferences that link PA groups (constant and to-be-reduced) to deforestation.

[4] Road networks evolve. For 2009-2012 erasures, we tried 2006 roads (DNIT, 2017) that differ little from 1996 as these are formal roads (surely the informal keep rising). The threshold obtained is approximately the same.

4.1.2 Constant Protection vs. Reduced Protection (in terms of 2010-2015 deforestation)

Table 3B considers only locations protected before 2009. Among them, only 5.6% are reduced in size between 2009 and 2012 and 2010-2015 tree-cover loss was lower in constant-sized PAs (0.3%) than in reduced PAs (5%). Size reductions were located where tree-cover losses were more likely, e.g., nearer 2006 roads and cities. While reduced PAs have more endemic species, and stricter protection, they are also smaller. Thus, more of their area is vulnerable to invasions. Differences must be controlled for to avoid spurious inferences about the effects of PA size reductions.

4.2 Improving Balances

Tables 4A and 4B convey that the Mahalanobis and propensity-score matching significantly improved covariate balances. For testing PAs' pre-reduction impacts (Table 4A) or impacts of size reductions (Table 4B), remaining differences are less significant. This helps to reduce bias.

Given concerns about unobserved covariates that might not be balanced, we test the sensitivity of our propensity-score estimates to unobserved confounders that could influence protection and deforestation (Becker and Caliendo, 2007). The test from Rosenbaum (2002) states what value of unobserved confounders would raise the odds of both protection and deforestation by a factor of τ. If the average treatment effects are still significant for a τ increase, it indicates some insensitivity to such biases. We show that most of the average treatment effects for 2001-2008 for constant-sized PAs (Table 5A) and to-be reduced PAs (Table 5B), as well as most of the treatment effects for size reductions (Table 6), would remain significant with large increases in unobserved confounders. However, some would have to be interpreted with greater caution.

4.3 Pre-Reduction Impacts of Protection

Tables 5A and 5B reveal PA impacts on 2001-2008 deforestation, for all PAs and for PA subsets. Recall from Figure 3 that, e.g., PAs in areas with low economic pressure should have no impact pre-reduction or, correspondingly, of a size reduction. The same predicted lack of impact from a size reduction holds if – under pressure – a PA is fully deforested due to low enforcement. Yet if a PA faced and blocked pressure, reducing its size should indeed increase deforestation. An intermediate case is partial enforcement, with pre-reduction partial internal deforestation lowering both pre-reduction impact and the increase in deforestation after PA size is reduced.

4.3.1 2001-2008 Impacts of Constant-Sized PAs

We consider first constant-sized PAs, i.e., the PAs that remain untouched by any legal change through at least 2015. We essentially confirm prior results (Pfaff et al., 2015) about landscapes. Table 5A confirms average impact for 2001-2008, as well as significant variation in impacts, across these many Brazilian Amazon PAs. They lowered deforestation (versus baseline). While clearly the Mahalanobis-distance matching better reduces residual biases after matching, the pattern of impacts results is robust across specifications (and to unobservable selection bias).

Variation by subset is important (Figure 6), including to interpret impacts of size reductions. Average impact blends higher impacts closer to roads with lower impacts farther from roads, as expected. Further, impacts approach zero for the farthest locations – even for an enormous region, 'outside the arc of deforestation'. Thus, we see variation that includes the low-pressure region in Figure 3 and pressures in other regions that constant-sized PAs block to some extent.

4.3.2 2001-2008 Impacts of To-Be-Reduced PAs

Table 5B and Figure 6 show to-be-reduced PAs did not block pressures as did constant-sized PAs. In fact, it appears that PAs selected for at least partial size reduction during 2009-2012 actually increased 2001-2008 pre-reduction loss of forest cover relative to loss in unprotected forests. This indicates a lack of effective enforcement and may be part of the selection for PA size reduction (Tesfaw et al.(2018) and Keles et al. (2020) find deforestation raises the risk of size reduction).

Figure 6 suggests this is more pronounced for intermediate road distances, where enforcement is not strong enough to overcome profits (Figure 3's medium-pressure-and-enforcement case). We also see no difference between to-be-reduced PAs and unprotected below 10km from roads, which could indicate that not blocking high pressure yields PAs that function like unprotected.

The only to-be-reduced PAs that lower deforestation, on average, are those in Pará, perhaps due to local political economy. Pará was soon to initiate programs to manage deforestation (Sills et al., 2020) – including with distinct state-supply-chain interactions (Gibbs et al., 2019).

4.4 Impacts of PA Erasures

Table 6 provides estimated impacts of size reductions on 2010-2015 deforestation. On average, relative to similar constant-sized PAs, size reductions increased deforestation. Next to Table 5B, these results suggest that we are observing Figure 3's case of medium enforcement, since: those PAs that experienced internal deforestation were reduced in size; while that was part of a cycle in which, after PAs are reduced in size, absent enforcement deforestation went further.

Figure 3's medium-pressure-and-enforcement case is suggested by rising deforestation as we look across Table 6. The highest deforestation impacts of PA size reductions are for the intermediate road distances, where transport costs can reduce protection (Keles et al., 2020; Sims, 2014).

We also highlight the variation in impacts from PA size reductions, following prior PA impacts (Table 5B). Pará has no impact of size reduction. That fits the story of local political economy, where implementation of programs to regulate land-clearing activities (Gibbs et al., 2019; Sills et al., 2019) could explain the absence of impacts of PA size reductions. We also do not observe an impact of PA size reductions within 10km of roads, as these PAs seem to have been invaded. Figure 3's no-enforcement-no-impact case arises with pressure and low political will for PAs, as confirmed when looking over time at areas that were not enforced before or after PADDD.

Finally, as in Table 5A and in Table 5B, we see that Figure 3's low-pressure case clearly exists. With low pressures, neither constant-sized PAs nor to-be-reduced PAs faced pressures to block. For those conditions, we expected little impact from size reductions at least in the short run. That hypothesis is supported by Table 6's results for Amazonas, Roraima, and whole non-Arc. The over-time figures and panel models certainly also confirm no impact given low pressures.

5 Discussion

Global commitments by the Brazilian government on biodiversity and the reduction of GhG emissions already were difficult to reach (Campos-Silva et al., 2015; Gallo and Albrecht, 2019; Visconti et al., 2019), despite past investments in PAs which on average reduced deforestation (Amin et al., 2019). The task has grown harder with recent policy including implicit permission for economic activities to impinge on PAs, e.g., downgrading, downsizing and degazettement (Naughton-Treves and Holland, 2019; Qin et al., 2019). Few have studied PADDD, in particular controlling for where and when PAs are reduced in size, which affects impacts from PADDD. We extend understanding of PA size reductions by looking further across time and considering heterogeneous impacts on deforestation across space due to size reductions of Amazon PAs.

In evaluating the impact of 2009-2012 size reductions (downsizings and degazettements) upon forest-cover loss in the Brazilian Amazon during 2010-2015, on average and for key PA subsets, we go beyond Pack et al. (2016) and Tesfaw et al. (2018). Conceptually, we lay out why size reductions should increase forest losses by more or by less. Empirically, we evaluate impacts of PA size reductions by economic context. To reduce biases, we employ pixel-level matching.

We also identify which PAs were already blocking such economic pressures before reductions (noting prior internal deforestation raises the chances of PA size reductions (Pack et al., 2016; Tesfaw et al., 2018). Thus, we assess whether to-be-reduced PAs which had internal losses still had managed to have some prior impacts. If so, we expected size reductions to raise forest loss, while for PAs that had not previously lowered deforestation, we did not expect erasure impacts.

We find that at intermediate distance from roads, some PAs experience internal deforestation, get reduced in size as an initial consequence, and then, absent enforcement, lose more forest. Yet we also see size reductions in both low-pressure and high-pressure contexts in which we find neither prior PA impacts on deforestation nor impacts from size reductions of the PAs.

While our main conclusions are robust to the sensitivity tests we included within this paper, potential limitations must be mentioned. For our 2001-2008 time period, we do not break out policy shifts after 2004 (Arima et al., 2014; Souza-Rodrigues, 2019; Veríssimo et al., 2011). Thus, within that period, results for 2004 to 2008 could differ from those for 2001 to 2004 (although we use these prior results simply as indicators of pressures both faced and blocked).

Our results could depend on our outcome variable. We define 'forest' using a 30% value in the remotely sensed data. That corresponds to many definitions of tropical forests, yet it can be contested. Also, the data we use do not distinguish natural and secondary planted forests

(Chazdon et al., 2016; Convention on Biological Diversity, 2019; Hansen et al., 2013; Sexton et al., 2016; Tropek et al., 2014) or consider any annual tree-cover gains (Hansen et al., 2013).

Our results are relatively short term, though we extend periods studied previously (Pack et al., 2016; Tesfaw et al., 2018). Our results also do not consider spatial spillovers from PA erasures that extend beyond PA borders, e.g., by signaling a priority on development (Herrera et al. 2018). Depending on the cause of reductions (e.g., rural settlement, infrastructure, extraction), new infrastructure such as roads may have longer-run and spatially broader impacts upon economic activities (Tesfaw et al., 2018). Further research would be needed to examine this.

We also highlight that, since 2015, further PA size reductions have been proposed and enacted, consistent with a relaxation of environmental policies in the Amazon since 2012 (Carvalho et al., 2019; Ferrante and Fearnside, 2019; Rochedo et al., 2018). This indicates local perceptions of conservation costs. Thus, calls to extend the PA network and to effectively manage the PAs to limit environmental degradation now seem less likely to be answered (Kroner et al., 2019).

Understanding the spatial variation in risks and impacts of PA size reductions should inform efforts to invest in PA siting, enforcement, management, as well as efforts to guide any size reductions to where they would do less damage. These points hold globally, noting PADDD and other regulatory rollbacks are occurring broadly of late, in part with some connections to economic pressures from shutdowns due to the pandemic (Conservation International, 2020).

If our theory and all empirical results are a guide, size reductions are more likely to be proposed in area of high pressure (Keles et al., 2020; Kroner et al., 2019; Qin et al., 2019; Symes et al., 2016; Tesfaw et al., 2018). Yet enforced protection in these locations can have big impacts, even if some internal deforestation is inevitable. This suggests continued enforcement near cities, facilitated (Sims, 2014) by the same lower transport costs that drive higher pressures.

Tradeoffs may differ in rural areas – including where dams are topographically very sensible yet also may lead to PADDD (Thieme et al., 2020) as energy production conflicts with forest conservation. As the expansion of human activities over the Brazilian Amazon (Escobar, 2019) suggests forest losses from such PADDD events, even if new dams are deemed essential to meet energy demand (here without GhG emissions) clearly it could be possible to plan to make up for lost protection through similar protection elsewhere, in a well-managed network (Fuller et al 2010), or via other types of conservation (Golden Kroner et al., 2019; Qin et al., 2019).

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7 Appendix

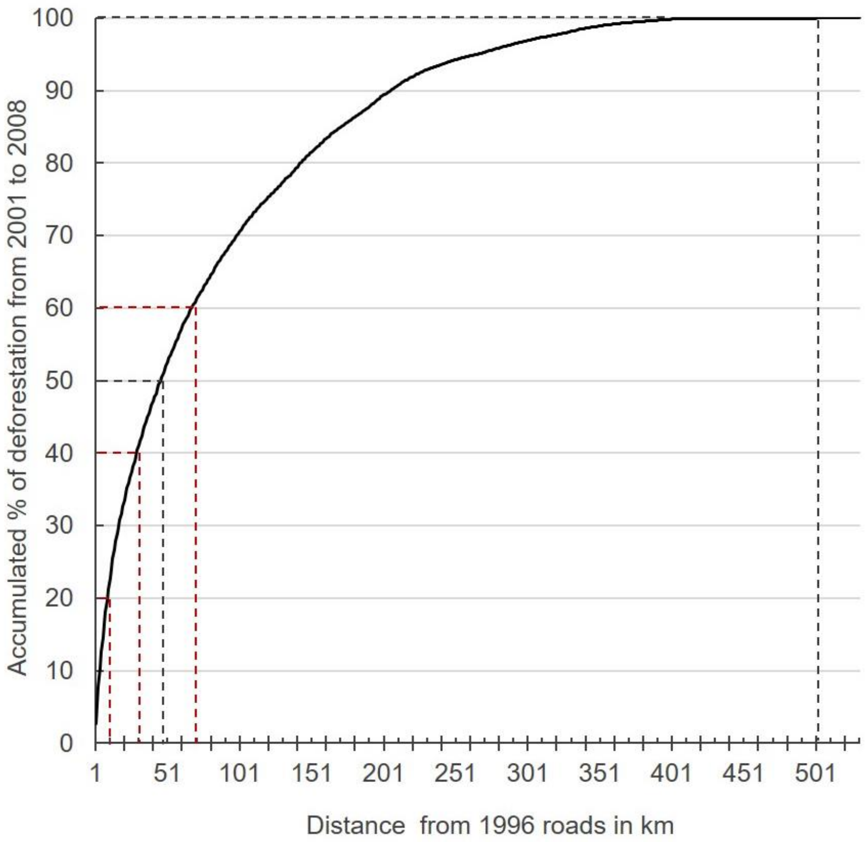

Figure 2-5 Accumulated Deforestation by Road Distance

In our first road-distance subset, 50% accumulated deforestation occur until 46km from roads. In our second type of road-distance subsets, 20% accumulated deforestation occur until 10km from roads, points here are highly accessible, then, the accessibility decrease: 20% additional deforestation occur until 30km from roads and 20% additional deforestation occur until 68km from roads. Above that threshold, points are considered to be highly inaccessible.

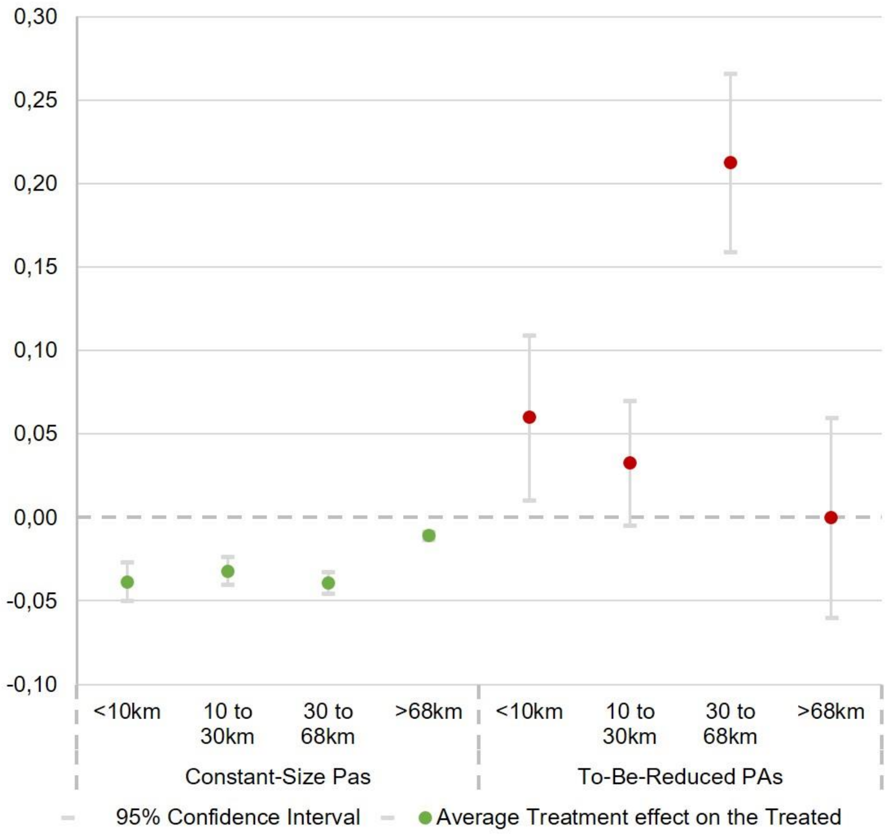

Figure 2-6 Pre-Erasure Protection Impacts according to distance to nearest road

Table 2-1 Source and description of covariates

	Source	Step 1	Step 2
Agricultural GDP	Vector format from the IBGE at the level of the municipality in current prices (1000 real) (IBGE, 2017)	Not used	Average from 2001 to 2008
Elevation	Gridded elevation data from the Shuttle Radar Topography Mission (SRTM) (Jarvis et al., 2008). 90m resolution resampled in 250 by 250 meters.	-	
Slopes	Gridded elevation data from the Shuttle Radar Topography Mission (SRTM) (Jarvis et al., 2008). 90m resolution resampled in 250 by 250 meters.	Degree from the horizontal (ArcGIS).	
Distance to the nearest river	Lake, pond and rivers, permanent and navigable. Vector format from the IBGE (IBGE, 2017).	Nearest Distance (km, ArcGis).	
Distance to the nearest road	Vector format from the Center for International Earth Science Information Network (CIESIN, 2015) and from the Brazilian Departamento Nacional de Infraestrutura de Transportes (DNIT, 2017).	Nearest distance from each obs. unit in km with ArcGis in 1996.	Nearest distance from each obs. unit in km with ArcGis In 2006.
Distance to the nearest city	Vector format from the Environmental Systems Research Institute (ESRI) (ESRI, 2019b)	Nearest distance from each obs. unit in km with ArcGis.	
Rainfalls	Gridded annual data from the version 2.0 of Climate Hazard Group InfraRed Precipitation with Station Data (CHIRPS) (Funk et al., 2015). 0,05 degrees of resolution.	mm. per year in 1995	Average from 2001 to 2008 in mm. per year
Soil suitability	Gridded data from the Global Agro-Ecological (FAO and IIASA, 2020) Rainfed soil suitability, high input farming 0.08 degree of resolution	-	
Number Of endemic species	Vector format from the WWF WildFinder database of species distributions (Olson et al., 2001; WWF, 2006). High endemism: from 21 to 47 endemic species; medium endemism: from 6 to 20 endemic species; low endemism: from 1 to 5 endemic species; no endemism (0 endemic species) is the baseline.	Not used	2006
PA size	WDPA (IUCN and UNEP-WCMC, 2016) PADDDtracker (Conservation International and World Wildlife Fund, 2017)	Not used	-
IUCN category	WDPA (IUCN and UNEP-WCMC, 2016), PADDDtracker (Conservation International and World Wildlife Fund, 2017): II: National Parks; V: Protected Landscape; IV: Habitat/Species Management Area; Ia (Strict Nature Reserve) is the baseline.	Not used	-

Table 2-2 Regressions for Tree-Cover Loss

PROBIT	Unprotected Forest Loss 2001-2008[1]			Protected Forest Loss 2010-2015[2]		
Road Distance in 1996	-0.058 (13.82)***	-0.015 (3.37)***	-0.035 (8.65)***			
Road Distance in 2006				-0.737 (23.28)***	-0.270 (8.08)***	-0.094 (2.79)***
City Distance	0.007 (3.87)***	-0.007 (3.15)***	-0.004 (1.73)*	-0.000 (0.04)	-0.026 (1.87)*	0.026 (1.85)*
River Distance	0.047 (8.26)***	0.114 (18.34)***		-0.146 (3.14)***	-0.021 (0.40)	
Slope	-0.024 (17.91)***	-0.023 (15.81)***	-0.025 (17.13)***	-0.008 (0.92)	-0.002 (0.23)	-0.002 (0.21)
Elevation	0.054 (22.64)***	-0.051 (16.69)***	-0.028 (10.27)***	-0.048 (2.19)**	-0.107 (3.94)***	-0.111 (4.37)***
Land suitable	-0.035 (5.50)***	0.017 (2.59)***	0.027 (4.02)**	-0.116 (3.01)***	0.037 (0.90)	0.037 (0.91)
Rainfall in 1996	-0.055 (78.15)***	-0.031 (34.15)***	-0.031 (34.63)***			
Rainfall 2001-2008				-0.920 (15.70)***	-0.920 (15.70)***	-0.611 (8.34)***
High # of Endemics				-0.171 (4.54)***	-0.377 (7.41)***	-0.377 (7.42)***
Strict IUCN Category				-0.508 (11.91)***	-0.508 (10.91)***	-0.599 (12.15)***
PA Size				-0.000 (0.57)	-0.000 (1.21)	-0.000 (1.23)
Amapa		-0.252 (7.85)***	-0.280 (8.74)***		-0.044 (0.25)	-0.031 (0.18)
Amazonas		-0.524 (28.91)***	-0.565 (31.48)***		-0.501 (5.02)***	-0.489 (5.13)***
Roraima		-0.162 (7.09)***	-0.213 (9.40)***		-0.750 (3.37)***	-0.736 (3.35)***
Para		0.428 (26.55)***	0.376 (23.73)***		0.002 (0.01)	0.018 (0.19)
Rondonia		0.655 (26.55)***	0.613 (35.81)***		0.573 (6.80)***	0.584 (7.35)***
Acre		0.206 (9.12)***	0.203 (8.97)***		0.236 (2.66)***	0.228 (2.56)**
Mato Grosso		0.652 (40.77)***	0.598 (38.09)***		0.083 (1.54)	-0.019 (0.15)
Tocantins		0.163 (7.92)***	0.083 (4.14)***		0.609 (5.87)***	0.627 (6.72)***
Maranhao		0.202 (10.59)***	0.794 (10.12)***		0.586 (2.48)**	0.591 (2.52)***
Constant	-0.575 (33.19)***	-1.159 (35.82)***	-1.093 (43.77)***	-0.370 (3.01)***	-1.075 (6.24)***	-2.088 (6.43)***
Pseudo R2	0.05	0.11	0.11	0.11	0.16	0.16
Observations	772,381	772,341	772,341	150,000	149,967	149,967

[1] Looking only at unprotected points, for the controls to compare to 2001-2008 protection.

[2] Looking only at protected points, for the controls to compare to 2009-20012 PA size reductions.

* p<0.1; ** p<0.05, *** p<0.01

Table 2-3A Descriptive Statistics, 1st Time Period: characteristics & 2001-2008 Deforestation

	Unprotected			Constant-Sized PAs					PAs Reduced In Size 2009-2012				
	Mean	Min.	Max.	Mean	DiM*	Combined KS*	Min.	Max.	Mean	DiM**	Combined KS**	Min.	Max.
Road Distance	103	0	521	164	0.00	0.00	0	518	38	0.00	0.00	0	129
City Distance	428	3	1111	370	0.00	0.00	5	926	302	0.00	0.00	27	587
River Distance	58	0	297	57	0.00	0.00	0	269	50	0.00	0.00	0	112
Slope	1.6	0	41	1.9	0.00	0.00	0	53	1.4	0.00	0.07	0	18
Elevation	211	0	1722	169	0.00	0.00	0	2410	194	0.00	0.00	25	498
Land Suitable	48.9	0	1	51.35%	0.00	0.00	0	1	75.8%	0.00	0.00	0	1
Rainfall	2185	819	4403	2369	0.00	0.00	1053	3532	1869	0.00	0.00	1493	2680
Forest Loss 2001-2008	6.2%	0	1	0.6%	0.00	0.00	0	1	17.9%	0.00	0.00	0	1

*P-values for differences in means and differences in distributions between Unprotected and Constant-sized.
**P-value for differences in means and differences in distributions between Unprotected and Reduced in Size.

Table 2-3B Descriptive Statistics, 2nd Time Period: characteristics & 2009-2012 Deforestation

	Constant-Sized PAs			PAs Reduced In Size 2009-2012				
	Mean	Min.	Max.	Mean	p*	Combined KS**	Min.	Max.
Road Distance	106	0	421	30	0.00	0.00	.02	96
City Distance	359	5	926	304	0.00	0.00	27	587
River Distance	48	0	303	47	0.45	0.00	0	112
Slope	1.9	0	53	1.4	0.00	0.00	0	18
Elevation	177	0	2410	192	0.00	0.00	25	498
Land Suitable	58.26%	0	1	76.85%	0.00	0.00	0	1
Rainfall	204	85	352	155	0.00	0.00	115	226
Forest Loss 2010-2015	0.24%	0	1	5.6%	0.00	0.00	0	1
High Endemic	26%	0	1	62%	0.00	0.00	0	1
Strict IUCN	36%	0	1	53%	0.00	0.00	0	1
PA Size	17,303	4.82	46428	7,685	0.00	0.00	0.09	26,647

*P-values for differences in means and differences in distributions between Unprotected and constant-size.
**P-value for differences in means and differences in distributions between Unprotected and Reduced in Size

Table 2-4A Improving Covariate Balances for PAs' Impacts on 2001-2008 Deforestation

	Unmatched	Mahalanobis Distance Matching			Propensity Score Matching		
	% bias	DiM*	% bias**	% reduced***	DiM	% bias	% reduced
PAs Erased During 2009-2012 versus Never Protected							
Land suitability	17.2	1.00	0.0	100.0	0.86	0.4	97.4
Distance to the nearest city	-56.2	0.55	-1.5	97.4	0.02	5.2	90.7
Distance to the nearest road	-79	0.13	-1.9	97.6	0.00	6.8	91.4
Distance to the nearest river	-12.8	0.96	-0.1	99.2	0.02	-6.3	50.2
Slope	-9.2	0.93	0.2	97.9	0.46	-1.8	79.8
Elevation	-7.8	0.90	0.2	97.0	0.42	-2.1	72.3
Rainfall	-50.7	0.08	-2.6	96.5	0.00	-11.2	85.0
Always Protected Through 2015 versus Never Protected							
Land suitability	1.8	1.00	0.0	100.0	0.31	-0.6	68.0
Distance to the nearest city	-24.9	0.32	-0.6	97.8	0.00	-8.8	64.8
Distance to the nearest road	40.1	0.36	0.6	98.5	0.00	-7.2	82.0
Distance to the nearest river	-4.0	0.01	1.5	62.7	0.00	-5.5	-36.8
Slope	10.6	0.24	0.5	95.1	0.00	-1.8	82.6
Elevation	-24.9	0.03	1.0	95.8	0.00	-2.3	90.8
Rainfall	16.9	0.35	-0.5	97.4	0.27	0.6	96.9

*P-value of simple difference in mean between treated and untreated matched samples.
Standardized bias is the difference between the treated and matched untreated covariate means, as a % of the square root of the average of treated and matched untreated sample variances (Rosenbaum and Rubin, 1985).
***The reduction in standardized bias is the difference between the standardized bias before and after matching.

Table 2-4B Improving Covariate Balances for PA size reductions Impacts on 2010-2015 Deforestation

	Unmatched	Mahalanobis Distance Matching			Propensity Score Matching		
	% bias	DiM*	% bias**	% reduced***	DiM	% bias	% reduced
Erased in 2009/2012 vs Constant-Size							
Land suitability	30.9	1.00	0.0	100	0.61	1.3	95.8
Distance to the nearest city	-30.6	0.08	5.9	80.7	0.85	0.5	98.2
Distance to the nearest road	-118.9	0.02	2.9	97.6	0.01	-3.7	96.9
Distance to the nearest river	6.3	0.01	6.2	4.3	0.26	2.7	56.8
Slope	-21.5	0.29	1.9	91.2	0.16	-3.6	83.2
Elevation	13.9	0.29	2.2	84.6	0.00	-11.4	18.8
Rainfall	-128.4	0.43	2.7	97.9	0.00	8.6	93.3
PA size	-85.8	0.52	-1.9	97.8	0.98	-0.1	99.9
Endemic species	76.6	1.00	0.0	100	0.89	-0.3	99.6
IUCN category	33.1	1.00	0.0	100	0.12	-4.8	85.3

Table 2-5A 2001-2008 Impacts of PAs Constant-sized through 2015

	All	<48 km	>48 km	<10 km	10-30 km	30-68 km	>68 km	In Arc	Not Arc	Rondônia	Pará	Roraima	Amazonas
Total	465,664	181,884	245,241	71,012	72,369	82,996	239,287	123,900	189,144	13,878	102,112	25,088	164,058
Treated	51,671	13,132	38,539	4,749	5,470	7,205	34,247	11,631	30,121	3,721	7,910	4,204	25,917
One to Two Mahalanobis Distance Matching, caliper (1)													
ATT[1]	-.03***	-.05***	-.02***	-.06***	-.04***	-.05***	-.01***	-0.11***	-.002***	-.17***	-.09***	-.002	-.003**
	(.002)	(.004)	(.002)	(.006)	(.005)	(.004)	(.002)	(.009)	(.001)	(.023)	(.009)	(.002)	(.001)
Bias[2]	2.0	1.4	2.5	1.8	1.8	1.6	2.7	3.3	2.7	1.5	4.3	4.6	3.1
Sensitivity Analysis													
Gamma[3]	6.3	3.3	10	2.4	2.9	9.1	10	10	3	10	10	1.5	2.7
P_mh-	0.048	0.032	0.00	0.042	0.030	0.048	0.03	0.00	0.037	0.07	0.00	.044	0.041
One to One Propensity Score Matching, caliper (0.01), without replacement													
ATT	-.04***	-.06***	-.03***	-.06***	-.06***	-.06***	-.03***	-.09***	-.005***	-.15***	-.07***	-.004**	-.003***
	(.001)	(.002)	(.0009)	(.005)	(.004)	(.003)	(.0009)	(.003)	(.0005)	(.007)	(.003)	(.002)	(.0005)
Bias	4.3	2.2	3.6	3.3	2.8	2.1	3.9	1.7	1.6	4.3	3.3	21.7	4.3
Sensitivity Analysis													
Gamma[3]	6.7	3.6	10	2.6	3.6	9.8	10	10	2.8	10	10	10	1.7
P_mh-	0.039	0.035	0.001	0.032	0.046	0.049	0.006	0.00	0.044	0.00	0.00	0.00	0.034

* $p<0.1$; ** $p<0.05$; *** $p<0.01$

[1]Average Treatment effect on the Treated. Standard errors are in parentheses.

[2]Absolute value of the difference of means in the treated and matched untreated subsamples as a percentage of the square root of the average sample variance in both groups. Here, we report the average for all covariates.

[3]Critical value of the Rosenbaum's τ. It indicates by how much unobserved confounding factors could negatively influence selection into treatment. P_mh- is the associated significance level.

Table 2-5B 2001-2008 Impacts of PA size reductions during 2009-2012

	All	<48 km	>48 km	<10 km	10-30 km	30-68 km	>68 km	In Arc	Not Arc	Rondônia	Pará	Roraima	Amazonas
Total	416,943	170,930	246,013	67,041	67,951	76,308	205,643	114,766	159,479	12,101	102,665	21,304	72,423
Treated	2,950	2,178	772	778	1,052	517	603	2,497	453	1,944	553	422	31
One to two Mahalanobis Distance Matching, common support, caliper (1)													
ATT[1]	.07***	.06***	.15***	.03	.05***	.25***	.09***	.07***	.005	.08***	-.11***	-.19	.06
	(.009)	(.011)	(.012)	(.018)	(.013)	(.025)	(.011)	(.012)	(.004)	(.028)	(.019)	(.012)	(.097)
Bias[2]	0.6	1.6	3.5	2.6	1.1	1.3	5.0	1.2	8.3	4.1	4.9	6.3	12.3
Sensitivity Analysis													
Gamma[3]	1.8	1.5	7.2	1	1.3	5.2	4.3	1.4	1	1.2	3.3	1.3	1
P_mh+	0.021	0.011	0.049	0.016	0.021	0.043	0.045	0.010	0.436	0.013	0.045	0.049	0.5
One to One Propensity Score Matching, common support, caliper (0.01), without replacement													
ATT	.09***	.09***	.11***	.07***	.05***	.23***	.08***	.07***	-.04**	.08***	-.08***	-.06***	0
	(.008)	(.009)	(.014)	(.016)	(.013)	(.022)	(.012)	(.010)	(.009)	(.012)	(.014)	(.011)	(0)
Bias	2.7	0.5	5.2	2.3	2.3	7.3	4.6	2.6	58.8	4.5	2.9	10.1	12.2
Sensitivity Analysis													
Gamma[3]	2.3	2.2	2.8	1.5	1.3	4.2	2.6	1.5	2.4	1.4	2.4	6.4	.
P_mh+	0.03	0.048	0.048	0.29	0.019	0.044	0.045	0.018	0.048	0.009	0.049	0.049	.

* p<0.1; ** p<0.05; *** p<0.01

[1]Average Treatment effect on the Treated. Standard errors are in parentheses.

[2]Absolute value of the difference of means in the treated and matched untreated subsamples as a percentage of the square root of the average sample variance in both groups. Here, we report the average for all covariates.

[3]Critical value of the Rosebaum's τ. It indicates by how much unobserved confounding factors could negatively influence selection into treatment. P_mh+ is the associated significance level.

Table 2-6 2010-2015 Post-Reduction Deforestation Impacts of 2009-2012 PA size reductions

	All	<48 km	>48 km	<10 km	10-30 km	30-68 km	>68 km	In Arc	Not Arc	Rondônia	Pará	Roraima	Amazonas
Total	128,813	36,409	92,404	9,787	15,790	24,737	78,449	53,789	61,484	6,702	47,087	6,822	56,858
Treated	2,781	2,073	708	759	999	440	583	2,241	533	1,649	592	422	111
One to two Mahalanobis Distance Matching, common support, caliper (1)													
ATT[1]	.05***	.05**	.038***	.03	.03***	.06***	.02**	.03***	.007	.04***	.011*	0	.069*
	(.004)	(.006)	(.007)	(.025)	(.008)	(.013)	(.008)	(.006)	.004	(0.009)	(0.007)	(0)	(.04)
Bias[2]	3.5	4.8	9.2	5.6	6.7	7.8	10.0	3.3	12.8	3.9	7.5	13.1	9.8
Sensitivity Analysis													
Gamma[3]	7.6	4	1.5	1	1.8	2	1	3	1	1	1	.	1
P_mh+	0.048	0.049	0.047	0.119	0.045	0.049	0.332	0.047	0.049	0.042	0.279	.	0.23
One to One Propensity Score Matching, common support, caliper (0.01), without replacement													
ATT	.04***	.05***	.03***	.04***	.05***	.14***	-.00	.06***	.02**	.11***	.005	0	0
	(.004)	(.006)	(.006)	(.010)	(.008)	(.017)	(.004)	(.006)	(.007)	(.011)	(.007)	(.)	(0)
Bias	3.6	2.4	5.8	2.6	3.0	5.5	10.1	2.7	22.6	5.4	3.3	10.8	32.1
Sensitivity Analysis													
Gamma[3]	3.7	4.2	5.4	3.2	3.7	8.3	1	8.9	1	10	1	.	1
P_mh+	0.044	0.045	0.049	0.046	0.038	0.048	0.12	0.049	0.066	0.002	0.36	.	.

* p<0.1; ** p<0.05; *** p<0.01

[1]Average Treatment effect on the Treated. Standard errors are in parentheses.

[2]Absolute value of the difference of means in the treated and matched untreated subsamples as a percentage of the square root of the average sample variance in both groups. Here, we report the average for all covariates.

[3]Critical value of the Rosebaum's τ. It indicates by how much unobserved confounding factors could negatively influence selection into treatment. P_mh+ is the associated significance level.

Chapter 3

Who benefits from PADDD in the Brazilian Amazon?

Authors

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Abstract (233 words)

When Protected Areas (PAs) avoid deforestation, they prevent the development of economic activities and intensify poverty. Rollbacks against PAs designations, (PA downgrading downsizing and degazettement) have been rising during the last decades to accommodate, mostly, industrial scale resource extraction and development. In this paper, we assess an issue that have been overlooked until know: whether and to what extent PADDD influenced income distribution of households and inequalities in the Brazilian Amazon, from 2000 to 2010. We use matching and difference-in-differences according to road distance and PADDD events influence on the final size of protection in order to control for location bias according to land profitability and the dynamic of the PA network. We find the higher-middle income class increased when the land was already profitable following strict reductions in protection. The development of off-farm opportunities near existing markets could have enriched middle-income class households (inter-class distributive effect) or could have spurred higher-middle income class households' in-migrations (intra-class distributive effect). However, we find inequalities decreased when the land was not profitable following a stabilization or increase of protection due to downgradings or the designation of new PAs. Enhanced PA management and new economic activities such as tourism within PAs might have enhanced the livelihood of poorest households (inter-class distributive effect), spurred lower-middle income class households' in-migrations (intra-class distributive effect), or losing access to forest resources could have induced poorest households' out-migrations (intra-class distributive effect).

Keywords

Protected areas, PADDD, Brazil, Amazon, inequalities, impact evaluation

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1 Introduction

Economic development challenges often make environmental conservation as a second purpose for bilateral or multilateral institutions (Kauppi et al., 2018; United Nations, 2020; Wunder, 2001), especially in tropical countries, which are characterized by high poverty, inequality and deforestation levels. Brazil contains the largest share of the Amazonian rainforest, with ~4 millions square kilometers (Barreto and Baima, 2019; Soares-Filho et al., 2010), but more than 750 000 square kilometers have been lost since 1970 onwards (INPE, 2019). In the Brazilian Amazon, deforestation rates have been rising from the 1990's until 2004 (Fearnside, 2005), along with economic growth, which yet have remained characterized by high levels of poverty, especially in rural areas (Celentano and Veríssimo, 2007). To halt environmental degradations, protected areas (PAs) have been extensively used as a tool to conserve forests and their ecosystem services such as biodiversity and carbon storage (Nogueira et al., 2018; Soares-Filho et al., 2010; UNEP-WCMC et al., 2020). Over the recent decades, the PA network has extended a lot until reaching 43% of the Brazilian Amazon in 2014 (Nogueira et al., 2018; Veríssimo et al., 2011). They have been showed to have participated in the drop in deforestation from 2004 to 2011 despite their location bias (Kere et al., 2017; Nolte et al., 2013; Pfaff et al., 2015a, 2015b).

PAs also could have supported local livelihoods (Naughton-Treves et al., 2005) by providing a wide range of economic benefits for local populations (Clements et al., 2014; Estifanos et al., 2020; Naidoo et al., 2019; Yergeau, 2020). Yet, except for Kauano et al. (2020), who do not find PAs influenced average economic growth per capita, their impact on economic development have been quite overlooked to our knowledge. Recurring fine-scale socio-economic and spatial data are indeed lacking for the Brazilian Amazon. In other tropical countries, PAs have been found to deter the development of agricultural or industrial economic opportunities and thus to contribute to exacerbate poverty (Ferraro et al., 2011; Hanauer and Canavire-Bacarreza, 2015; Miranda et al., 2016).

As a result, along with other weakening of environmental law (Campos-Silva et al., 2015; Fearnside, 2016; Ferreira et al., 2014), PAs have been legally challenged, especially from the mid-2000's. These large-scale legal rollbacks, which are known as PA downgrading downsizing and degazettement mostly have been largely enacted to accommodate industrial scale resource extraction and development and, to a lesser degree, local land pressures as well as land claims (Golden Kroner et al., 2019; Naughton-Treves and Holland, 2019; Qin et al., 2019). 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 through a legal boundary change and degazettement is a loss of legal protection for an entire PA (Mascia and Pailler, 2011).

PADDD especially occur where economic activities would have otherwise been developed (Keles et al., 2020; Symes et al., 2016; Tesfaw et al., 2018): on lands that are suitable for the development of economic activities and when their environmental costs are low. PADDD are thus more likely to be proposed or enacted near to urban centers, for PAs that are rather large and costly to manage and when deforestation already occurred to some extent. For these reasons, even though higher deforestation levels has been observed following PADDD, Pack et al. (2016) and Tesfaw et al. (2018) did not find these events increased forest cover losses in the Brazilian Amazon from 2002 to 2011 and in Rondônia in 2010.

The aim of this paper is to fill up the existing gap regarding the influence of the dynamic of the PA network on economic development: we investigate whether and where PADDD events influenced local socioeconomic outcomes in the Brazilian amazon. Like for higher observed deforestation rates, better socio-economic conditions following PADDD could simply be related to the pre-existence of favorable economic conditions, which also led to deforestation activities. In that case, PADDD would have no additional effect on economic development. The same result could be obtained if PADDD occurred in lands that were profitable and that would not have attracted economic activities had they never faced PADDD. Yet, PADDD were enacted for various reasons, from allowing tourism activities, to implementing large-scale industrial agriculture, which could influence their impact.

Our contributions are the following. First, we use an original dataset (Masson, 2020) that relies on GIS and socio economic data in 2000 and 2010 in the Brazilian Amazon and depicts the distribution of revenue and households within 100km^2 grids. They will help to supplement the lack of fine-scale spatialized socio economics and geographic data in the Brazilian Amazon. We are thus able to work on the distribution of income and on inequalities to appreciate to what extent and which part of the population most benefited from PADDD. Second, in order to account for location bias, we apply a propensity score matching on the pre-treatment period (2000). A difference-in-differences estimations on the pre-matched sample gives us average treatment effect on the treated for the impact of PADDD, in addition to control for both time-unvarying and varying confounding factors. We think location bias might differ according to PADDD events influence on the final size of protection and the level of economic pressures. As a result, we separate grids in which protection decreased through downsizing and/or through degazettement from grids in which protection stabilized or increased through downgrading and/or through a combination of downsizings, degazettements and new PA designations. We also distinguish grids according to economic pressures using proximity to existing roads.

On the one hand, near roads, we find that PADDD events that reduced protection contributed to enlarge middle and higher-middle income classes. We distinguish two possible transmission channels, both fostered by existing proximity to roads and the development of infrastructure and economic activities. The first one is based on the enrichment of lower classes households, leading to an inter-class distributive effect; the second one is based on the in-migration of higher-middle income households, leading to an intra-class distributive effect. On the second hand, far from roads, we find that if the size of protection increased or stabilized, inequalities were reduced though a growth of the lower middle-income class and a reduction of the poorest class. An inter-class distributive effect could have been driven by the enrichment of lower-income households, who might have taken advantage of low-skilled jobs provided by tourism-related economic opportunities in newly accessible areas or who might have benefited from better-secured access to forest resources. An intra-class distributive effect could have been driven by an out-migration of households without revenue if new economic activities prevented them to access forest resources or if PAs remained poorly managed; and by an in-migration of lower-middle income households to access newly accessible off-farm jobs.

The rest of the manuscript is organized as follows. In Section 2, we present relevant background about income distribution and PAs dynamic in the Brazilian amazon, as well as relevant literature for considering potential effect of PADDD on local households. Section 3 presents the data we use and our empirical strategy. In Section 4, we provide our estimates for impacts of PADDD events on income distribution as well as robustness checks. Section 5 discusses and Section 6 concludes.

2 PAs, PADDD events and Income Distribution

2.1 PADDD as a tool to foster economic development

From the 1960's to the 1980's, human settlements, economic activities and large-scale infrastructure projects were promoted by the military dictatorship in the Brazilian Amazon through credits and subsidies. Along with insecure land tenure and consequent land grabbing, deforestation increased (Araujo et al., 2009; Brown et al., 2016). In addition to biodiversity losses, the surge in deforestation -approximately 20 000 squared kilometers of forests were cleared in 2000 (INPE, 2019)- caused social insecurities due to land use conflicts and prevented indigenous people to pursue their traditional livelihood (Silva et al., 2017). The economy stabilized in the 90's as the country became integrated in globalizing market as a major supplier of beef and soybeans (Arima et al., 2014). Deforestation was mainly the consequence of profitable extractions of forest resources on places that were then converted into crops and pastures. The Brazilian Amazon would have follow a "boom and bust" development pattern (Celentano et al., 2012; Silva et al., 2017): the extraction of forest resources is first characterized by an increase in economic development due to larger income and better access to jobs. However, this increase is short and unsustainable due to poor governance conditions, lack of infrastructure and the depletion of forest resource on which local population depended (Celentano et al., 2012; Silva et al., 2017). The Brazilian Amazon has been characterized by high poverty rates, even though inequalities are lower than in other Brazilian regions, as there is a dominance of rural smallholders who suffer from poor infrastructure, public services such as education and lack of access to markets (Clements et al., 2014; Guedes et al., 2012; Tritsch and Arvor, 2016).

Local and international communities started to raise the importance of the Amazon rainforest to sustain global climate and local livelihoods (Naughton-Treves et al., 2005; Veríssimo et al., 2011; Pailler, 2018) which lead to several reinforcements of the environmental law (Arima et al., 2014). Among other tools, PAs were established to conserve forest habitats and the Brazilian Institute of the Environment and Renewable Natural Resources (IBAMA) was created to enforce environmental laws (Arima et al., 2014). However, PA management were underfunded and enforcement remained poor (Veríssimo et al., 2011), which led to a peak in deforestation in 2004 with 26.800 square kilometers of land being cleared according to INPE (2019). Deforestation decreased from 2004 to 2012 after the Brazilian government started to be more "conservation centered" (Duchelle et al., 2014; Tritsch and Arvor, 2016) and implemented various policy measures; in addition to the 2008 economic crisis, which decreased global demand for beef and soybeans (Assunção et al., 2015). The Amazon PA program was implemented in 2002 in order to extend the PA network and to improve PA management. From 2004 to 2011, the succession of the two phases of the Action Plan to Prevent and Control Deforestation in the Amazon (PPCDAm) (Arima et al., 2014; Souza-Rodrigues, 2019; Veríssimo et al., 2011) added new binding measures, with increased enforcement by the Real-Time System for Detection of Deforestation (DETER) which was implemented by the National Institute for Space Research (INPE).

According to Caviglia-Harris et al. (2016) and Tritsch and Arvor (2016) the development of the Brazilian Amazon would rather follow a "forest transition" pattern. Like in other Brazilian regions, several social progresses were made and contributed to an increase in the middle class as poverty was reduced (Clément et al., 2018). However, the Brazilian Amazon would remained characterized by an active deforestation frontier (Celentano et al., 2012; Silva et al., 2017) with high poverty levels and a middle class that rather stabilized (Clément et al., 2018). Rural households are indeed threaten by forest losses as they heavily rely on forest resources (Wunder et al., 2014) and by urbanized activities to implement large-scale infrastructure and development projects (Celentano et al., 2012). When they can migrate to urban areas to find off-farm jobs (Guedes et al., 2012), their welfare does not necessarily increase (Silva et al., 2017) as governance conditions, infrastructure and public services are poorly developed (Celentano et al., 2012). In addition, PAs have been found to negatively affect the growth rate of the industrial value added in the Brazilian Amazon (Kauano et al., 2020) and to exacerbate poverty in several tropical countries (Miranda et al., 2016; Ferraro et al., 2011; Hanauer and Canavire-Bacarreza, 2015). As more than 40% of the Brazilian Amazon was covered by PAs in 2010 (Veríssimo et al., 2011), this has led to several backlash of the Brazilian federal government against conservation policies supported by interest groups from industrial sectors and local interests (Bernard et al., 2014; Marques and Peres, 2015).

Reducing PA sizes, either entirely (i.e. degazettement) or to some extent (i.e. downsizing) and allowing more economic activities within PAs (i.e. downgrading)[1] has been seen as a solution to foster economic development (Golden Kroner et al., 2019; Kauano et al., 2020; Qin et al., 2019). While such events started in the 70's, the phenomenon has accelerated from the mid-2000's (Bernard et al., 2014). Overall, from 2001 to 2010, 21 PADDD events were enacted[2] in the Brazilian Amazon affecting more than 43,000 square kilometers of forests. 15 events were enacted to allow infrastructure construction such as the Jirau hydroelectric dam and the federal highway BR-421 in Rondônia or the expansion of agribusiness activities in Mato Grosso. 2 where enacted to allow internal settlements of large property owners in Rondônia and Amazonas. One has been enacted due to its overlap with an indigenous land and one other to allow tourism within the park in Roraima. The remaining three have been enacted for no

[1] PA downgrading often lead to a change of IUCN category, from strict to mixed use for example. However, in our sample, only PAs that were Strict Nature Reserve (IUCN category Ia) were transformed into National Park (IUCN category II). While no human visitation or use were allowed in 2000, some economic opportunities considered as environmentally and culturally compatible with ecosystem conservation, for example linked to tourism, were then allowed.

[2] It must be acknowledged that these events are only those that are known from PADDDtracker.org Data Release Version 1.1 (Conservation International and World Wildlife Fund, 2017). In addition, others are proposed, i.e., not passed into law yet. In this paper, we only focus on enacted events as we seek to evaluate whether they affected income distribution from 2000 to 2010.

know reason. It is worth to notice that not all PADDD events do lead to a strict decrease of protection, for example, when PAs are downgraded to allow some human activities within them or when new PAs are designed years later to replace degazetted or downsized PAs.

Despite these various reasons, PADDD especially occur where the opportunity cost of conservation is high i.e. when PAs are accessible, rather large and costly to manage and when it is not too damaging for the environment i.e. when prior internal deforestation already occurred (Keles et al., 2020; Symes et al., 2016; Tesfaw et al., 2018). This make it legitimate to wonder whether such enacted events actually increased economic development in the Brazilian Amazon, as they are most likely located in places where households were already better off.

2.2 Who might benefits from PADDD events?

No paper has yet assessed the effect of PADDD on socioeconomic outcomes, or more specifically, on income distribution. However, PAs are often considered to reduce the availability of land for economic activities (e.g. agriculture) (Pfaff et al., 2009; Sims, 2014, 2010). Using a Von Thünen land rent model, Robalino (2007) investigates whether land conservation policies may affect landless agricultural wages and landowners rents. When agricultural activities are not permitted within PAs, even though agricultural rent may first decrease, the increases in price due to the migration of workers to cities make the agricultural rent actually increasing. While landowners are better off, the real wage of landless agricultural workers decreases, which leads to modifying income distribution.

Empirically, PAs have been found to exacerbate poverty when they reduce the rent from economic activities: near to roads or cities, where lands are more profitable, and when they are managed effectively. In the Peruvian Amazon, old and mixed-use PAs would have both avoid deforestation and exacerbated poverty of households living nearby (Miranda et al., 2016). This was also the case in Costa Rica, Thailand and Bolivia for PAs located on flat lands and near markets, which tends to increase agricultural suitability (Ferraro et al., 2011; Hanauer and Canavire-Bacarreza, 2015). On the contrary, no effect was found farther from economic pressures (e.g. far from cities) and on less suitable lands (e.g. on steep slopes) in Costa Rica and in the Brazilian Amazon (Ferraro et al., 2011; Kauano et al., 2020).

However, if they attract tourism activities (Paul J Ferraro and Hanauer, 2014), PAs might create off-farm job opportunities and foster the development of infrastructure such as roads (Canavire and Hanauer, 2012; Naidoo et al., 2019; Robalino and Villalobos, 2015; Sims, 2010; Yergeau, 2020). Even though these activities may not directly benefit poorest households and may actually increase inequalities (Sims, 2010); they could generate positive externalities on the well-being of households (Yergeau, 2020). Still, such economic benefits must offset the forgone rents from land clearing activities (Sims, 2010), which are lower far from economic pressures or in less profitable lands (Ferraro and Hanauer, 2011; Hanauer and Canavire-Bacarreza, 2015). At intermediate distance of economic pressures, or on less profitable lands near economic pressures (Ferraro and Hanauer, 2011; Hanauer and Canavire-Bacarreza, 2015), PAs might still prevent some land clearing activities (Sims, 2010) while attracting more tourism activities, generating win-win situations where both deforestation and poverty are reduced (Duchelle et al., 2014).

In addition, PAs might contribute to enhance local livelihoods and act as a safety net for poorest households (Andam et al., 2010; Wunder et al., 2014) when they ensure land rights and better access to forest resources (Clements et al., 2014; Estifanos et al., 2020; Naidoo et al., 2019; Duchelle et al., 2014). PAs could also safeguard the provision of ecosystem services such as flood control or maintenance of water quality on which the well-being of local populations depends (Coad et al., 2008).

PADDD are often proposed and enacted near roads when they generate positive economic benefits without generating high environmental costs since some internal deforestation already occurred (Keles et al., 2020; Tesfaw et al., 2018; Pack et al. 2016). If PAs contributed to avoid some land clearing activities, we could expect that enacted PADDD fostered the development of economic activities. Since lands are made accessible, the economic well-being of households living inside or nearby could increase and an in-migration of wealthy households could be spurred (Hering and Paillacar, 2016; Redding and Schott, 2003; Sjaastad, 1962). Depending on the type of activities that are developed following PADDD events, agricultural rents might increase first, as well as incomes, but prices too, which might not benefit poorest households.

Far away from roads, PADDD are proposed and enacted despite lower economic benefits at lower environmental costs (Keles et al., 2020; Sims, 2014). Even though these events would make no difference in term of economic activities, PAs could have provided poor households an access to forest resources or to some tourism related economic activities. This may also be detrimental to the economic well-being of households living inside or nearby, who might have no other source of incomes and could spur out-migration behaviors.

Since PADDD were enacted for various reasons and may have different impacts in terms of economic development, we evaluate how income distribution and inequalities have varied from 2000 2010, following PADDD at the grid level. To approach these heterogeneities, we distinguish economic pressures on land using proximity to roads and we separate grids in which protection decreased from grids in which protection actually increased or stabilized despite PADDD.

3 Empirical Strategy

In this section, we first present our observation units and data. We have measured the extent of protection in 2000 and 2010 on grids of 100km² to evaluate whether it influenced income distribution and inequalities, controlling for key confounding factors. Then, we precisely describe our empirical strategy: a difference-in-differences estimations based on pre-matched data.

3.1 Data

3.1.1 Units of observation

To investigate whether PADDD events affected the distribution of households' income in the Brazilian Amazon, we use a unique database combining highly remote sensing and socio-economic data over the Brazilian Amazon in 2000 and in 2010 (Masson, 2020). Socioeconomic data have been measured each year by the IBGE (2016) at the census tract level. Since census tract frontiers are established according to population size when data are collected, the frontiers are not stable through time. To deal with this issue, both years' layers are overlapped using a $100km^2$ grid. Consequently, socio-economic data, as well as remote sensing data are aggregated at the grid level.[3]

3.1.2 Treated variable: PADDD events

We extracted PAs locations and frontiers from the World Database on Protected Areas (WDPA) (IUCN and UNEP-WCMC, 2016) and only kept conservation units from the National System of Protected Areas (Sistema Nacional de Unidaded Conservação - SNUC). We consider a grid to be protected if at least 25% of its surface is protected before 2000. As the SNUC records the degree of permitted intervention within each conservation unit, we distinguish between grids covered with strict protection (no human intervention is allowed) and those covered with sustainable use protection (allowing human activities to some extent).

PADDD data comes from PADDDtracker.org Data Release Version 1.1 (Conservation International and World Wildlife Fund, 2017a), which describes the locations and frontiers of known proposed and enacted PADDD events until 2015. We study PADDD that were enacted from 2001 to 2010 and that only concerns conservation units designated before 2000 (see Figure 1). As a result, we compute the total grid surface affected by PADDD and subtract it to the grid surface protected before 2000. We obtain the surface of the grid that remains protected in 2010.

PADDD events may be enacted more than once in a grid during our study period. In addition, some PAs may be designated up to 2010 and may overlap with PADDD (Golden Kroner et al., 2019). As a result, we differentiate grids in which PADDD reduced the size of protection through downsizings and/or degazettements, from those in which the protected surface increased or stabilized due to downgradings and/or a combination of downsizings, degazettements and new PA designations until 2010.

[3] Further details about the compilation of this dataset and all associated variables are provided in Masson (2020).

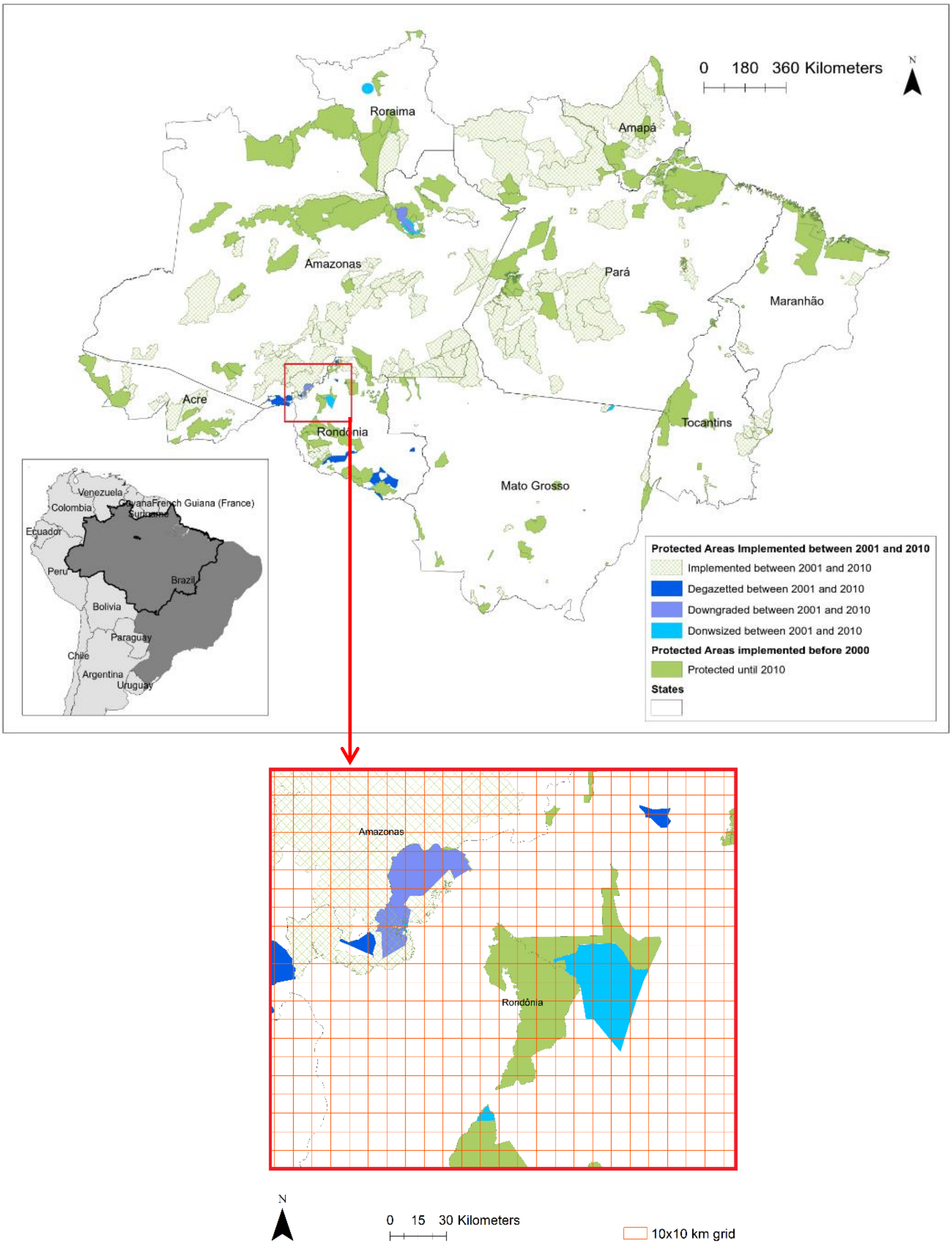

Figure 3-1 Protected Areas and PADDD in the Brazilian Amazon

3.1.3 Outcome variables: income distribution

We use several outcome variables measured at the census tract level by the IBGE (2016) and aggregated at the grid level by Masson (2020) which depict the number of households belonging to each income class within each grid. To account for the high increase in population density in the Brazilian Amazon from 2000 to 2010, these variables have been reported to the total population density within each grid.

The proportion of households without revenue and with less than half a minimum wage (MW) are considered to be in the lower income classes. They were 27.5% on average within each grid. The proportion of households earning from one-half to one MW, from one to two and from two to three MW are considered to be in the middle-income classes (lower, middle and higher middle-income classes) and were 53% on average. Finally, we consider that the proportion of households earning from three to five and from five to ten MW belong to the higher income classes. [4] They were on average 5% within each grid. It is worth to note that the minimum wage vary from 147.3 reals in 2000 to 510 reals in 2010.

To measure inequalities, we chose to compute Gini coefficients, as well as Lorenz curves (reported for treated and control groups in Figure 4 in the Appendix) for each grid using the average distribution of each income class and of households within each grid.

3.1.4 Confounding variables

We use several covariates affecting both the likelihood of PAs to be selected for PADDD and local economic development. The nearest distance of the center of the grid to 1996 roads (CIESIN, 2015) illustrates access to markets (Celentano et al., 2012; Hanauer and Canavire-Bacarreza, 2015; Miranda et al., 2016). Average elevation per grid (Masson, 2020), rainfalls (Funk et al., 2015) as well as forest cover in 2000 (Hansen et al., 2013) are used to depict agricultural profitability (Canavire-Bacarreza and Hanauer, 2013; Celentano et al., 2012; Ferraro et al., 2011; Hanauer and Canavire-Bacarreza, 2015; Miranda et al., 2016; Miyamoto, 2020; Naidoo et al., 2019). We use the share of the grid surface under integral or sustainable use protection as well as the presence of indigenous lands (FUNAI, 2015) to control for households' access to lands (Hanauer and Canavire-Bacarreza, 2015; Kauano et al., 2020; Naidoo et al., 2019). Average nightlights (NOAA, 2016) per grid help accounting for population density and the presence of economic activities (Ghosh et al., 2013). Raster format datasets such as for nightlights, rainfalls, elevation and forest cover were uniformly distributed at the grid level using a methodology described in Masson (2020).

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[4] We use these thresholds because the Brazilian middle class enlarged from 2000 to 2010 as compared to the poor class. The IBGE considers households to be in extreme poverty and in poverty when they have no revenue or when they earn half a minimum wage while households are considered to be in the richer classes when they earn more than 5 times the minimum wage. Since we focus on the Brazilian Amazon, where households are mainly rural and poor (Clément et al., 2018), it seems more appropriate to consider that households earning more than 3 times the minimum wage belong to the high income classes.

3.2 Difference-in-differences Matching

3.2.1 Pre-matching strategy

We estimate PADDD events' impacts on income distribution of Brazilian Amazon households and on inequalities. Contrary to the PA network on average, PAs selected for PADDD are often located in highly profitable lands (Keles et al., 2020; Symes et al., 2016; Tesfaw et al., 2018). These lands are also those where a larger amount of higher-income households might be encountered (Barbier, 2020; López Sandoval et al., 2017; Symes et al., 2016). Naïvely comparing PAs selected for PADDD to PAs may make us falsely conclude that these events increased the proportion of households in the higher income classes or to overestimate that result.

First, we use a matching strategy to build a comparison group as similar as possible to the treated group. Depending on the success of the matching, the remaining differences should be attributed to the treatment (Ferraro and Hanauer, 2014). The propensity score method attributes to each control observation a probability of being treated regarding a set of covariates affecting both the selection into treatment and the outcome (Caliendo and Kopeinig, 2008). Each treated observation is then matched to the control observation having the most similar probability of being treated. As noted above, these covariates affect both the likelihood of PAs to be selected for PADDD event and local economic development: nearest distance of the center of the grid to 1996 roads, average elevation per grid, forest cover in 2000, PAs types, presence of indigenous lands and average nightlights. These covariates are either fixed over time or measured pre-treatment to avoid endogeneity. We also use states, since they feature very distinct performances and policies in terms of economic development and environmental protection (Celentano and Veríssimo, 2007; Silva et al., 2017).

To ensure enough similarity between treated and control groups, we impose that treated observations should be only matched to the first most similar control observation within a caliper of 0.05 standard deviation of the estimated propensity score. Our standard errors use the variance approximation of Abadie and Imbens (2006) in case hidden biases remain due to unobservable confounders. To avoid leakage effects, which may influence the outcome of the control group near treated observations and bias our estimated coefficients, we exclude a 10km buffer zone (Naidoo et al., 2019) around each PA and PADDD event. We use standardized mean differences between covariates before and after matching to assess whether we appropriately reduce the differences between treated and control observations.

The pre-matching strategy is used to find similar control groups for each treated groups: i) the whole group of treated grids, ii) grids in which the protected surface decreased after treatment and iii) grids in which the protected surface increased or stabilized after treatment.

3.2.2 Difference-in-differences

The pre-matching strategy helps us to account for time-varying confounders (Abadie, 2005; Lechner, 2011) but it does not ensure that fixed unobservable biases would not drive the results. We take advantage of the panel structure of our data to apply a Difference-in-differences estimation (Chabé-Ferret and Subervie, 2013; Paul J. Ferraro and Hanauer, 2014; Heckman et al., 1997).

The Difference-in-differences design allows estimating average treatment effects on the treated by subtracting the difference in the outcome variable for the control group before and after treatment to the difference in the outcome variable for the treated group before and after treatment. If both treated and untreated groups followed the same time trends, the difference may be attributed to the treatment (Abadie, 2005; Lechner, 2011).

Our regression takes the following form:

$$Y_{it} = \beta 1 PADDD_i + \beta 2 Post_t + \beta 3 PADDD_i{*}Post_t + \alpha_i + \varepsilon_i$$

Y_{it} represents our outcome variables: the proportion of households in a certain income class in grid i at time t. $PADDD_i$ is our binary variable for treatment. It takes the value one if the grid is subject to any form of PADDD and zero otherwise. This variable helps to account for fixed unobservable differences between treated and untreated observations that could influence our outcomes. $Post_t$ is our binary variable representing post-treatment year (2010). It takes the value one in 2010 and zero in 2000 and takes into account any common time trend that could influence the distribution of Brazilian Amazon households' incomes. $PADDD_i{*}Post_t$ is the interaction term of the Difference-in-differences with $\beta 3$ its associated coefficient. It equals one if the grid has been subject to any form of PADDD in 2010 and zero otherwise. We use fixed effects α_i at the level of the municipality as they differ regarding their political and socioeconomic conditions. Standard errors are clustered at the PA level for the control group and at the PADDD event level for the treated group in order to account for spatial autocorrelation between same PAs and same PADDD events.

The Difference-in-differences is estimated for each of the pre-matched samples: i) the whole group of treated grids, ii) grids in which the protected surface decreased after treatment and iii) grids in which the protected surface increased or stabilized after treatment. Those samples are then separated according to the average distance of the center of the grid to the nearest 1996 road to distinguish between profitable unprofitable lands.

3.3 Robustness checks

If we want the difference-in-differences coefficient to be valid, the treated and control groups must have followed the same time trends, conditional on a set of time-varying confounders (Chabé-Ferret and Subervie, 2013; Lechner, 2011).

This conditional parallel trend assumption is hardly testable, as it would need to observe how income distribution, only observed in 2000 and 2010, varied at least two periods before treatment. Therefore, we decide to visually inspect average nighttime lights per grid in 1992 and 2000, since it is commonly used in the literature to approximate the level of local economic development (see e.g. Ghosh et al. (2013)). In addition, the pre-matching strategy may help to satisfy those conditions as the samples are conditioned on pre-treatment confounders.

It is however possible that a difference-in-differences strategy applied to matched sub-samples generates biases, if selection into treatment is conditional on both fixed and time-varying confounders (Chabé-Ferret and Subervie, 2013). Moreover, matching within calipers may increase the variance of the estimator and generate biases since fewer observations are matched. The average treatment effect on the treated might not be representative of the true treatment effect for the full sample (Caliendo and Kopeinig, 2008). We check the stability of our results using a difference-in-differences on the total sample. Standard errors are clustered at the PA level for the control group and at the PADDD level for the treated group. We use fixed effects at the grid level to account for fixed differences between them.

4 Results

4.1 Descriptive Statistics

4.1.1 PAs before 2000 and selection into PADDD

On average

Among 4,556 grids having at least 25% of their surface protected by 2000, 300 faced some type of PADDD events between 2001 and 2010. 203 ended up with no or with a lower protected surface due to downsizings and/or degazettements. The remaining 97 grids had their protected surface increased or stabilized because of downgradings (66) and/or new PA designations (30). We observe several differences concerning income distribution both over time and between treated and untreated grids (see Table 1A and Figure 2A).

Table 3-1A Description of outcome variables, according to type of treatment

Outcomes	Untreated Grids (N=4,351)		All treated (N=300)			Protection decreases (N=203)			Protection increases or stabilizes (N=97)		
	Mean		Mean		DiM* vs untreated	Mean		DiM* vs untreated	Mean		DiM* vs untreated
	2000	2010	2000	2010	2000	2000	2010	2000	2000	2010	2000
Proportion of households:											
- without revenue	23.6%	34.5%	19.0%	22.7%	0.06	10.4%	23.1%	0.00	36.8%	21.9%	0.00
- earning less than ½ MW	4.3%	14.3%	2.9%	10.3%	0.00	2.5%	7.6%	0.00	3.6%	15.9%	0.21
- earning from ½ to 1 MW	30.4%	28.7%	18%	30.8%	0.00	17.2%	27.9%	0.00	19.7%	36.8%	0.00
- earning from 1 to 2 MW	20.6%	12.7%	20%	21.3%	0.73	21.2%	24.4%	0.31	17.7%	15%	0.03
- earning from 2 to 3 MW	4.5%	2.5%	6.2%	6.9%	0.00	7.5%	8.6%	0.00	3.6%	3.6%	0.06
- earning from 3 to 5 MW	3.1%	1.6%	5.3%	3.6%	0.00	6.7%	4.2%	0.00	2.4%	2.3%	0.03
- earning from 5 to 10 MW	1.8%	0.9%	3.1%	1.6%	0.00	4%	2.1%	0.00	1.1%	0.1%	0.00
Average number of households	5.6	85,589	4.5	1,588	0.14	5.3	2,184	0.44	0.15	341	0.03
Gini Index	0.43	0.48	0.38	0.46	0.00	0.32	0.47	0.00	0.52	0.46	0.00

On average, the proportion of households belonging to the lower-income classes (without revenue and earning less than half a MW) increased over time in both untreated and treated grids in which the protected surface decreased. Interestingly however, in grids in which the protected surface increased or stabilized, the proportion of households without revenue decreased, while it was significantly higher before treatment compared to other groups.

The proportion of households belonging to the middle income classes (earning from half to three minimum wages) grew in treated grids in which the protected surface while it declined in untreated grids. While the proportion of households in the higher-middle income class was larger in treated than in untreated grids, that of the lower-middle income class was yet much smaller. We can additionally notice that the proportion of households in the lower-middle income class increased the most in treated grids in which the protected surface increased or stabilized.

The proportion of households belonging to the higher income classes decreased over time, especially in treated grids in which the protected surface decreased where they were more numerous. Finally, inequalities were lower in treated grids in which the protected surface decreased but increased over time as compared to treated grids in which the protected surface increased or stabilized. These observations are consistent with PAs being selected for strict size reductions (downsizings and/or degazettements) where the land is profitable (Golden Kroner et al., 2019; Keles et al., 2020; Symes et al., 2016; Tesfaw et al., 2018).

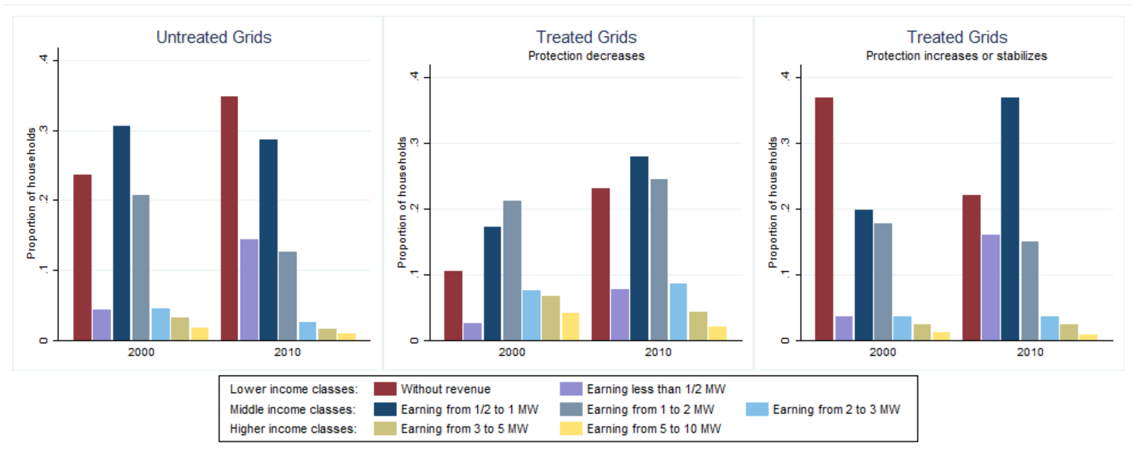

Figure 3-2A Income distribution in 2000 and 2010 in untreated and treated groups

According to road distance

These trends may differ according to the distance of the grid to the nearest road, used here to indicate economic pressures since more accessible lands are also more profitable (Jusys, 2018; Laurance et al., 2009; Pfaff, 1999) (see Table 1B and Figure 2B). On 1,722 grids located below 41.12 km from roads, the median of the sample, 178 faced some type of PADDD events from 2000 to 2010. Beyond that distance, they were only 122 grids over 2,834.

Near roads, in both untreated and treated grids in which the protected surface decreased, the proportion of households without revenue was lower before any PADDD event occurred than far from roads and increased in a lower extent. Besides, the proportion of households in the middle and higher middle-income classes was higher and increased for both type of treated grids while it decreases in untreated grids. Inequalities were lower too in treated grids, but increased a lot, as compared to untreated. This can also be observed in the Lorenz curve in Figure 4b in the Appendix. Lands would have been more profitable near roads than farther away, giving households better access to economic opportunities, especially in grids selected for strict size reductions.

Far from roads, while the proportion of households without revenue was higher before treatment in treated grids in which the protected surface increased or stabilized than in any other groups, it decreased until 2010. Here, we also observe an increase in the proportion of households belonging to the lower-middle income class, while it was significantly lower before treatment than in any other groups. Inequalities were higher far from roads but did not increased as much as near roads. This can also be observed in the Lorenz curve in Figure 4c in the Appendix. It seems that lands were not as profitable as they were near roads, but that allowing more activities within PAs or designing new PAs contributed to improve the situation.

Table 3-1B Description of outcomes variables, according to roads distance

Outcomes	Near roads					Far from roads				
	All Treated			Untreated		All Treated			Untreated	
	Mean		DiM* vs untreated	Mean		Mean		DiM* vs untreated	Mean	
	2000	2010	2000	2000	2010	2000	2010	2000	2000	2010
Proportion of households:										
- without revenue	11.0%	15.6%	0.00	18.7%	24.3%	30.6%	33.1%	0.06	26.4%	40.3%
- earning less than ½ MW	2.4%	7.7%	0.00	4.5%	12.7%	3.6%	14%	0.19	4.2%	15.2%
- earning from ½ to 1 MW	19.5%	31%	0.00	32.7%	33.8%	15.9%	30.5%	0.00	29.2%	25.8%
- earning from 1 to 2 MW	22.4%	28.6%	0.13	23.6%	16.3%	16.5%	10.8%	0.07	18.9%	10.6%
- earning from 2 to 3 MW	7.6%	9.7%	0.00	5.2%	3.3%	4.1%	2.9%	0.45	4.1%	2.1%
- earning from 3 to 5 MW	7.2%	4.5%	0.00	3.9%	2.1%	2.5%	2.3%	0.37	2.7%	1.3%
- earning from 5 to 10 MW	3.4%	2.2%	0.00	2.5%	1.5%	2.5%	0.7%	0.00	1.4%	0.6%
Gini Index	0.33	0.43	0.00	0.43	0.48	0.46	0.50	0.17	0.43	0.48

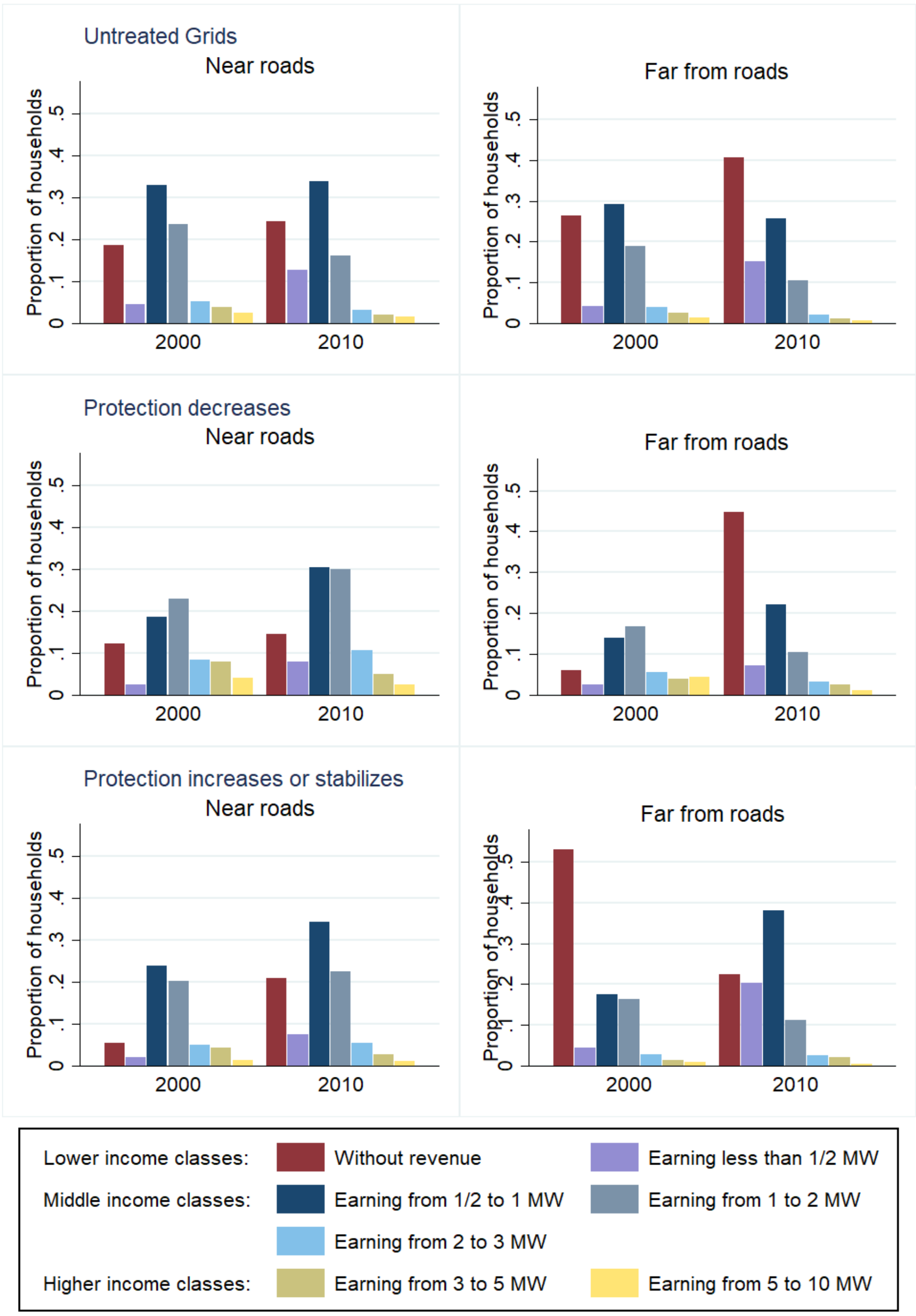

Figure 3-2B Income distribution in 2000 and 2010 in untreated and treated groups according to road distance

4.1.2 Covariate balance and quality of pre-matching strategy

We provide descriptive statistics as well as differences in means between untreated and treated observations regarding confounding variables both before and after pre-matching (see respectively Tables 4 and 5 in the Appendix).

Before any PADDD event occurred, treated grids in which the protected surface decreased had a lower share of their surface under integral protection than untreated grids but encroached more on indigenous land than any other groups. Households probably already had better access to lands. They were also in lower average rainfalls locations, at higher elevation and nearest to roads, which may be appropriate conditions for the development of economic activities, such as dams. Better access to lands and markets might also constitute better livelihood conditions for local households (Duchelle et al., 2014; Kauano et al., 2020; Naidoo et al., 2019; Ferraro et al., 2011; Hanauer and Canavire-Bacarreza, 2015).

Before treatment, treated grids in which the protected surface increased or stabilized had a larger amount of their surface under strict protection and encroached less with indigenous lands than other groups, which may indicate poorer access and use of lands. Even though they were at lower elevation, nearest to roads and had a lower amount of rainfalls than untreated grids, they also had more forest cover than both groups, a higher amount of rainfalls and were farther from roads than treated grids in which the protected surface decreased. That may indicate a lack of profitability as compared to treated grids in which the protected surface decreased.

These differences in means are all significant (Table 4 in the Appendix) before matching, which confirms selection biases for both treated groups (Keles et al., 2020; Symes et al., 2016; Tesfaw et al., 2018). After matching, we considerably reduce differences in observed covariates and thus average biases between treated and untreated groups (see Figure 3 below and Table 5 in the Appendix).

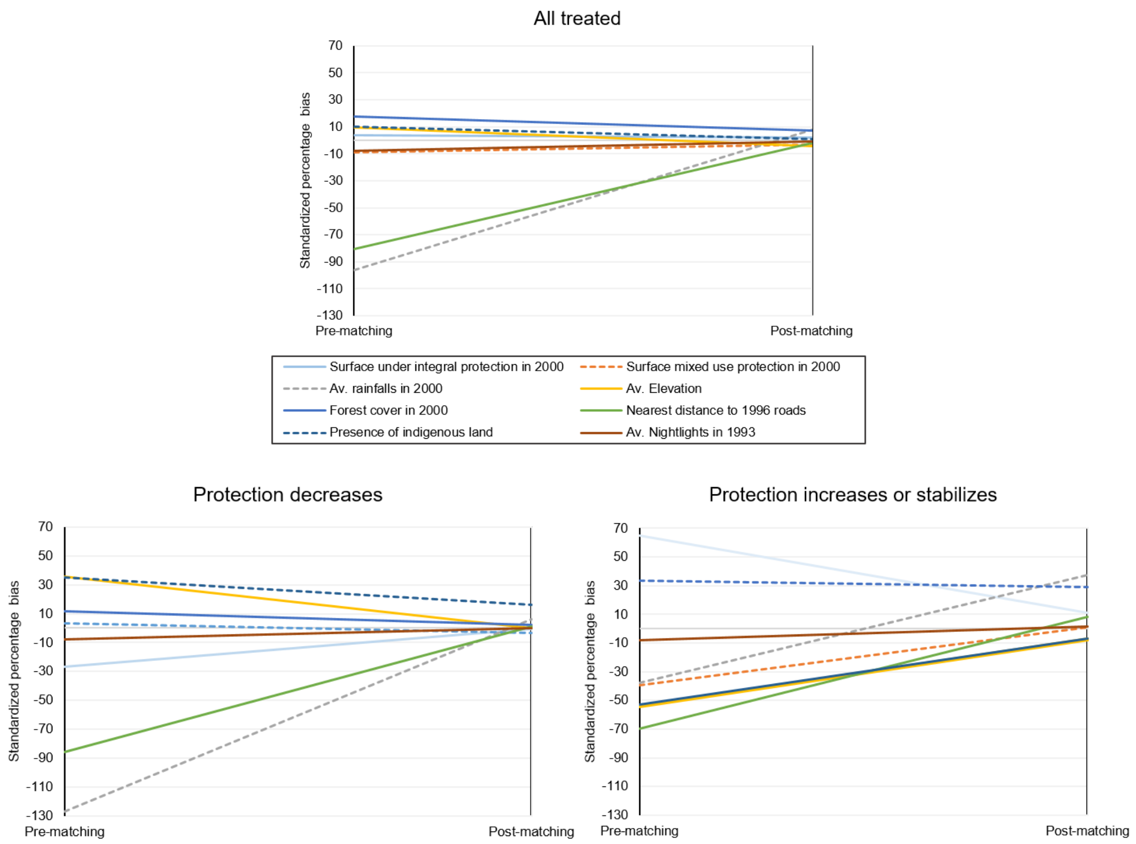

Figure 3-3 Reduction of average differences across covariates after matching

4.2 Impact of PADDD on income distribution

4.2.1 Average results

The coefficients of the difference-in-differences estimations on pre-matched samples for all treated grids, grids in which protected surface decreased, and grids in which the protected surface increased or stabilized are presented in Table 2A below.

On average, PADDD contributed to increasing the proportion of households in the middle-income classes, earning from half to twice a MW. Following PADDD events that decreased grids protected surfaces; both the middle-class and the higher middle-income class had their proportion extended as compared to untreated grids. Both proportions actually decreased over time in untreated grids, and the proportion of households in the higher-middle income class was already higher before treatment in treated grids (see Table 1a and Figure 2a). In grids in which, despite PADDD, the protected surface increased or stabilized, only the proportion of households in the lower middle-income class was positively affected. Yet, their proportion was lower on average in treated grids than in untreated grids before treatment, and actually, their proportion decreased over time in untreated grids.

Even though it looks like inequalities were reduced, benefiting either the higher-middle income or the lower-middle income class, the absence of significance of the Gini index does not allow us to confirm that was the case on average. As previously noticed, the proportion of households in each class of revenue does not follow the same time trend when the sample is broken down according to road distance, a proxy of access to market and land profitability (Jusys, 2018; Laurance et al., 2009; Pfaff, 1999). As a result, we distinguish grids located below and grids located after the median road distance of the sample.

Table 3-2A Difference-in-differences estimations on pre-matched samples, average results

proportion of households:	without revenue	with less than ½ MW	From ½ to 1 MW	From 1 to 2 MW	From 2 to 3 MW	From 3 to 5 MW	From 5 to 10 MW	Gini Index
DiD All events	-0.024	-0.001	0.078	0.080	0.021	-0.004	-0.009	-0.011
	(0.27)	(0.09)	(1.98)*	(2.50)**	(1.55)	(0.44)	(1.27)	(0.15)
R2	0.45	0.49	0.42	0.38	0.46	0.33	0.32	0.34
adj. R2	0.39	0.44	0.36	0.31	0.40	0.27	0.25	0.28
N	1,092	1,092	1,092	1,092	1,092	1,092	1,092	1,092
DiD Protected surface decreases	0.057	-0.029	0.026	0.103	0.040	-0.010	-0.013	0.038
	(0.50)	(0.97)	(0.77)	(2.41)**	(1.65)**	(0.83)	(1.44)	(0.47)
R2	0.41	0.53	0.46	0.36	0.45	0.33	0.32	0.32
adj. R2	0.34	0.47	0.39	0.28	0.38	0.25	0.24	0.24
N	748	748	748	748	748	748	748	748
DiD Protected surface increases or stabilizes	-0.165	0.041	0.176	0.048	0.005	0.011	0.004	-0.12
	(1.16)	(1.07)	(2.35)**	(1.18)	(0.27)	(0.78)	(0.50)	(0.10)
R2	0.54	0.50	0.41	0.32	0.42	0.38	0.33	0.39
adj. R2	0.47	0.42	0.32	0.22	0.33	0.28	0.2	0.29
N	352	352	352	352	352	352	352	352

We use fixed effects at the level of the municipality and clustered standard errors at the level of the PADDD event and protected area.

4.2.2 Spatial heterogeneity

In this section, the coefficient of the difference-in-differences estimations on pre-matched samples for all treated grids, grids in which protected surface decreased, and grids in which the protected surface increased or stabilized in Table 2B, according to road distance.

Near roads, first, we confirm that PADDD events contributed to increasing the proportion of households in the middle-income classes, which actually stabilized or decreased in untreated grids. Yet, the proportion of lower-middle income households was significantly smaller in treated grids as compared to untreated (see Table 1B and Figure 1B) before treatment. On average, we cannot confirm that it positively influenced income distribution.

In grids in which the size of protection was reduced, as expected, only the proportion of households belonging to middle and higher middle-income classes increased, and only near roads, whereas again, their proportion decreased in untreated grids. Yet, income does not seem to be more evenly distributed.

In grids in which, despite PADDD, the protected surface increased or stabilized, as expected, only the proportion of households in the lower middle-income class accelerated far from roads. It was significantly lower before treatment, and, in addition, their proportion decreased in untreated grids. It is interesting to notice that the proportion of households without revenue decreased as compared to untreated grids, while their proportion were the highest among any other groups before treatment, where their proportion increased. We can observe that income distribution is more evenly distributed.

Table 3-2B Difference-in-differences estimations on pre-matched samples, according to road distance

proportion of households:	without revenue		with less than ½ MW		With ½ to 1 MW		with 1 to 2 MW		with 2 to 3 MW		with 3 to 5 MW		with 5 to 10 MW		Gini Index	
	near	far	near	far	near	far	near	far	near	far	near	far	near	far	near	far
DiD All events	-0.014	-0.035	-0.031	0.040	0.110	0.045	0.157	-0.017	0.046	-0.010	-0.008	0.000	-0.006	-0.015	0.038	-0.071
	(0.35)	(0.22)	(1.25)	(0.65)	(2.66)**	(0.66)	(3.61)***	(0.341	(2.37)**	(0.66)	(0.71)	(0.02)	(0.70)	(1.36)	(0.86)	(0.41)
R2	0.42	0.42	0.70	0.55	0.42	0.52	0.32	0.50	0.45	0.54	0.30	0.42	0.30	0.35	0.40	0.34
adj. R2	0.34	0.34	0.49	0.47	0.34	0.46	0.22	0.44	0.37	0.48	0.21	0.34	0.20	0.26	0.32	0.25
N	638	454	638	454	638	454	638	454	638	454	638	454	638	454	638	454
DiD Protected surface decreases	-0.058	0.322	-0.062	0.027	0.058	-0.030	0.161	-0.018	0.063	-0.001	-0.012	-0.003	-0.010	-0.020	-0.03	0.19
	(1.58)	(0.67)	(2.25)**	(0.82)	(1.38)	(0.67)	(3.02)**	(0.36)	(3.05)**	(0.03)	(0.97)	(0.14)	(1.27)	(0.98)	(0.77)	(0.96)
R2	0.42	0.51	0.62	0.48	0.47	0.50	0.27	0.51	0.48	0.41	0.30	0.50	0.31	0.37	0.44	0.31
adj. R2	0.33	0.43	0.50	0.50	0.39	0.43	0.16	0.43	0.40	0.32	0.20	0.43	0.20	0.28	0.36	0.20
N	502	246	502	246	502	246	502	246	502	246	502	246	502	246	502	246
DiD Protected surface increases or stabilizes	0.147	-0.387	-0.025	0.075	0.080	0.236	0.125	-0.026	0.014	-0.004	-0.004	0.020	0.006	-0.002	0.159	-0.522
	(2.26)**	(3.21)***	(0.69)	(1.91)*	(1.00)	(3.48)***	(1.48)	(0.58)	(0.28)	(0.33)	(0.19)	(1.67)	(0.45)	(0.26)	(2.24)**	(3.29)***
R2	0.46	0.61	0.63	0.55	0.35	0.50	0.25	0.51	0.20	0.71	0.24	0.54	0.26	0.58	0.37	0.64
adj. R2	0.31	0.54	0.54	0.47	0.19	0.42	0.06	0.42	0.00	0.66	0.04	0.47	0.08	0.50	0.22	0.57
N	161	191	161	191	161	191	161	191	161	191	161	191	161	198	161	198

We use fixed effects at the level of the municipality and clustered standard errors at the level of the PADDD event and protected area.

4.2.3 Robustness Checks

First, we can notice in Figure 5A and 5B in the Appendix that treated grids in which protection decreased and untreated grids all seems to have followed the same time trends in terms of economic development, as depicted by average nighttime lights per grids (Ghosh et al., 2013). While that could indicate our difference-in-differences coefficients would be valid, it is likely this measure might not be representative enough of how households' income varied before treatment. In addition, it remains possible that a difference-in-differences strategy applied to matched subsamples actually increases time trends differences before treatment by reducing differences in 2000 (Chabé-Ferret and Subervie, 2013). Moreover, matching might decreases the number of observations on which we apply the difference-in-differences estimations. The coefficients might thus not represent average treatment effects on the total sample (Caliendo and Kopeinig, 2008).

As a result, we present below in Table 3A, the coefficients of the difference-in-differences estimations for all treated grids, grids in which protected surface decreased, and grids in which the protected surface increased or stabilized. In Table 3A, these coefficients are presented according to road distance.

Table 3-3A Difference-in-differences , average results

proportion of households:	without revenue	with less than ½ MW	From ½ to 1 MW	From 1 to 2 MW	From 2 to 3 MW	From 3 to 5 MW	From 5 to 10 MW	Gini Index
DiD	-0.072	-0.025	0.146	0.094	0.028	-0.001	-0.006	0.037
All events	(0.47)	(0.65)	(2.88)***	(2.36)**	(1.43)*	(0.11)	(0.58)	(0.10)
R2	0.67	0.64	0.65	0.64	0.66	0.62	0.61	0.50
adj. R2	0.33	0.29	0.29	0.28	0.32	0.24	0.22	0.01
N	9,104	9,104	9,104	9,104	9,104	9,104	9,104	9,104
DiD	0.020	-0.048	0.124	0.113	0.032	-0.009	-0.011	0.105
Protected surface decreases	(0.14)	(1.42)	(2.89)***	(2.51)**	(1.66)*	(0.62)	(0.90)	(0.10)
R2	0.67	0.65	0.65	0.64	0.66	0.62	0.61	0.50
adj. R2	0.34	0.29	0.29	0.28	0.32	0.25	0.22	0.01
N	8,932	8,932	8,932	8,932	8,932	8,932	8,932	8,932
DiD	-0.288	0.035	0.209	0.058	0.024	0.017	0.005	-0.118
Protected surface increases or stabilizes	(1.70)*	(0.79)	(2.40)**	(1.45)	(1.32)	(1.21)	(0.90)	(0.12)
R2	0.67	0.64	0.65	0.65	0.65	0.61	0.60	0.50
adj. R2	0.34	0.29	0.29	0.29	0.29	0.23	0.21	0.01
N	8,695	8,695	8,695	8,695	8,695	8,695	8,695	8,695

We use fixed effects at the level of the grid and clustered standard errors at the level of the PADDD event and protected area.

In table 3a, we confirm that the proportion of households in the middle-income classes, earning from half to two times the MW increased following PADDD. This is also true following PADDD that decreased grids' protected surface where, in addition, the proportion of households in the higher middle-income class increased. In grids in which, despite PADDD, the protected surface increased or stabilized, we also confirm that the proportion of households in the lower middle-income classes, earning from half a MW to one MW is positively affected.

On average, it is only near roads that PADDD positively influence income distribution toward middle-income class households (Table 3b). In grids in which the size of protection is reduced, we confirm that the proportion of households belonging to middle and higher middle-income classes increase near roads only. When, despite PADDD, protected surface increased or stabilized, we confirm it is only far from roads that the proportion of households belonging to the lower middle-income class increased. We are also able to confirm that the proportion of households without revenue decreased far from roads, which contributes to decreasing inequalities levels.

Table 3-3B Difference-in-differences estimations, according to road distance

proportion of households:	without revenue		with less than ½ MW		With ½ to 1 MW		with 1 to 2 MW		with 2 to 3 MW		with 3 to 5 MW		with 5 to 10 MW		Gini Index	
	near	far	near	far	near	far	near	far	near	far	near	far	near	far	near	far
DiD All events	-0.008	-0.116	-0.027	-0.006	0.105	0.181	0.138	0.026	0.043	0.008	-0.009	0.012	-0.003	-0.009	0.065	-0.00
	(0.56)	(2.03)	(0.98)	(0.10)	(1.86)*	(1.94)*	(2.66)***	(0.54)	(1.82)*	(0.40)	(0.68)	(0.71)	(0.25)	(0.61)	(0.04)	(0.23)
R2	0.62	0.66	0.65	0.65	0.67	0.63	0.61	0.64	0.67	0.64	0.61	0.62	0.59	0.65	0.64	0.45
adj. R2	0.25	0.32	0.31	0.29	0.33	0.25	0.22	0.28	0.35	0.28	0.21	0.23	0.19	0.29	0.29	-0.08
N	3,436	5,668	3,436	5,668	3,436	5,668	3,436	5,668	3,436	5,668	3,436	5,668	3,436	5,668	3,436	5,668
DiD Protected surface decreases	-0.033	0.248	-0.027	-0.065	0.106	0.115	0.147	0.021	0.047	-0.003	-0.011	0.001	-0.005	-0.025	0.048	0.244
	(0.68)	(0.73)	(0.90)	(1.35)	(1.83)*	(1.71)*	(2.79)***	(0.31)	(2.18)**	(0.10)	(0.73)	(0.02)	(0.42)	(0.98)	(1.04)	(0.94)
R2	0.63	0.67	0.65	0.67	0.67	0.63	0.61	0.64	0.68	0.64	0.61	0.62	0.60	0.60	0.64	0.46
adj. R2	0.25	0.34	0.31	0.33	0.33	0.27	0.22	0.28	0.36	0.27	0.22	0.24	0.20	0.20	0.29	.0.08
N	3,384	5,548	3,384	5,548	3,384	5,548	3,384	5,548	3,384	5,548	3,384	5,548	3,384	5,548	5,548	5,548
DiD Protected surface increases or stabilizes	0.110	-0.468	-0.026	0.059	0.073	0.272	0.106	0.037	0.023	0.024	0.003	0.023	0.006	0.004	0.143	-0.231
	(1.56)	(4.55)***	(0.59)	(1.15)	(0.86)	(3.96)***	(1.01)	(1.16)	(0.48)	(2.33)**	(0.15)	(2.28)**	(0.70)	(0.56)	(0.06)**	(0.84)***
R2	0.62	0.68	0.65	0.65	0.66	0.63	0.62	0.64	0.64	0.64	0.64	0.61	0.60	0.59	0.64	0.46
adj. R2	0.24	0.35	0.30	0.30	0.32	0.26	0.24	0.28	0.28	0.28	0.28	0.23	0.19	0.18	0.28	-0.07
N	3,147	5,548	3,147	5,548	3,147	5,548	3,147	5,548	3,147	5,548	3,147	5,548	3,147	5,548	3,147	5,548

We use fixed effects at the level of the municipality and clustered standard errors at the level of the PADDD event and protected area.

5 Discussion

5.1 Near roads, middle income and higher-middle income classes expand due to decrease of protection

Near roads, in grids in which the size of protection was reduced, income distribution changed in favor of middle and higher-middle income classes. Actually, both the total number of households and the number of households within each income class increased from 2000 to 2010 (see Table 1A). As a result, treatment actually accelerated the growth of their number as compared to untreated grids.

Grids in which the protected surface decreased were mostly located near roads (71%), on land that were suitable for the development of economic activities (Keles et al., 2020; Pack et al., 2016; Tesfaw et al., 2018). Reducing protection could thus generate economic benefits, since land clearing activities would not have been developed had PAs been retained and enforced (Keles et al., 2020; Pack et al., 2016; Tesfaw et al., 2018). The proportion of households in middle and higher-middle income classes were indeed larger before treatment in grids selected for strict size reductions (see Figure 2B). PADDD events have been enacted there to develop large-scale infrastructure like dams or, to a lower extent, highways (91%) and to legitimize and foster the settlement of large-scale property owners (8%) (Conservation International and World Wildlife Fund, 2017).

An inter-class distributive effect might exist if poorest class households or lower-middle income households more easily accessed off-farm job opportunities related to the development of dams. The proximity of existing roads and the development of highways as well as other infrastructure related to dams construction could have made market access easier. Hence, landowners might have benefited more easily from an increase in agricultural rents (Ferraro and Hanauer, 2011; Hanauer and Canavire-Bacarreza, 2015; Robalino, 2007). An inter-class distributive effect might exist if higher-middle income households from untreated grids in-migrated within treated grids following PADDD. Migrations could be easier for households in the higher-middle income class (Redding and Schott, 2003), especially when travel-time to cities and off-farm jobs are lowered by the improvement of the road network (Hering and Paillacar, 2016).

5.2 Far from roads, maintaining protection and allowing economic activities decreased inequalities.

Far from roads, when, despite PADDD, the protected surface increased or stabilized, inequalities were reduced, driven by an increase in the proportion of lower middle-income class households and by a reduction in that of households without revenue. As previously observed in Table 1A, treatment actually accelerated the growth of the number of households belonging to the lower-middle income class and slowed down the growth of households without revenue.

Grids in which, despite PADDD, the protected surface increased or stabilized were mostly located far from roads (66%) and had characteristics that made the land hardly suitable for the development of economic activities. They were either strictly reduced and later replaced with new PAs, either downgraded, i.e., had a new status allowing economic activities. The proportion of households without revenue, as well as inequalities, were significantly higher in treated grids than in untreated grids before treatment, while it was the opposite regarding the proportion of households in the lower-middle income class (see Figure 2B). This could indicate that PAs were poorly managed and did not support local livelihoods (Miranda et al., 2016; Duchelle et al., 2014). These events were mostly enacted to allow tourism activities or to allow indigenous people to access lands (47%) and to legitimize large-scale property owners and internal populations (31%). They allowed the development of dams (20%) in a lower extent.

An inter-class distributive effect could appear if poorest households accessed lower-middle income classes. Far from roads, the provision of infrastructure and tourism-related economic opportunities that did not previously existed could have occasioned such effect (Canavire and Hanauer, 2012; Naidoo et al., 2019; Robalino and Villalobos, 2015; Sims, 2010; Yergeau, 2020). Designing better-managed PAs, as it seems to be the case after 2000 in Brazil with the SNUC Law No. 9.985/2000[5], might have also triggered such effect, through better land rights and safer access to forest resources, fostered by infrastructure development (Clements et al., 2014; Duchelle et al., 2014; Estifanos et al., 2020; Naidoo et al., 2019). An intra-class distributive effect could occur if households without revenue out-migrated toward untreated

[5] In 2000, a public institution (SNUC) was created and allowed common bases and guidelines for ``the creation, administration and management of the [environmental] units''.

grids. This could have been driven by loss of access to forest resources because of tourism activities, dams construction, or if newly implemented PAs did not provide enough land rights (Sjaastad, 1962). Households in the lower middle-income classes could rather in-migrate within treated grids to access low-skilled economic opportunities, facilitated by enhanced market access that does not previously existed (Hering and Paillacar, 2016).

6 Discussion

In this paper, we explore the effect of PADDD events occurring between 2001 and 2010 on income distribution in the Brazilian Amazon in 2010. Brazil contains a large part of the Amazonian tropical forests (Barreto and Baima, 2019), but a large part have been cleared in the first half of the 2000's (16,3% in 2003) (Fearnside, 2005). PAs have been extensively developed from the 2000's until reaching almost half of the territory (Soares-Filho et al., 2010). They have been considered to contribute to large poverty rates and inequalities since land-use constraints prevent the development of agricultural and industrial economic opportunities. To answer local land pressures, land claims and to develop large-scale infrastructure, PADDD events have been largely proposed and enacted since the mid-2000's. These events are mostly enacted in land that were suitable for the development of such activities (Keles et al., 2020; Symes et al., 2016; Tesfaw et al., 2018), where lands would have otherwise been cleared.

As far as we know, no PADDD events impact evaluation on economic development yet exists. In addition, lack of reliable data have made the evaluation of PAs effects on economic development in the Brazilian amazon largely under-explored. To deal with these two issues, we provide an ex-post quasi experiment, using propensity score matching on the baseline sample data, and a difference-in-differences estimation on a pre-matched sample data. We use an original dataset from Masson (2020) that covers the whole Brazilian Amazon from 2000 to 2010 and allies highly remote sensing and socio-economic data. To deal with selection bias, we break down our sample according to PADDD influence on the final size of protection and perform a spatial heterogeneity analysis using road distance.

PADDD events, on average, have contributed to the enlargement of the middle-income classes from 2000 to 2010 as compared to untreated grids in which PAs remained legally protected. Many other contextual factors might simultaneously explain the enlargement of the middle-income class in Brazil from the 2000's. Following the economic crisis occurring in the 1980's, the country has been characterized by a large and steady economic growth induced, among other factors, by numerous creation of off-farm jobs, by a raise in minimum wage – which was multiplied by 5 in nominal terms - and by social cash transfer programs (Clément et al., 2018).

An important distinction to be made is whether PADDD arises near or far from development opportunities that we approximate with distance to roads. Another important distinction is whether we observe PA size reductions or if the PA is replaced by another type of protection. In grids in which protection has been reduced, the higher-middle income class extended, near roads. In grids in which the protection stabilized or increased due to downgradings or the designation of new PAs, inequalities were reduced far from roads, due to the expansion of the lower-middle income class and the slowdown of the poorest class growth.

Near roads, the development of off-farm jobs and the increase of agricultural rents has been facilitated by the proximity to markets. The higher-middle income class could have enlarged following the reduction of protection because of an enrichment of households – inter-class distributive effect - (Ferraro and Hanauer, 2011; Hanauer and Canavire-Bacarreza, 2015; Robalino, 2007), or due to the in-migration of higher-middle income class households –intra-class distributive effect- (Redding and Schott, 2003). Far from roads, the development of tourism activities or of better-managed PAs helped to increase off-farm opportunities, access to markets and possibly to forest resources. Since inequalities decreased, it is likely that the enlargement of lower-middle income class came from the enrichment of households without revenue –inter-class distributive effect-, or from the in-migration of lower-middle income households –intra-class distributive effect- (Clements et al., 2014; Duchelle et al., 2014; Estifanos et al., 2020; Naidoo et al., 2019).

Several limitations can be underlined and might constitute future extension of this research. First, the data we use do not allows us to distinguish between inter-class and intra-class distributive effects as they only depict the distribution of households within some income classes in 2000 and in 2010. Both the population density and the minimum wage change over time and we would rather need the exact income of each households living within each grids as well as migration data each year. Second, we work on PADDD enacted between 2001 and 2010 regarding PAs implemented before 2000. These PAs were not as effectively managed (Veríssimo et al., 2011) and were not as effective in reducing deforestation (Kere et al., 2017; Nolte et al., 2013) as those implemented from 2000, which could explain enhanced economic development after their reduction. This is indeed confirmed by the positive impact that increasing protection had after 2000. In addition, many PADDD events have been enacted from 2010 to 2017 and may occur on PAs implemented after 2000 (Conservation International and World Wildlife Fund, 2017b). It would thus be interesting to replicate PADDD events impact evaluation on economic development, for more recent periods, and other countries, since the context matters too (Qin et al., 2019; Symes et al., 2016). Third, we study the income distribution, which is a monetary measure of households' economic well-being and does not account for many important features of local livelihoods (Guedes et al., 2012).

Our results allow us to underline that the conditions under which PADDD are enacted, as well as their types, influence income distribution, and do not influence all households categories similarly. This has to be coupled with impact evaluation of PADDD events on deforestation, to understand where and how to better protect without impeding economic development where it is necessary (Naughton-Treves and Holland, 2019; Tesfaw et al., 2018; Thieme et al., 2020). Indeed, finding that PA size reductions increase the proportion of higher-middle income households without reducing inequalities is not necessarily bad from a conservation perspective since better economic development could induce a forest transition (Caviglia-Harris et al., 2016; Tritsch and Arvor, 2016). However, a risk would be that PADDD only increase short-term economic gains, even if it reduced inequalities, without any long-term returns because of poor governance and loss of ecosystem services (Celentano et al., 2012; Silva et al., 2017), especially considering uncertainties related to climate change.

7 References

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8 Appendix

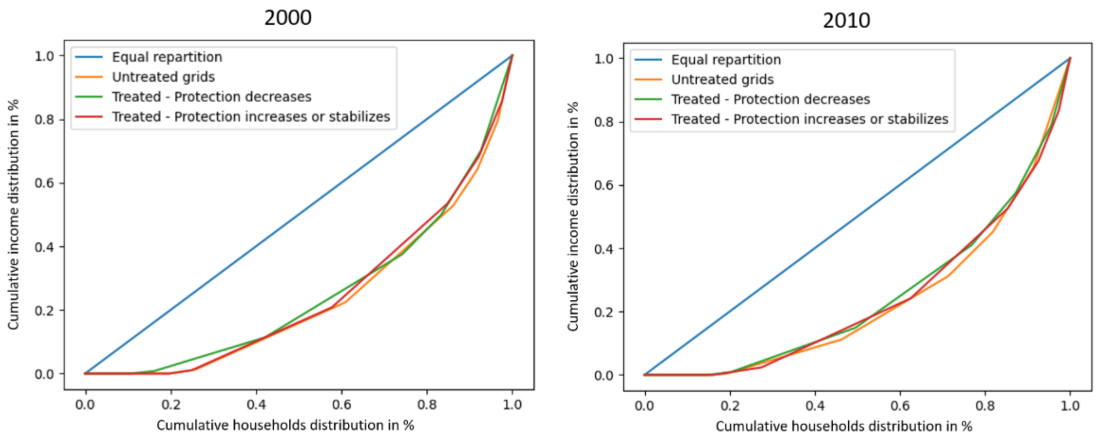
Table 3-4 Descriptive statistics of confounding variables before pre-matching

Covariates	Untreated Grids			Treated Grids											
				All types				Protection decreases				Protection increases or stabilizes			
	Mean	Min.	Max.	Mean	DiM*	Min.	Max.	Mean	DiM*	Min.	Max.	Mean	DiM*	Min.	Max.
Surface under integral protection in 2000	23.88	0	100	25.33	.48	0	100	14.88	.00	0	100	49.95	.00	0	100
Surface mixed use protection in 2000	56.86	0	100	41.56	.01	0	100	46.74	.52	0	100	29.36	.00	0	100
Av. rainfalls in 2000	2459.5	465.9	4959.9	1930.9	.00	1253.1	2808.8	1805.7	.00	1265.9	2808.8	2226	.00	1253	2797.9
Av. Elevation	142.84	0	1815	155.77	.13	0	526	188.61	.00	19	526	78.37	.00	0	376
Forest cover in 2000	77.01	0	100	81.5	.00	0	100	79.17	.09	0	100	86.99	.00	0	100
Nearest distance to 1996 roads	133.78	0	523.02	41.12	.00	.16	126.6	36.23	.00	.16	126.6	52.66	.00	2.18	112.6
Presence of indigenous land	0.42	0	1	.48	.05	0	1	.60	.00	0	1	.19	.00	0	1
Av. Nightlights in 1993	.06	0	27	.04	.32	0	2	.04	.44	0	2	.02	.52	0	2

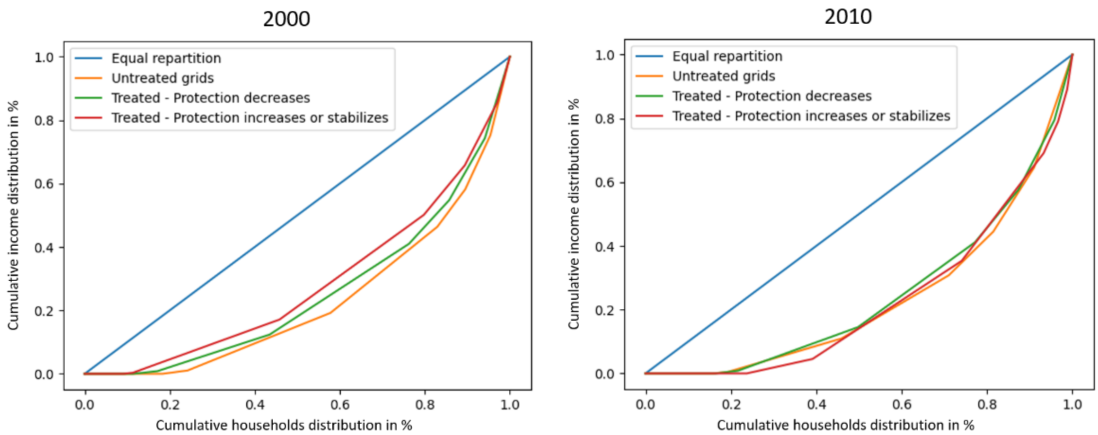
Table 3-5 Covariate balance of confounding variables after pre-matching

Covariates	% bias untreated	DiM*	% bias	% reduced
Untreated Grids vs Treated Grids: All types				
Surface under integral protection in 2000	3.8	.82	2.0	48.8
Surface mixed use protection in 2000	-9.1	.68	-3.3	63.9
Av. rainfalls in 2000	-96.1	.29	7.6	92.1
Av. Elevation	9.5	.48	-4.2	56.2
Forest cover in 2000	17.8	.33	7.2	59.8
Nearest distance to 1996 roads	-80.9	.56	-1.9	97.7
Presence of indigenous land	10.2	.93	0.7	93.4
Av. Nightlights in 1993	-7.7	.51	-0.7	90.9
Protected surface decreases				
Surface under integral protection in 2000	-26.5	.94	-0.7	97.5
Surface mixed use protection in 2000	3.5	.72	-3.4	3.2
Av. rainfalls in 2000	-127.1	.43	6.0	95.3
Av. Elevation	35.9	.99	0.1	99.8
Forest cover in 2000	11.6	.80	2.2	81.3
Nearest distance to 1996 roads	-85.7	.78	0.9	99.0
Presence of indigenous land	35.2	.10	16.0	54.6
Av. Nightlights in 1993	-7.6	1.00	0.0	100
Protected surface increases or stabilizes				
Surface under integral protection in 2000	64.9	.46	11.3	82.6
Surface mixed use protection in 2000	-39.7	.96	0.7	98.2
Av. rainfalls in 2000	-37.6	.00	37.4	0.7
Av. Elevation	-54.0	.30	-7.6	85.9
Forest cover in 2000	33.2	.05	29.0	12.7
Nearest distance to 1996 roads	-69.8	.20	8.1	88.4
Presence of indigenous land	-53.0	.59	-7.1	86.7
Av. Nightlights in 1993	-8.0	.41	1.5	81.5

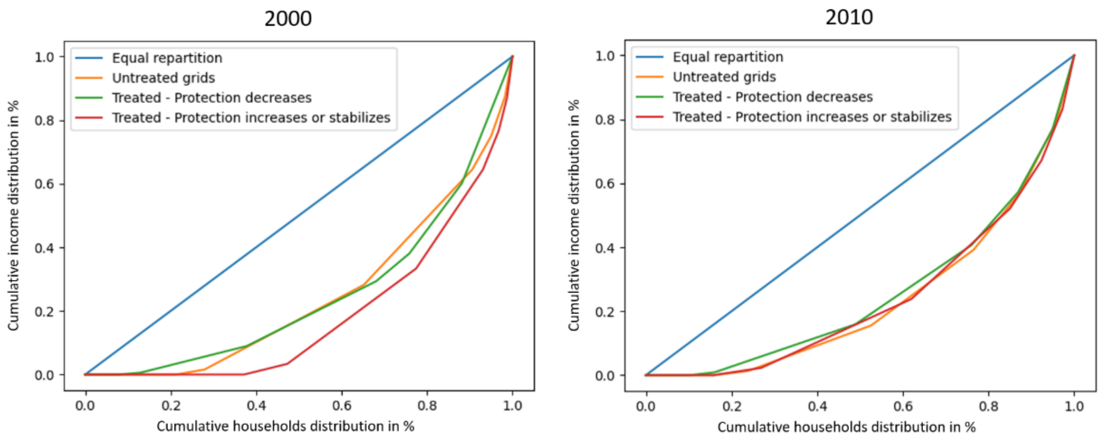
*P-values for differences in means between Untreated and Treated Grids after pre-matching



All sample (a)



Near roads (b)



Far from roads (c)

Figure 3-4 Lorenz Curve for treated and untreated grids in 2000 and 2010

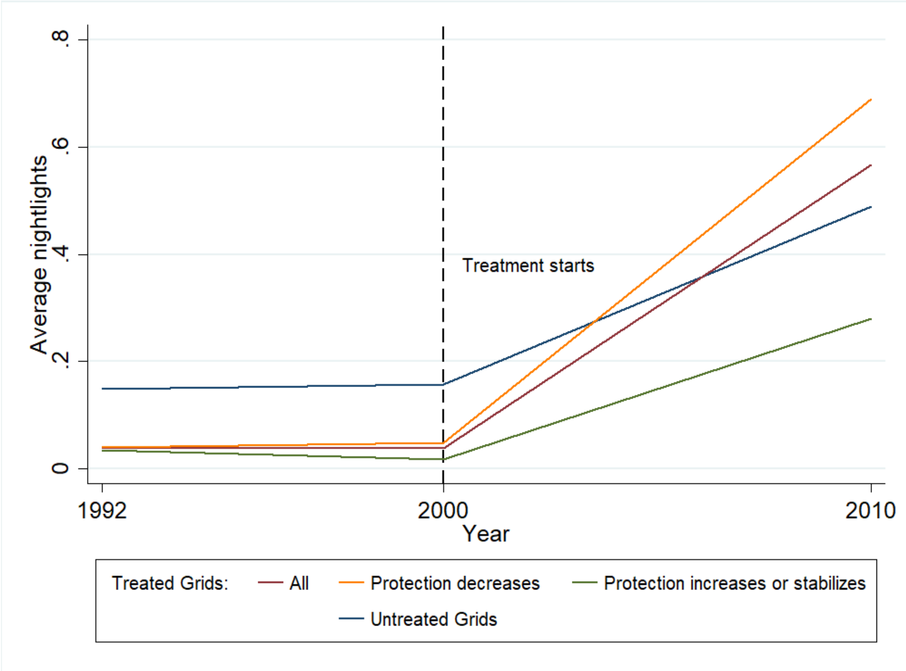


Figure 3-5A Average nightlight from 1993 to 2010 in untreated vs treated grids

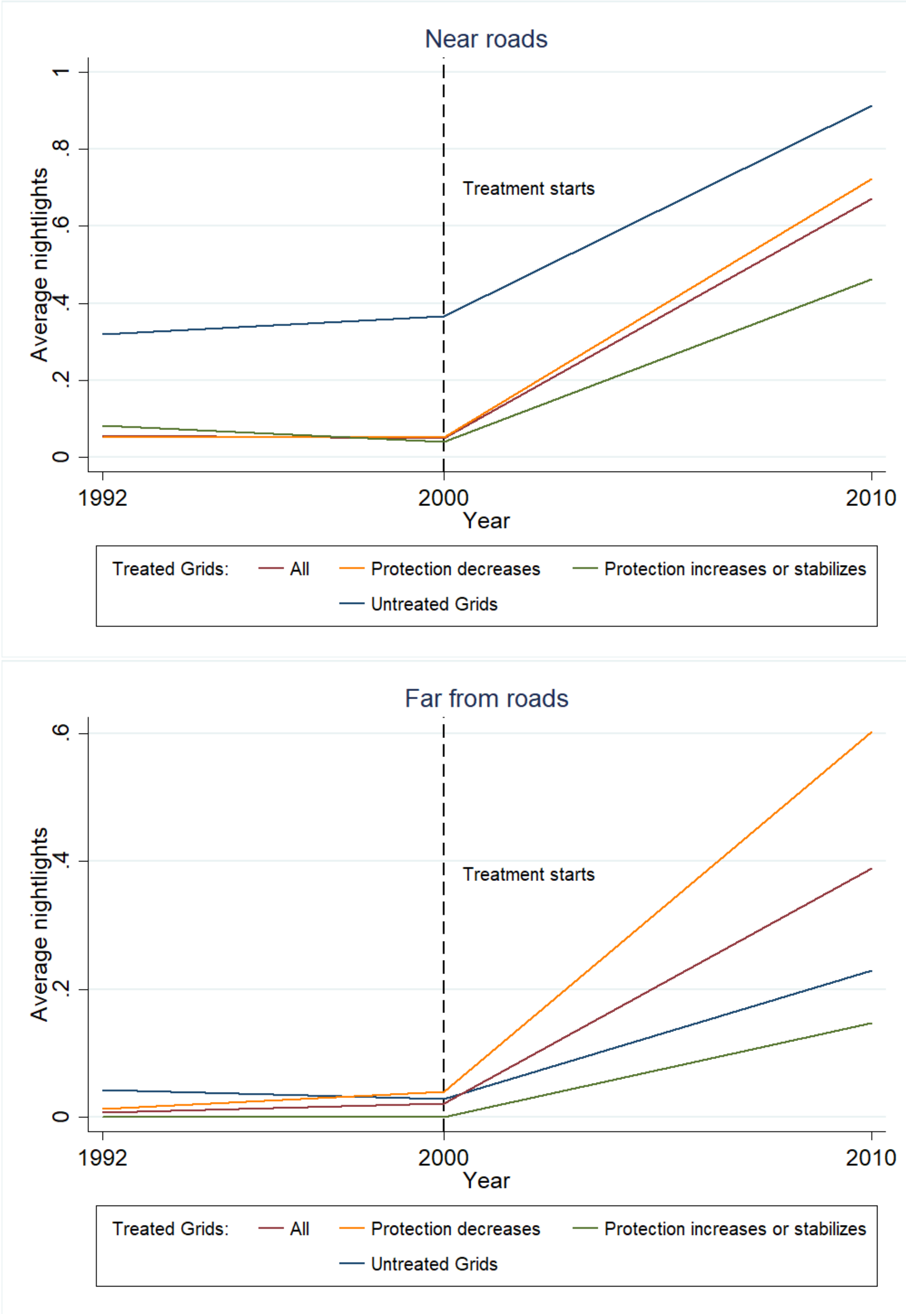


Figure 3-5B Average nightlight from 1993 to 2010 in untreated vs treated grids, according to road distance

General Conclusion

Conflicts between conservation and development objectives have led PAs to be located where economic pressures were weak, resulting in limiting their capacity to provide additional deforestation reductions. Besides, when PAs have had potential to contain economic pressures, thus limiting deforestation, they have often been invaded, due to lack of effective management. In addition, the risk that PAs could hamper economic development may have resulted in the surge of PA downgrading, downsizing and degazettement since the second half of the 2000's, when Brazil federal government have been focused on economic development after achieving historically low deforestation rates.

Yet, Brazilian Amazon PAs remain at the crossroad of environmental and development challenges, since their designation and management are still expected to contribute to several targets of the post-2020 biodiversity framework (CBD, 2019), that will be adopted under the 15th COP in China, of the Paris Agreement, and of the 2030 agenda for sustainable development.

Through this thesis, my aim was to assess whether, where, and to what extent PADDD events have been related to conflicts between conservation and development objectives in the Brazilian Amazon. I first assessed how PAs downsizing and degazettement, that reduced protection, resulted from trade-offs between conservation and development objectives according to PAs' and lands' characteristics (chapter 1), and then, whether and to what extent they influenced deforestation activities (chapter 2) and income distribution (chapter 3) based on relevant landscape and PAs heterogeneities from chapter 1. Below, I recall the main results of this thesis, emphasize their implications for public policies, as well as for science and, lastly, develop perspectives for further work.

1 Summary of main results

1.1 Trade-offs between conservation and development objectives

In the first chapter, we conceptualized the bargaining between conservation and development agencies that leads to PA size reductions across the landscape building on Tesfaw et al. (2018) original framework.

Both benefits from PA size reductions for development agencies and from avoiding PA size reductions for environmental agencies increase with the conservation opportunity costs, i.e. economic pressures. Reducing the size of perfectly enforced PAs might be socially efficient when economic gains are high while environmental losses are low. Such conditions would actually be difficult to meet since reducing PAs size when the OC of conservation is high has actually both high development gains and high conservation costs. To allow for spatial variations, we assumed that illegal deforestation within PAs varies across the landscape and shapes development and environmental agencies' views regarding PA size reductions. Indeed, previous illegal deforestation within PAs lowers development gains from PA size reductions and environmental gains from avoiding them, since a lower amount of forest remains whether for protection or for productive uses. With that in mind, if enforcement is more likely to outweigh economic pressure near roads - which is empirically confirmed in our data from 2003 to 2006 - development profits from PA size reductions and environmental benefits of avoiding them are high there and both decrease as we go away from cities where illegal invasions are more likely to take place. As a result, empirically investigating where PA size reductions mostly occur gives insight on the bargaining power of each agency.

1.2 The influence of development agencies

The first critical empirical result from chapter 1 is that PA size reductions were more likely to be enacted and proposed from 2006 to 2015 near cities, which is consistent with development actors having more bargaining power, since PA size reductions bring more environmental damages near cities. Far from cities or on less profitable lands, development agencies have fewer benefits from avoiding PA size reductions since the opportunity cost of conservation is much lower.

In chapter 3, our aim was to evaluate whether PADDD events had their intended effect on local economic development. We took advantage of GIS data on households' income distribution collected by Masson (2020) on 100 km² grids in 2000 and 2010 and used difference-in-differences estimations on pre-matched data to account for location bias. In addition, we divided samples according to their road distance to distinguish between high and low profitability areas. It was also important to distinguish between grids in which protection were reduced due to downsizing and degazettement and grids in which protection actually increased or stabilized due to downgrading or combinations of degazettements, downsizings and new PA designations.

Near roads, PADDD events that reduced protection accelerated the expansion of both middle income and higher-middle income classes. This result could come from an inter-class distributive effect: lower-middle income classes became richer, taking advantage of new off-farm job opportunities related to dams and highways construction or benefiting from a rise in agricultural land rents. This result could also be explained by an intra-class distributive effect: middle and higher-middle income classes in-migrated in treated grids, taking advantage of the existing proximity to roads.

Far from roads, we found no effect of PADDD events that reduced protection. However, PADDD events after which protection actually increased or stabilized helped decreasing inequalities, by slowing down the expansion of households without revenue and accelerating that of the lower-middle income class. These results could come from an inter-class distributive effect: poorest households accessed lower-middle income classes due to new allowed tourism activities within PAs or due to improved PA management securing land rights and access to forest resources. An intra-class distributive effect could have occurred too if improved infrastructure and access to markets helped lower middle-income class households to in-migrate to access new job opportunities. However, if newly designed PAs were not better managed, households without revenue could have out-migrated due to loss of land rights and secured access to forest resources.

1.3 The influence of environmental agencies

The second critical empirical result from chapter 1 is that PA size reductions were more likely to be enacted and proposed from 2006 to 2015 when they were already deforested, suggesting lower PA enforcement, and when they were costly to manage, as demonstrated by the influence of their size and IUCN management categories. This result illustrates that development agencies do not solely dictate PA size reductions: environmental agencies may not bargain to maintain protection for PAs that failed to block economic pressures or that are too costly to manage.

In chapter 2, we evaluated where and to what extent PA size reductions enacted from 2009 to 2012 induced additional deforestation from 2010 to 2015. First, we expected no impact of PA size reductions for PAs that were already entirely invaded, either because of lack of enforcement or because of lack of economic pressures. PA size reductions would raise deforestation to the extent that PAs previously blocked deforestation. Their capacity to limit deforestation depends on how PAs reduced the rent associated with agricultural activities, which is a function of land profitability and distance to cities.

We computed average treatment effects on the treated using a matching strategy at the pixel level that accounts for location bias. We went beyond previous works (Pack et al., 2016; Tesfaw et al., 2018) both in terms of space, as assess landscape and PAs heterogeneities across the entire legal amazon, and time, as we use more recent PADDD events and deforestation observations.

PAs reduced between 2009 and 2012 were already accelerating deforestation between 2001 and 2008 on average as compared to similar unprotected lands, confirming our result in chapter 1: failing PAs were selected to be reduced in size. Once reduced, on average, they contributed to additional deforestation activities between 2010 and 2015 as compared to similar constant-sized PAs. We found these average effects were driven by results for intermediate distances to roads. We found no previous nor post impacts of PA size reductions outside the arc of deforestation, due to lack of economic pressures; and nearest to roads, due to full invasions despite good expected enforcement capacities.

2 Scientific and Policy Implications

The main conclusions of this thesis allow us to underline useful insight for both science and public policy regarding PAs evaluation and management decisions

2.1 Improving PADDD events assessments

First and foremost, as stated in previous research (Mascia et al., 2014; Tesfaw et al., 2018), PAs impact evaluations must consider the dynamic of the PA network in analyses according to land and PAs characteristics. Recent versions of most databases, such as the world database on PAs, do not acknowledge how the status of standing PAs varied in time. It is for instance common that PAs that have been reduced do not appear anymore, without indications on their past status, or that some PADDD events are simply not recorded. Further efforts to complement world-scale initiatives to document PADDD events, like PADDDtracker.org, are thus needed to build comprehensive impact analyses and help decision-making (Cook et al., 2017; Forrest et al., 2015; Golden Kroner et al., 2019; Pack et al., 2016; Qin et al., 2019).

2.2 Allocating PADDD and PAs enforcement

Despite the fast expansion of PAs surface, mostly during the second and third phases of the PPCDAm, they have remained underfunded (Bernard et al., 2014; Silva et al., 2021; West and Fearnside, 2021), especially since Brazil underwent several fiscal and political crisis during the recent decade (Silva et al., 2021). Greater investments to reinforce PAs management would be key discussion elements of the post-2020 biodiversity framework (Silva et al., 2021). In this context, precisely addressing why and where PAs have been effective, and to what extent reducing their size influenced conservation and development objectives can help: i) to guide investments to improve enforcement practices where they are necessary and ; ii) to choose optimal sites and management types for new PAs designations (Bernard et al., 2014; Mascia and Pailler, 2011; Symes et al., 2016).

PAs located far from economic pressures had no impact, which resulted, in turn, in no impact of their size reductions. While that suggests conservation benefits are low, it is worth keeping in mind that economic pressures are likely to increase over time and encroach on isolated areas (Pack et al., 2016; Symes et al., 2016). For instance, it has been demonstrated that roads, especially unofficial ones, have been developing at a high pace in states outside the arc of deforestation such as Acre, even in remote municipalities, and within PAs (Nascimento et al., 2021; Symes et al., 2016). In addition to driving deforestation activities (Barber et al., 2014; Nascimento et al., 2021), they may have detrimental social impacts and lead to conflicts with indigenous people as they would often lack management plans (Bernard et al., 2014; Fearnside and de Alencastro Graça, 2006; Ferrante et al., 2020). In low economic pressure areas, PA size reductions did not either modify income distribution and would instead positively influence local economic development if they allow nature-related economic activities such as tourism that include local communities. In that sense, it could be meaningful to reverse PADDD events and ensure better management and inclusion of local people, while PAs still have some conservation value (e.g. biodiversity) (Golden Kroner et al., 2016; Qin et al., 2019).

Near to economic pressures, conservation costs from PA size reductions would rather be low too, since when PAs are entirely invaded, selecting them for size reduction has no deforestation impact, leading to few reasons to maintain protection. Besides, it appears that size reductions could help enlarge higher-middle income classes' households – even though our data only covers 2001 and 2010 years - which limits the scope of such implication. Yet, these PAs seem less costly to defend against illegal invasions, since low travel costs to cities make enforcement outweighing economic pressures. Investments should thus be dedicated to implement enforcement to increase PAs deforestation impact and avoid PADDD events as a consequence of their poor management (Bernard et al., 2014; Forrest et al., 2015; Pack et al., 2016; Symes et al., 2016; Tesfaw et al., 2018). PA size reductions rather seem to show larger environmental damages at intermediate distance from roads, where enforcement is more costly to implement. It seems thus relevant to reinforce enforcement eventually by taking advantage of budget made available from reducing failing should it occurs PAs (Fuller et al., 2010; Kareiva, 2010; Mascia et al., 2014).

2.3 Building better practices

Some practices could help consolidate PAs effectiveness in high to medium pressure areas and set up better-managed PAs. Choosing the appropriate level of governance is an important first condition. Herrera et al. (2019) found federal PAs to stem deforestation better than state PAs, because of the larger amounts of available resources and larger environmental interests from federal governance level. Second, particular attention has to be devoted to PAs types. Allowing low-impact activities through new PAs designations or downgrading might actually reinforce public support for their maintenance (Naughton-Treves and Holland, 2019), especially in case of land tenure conflicts (Mascia et al., 2014). It has been theoretically and empirically demonstrated that sustainable use PAs and enforcement by local people could be more effective to reduce PAs illegal invasions while preserving access to forest resources (Kere et al., 2017; Robinson et al., 2014; Soares-Filho et al., 2010).

Other forms of conservation instruments may adequately answer both development and conservation objectives (Mascia et al., 2014; Mascia and Pailler, 2011) under tight budget constraints, either when PADDD events cannot be avoided or to improve PA effectiveness and actually avoid them. First, market-based approaches like Payment for environmental services (PES) have been found to foster land management practices in Mexico's land titled communities without damaging social behaviour and potentially thus, expected economic development (Alix-Garcia et al., 2018). When combined with PAs (Sims and Alix-Garcia, 2017), PES could actually facilitate enforcement acceptability, even though it remains vulnerable to location bias and insecure land tenure (Börner et al., 2020; Cisneros et al., 2019). Second, bottom-up approach, like community-based forest management (Bowler et al., 2012), which are not common in the Brazilian Amazon (Medina et al., 2009), are sometimes found to prevent deforestation and improve local welfare under certain conditions (e.g. trust conditions of involved communities)(Arts and de Koning, 2017). Third, other types of top-down instruments, such as the federal priority list, which was launched in 2006 and blacklisted several highly deforested municipalities in the Brazilian Amazon, would have actually eased PAs enforcement without harming agricultural production (Assunção and Rocha, 2019; Cisneros et al., 2015). The priority list would have actually enhanced cattle ranching productivity by reducing net benefits from clearing available land (Koch et al., 2019). Further, Sills et al. (2020) demonstrate that when blacklisting is supplemented with state-level environmental governance capacity building programs (e.g. the green municipality program in Pará), it reinforced economic development and made enforcement efforts more supported by the public.

Developing better practices will surely need regularizing land tenure within PAs since it is a major driver of illegal internal deforestation (Bernard et al., 2014; Pack et al., 2016; Reydon et al., 2020) ; building better land governance to avoid land speculation-induced deforestation ; and strengthening the detection of illegal deforestation (Bernard et al., 2014; Börner et al., 2015; Pack et al., 2016).

3 Limits and opportunities for future research

PADDD events relations to conservation-development trade-offs in the Brazilian Amazon, in terms of decisions and impacts is an extended research question that we answered to some extent. Therefore, this thesis has several boundaries that can guide future research, and reinforce their policy implications and consistent application in decision-making. Below, we emphasize several issues that remain yet to be addressed.

3.1 Accounting for various PADDD types and cause, contexts and methods

It is likely that neither all types nor causes of PADDD events had the same relations with conservation development trade-offs, or the same impacts, in terms of both deforestation and income distribution. In chapter 1 and chapter 2, we only consider events that we expected to actually reduce protection (i.e. degazettement and downsizing). Indeed, the literature on PAs impact do not agree on the relation between PAs effectiveness and PAs types (see e.g. Jusys (2018) and Pfaff et al. (2014)) for various impacts of sustainable use PAs). Further research should specially assess downgrading, in terms of both risks and impacts, especially since they represent a large share of total PADDD events (Cook et al., 2017; Golden Kroner et al., 2016). For example, the economic and conservation outcomes of PADDD events depend on the type of activities that are developed when a PA is downgraded (Cook et al., 2017; Naughton-Treves and Holland, 2019). Future research could provide useful insights on the differences between degazettement and downsizing, in terms of both risks and impacts. For example, factors such as the political acceptability of the event might shape the choice between degazettement and downsizing when the conservation costs of PADDD events are high. In addition, as stated in Mascia et al. (2014) and Tesfaw et al. (2018), PADDD events can occur for a large variety of reasons, which should also be the focus of future work. PADDD events that accommodate large-scale infrastructure development are likely not influenced by the same bargaining process, nor have the same impacts on deforestation and income distribution than those enacted for rural settlements.

Except in Brazil, few research have assessed the underlying factors related to the dynamic of the PA network. Existing efforts to broaden documentations on PADDD events will surely help further research to be engaged and to disentangle specific country level factors (Qin et al., 2019; Symes et al., 2016) (see e.g. Cook et al. (2017) for Australia ; Qin et al. (2019) for several iconic PAs in the USA, Oman, Ecuador, and the Democratic Republic of Congo ; and Golden Kroner et al. (2019) for several developing and emerging countries). Research on their impacts, following examples from Forrest et al. (2015) and Golden Kroner et al. (2016), while taking selection bias into account, should also be developed.

The impermanence of PAs also raise the same questions regarding other conservation instruments. Except for PES, for which permanence has already been raised (Engel et al., 2008; Velly et al., 2017), this issue has been overlooked regarding marine PAs (Albrecht et al., 2020), indigenous lands or community based forest management for example (Forrest et al., 2015; Golden Kroner et al., 2016; Qin et al., 2019; Tesfaw et al., 2018).

3.2 Accounting for changes in federal objectives

PADDD events can be either proposed or enacted, with variable time between the two decisions, which can influence both their risk factors and their impacts (Mascia et al., 2014; Pack et al., 2016). Some proposed PADDD events may never be enacted because they are forgotten or rejected while other ones can be enacted several years later or directly (Pack et al., 2016). The likelihood that a proposal become enacted, and the time between both decisions could be related to time-varying economic pressures (e.g. an increase of deforestation following PADDD proposals may facilitate their enactment) or to changes in the views and in the bargaining between agencies (Pack et al., 2016; Tesfaw et al., 2018), for example due to electoral reasons (Pailler, 2018). However, it is common to not have proposals data for some enacted PADDD events. In chapter 1, we use both types of events, since they both represent an intention to reduce protection. Our work is static and only focuses on lands heterogeneities and PA characteristics. Future research, along with a better tracking of PADDD proposals and enactments could help to acknowledge these questions. Panel data and duration models applied to political business cycles (Pailler, 2018) seem particularly suited to such work.

Beyond the issue of proposed versus enacted PADDD events, federal and states governments can change their views toward development or conservation objectives (Tesfaw et al., 2018), as evidenced by 2004, 2008 and 2012, which corresponds to turning point dates in federal objectives (Assunção et al., 2015; West and Fearnside, 2021). As a result, PAs effectiveness as well as the degree of economic pressures, that are often measured using roads, can greatly evolve (Nascimento et al., 2021; Pfaff et al., 2015). While, in chapter 1, we considered that a summary of economic pressures and PAs characteristics before 2006 influenced decisions after that date, in chapter 2, we only focused on 2009-2012 PADDD events, which helps to be consistent in federal policies. In chapter 3, however, we considered all PADDD events occurring between 2001 and 2010. Methods based on panel data, or matching and difference-in-differences estimations for several distinct periods, with enough observations and a visualization of time-trend for pre-deforestation and income data would be needed to extend research.

3.3 Beyond deforestation and income distribution

Beyond the definition of forests, which has already been discussed, and could also be the subject of additional research (Chazdon et al., 2016), we did not assess long-term deforestation rates, nor do we assess the issue of forest degradation. Deforestation and forest degradation could increase years after PADDD events enactment because of associated infrastructure development, or if actual project development did not start at the exact time of their enactments (Pack et al., 2016; Tesfaw et al., 2018). In addition, it is worth to notice that spillover effects might also occur following PADDD events. PADDD events may act as a signal to foster economic development and attract agents seeking to deforest (Tesfaw et al., 2018), or to reinforce economic pressures around other conservation instruments to lower the conservation costs of PADDD events. Future research needs to investigate on the type of spillovers effects that could follow PADDD events and assess their effects on deforestation and forest degradation.

Beyond deforestation and GhG emissions, other ecosystem services are of utmost importance and are part of international agreements, such as the conservation of biodiversity. Except Golden Kroner et al. (2016), no paper has yet assessed the influence of PADDD events on landscape fragmentation. Even though PADDD events only concern small PAs, part of PAs, or have a limited influence on deforestation, ecological corridors might be threatened (Cook et al., 2017; Tesfaw et al., 2018), which may jeopardize the maintenance of biodiversity. This could be done while controlling for selection biases, using empirical techniques such as done in Sims (2014) regarding PAs in Thailand.

Finally, our measure of local economic development is narrow. First, we use income distribution to assess inequalities following PADDD events, since they may have various effects on households depending on their types and causes. Our measures have several limitations since we do not precisely access households' income at the grid level. The lack of yearly demographic data does not allow us to assess accurately how each income class have grown following PADDD events. Future research may measure inequalities at an aggregate level to get more precisions. Second, we only imperfectly assess one monetary dimension of well-being. Many other factors influence households' livelihoods and participate in diminishing poverty, like material assets, social networks, health, or education, among others (Woodhouse et al., 2015). Future research would need to build multidimensional index to analyse whether livelihoods improved (see e.g. Guedes et al. (2012)) and to involve local communities or conduct interviews with stakeholders to better acknowledge their experience of PADDD events and perceived well-being.

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Abstract

Protected Areas (PAs) have been a key conservation tool in the Brazilian Amazon region, which has an important role in mitigating climate change while facing strong development challenges. However, conservation-development conflicts over land-uses have led PAs to lack effectiveness, either because of isolated location (location bias), or because of ineffective enforcement. Besides, PAs deter economic development when they set aside land, which have led to an acceleration of PA downgrading, downsizing and degazettement (PADDD) over the last decades. The objective of this thesis is to examine the relation between PADDD events and conservation-development trade-offs in the Brazilian Amazon according to PAs' and landscape's heterogeneities. I evaluate their relation first, in terms of decisions and second, in terms of impacts on deforestation and on economic development. In the first chapter, we assess drivers of PADDD events by examining how conservation-development interactions induce PA size reductions. Between 2006 and 2015, they were more common near cities, which reflects the influence of development agencies. Indeed, PA size reductions could generate high environmental costs since PAs were less deforested. PA size reductions were also more likely when they were already deforested and costly to manage, reflecting the influence of environmental agencies. In the second chapter, we assess the impact of 2009-2012 PA size reductions on 2010-2015 deforestation using matching strategies according to economic pressure in the landscape and past PAs effectiveness. Our results are consistent with what we expected conceptually: PA size reductions had no impact when PAs did not faced or did not constraint economic pressures, and raised deforestation when PAs blocked economic pressures to some extent. In the third chapter, we evaluate the effect of 2001-2010 PADDD events on income distribution and inequalities. We use matching and difference-in-differences estimations according to economic pressures in the landscape and to PADDD events' effect on the final size of protection. Where economic pressures were high, new economic opportunities that followed strict protection reductions have contributed to the growth of the higher-middle income class. Far from economic pressures, we found a decrease in inequalities when PADDD events increased protection size, probably due to better PA management or the development of tourism activities. This thesis calls for more robust PAs analysis taking into account their impermanence and landscape heterogeneities. These results could help to allocate scarce resources to consolidate the protected area network where it would be most effective and to choose more optimal sites and management types for new PA designations.

Keywords: Protected Areas, PADDD, conservation, deforestation, development, inequalities, impact evaluation

Résumé

Les aires protégées (AP) sont un outil de conservation essentiel dans l'Amazonie brésilienne qui est une région clé de la lutte contre le changement climatique et faisant face à de nombreux enjeux de développement économique. Les conflits entre objectifs de conservation et de développement pour l'utilisation des terres ont toutefois diminué l'efficacité des AP, soit du fait de leur isolation (biais de localisation), soit car leurs frontières n'étaient pas suffisamment défendues. En outre, le fait que les aires protégées puissent décourager le développement économique a conduit à une accélération des déclassements, réductions et suppressions des AP (PADDD) dans les dernières décennies. L'objectif de cette thèse est d'examiner la relation entre les événements PADDD et les arbitrages entre objectifs de conservation de l'environnement et de développement économique en Amazonie brésilienne, selon les caractéristiques des AP et l'hétérogénéité des terres. J'évalue cette relation, tout d'abord en termes de décisions et ensuite, en termes d'impacts sur la déforestation et sur le développement économique. Dans le premier chapitre, nous évaluons les moteurs des événements PADDD en examinant comment les interactions conservation-développement induisent une réduction de la taille des AP. De 2006 à 2015, les réductions de taille d'AP étaient plus courantes près des villes, reflétant l'influence des agences de développement. Les AP ayant été moins déboisées, les réduire en taille pouvait générer des coûts environnementaux élevés. Les réductions de taille d'AP étaient également plus courantes lorsque les AP étaient déjà déboisées et coûteuses à gérer, reflétant l'influence des agences environnementales. Dans le deuxième chapitre, nous évaluons l'impact des réductions de taille d'AP de 2009-2012 sur la déforestation de 2010-2015, en utilisant des stratégies d'appariement en fonction de la pression économique exercée sur les terres et de l'efficacité antérieure des AP. Nos résultats sont conformes à notre cadre conceptuel : les réductions de taille d'AP n'ont pas eu d'impact lorsqu'elles n'avaient pas fait face à des pressions économiques ou lorsqu'elles ne les avaient pas contrainte, et ont augmenté la déforestation lorsqu'elles limitaient les pressions économiques dans une certaine mesure. Dans le troisième chapitre, nous évaluons l'effet des événements PADDD de 2001 à 2010 sur la distribution des revenus et sur les inégalités. Nous utilisons des stratégies d'appariement et une estimation en double-différence en fonction de la pression économique exercée sur les terres et de l'effet des événements PADDD sur la taille finale de la protection. Là où les pressions économiques étaient fortes, les nouvelles opportunités économiques qui ont suivi les réductions de taille d'AP ont contribué à la croissance de la classe des revenus moyens-supérieurs. Loin des pressions économiques, les inégalités ont diminué lorsque les événements PADDD ont augmenté la taille de la protection, probablement en raison d'une meilleure gestion des aires protégées ou du développement d'activités touristiques. Cette thèse appelle à une analyse plus robuste des AP en tenant compte de leur instabilité et de l'hétérogénéité des paysages. Ces résultats pourraient aider à allouer des ressources rares pour consolider le réseau d'AP là où il serait le plus efficace et à choisir de meilleurs sites et types de gestion lors de la mise en place d'AP.

Mots clés : Aires Protégées, PADDD, conservation, déforestation, développement, inégalités, évaluation d'impact