UNIVERSITY OF SÃO PAULO
ESCOLA POLITÉCNICA

Juliana Siqueira-Gay

From intensive land use to fragmented landscapes: perspectives on cumulative impacts of mining on forests in the Brazilian Amazon

São Paulo
2021

JULIANA SIQUEIRA-GAY

From intensive land use to fragmented landscapes: perspectives on cumulative impacts of mining on forests in the Brazilian Amazon

Versão Corrigida

Ph.D. Thesis presented to the Graduate Program of Minerals Engineering at Escola Politécnica at University of São Paulo to obtain the degree of Doctor of Science.

Concentration area: Minerals engineering

Advisors:
Professor Dr. Luis Enrique Sánchez
Dr Laura Sonter

São Paulo
2021

Este exemplar foi revisado e corrigido em relação à versão original, sob responsabilidade única do autor e com a anuência de seu orientador.

São Paulo, 05 de julho de 2021

Assinatura do autor: Juliana S.

Assinatura do orientador: L Sánchez

Catalogação-na-publicação

Siqueira-Gay, Juliana
From intensive land use to fragmented landscapes: perspectives on cumulative impacts of mining on forests in the Brazilian Amazon / J. Siqueira-Gay -- versão corr. -- São Paulo, 2021.
198 p.

Tese (Doutorado) - Escola Politécnica da Universidade de São Paulo. Departamento de Engenharia de Minas e de Petróleo.

1.Desmatamento 2.Mineração 3.Avaliação de impactos ambientais 4.Avaliação de impactos cumulativos 5.Conservação I.Universidade de São Paulo. Escola Politécnica. Departamento de Engenharia de Minas e de Petróleo II.t.

Folha de avaliação

Siqueira-Gay, Juliana. **From intensive land use to fragmented landscapes: perspectives on cumulative impacts of mining on forests in the Brazilian Amazon**. 2021. 198p. Tese (Doutorado em Engenharia Mineral) – Escola Politécnica, Universidade de São Paulo, São Paulo, 2021.

Aprovado em: 25.05.2021

Banca Examinadora

Prof. Dr. ____

Instituição: ____

Julgamento: ____

Prof. Dr. ____

Instituição: ____

Julgamento: ____

Prof. Dr. ____

Instituição: ____

Julgamento: ____

Prof. Dr. ____

Instituição: ____

Julgamento: ____

Prof. Dr. ____

Instituição: ____

Julgamento: ____

A meu amado companheiro Alfredo Gay Neto

Acknowledgements

Thank you to Luis Enrique Sánchez for supervising me in the last years and for sharing his vast knowledge about impact assessment. I would like to express my gratitude to Laura Sonter for being a great mentor by inspiring, guiding, and providing me all kinds of support I needed.

I also wish to express my appreciation to the examination board of my qualifying exam for all constructive critics that helped me a lot in the second stage of the thesis development. Thank you to Pedro Walfir Souza-Filho for receiving me at Instituto Tecnológico Vale and teaching me about remote sensing. Many thanks to Amarilis Lucia Casteli Figueiredo Gallardo for comforting me in the difficult moments of this thesis and for the many professional opportunities she kindly invited me.

Many thanks to all my collaborators for all their support and great talks about the topics addressed in this thesis. Especially to Jeremy Simmonds and Jean Paul Metzger for sharing their expertise in very nice discussions about conservation planning. Thanks to the group of early carrier researchers of the São Paulo School of Advanced Science on Scenarios and Modelling on Biodiversity and Ecosystem Services to Support Human Well-Being for the great discussions and opportunity to work together.

Thank you to Ana Dibo for the friendship and fun moments over the last years. Thank you to Carla Grigoletto Duarte for inspiring me since I was in my scientific initiation and for the opportunities we have to work together. Thank you to Tais Fernandes for the good moments, supportive talks, and shelter during my time in Belo Horizonte. Thank you to Josianne Rosa for reviewing the early draft of the thesis project and to Barbara Souza for the companionship as a Ph.D. student over the last years.

My warm thanks to each person who made my lonely days better during my daily life at USP and Queensland. My gratitude to my family for always being there for me and stimulating me to go further. Thank you to Alfredo Junior and Mara, for always encouraging me and to Nilce Siqueira, for being a great example of resilience and supporting me in the most important steps of my life. Especially thank you to Alfredo, for inspiring me and for being by my side in all good and bad moments.

This study was financed in part by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior - Brasil (CAPES) - Finance Code 001.

Resumo

Siqueira-Gay, Juliana. **From intensive land use to fragmented landscapes: perspectives on cumulative impacts of mining on forests in the Brazilian Amazon**. 2021. 189p. Tese (Doutorado em Engenharia Mineral) – Escola Politécnica, Universidade de São Paulo, São Paulo, 2021.

A mineração causa impactos cumulativos nas florestas não apenas pelo desmatamento direto da construção de suas instalações, mas também por meio da construção de infraestrutura associada, como linhas de transmissão, estradas de acesso, estradas de ferro, atraindo mão de obra e expandindo núcleos urbanos. Esses impactos se estendem para além das concessões de lavra, causando consequências em toda paisagem. Propostas para expansão da mineração na Amazonia brasileira, que incluem iniciativas de redução, recategorização e extinção de unidades de conservação (PADDD em inglês), têm sido discutidas dada a importância dos ecossistemas a serem afetadas e sua relevância para conservação da biodiversidade. Esta tese tem como objetivo investigar os impactos cumulativos da mineração e infraestrutura associada nas florestas e testar a influência das áreas protegidas nesses impactos na Amazônia brasileira. Impactos cumulativos da mineração nas florestas foram investigados por meio da análise histórica de mudanças de uso e cobertura da terra e calibração de modelo de autômatos celulares para simular diferentes cenários de PADDD em áreas sob pressão para desenvolvimento de projetos de mineração. Como resultados, evidenciou-se que as mudanças de uso e cobertura da terra que desencadeiam impactos cumulativos podem variar de acordo com a indústria relacionada e infraestrutura necessária para implantação e operação do projeto, causando alterações na paisagem como a fragmentação de habitats. As florestas a serem afetadas protegem ecossistemas únicos que provêm diversos benefícios para as comunidades em escala local, regional e global, evidenciando a necessidade de se avaliar de forma abrangente os impactos resultantes do desenvolvimento mineral em regiões com alta biodiversidade. Essa pesquisa propõe cinco principais recomendações para avaliação dos impactos cumulativos nas florestas em áreas de mineração: (i) a determinação dos limites espaciais deve considerar a dinâmica do uso da terra em toda a paisagem; (ii) os limites temporais devem contabilizar taxas de mudanças que influenciam na interação e nos processos de cumulatividade dos impactos; (iii) a relação causal dos impactos deve considerar a importância das estradas como fonte de impactos bem como demais efeitos da perda de floresta sobre a biodiversidade e os serviços ecossistêmicos; (iv) a avaliação da importância dos impactos deve considerar adequadamente a relevância dos impactos diretos e indiretos; (v) impactos cumulativos nas florestas podem ser mitigados por meio da proteção da paisagem e otimização do desenho de estradas. Negligenciar impactos cumulativos sobre as florestas em regiões de mineração afetaria permanentemente os ecossistemas e os serviços que eles provêm.

Palavras-chave: Mineração; Avaliação de Impactos Ambientais; Cenários; Análise espacial; Fragmentação de habitat, Amazônia; Áreas protegidas; Unidades de conservação; Terras indígenas.

Abstract

Siqueira-Gay, Juliana. **From intensive land use to fragmented landscapes: perspectives on cumulative impacts of mining on forests in the Brazilian Amazon**. 2021. 189p. Tese (Doutorado em Engenharia Mineral) – Escola Politécnica, Universidade de São Paulo, São Paulo, 2021.

Mining entails cumulative impacts on forests not only by directly clearing land for minerals facilities construction, but also by building associated infrastructure, such as transmission lines, access roads, railways, pipelines, and terminals to transport ore, as well as attracting workforce and expanding urban areas. The impacts extend far from the leases, causing landscape-wide consequences. Proposals to expand mining in the Brazilian Amazon, including initiatives of protected areas downgrading, downsizing and degazettement (PADDD), have been discussed given the importance of the potentially affected ecosystems and their relevance for biodiversity conservation. This thesis aims at investigating the cumulative impacts of mining and associated infrastructure on forests and testing the influence of protected areas on these impacts in the Brazilian Amazon. As results, it was found that these impacts were investigated by analyzing historic land use and cover changes and calibrating a cellular automata model to simulate scenarios of PADDD in regions under pressure for development. The interplay of land use and land cover changes leading to cumulative impacts varied according to the associated industry and infrastructure required for the mining operation. Not only the extent of the cumulative impacts is relevant, but further effects in the landscape, such as habitat fragmentation. The affected forests protect unique ecosystems and provide benefits for communities at the local, regional, and global scale, evidencing the need to comprehensively assessing the long-lasting and difficult to mitigate impacts in mineral-rich areas. This research unfolded five main recommendations for addressing cumulative impacts on forests in mining regions: (i) spatial boundaries determination should tackle the dynamic of land use and land cover change across the landscape; (ii) temporal boundaries should undertake rates of changes influencing interaction and accumulation of impacts; (iii) causal relation of impacts should take into account the importance of roads as source for impacts and further effects of forest loss on biodiversity and ecosystem services; (iv) the impact evaluation should properly consider the relevance of direct and indirect impacts; (v) protecting the landscape and optimizing roads design are opportunities for impacts mitigation. Neglecting the cumulative impacts on forests in mining regions would perversely affect ecosystems and the services they provide.

Keywords: Mining; Environmental and Social Impact Assessment; Scenarios; Spatial analysis; Renca; Amazon forest; Protected Areas; Conservation Units; Indigenous lands.

List of figures

List of tables

List of abbreviations

ALOS	Advanced Land Observing Satellite
AMN	National Mining Agency
ANM	Agencia nacional de mineração
BAU	Business as usual scenario
CEA	Cumulative Effects Assessement
CPRM	Companhia de Pesquisa de Recursos Minerais
EIA	Environmental Impact Assessments
EIS	Environmental Impact Studies
FP	Full protection scenario
FPIC	Free, Prior and Informed Consent
FullDev	Full development scenario
FUNAI	Fundação Nacional do Índio
GEOBIA	Geographic Object-Based Image Analysis
IL	Indigenous lands scenario
INPE	National Institute for Space Research
LULC	Land use and land cover
MNN	Mean Nearest Neighbor
MPE	Edge density
MPS	Mean Patch Size
NGOs	Non governamental organizations
NumP	Number of patches
PADDD	Protected areas downgrading, downsizing and degazetting
PALSAR/PALSAR-2	Phased L-band Synthetic Aperture Radar
PAs	Protected areas
PL	Projeto de lei
RAISG	Amazon geo-referenced socioenvironmental information network
Renca	Reserva Nacional de Cobre e Associados
RESEX	Reserva Extrativista
SAR	Synthetic Aperture Radar
SNUC	Sistema Nacional de Unidades de Conservação
SRTM	Shuttle Radar Topography Mission
SU	Sustainable use scenario
SU & IL	Sustainable use and indigenous lands scenarios
UNHRC	United Nations Human Rights Council

List of publications during candidature

*Publications resulted from this thesis

Peer-reviewed articles

Siqueira-Gay, J., Giannotti, M. & Sester, M. Learning about spatial inequalities: Capturing the heterogeneity in the urban environment. *Journal of Cleaner Production* 237, 117732 (2019).

Siqueira-Gay, J., Gallardo, A. L. C. F. & Giannotti, M. Integrating socio-environmental spatial information to support housing plans. *Cities* 91, 106–115 (2019).

Athayde, S., Duarte, C.G., Gallardo, A.L.C.F., Moretto, E.M., Sangoi, L., Dibo, A.P.A., **Siqueira-Gay, J.**, Sánchez, L.E. Improving policies and instruments to address cumulative impacts of small hydropower in the Amazon. *Energy Policy* 132, 265–271 (2019).

Siqueira-Gay, J. & Sánchez, L. E. Mainstreaming environmental issues into housing plans: the approach of Strategic Environmental Assessment. *Environmental Impact Assessment Review* 77, 145–153 (2019).

Siqueira-Gay, J., Yanai, A.M., Lessmann, J., Pessôa, A.C.M., Borja, D., Canova, M., Borges, R.C. Pathways to positive scenarios for the Amazon forest in Pará state, Brazil. *Biota Neotropica* 20, e20190905. (2020).

***Siqueira-Gay, J.**, Sonter, L. J. & Sánchez, L. E. Exploring potential impacts of mining on forest loss and fragmentation within a biodiverse region of Brazil's northeastern Amazon. *Resources Policy* 67, 101662 (2020).

***Siqueira-Gay, J.** & Sánchez, L. E. Keep the Amazon niobium in the ground. *Environmental Science & Policy* 111, 1–6 (2020).

***Siqueira-Gay, J.**, Soares-Filho, B., Sánchez, L. E., Oviedo, A. & Sonter, L. J. Proposed Legislation to Mine Brazil's Indigenous Lands Will Threaten Amazon Forests and Their Valuable Ecosystem Services. *One Earth* 3, 356–362 (2020).

***Siqueira-Gay, J.**, Sánchez, L.E. The outbreak of illegal gold mining in the Brazilian Amazon boosts deforestation. *Regional Environmental Change*. 2, 28 (2021).

Souza, B. A.; Rosa, J. C.; **Siqueira-Gay, J.**; Sánchez, L. E. Mitigating impacts on ecosystem services requires more than biodiversity offsets. *Land Use Policy*. 105, 105393 (2021).

Rolo, D. A. M. O.; Gallardo, A. L. C. F.; Ribeiro, A. P.; **Siqueira-Gay, J**. Local Society Perception on Ecosystem Services as an Adaptation Strategy in Urban Stream Recovery Programs in the City of São Paulo, Brazil. *Environmental management*. (2021).

Unpublished articles included as chapters of this thesis

*Siqueira-Gay, J.; Santos, D.; Nascimento Jr, W. R.; Souza-Filho, P.W.; Sánchez, L.E. Investigating changes driving cumulative impacts on native vegetation in the Brazilian Amazon. *In review*

*Siqueira-Gay, J. Metzger, J.P.; Sánchez, L.E.; Sonter, L.J. Strategic planning needed to mitigate landscape-wide impacts of opening protected areas to mining. *In prep.*

Conference papers and abstracts

***Siqueira-Gay, J.**; Sánchez, L.E. Processes of land use and cover change in two mining regions: implications for conservation in the amazon forest. XIX Simpósio Brasileiro de Sensoriamento remoto. 2019.

***Siqueira-Gay, J.**; Sánchez, L.E. Assessing mining impacts on native vegetation. IAIA21 Smartening Impact Assessment in Challenging Times. 2021.

Table of contents

1 Introduction

1.1 Research problem

A cumulative impact[1] is “a change in the environment caused by multiple interactions among human activities and natural processes that accumulate across space and time” (CCME, 2014). Although a mine footprint may be relatively small (BRIDGE, 2004), and individual mining project and associated facilities have potential to cause major land use and land cover (LULC) changes and landscape-wide consequences (RAITER et al., 2014), affecting kilometers away from mining leases (SCHUELER; KUEMMERLE; SCHRÖDER, 2011; SONTER et al., 2017). Mining potentially cause cumulative impacts on forests by implementing and expanding its facilities and by constructing infrastructure required to process ore and to transport concentrate, equipment, and materials (WORLD BANK, 2019a).

Mining regions usually shelter high biological diversity (KOBAYASHI; WATANDO; KAKIMOTO, 2014; MURGUÍA; BRINGEZU; SCHALDACH, 2016), and threatened ecosystems (FERNANDES et al., 2018b; SOUZA-FILHO et al., 2019). These high biodiversity values may be affected by the direct, indirect, and cumulative clearing of forests resulted from facilitating new projects in these regions (SONTER et al., 2018). Moreover, these cumulative losses may act as a source for other impacts, in particular when projects require linear infrastructures, such as roads, railways, and pipelines, potentially causing habitat fragmentation[2] (JUNKER et al., 2015; MALAVIYA et al., 2010), an important drive of biodiversity loss (HADDAD et al., 2015; LINDENMAYER; FISCHER, 2006; PARDINI; NICHOLS; PÜTTKER, 2018).

Protected Areas (PAs) - a clearly defined geographical space, recognized, dedicated, and managed, through legal or other effective means, to achieve the long term conservation of nature associated with ecosystem services and cultural values (IUCN Definition 2008) - represent one of the most important strategies for conservation by setting aside areas to maintain species richness, ecological processes and functioning as well as to provide direct human benefits (DUDLEY;

---

[1] “Cumulative environmental change”, “cumulative effects” and “cumulative impacts” are synonymous in the literature of environmental impact assessment.

[2] Fragmentation is the process of landscape breaking apart through habitat loss producing an increased number of patches, decreased patch area, and increased isolation of patches (Fahrig, 2003).

PETER; SUE, 2013). The growing pressure for PAs downgrading, downsizing, and degazetting (PADDD) in Brazilian Amazon for natural resources extraction (BERNARD; PENNA; ARAÚJO, 2014; KRONER et al., 2019; NAUGHTON-TREVES; HOLLAND, 2019) and infrastructure projects (HYDE; BOHLMAN; VALLE, 2018; MASCIA et al., 2014) have already impacted biodiverse regions causing long-lasting impacts (RUDKE et al., 2020).

Environmental Impact Assessments (EIA) of mining projects, broadly used in support of decision-making about new projects in these sensible regions, often fail in considering cumulative impacts (NERI; DUPIN; SÁNCHEZ, 2016; SINGH et al., 2020) especially those on forests caused by new mining projects in remote and intact regions (WORLD BANK, 2019a). Although fragmentation is already identified in many Environmental Impact Studies (EIS) and related literature of linear infrastructure projects, such as powerlines (BIASOTTO; KINDEL, 2018; CARDOSO JÚNIOR; MAGRINI; DA HORA, 2014), little attention is given to fragmentation effects caused by mining and related activities (SINGH et al., 2020; WORLD BANK, 2019a).

The Brazilian Amazon is the site of numerous potential mining operations (ANM, 2020) and undergoing an array of policy changes set to weaken environmental regulation and mitigation requirements (EDWARDS; LAURANCE, 2015; FEARNSIDE, 2016). Initiatives of downgrading, downsizing and degazettement (PADDD) proposed in the last years (MASCIA et al., 2014) have been threatening conservation in high biodiverse areas by reducing restrictions to human activities insider PAs, decreasing their size, and removing legal protection. The Brazilian Amazon is one of the worldwide hotspots in this matter and PADDD is driven by initiatives for natural resource extraction, including infrastructure and mining (KRONER et al., 2019; NAUGHTON-TREVES; HOLLAND, 2019). Previous cases of PADDD led to increased deforestation and fragmentation due to road construction (KRONER; KRITHIVASAN; MASCIA, 2016) and mining activities (RUDKE et al., 2020), calling attention to the importance of discussing the implications of these policy changes on biodiversity and ecosystem services.

1.2 Hypotheses

Two hypotheses, here considered as facts grounding the investigation (MARTINS, 2009), are proposed to be tested:

1) Infrastructure associated with mining plays an important role in driving cumulative impacts on forests.

2) Facilitating mining and associated infrastructure within PAs cause cumulative impacts on forests.

1.3 Knowledge gaps

This thesis aims at contributing to fill the following knowledge gaps found in the literature:

Knowledge gap 1: How do mining and associated infrastructure drive cumulative impacts on forests

Few studies focused on understanding how an impact becomes cumulative (DUINKER et al., 2012). The spatial concentration of projects and, particularly in mining regions, the geological control of mineral deposits highlights the challenges related to assessing cumulative impacts in these areas (LECHNER et al., 2017). For better informing decision-making and integrated assessment, the investigation of the causality of cumulative impacts is needed (VOEGELI; HEDIGER; ROMERIO, 2019). Although the cumulative impacts of mining are already discussed in the literature (FRANKS; BRERETON; MORAN, 2013; NERI; DUPIN; SÁNCHEZ, 2016), few studies were found exploring causality and pathways analysis to better understand how the impact occurs, especially, their causes and potential interactions. Especially in the Amazon forest, some studies focused on impacts of linear structures, such as roads (SOARES-FILHO et al., 2004, 2006), power lines (HYDE; BOHLMAN; VALLE, 2018) as well as dams and hydropower projects (ATHAYDE et al., 2019; FEARNSIDE, 2014; LATRUBESSE et al., 2017). However, few conceptualize causality and quantify the cumulative impacts of mining on forests regionally as well as further effects in the landscape structure, especially considering biodiverse-rich areas.

Knowledge gap 2: How changes in protected areas influence the extent of cumulative impacts of mining and associated infrastructure on forests

Movements for PADDD have been causing major impacts worldwide (FORREST et al., 2015; KRONER; KRITHIVASAN; MASCIA, 2016). Especially in Amazon, resource extraction and infrastructure have been playing an important role in threatening PAs regulations (PACK et al., 2016). Although some studies explored the loss of biodiversity and ecosystem services in scenarios of mining projects and shared infrastructure (EVANS; KIESECKER, 2014; JUNKER et al., 2015;

RUNGE et al., 2017), few were found discussing scenarios in a mining region with high biodiversity values covered by PAs. Although the effects of a mineral transportation infrastructure are assessed in terms of biodiversity (RUNGE et al., 2017), few studies focused on testing the role played by PAs on cumulative impacts on forests and discussing further consequences for EIAs and conservation strategies.

1.4 Research questions

This research addresses the two abovementioned knowledge gaps by seeking to respond to five research questions:

Questions related to hypothesis 1 and knowledge gap 1:

1. What are the most important ways mining and associated infrastructure lead to cumulative impacts on forests?
2. What are the main land use and land cover changes driving cumulative impacts on forests?
3. Which cumulative impacts on forest mining and its associated infrastructure could cause?

Questions related to hypothesis 2 and knowledge gap 2:

4. What is the role played by PAs in mitigating the cumulative impacts of mining and associated infrastructure on forests and biodiversity?
5. What are the cumulative impacts on forest and ecosystem services related to the policy change to expand mining into indigenous lands?

1.5 Objectives

This thesis aims at investigating the cumulative impacts of mining and associated infrastructure on forests and testing how PAs influences these impacts in the Brazilian Amazon.

The specific objectives related to hypothesis 1 and knowledge gap 1 are:

1) Conceptualize how infrastructure associated with mining drives cumulative impacts on forests
2) Identify the land use and cover changes leading to cumulative impacts in mining regions
3) Quantify the forest loss and fragmentation drove by mining and associated infrastructure

The specific objectives related to hypothesis 2 and knowledge gap 2 are:

4) Analyze the cumulative impacts on forests and biodiversity caused by facilitating mining and associated infrastructure inside PAs boundaries in different scenarios
5) Evaluate the cumulative impacts on forests and ecosystem services derived from policy change into an entire category of PA

1.6 Thesis framework

To summarize the main hypotheses, knowledge gaps, objectives, research questions and main findings, a thesis framework is presented (Figure 1).

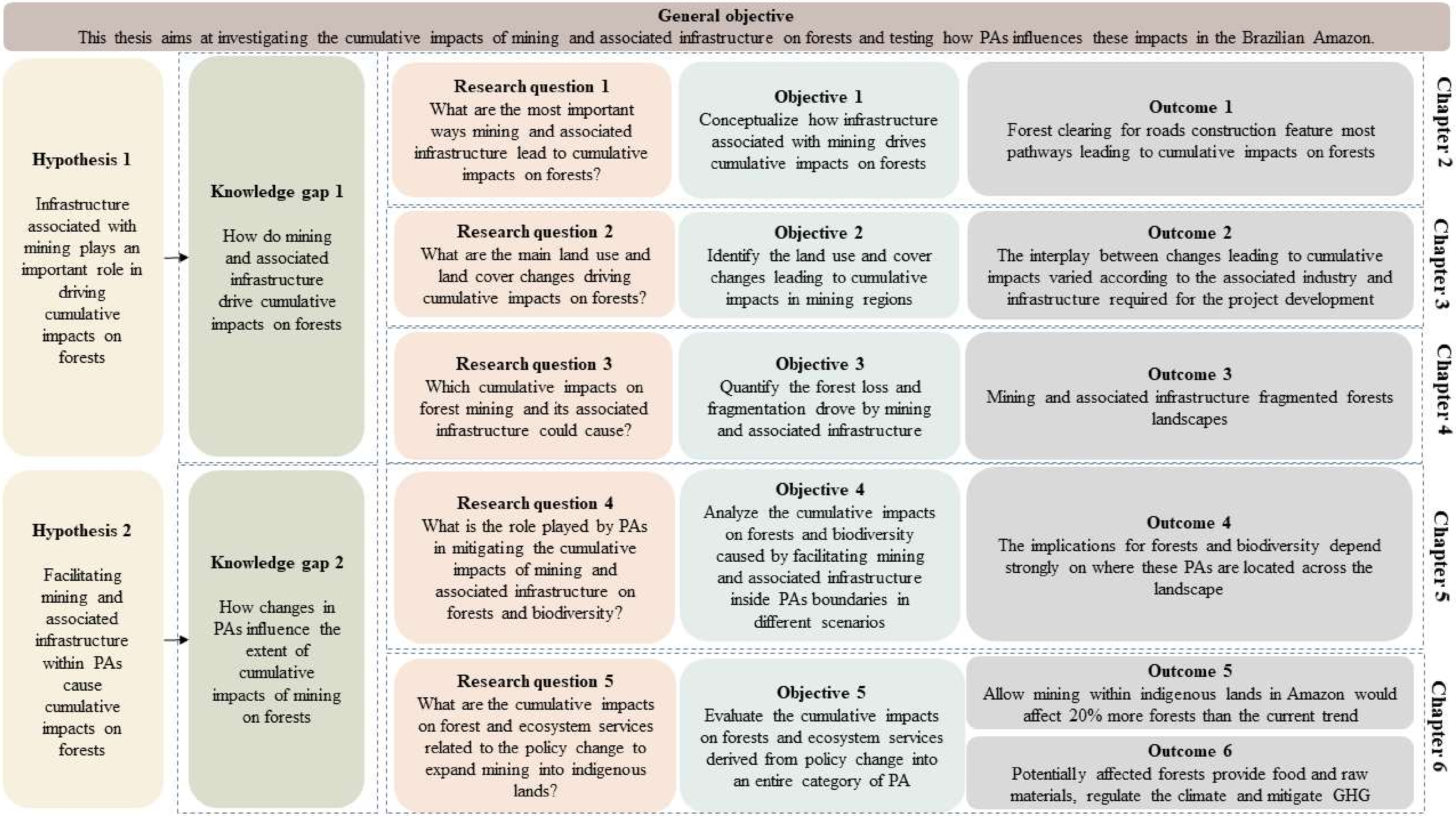


Figure 1. Thesis framework

1.7 Thesis outline

This thesis is divided into eight chapters that resulted from three already peer-reviewed articles, one submitted and one in preparation. One peer-reviewed article related to results of this thesis was added in appendix 5. This first introductory chapter, describes the general research problem and the main hypotheses, knowledge gaps, research objectives, and questions as well as presenting the thesis structure. Chapters 2, 3, 4, 5, and 6 bring analysis and results. Chapter 7 present the results discussion and Chapter 8 the conclusion.

Chapter 2, entitled "Conceptualizing cumulative impacts on forests", aims at understanding the direct, indirect, and cumulative impacts resulting from a mining project in a tropical region. It contributed to a better understanding of these relations and evidenced the causality of impacts. The direct impacts are those caused by the project activities implemented by the developer or that can be controlled by the developer (SÁNCHEZ, 2020), such as clearing native vegetation for opening the pit and construction of processing facilities (IFC, 2012) (Figure 2 A). The indirect impacts are those related to associated infrastructure required to support the project operation but are caused by third parties (IFC, 2012), such as road constructions and urban areas that grow to shelter workforce and people attracted by the mining project (Figure 2 B). These impacts are also named secondary impacts, which are those that resulted from a direct impact (COOPER, 2004). The direct and indirect impacts interact and the entire landscape may be affected by cumulative forest loss and further changes in the landscape structure. The cumulative forest loss is a source for other impacts (NOBLE, 2015), affecting biodiversity and ecosystem services.

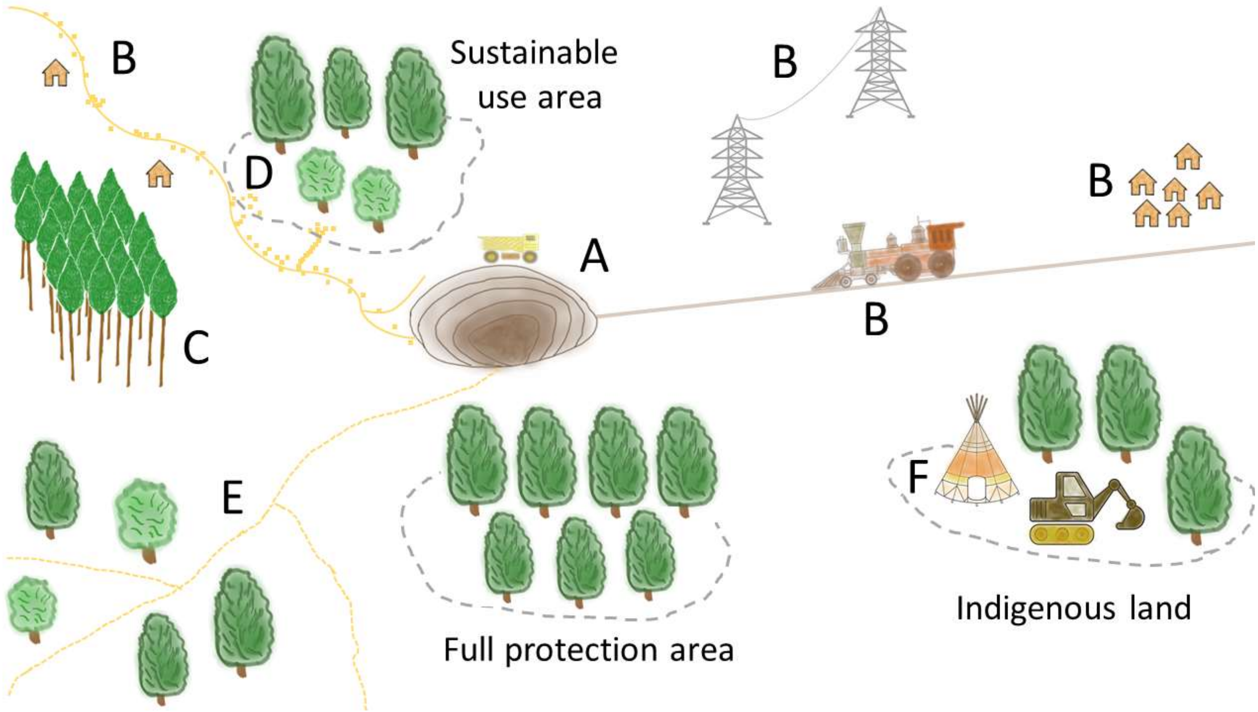


Figure 2. Hypothetical landscape showing the general aspects of mining in tropical mining regions addressed in this thesis. (A) Open pit representing the direct impacts of forest clearing; (B) Access roads, transportation infrastructure, transmission lines and urban settlements to show the indirect impacts; (C) Silviculture expansion as a relevant transition of interest in a mining region; (D) sustainable use PA impacted by deforestation in a mining region; (E) new roads construction to access mining deposits impacting forests; (F) Mining within indigenous lands.

Chapters 3, 4 and 5 are focused on analyzing a region covered by PAs and mining projects in the Brazilian Amazon. Chapter 3 entitled "Land use and land cover changes driving cumulative impacts" map historic LULC changes to identify the main drivers of change[3] and their spatial relevance. Silviculture growth (Figure 2 C) cleared more forests than pastureland expansion when associated with kaolin mining and pulp mill activities. In contrast, in regions with gold and iron ore activities, pastureland expansion was more relevant, clearing areas surrounding new roads. The

[3] Drivers of change can be divided into two groups: (i) the biophysical, including processes of the natural environment, such as weather and climate, plant succession, and volcanic eruptions; (ii) socio-economic drivers, comprising the demographic, social, economic, political and institutional factors and, processes such as population change, the market, and various public sectors and related policies and rules (Briassoulis, 2000).

indirect impacts, such as those associated with pastureland expansion surrounding roads, clear more forests than direct impacts of mining and industrial footprints. Chapter 4, entitled “Exploring mining impacts of forest loss and fragmentation”, quantifies (i) the cumulative forest loss within PAs, (ii) the relevance of deforestation surrounding roads, (iii) effects on the landscape structure. Forest loss was identified within PAs of sustainable use (Figure 2 D), threatening the integrity of ecosystems in this region, and areas surrounding roads present the majority of deforestation in the region.

Chapter 5, entitled “The influence of protected areas on cumulative impacts of mining and associated infrastructure on forests”, models scenarios of mining and road development (Figure 2 E) considering initiatives of PADDD. The scenario proposal seeks to analyze the role of PAs in avoiding and minimizing impacts on forests and biodiversity by analyzing both, deforestation and fragmentation effects due to new road construction to reach mineral deposits that could be developed in the region. The implications for forests depend strongly on where these PAs are located across the landscape. Roads to reach few and remote mineral deposits would entail more deforestation within areas with high biological importance and fragmentation than connecting the existing road system to reach more deposits.

Chapter 6, entitled “Impacts of policy change to expand mining into indigenous lands”, features a quantitative investigation of cumulative impacts of forest and ecosystem services in the context of expanding mining over indigenous lands in the Legal Amazon (Figure 2 F). The forests to be potentially affected and the monetary losses of the loss of 4 ecosystem services were estimated. If a legislative proposal to facilitate mining indigenous lands is approved 20% more forests would be impacted, affecting the benefits they provide to people.

Chapter 7 brings the summary of the main findings and the discussion about how the results contribute to a broader literature with perspectives for addressing cumulative impacts on forests with implications for decision making in two perspectives: EIAs and conservation planning. Finally, the main conclusions, thesis limitations, and further research directions are presented in Chapter 8.

1.8 General methodology

Each chapter of this thesis presents a dedicated methodological approach. The motivation presented in all chapters was on studying areas covered by PAs under pressure for mining development. Two regions were studied in more detail, the niobium reserves at the Pico da Neblina National Park and the National Reserve of Copper and Associates (Renca), and the final chapter brings a general overview of one category of PA considering the entire Legal Amazon (Figure 3).

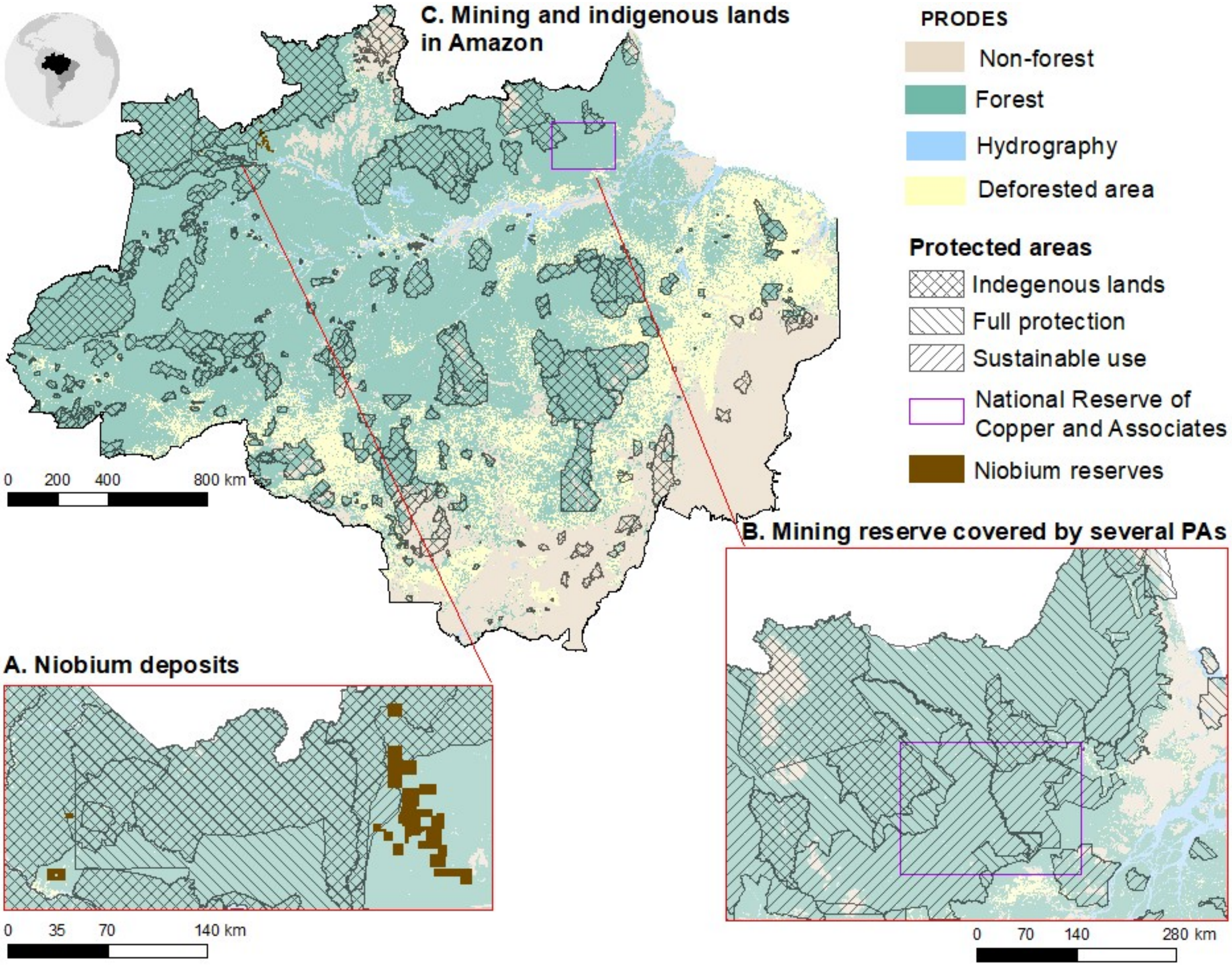


Figure 3. Areas used as reference for the studies presented in this thesis

A. Niobium deposits within PAs

The aim of this investigation was twofold: (i) quantify the forest extent to be affected in scenarios of mining development considering both the direct and indirect impacts in the areas surrounding the mineral deposits as well as the area to be impacted by roads; (ii) propose of a causal diagram to depict different actions of each scenario would affect forest loss and further impact biodiversity and ecosystem services.

B. Mining reserve covered by several PAs

Chapters 3, 4, and 5 are focused on the Renca region. Renca is a 47,000 km^2 mineral-rich area situated in Pará and Amapá States. Among recent changes in Brazilian environmental policies and ongoing initiatives to streamline regulations, the federal government moved in August 2017 to defreeze Renca. Facing considerable criticism from NGOs and academics about the devastating role that mining could play in a well-conserved area with high biodiversity value, making media headlines, and facing citizens' protests, one month after the announcement, the government reversed its decision. Although this proposal was eventually overturned, the federal government in office since January 2019 already stated the intention of extinguishing the prohibition of granting mineral leases inside Renca.

Chapter 3 classifies multisensory products to quantify the historic LULC and main drivers of change. Two regions are investigated and the main changes compared and discussed to shed light about cumulative impact assessment in mining regions.

Chapter 4 analyses historic changes and landscape effects due to mining projects close to Renca's borders using publicly available LULC maps. The results are divided into (i) forest loss inside PAs to calculate past pressure over these areas using historical maps of the region overlaid with PAs boundaries; (ii) forest loss surrounding the two main mining areas considering the historic maps in the region up to 70km far from the mineral deposits; (iii) forest loss close to roads by calculating the forest extent to be affected from 5 to 30km of existing roads; (iv) the effects in the landscape structure using four landscape metrics.

Chapter 5 present a spatially explicit model developed to simulate scenarios of future mining development given differences in PAs regulations. Three approaches are presented comparing five PADDD scenarios: (i) quantifying the mining and non-mining deforestation rates and relevance of impacted areas for biodiversity conservation; (ii) quantifying the influence of roads in the forest extent driving changes within PAs and biodiverse areas; (iii) quantifying fragmentation effects by using landscape metrics.

C. Mining and indigenous lands in Amazon

Chapter 6 present the investigation based on the extent of forests and ecosystem services that could be directly and indirectly affected by a policy change. It is grounded in land cover information of the entire Brazilian Amazon provided by Prodes dataset and the spatial explicit valuation evaluation of ecosystem services produced by Strand and colleagues (STRAND et al., 2018). The area to be affected was estimated using an upper limit of 70 km and a conservative value of 10 km of mineral deposits.

2 Conceptualizing cumulative impacts of mining on forests

Objective

This chapter aims at conceptualizing the cumulative impacts on forests, biodiversity and ecosystem services caused by mining and associated infrastructure.

Preface

This chapter investigates niobium rich areas in western Amazon located within an indigenous land and a full protection area. By exploring scenarios of development in the region, a causal diagram is proposed to better understand how mining has the potential to entail cumulative impacts on forest loss, integrating the main actions of each scenario and further consequences to biodiversity and ecosystem services. It revealed that, in a mining region, roads play an important role in most pathways leading to cumulative impacts by direct and indirect clearing and forest degradation. The cumulative forest loss is source of effects for biodiversity by directly removing habitats and entailing fragmentation, and for ecosystem services by clearing important areas responsible to provide regulation, provision and cultural ecosystem services.

This chapter supports the thesis (Figure 1) on the better understanding of the cumulative impacts on forests, including biodiversity and ecosystem services, in mining tropical regions by (i) identifying the major pathways affecting forests; (ii) analyzing the qualitative importance of each pathway; (iii) formalize the understanding of the impacts of forest loss on biodiversity and ecosystem services.

Related publication

Siqueira-Gay, J., Sánchez, L.E., 2020. Keep the Amazon niobium in the ground. Environmental Science & Policy 111, 1–6. https://doi.org/10.1016/j.envsci.2020.05.012

Abstract

Political pressure to expand mining to protected areas in near-pristine regions urges strategic-level anticipation of the impacts of infrastructure provision necessary to support mining development. Undeveloped niobium deposits in Northwestern Amazon were broadcasted as of primary interest of Brazilian politicians, what called attention to the risks to ecosystem conservation and indigenous people in this key biodiversity area. Given this current threat, we investigated different scenarios of mining expansion in the region and found that it would entail significant cumulative forest loss, affecting biodiversity and ecosystem services. If conciliation of niobium exploitation and conservation is possible, it would require strengthening the assessment of cumulative impacts under the current environmental impact assessment process by: (i) providing terms of reference with straight- forward requirements and criteria; (ii) produce a public database and standardize procedures for data acquisition; (iii) use and development of retrospective and prospective investigative protocols. These ingredients are lacking in the current political trend of weakening environmental legislation and governance, therefore Amazon niobium should be kept in the ground.

2.1 Introduction

PAs in the Brazilian Amazon are threatened by a movement for downgrading – permitting more activities inside their boundaries; downsizing – redesigning boundaries to reduce their area; and degazetting - weakening their protective status (BRAGAGNOLO et al., 2016; PACK et al., 2016), harming biodiversity conservation in the largest tropical forest in the world. It is known that extractives industries play an important role either in pushing changes in reserves boundaries (NAUGHTON-TREVES; HOLLAND, 2019) or in maintaining them (SOUZA-FILHO et al., 2016).

Considering the growing global demand for metals and industrial minerals (DURÁN; RAUCH; GASTON, 2013), the proximity of existing mines and mineral deposits to PAs is a matter of concern, due to evidences of: (i) deforestation both outside and inside PAs due to large scale projects (MURGUÍA; BRINGEZU; SCHALDACH, 2016) and artisanal gold mining (Asner and Tupayachi, 2016; Swenson et al., 2011); (ii) fragmentation (JUNKER et al., 2015; SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020); (iii) loss of heritage (REDONDO-VEGA et al., 2017) and;

(iv) pollutants discharge from small scale and artisanal gold mining (LOBO; COSTA; NOVO, 2015) and industrial mining (RUDKE et al., 2020).

In the context of weakening environmental legislation and enforcement in Brazil (PEREIRA et al., 2019), the impacts of opening up areas with large known and potential mineral deposits received great attention in the last years (VILLÉN-PÉREZ et al., 2017; WWF, 2017a), especially due to the cumulative impacts on forests that could result from future developments in near pristine regions (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020). In this investigation, we aim at analysing landscape-scale cumulative effects of forest loss resulting from potential development of unexploited rare earths and niobium (Nb) reserves inside a PA.

2.2 A fallacious narrative could trigger unprecedent devastation

Nb is a metal used essentially as an alloy, combined with iron in several industrial applications, such as aeronautics, aerospace, fabrication of pipelines and oil rigs (LIMA, 2010). It is relatively new to the world metals market, being used industrially since the 1960s and prized for lending to these products properties such as high strength and corrosion resistance.

Nb, however, is not the only metal that lends important properties to metal alloys. Tantalum, in particular, but also molybdenum, vanadium and tungsten have, to a certain extent, similar properties. Industries that use Fe-Nb have, therefore, options to substitute raw materials, what means that any substantial price increase could stimulate substitution, thus jeopardizing Nb market share. A number of mining projects are underway in different countries having Nb as a byproduct (PADILLA; GHALAYINI; PAPP, 2019), but not projects aimed primarily at developing Nb reserves.

Differently of most metals, Nb prices do not fluctuate heavily (SIMANDL et al., 2018), although they vary according to the demand and are generally privately negotiated between the seller and the buyer (BGS, 2011) and have been rising since 2006 (MACKAY; SIMANDL, 2014). Although there is a small number of producers (oligopolistic market), prices are largely determined by the balance between supply and demand, to the extent that substitute metals do not provide a cheaper alternative to consumers (SIMANDL et al., 2018).

Brazil concentrates 98% of world reserves of Nb and supplies 93% of the world market (ANM, 2019; USGS, 2019) with the output of its two Nb mines, the largest in Araxá, Minas Gerais and another in Catalão, Goiás. Global mine output for niobium grew 3.1 times in the last 20 years (as compared to 2.5 times for iron and 1.6 for copper, according to data compiled by the US

Geological Survey in the Mineral Commodity Summaries annual series (USGS, 2019) and the operating mine in Araxá has proven reserves of about 200 years at current production levels, according to reserves and production data published by the Brazilian National Agency of Mining (ANM, 2019).

Undeveloped reserves are known in a polymetallic mine located north of Manaus (ANM, 2019), but mineral resources are known in other places, one of the most publicized being the Seis Lagos carbonatite, in Northwestern Amazon (PHILLIPS, 2019). The huge reserves in Araxá, unparalleled for other mineral commodities, in an operating mine well served by infrastructure makes very hard that any greenfield project could easily prosper, especially if situated in a remote location.

In Brazil, however, Nb gained unexpected relevance due to politically motivated construction of myths about its near miraculous capacity of generating wealth, as voiced by President Bolsonaro in several social media appearances and in the 2019 G20 summit in Japan (PHILLIPS, 2019). Politicians' illiteracy about mineral economics feeds beliefs that Brazil could simultaneously promote a significant increase in the national production of Nb and achieve higher prices (MACKAY; SIMANDL, 2014), thus generating jobs and collecting more taxes (LIMA, 2010), even if at the expense of biodiversity.

Whilst developing these mineral deposits goes against the economic rationale of matching supply and demand of commodities in international markets, it is conceivable that political will could build a narrative "demonstrating" that opening up the region for mining is in the national interest, thus paving the way for subsidies and public investments in infrastructure that could have devastating consequences for biodiversity and indigenous peoples (WALKER et al., 2019).

Northwestern Amazon is a highly biodiverse region that provides important ecosystem services, such as climate regulation, raw materials and food (STRAND et al., 2018). The extent of the possible devastation is anticipated in the next section by exploring scenarios of infrastructure development to reach two areas with known deposits.

2.3 Potential futures for niobium exploitation in western amazon

2.3.1 The cumulative impacts on forests considering potential futures for niobium exploitation in Western Amazon

The Pico da Neblina Key Biodiversity Area overlaps both the Biological Reserve of Seis Lagos and the Balaio indigenous lands, both within a biodiverse and well-preserved region that encompasses other Nb geological prospects, the area in the north of Santa Isabel do Rio Negro (Figure 4).

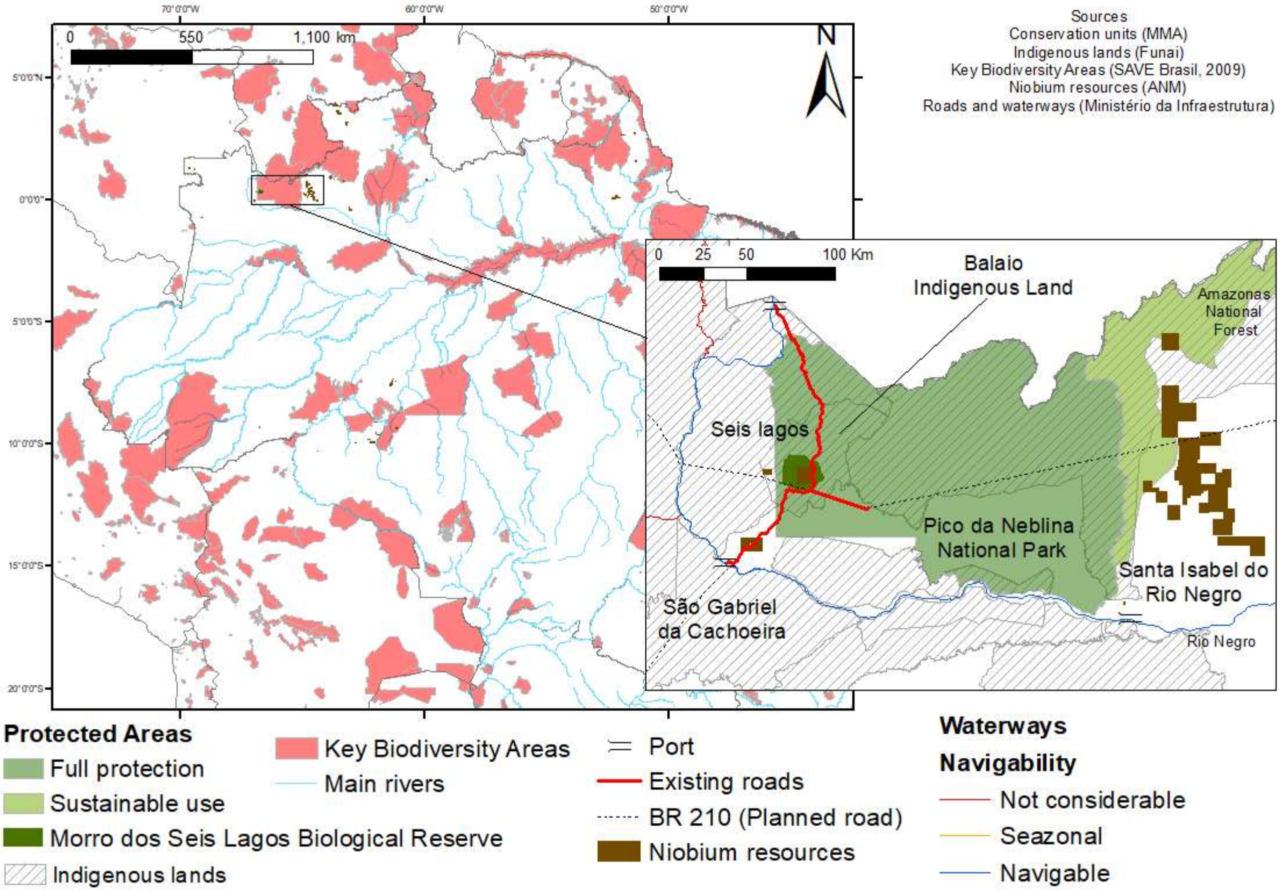
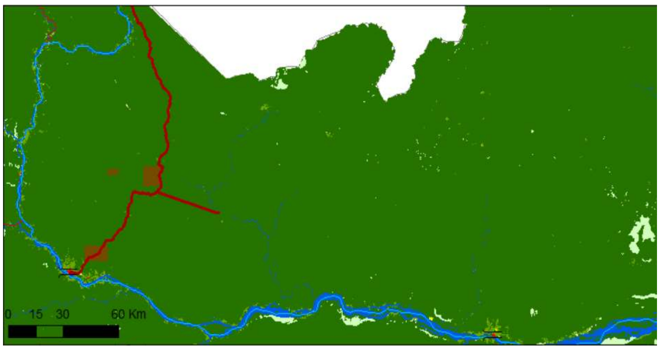
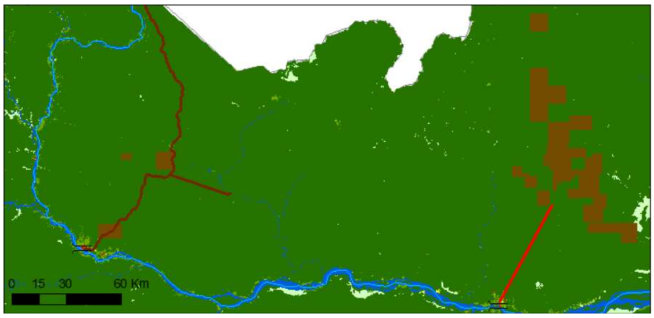
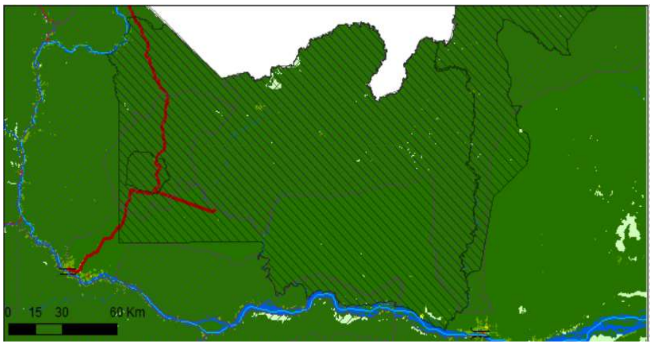


Figure 4. Location of niobium mineral deposits in relation to Key Biodiversity Areas and Protected Areas in the Brazilian Northwestern Amazon. Source: (SIQUEIRA-GAY; SÁNCHEZ, 2020)

The access and the provision of logistic infrastructure is a hurdle for commercial exploitation in the region (TAKEHARA, 2019). The BR- 210 road, known as “Perimetral Norte” (Figure 4), was planned during the military government to connect the northern portion of the states of Amazonas, Roraima, Pará and Amapá. However, the construction stopped in 1977 due to conflicts with the Waiãpi people in eastern state of Amapá. The alternative infrastructure to flow minerals is the small port in São Gabriel da Cachoeira, where the Rio Negro presents regular navigability (Figure 4). About 200 km to the east of Seis Lagos, in north of the village of Santa Isabel do Rio Negro, Nb exploration permits were granted in an area that partially overlaps another IL and lies in the vicinities of the Amazonas National Forest (Figure 4).

Based on the review of public documents and the current discussion in the literature about infrastructure in resources extraction regions (EVANS; KIESECKER, 2014; RUNGE et al., 2017), we develop four scenarios for Seis Lagos region (Figure 5) to explore: (i) possible changes in mining legislation in PAs (EL BIZRI et al., 2016); and (ii) different infrastructure arrangements to access the region based on roads construction and navigable river, important drivers of deforestation in Amazon (LAURANCE; GOOSEM; LAURANCE, 2009; WALKER et al., 2019). The geographical information in support of the different project arrangements, considering current and planned infrastructure, Nb mineral deposits and PAs is presented in Table S 1 in Appendix 1. To determine the area that could be potentially affected by significant deforestation caused by roads we assumed two possible realities according to Barber et al. (2014): (i) a 5.5 km buffer to estimate the effects considering all types of planned and unplanned roads; and (ii) a 32 km buffer to estimate the effects of large projects, such as highways. To estimate the area to be potentially affected by significant deforestation caused by mining we used a 70 km buffer around mining leases (SONTER et al., 2017). For more information about the affected areas see Figure S 1 in Appendix 1.

Scenarios	**Description**	
(A) Development without construction of new roads	Considering the possibility of changing the current regulation and allowing mining activities inside PAs or redrawing their boundaries, this scenario considers the use of existing infrastructure to access the deposits. As the existing road is unpaved, works are needed for its improvement to meet requirements for conveying materials and supplies to a hypothetical mine and to transport ore to the port in São Gabriel da Cachoeira, where a terminal should be built. Given that no new large infrastructure would be constructed, the	

	Santa Isabel do Rio Negro deposits remain unexplored in this scenario.	
(B) Construction of shared infrastructure	Assuming changes in regulation to allow mining in PAs, as in (A) above, this scenario considers improvement of the existing road and construction of the planned BR-210 connecting the Santa Isabel do Rio Negro and Seis Lagos deposits. This new road would serve to run off the production of Santa Isabel do Rio Negro sharing the road to a new terminal in the port of São Gabriel da Cachoeira.	
(C) Independent development	Assuming changes in regulation to allow mining in PAs, as in (A) and (B) above, and improving existing roads, this scenario considers the exploitation of the two deposits separately, requiring the construction of a new road connecting each hypothetical mine to ports to be constructed in Santa Isabel do Rio Negro.	
(D) Keep Amazon Niobium in the ground	This scenario considers no changes in the regulation to allow mining inside PAs, named Business as Usual (BAU) scenario. No new infrastructure construction and therefore, no development of Nb deposits.	

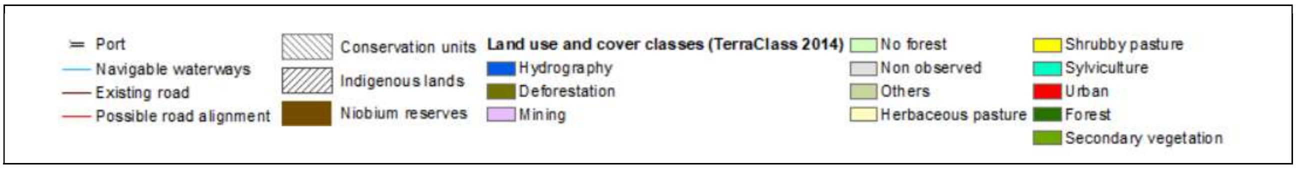


Figure 5. Scenarios of potential niobium exploitation and non-exploitation in Brazilian Northwestern Amazon.

Source: (SIQUEIRA-GAY; SÁNCHEZ, 2020)

Developing the Santa Isabel do Rio Negro deposits (B and C) would entail the loss of 60% more forest area than the area that would be affected by the sole development of the Seis Lagos deposit (A) (Figure 6). In scenario (D) no mining infrastructure would be constructed, and the PAs would block the effects of roads, evidencing the BAU scenario as representing the least forest extent to be affected. In the scenario of shared infrastructure (B), the construction of roads would affect three times more forests (from 7,613 km² up to 45,395 km²) as compared with the scenario (A) (approximately 2,462 km² and 16,000 km²) for both values of buffers surrounding roads. The area to be potentially affected by shared roads (B), totaling 7,613 km² with the 5.5 km buffer corresponds to twice the area that would be affected by independent roads construction (C), that reaches 3,546 km² with the 5.5. km buffer, revealing an opportunity for future impacts reduction in case of developing both regions.

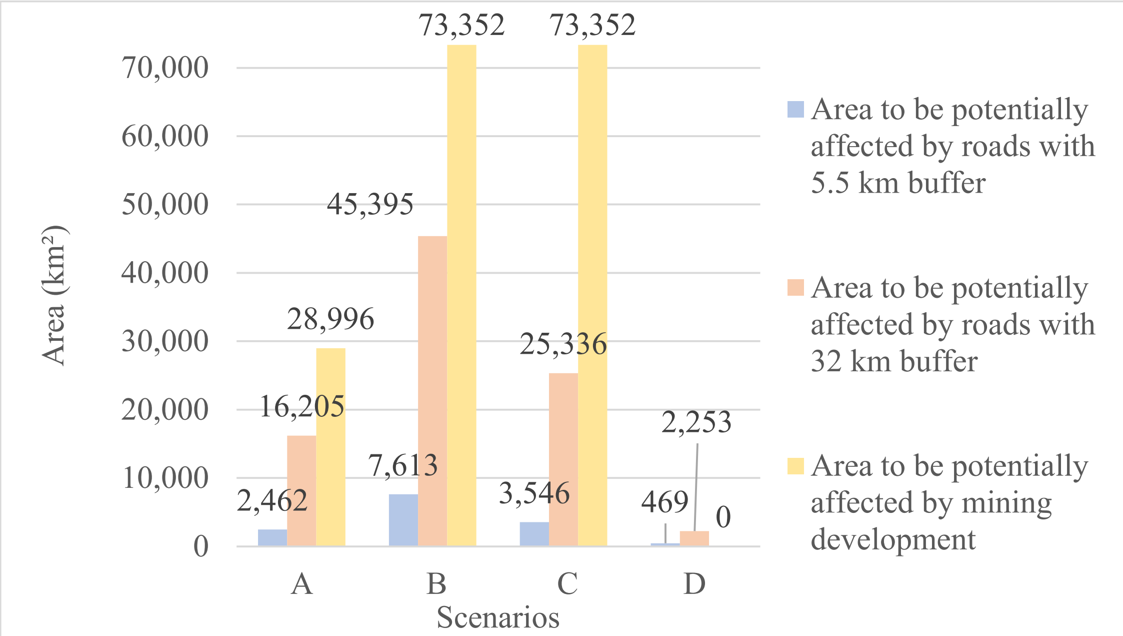


Figure 6. Area to be potentially affected by significant deforestation in each scenario. The bars show the area to be potentially affected by roads (considering 5.5 km and 32 km buffers) and the area to be potentially affected by mining (using 70 km buffer). Source: (SIQUEIRA-GAY; SÁNCHEZ, 2020)

More investigations could be done regarding mining impacts on biodiversity and ecosystem services, especially quantifying other cumulative impacts resulted of these and other developments of new mines in tropical regions. Moreover, the composite outcomes of offsets that could be required in connection with those projects, as well as the influence of other drivers of deforestation, could be considered in modeling long-term forest area balance in the region.

2.3.2 Pathways leading to cumulative impacts on forests for niobium exploitation in Western Amazon

Few studies examine cumulative dimensions of environmental impacts (BRISMAR, 2004; DUINKER et al., 2012). Pathways are the link between the source or cause and an effect, and the magnitude of the combined effects can be equal to the sum of the individual effects (additive effect) or can be an increased effect (synergistic). As cumulative impacts occur from (i) interactions between actions; (ii) between actions and the environment, and (iii) between ecological components. Pathways are used to describe the cumulative process from the cause to the effects (HEGMANN et al., 1999).

Noble (2015) exemplifies pathways illustrated in the Cold Lake oil sand project. Four pathways were identified: (i) operation and maintenance of roads will lead to compaction of the roadbed; (ii) compaction will cause an increase in surface runoff from the road; (iii) increased runoff of roads will result in erosion of exposed soils, resulting in erosion of exposed soils, resulting in an increase in sediment generation and transport. Spaling and Smit (1995) identified two pathways caused by drainage: (i) it provides a way for spatial movement of environmental components from one location to another; (ii) it provides a mechanism for accumulation of environmental change among geographic scales.

Functional pathways lead to cumulative impacts through (i) the persistent addition of a material, a force or an effect from a single source or process; (ii) compounding effects involving two or more processes. The first circumstance can result, additively, in a slow and dissipative effect or, synergistically, in a magnification of the effect produced by one single source of perturbation. The circumstance of two or more projects can additively produce multiple impacts or synergistic relationships (PETERSON; CHAN; PETERSON, 1987). For example, one or more projects can affect water quality and aquatic life by increasing the water temperature and dissolving less oxygen in a way that the effect of accumulated contaminants is multiplied, resulting in an impact greater than the sum of the impacts of individual projects (NOBLE, 2015).

One type of cumulative impact is the habitat “nibbling” loss, i.e., the progressive loss of habitat or land (HEGMANN et al., 1999; TREWEEK, 1999). Peterson et al. (1987) identify the pathway of nibbling as the additive interaction of one or two projects or source of the disturbance. This type of effect is difficult to tackle at the project level (HEGMANN et al., 1999). While some metrics of habitat loss and fragmentation can be applied, it is difficult to determine the significance of a change caused by only one action. Given this complexity, the pathways determination has an important role in determining thresholds of significance (PETERSON; CHAN; PETERSON, 1987) as well as, identifying the root causes and actions for better assess cumulative and high order impacts (BRISMAR, 2004).

Perdicoúlis and Glasson (2012) emphasize the importance of clearly explaining causality in EIA with causal networks, linking elements in the environment and activities as well as representing affected elements and related interactions. Especially for Cumulative Effects Assessment (CEA), Perdicoúlis and Piper (2008) emphasize the importance of explaining causality, identifying important cause and effect relationships between activities and resources with the main causal analysis: (i) represent time, space and impact magnitude; (ii) have evaluation capacity of analysis and data required for assessing impacts; (iii) be easy to use, to learn to use, and to interpret, and be transferable between all EIA levels.

2.3.2.1 Causal diagram

For integrated assessment and resource management, and for better informing decision making and stakeholder consultation, investigating and communicating the causality of cumulative impacts is needed (VOEGELI; HEDIGER; ROMERIO, 2019). Network analysis is a powerful tool to identify cause-effects relations, encompassing pathways or process of accumulation. To feature the elements and representations in causal networks, Perdicoúlis et al. (2014) developed a graph format and a syntax to feature the sources and action leading cumulative effects. They propose: (i) node content is a box with elements and respective actions; (ii) arrows are changes, i.e., pathways of causality of cumulative impact; (iii) final node with a receiving element (a valued ecosystem component) and cumulative change, with a colored background and the text in italic format. The indirect actions are those influencing other direct actions leading to forest loss. To represent the cumulative effects of mining development in the region, a diagram was developed to represent the pathways by which mining could entail cumulative forest loss (Figure 7).

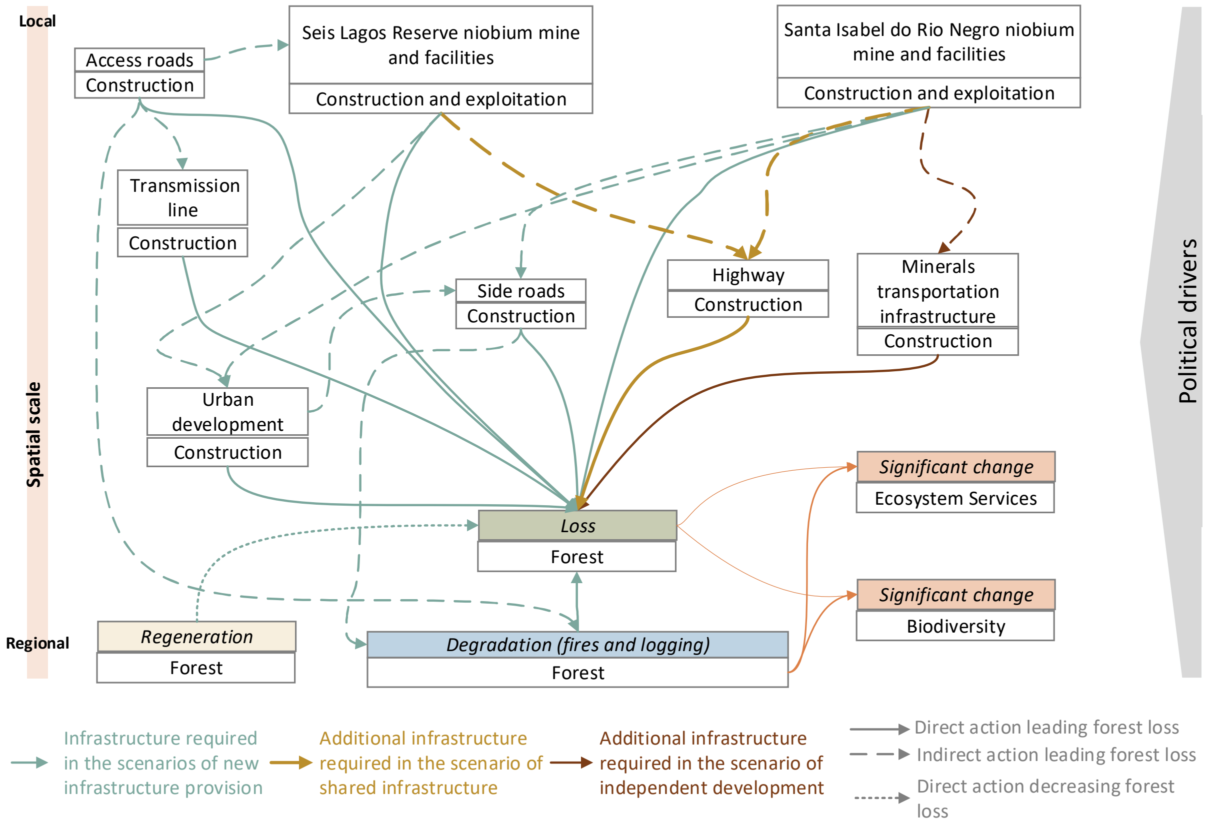


Figure 7. Causal diagram of the potential relationships among major actions of mining development and cumulative impacts on biodiversity and ecosystems services. Source: (SIQUEIRA-GAY; SÁNCHEZ, 2020)

The following pathways leading to forest loss in mining regions can be traced: (1) by building facilities for mining and mineral processing (SOUZA-FILHO et al., 2016; WORLD BANK, 2019a); (2) by constructing access roads to mine facilities and local traffic (WORLD BANK, 2019a); (3) by human occupation caused by economic and regional development of mining activities (WORLD BANK, 2019a); (4) by requiring the provision of infrastructure for energy supply, such as transmission lines (BIASOTTO; KINDEL, 2018); (5) by side roads construction to connect urban settlements (LAURANCE; GOOSEM; LAURANCE, 2009); (6) by highway construction to connect the two mines in the scenario of shared infrastructure (LAURANCE; GOOSEM; LAURANCE, 2009); (7) by the construction of ore transportation infrastructure to connect Santa Isabel do Rio Negro mine to the port in the case of the scenario of independent development (RUNGE et al., 2017; SONTER et al., 2017; WORLD BANK, 2019a); (8) by causing forest degradation (WALKER et al., 2020). The cumulative forest loss has potential to affect (9) biodiversity (TREWEEK, 1999) and (10) ecosystem services (DOBSON et al., 2007). All these pathways can be intensified or weakened depending on the type of mine and technology used and the forest affected (RUDKE et al., 2020).

Based on the causal diagram, we emphasize: (i) roads construction play an important role causing forest loss potentially affecting landscape structure; (ii) in addition to loss, these roads have potential to degrade remaining forest, as already seen in other regions in Amazon; (iii) the direct forest loss caused by mining facilities represent only one potential pathway of cumulative forest loss while the required infrastructure for mining operation, such as access roads, ore transportation and transmission lines, are in the majority of pathways; (iv) given the network complexity, the interactions between losses potentially affect landscape patterns and structure (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020), harming biodiversity and the provision of ecosystem services. Therefore, there is the need to provide a regional and comprehensive assessment to capture not only the direct impacts of mining activities but those caused by related infrastructure as well.

2.4 Is there a way forward to better appraise future niobium mining projects in northwestern amazon and ensure conservation?

Making room for a possible development of niobium mines in Seis Lagos or in Santa Isabel and at the same time conserving key biodiversity features is a huge challenge whose outcomes depend on many variables. Nevertheless, simply opposing to development can result in projects going ahead without any appropriate safeguards. If legislation changes to permit mining development within PAs, the current EIAs in Brazil – arguably the most important national policy tool for development control – is ill-equipped to provide for effective protection of biodiversity (BIASOTTO; KINDEL, 2018; DIBO; NOBLE; SÁNCHEZ, 2018).

The effectiveness of EIAs and other environmental policies depend on a vast array of factors (HANNA; NOBLE, 2015) whose consideration is far beyond the scope of this research. One such factor, however, is science- and evidence-based impact assessment (MACKINNON; DUINKER; WALKER, 2018), addressed below. Key shortcomings in Brazilian practice include: (i) cumulative impacts are overlooked (DUARTE et al., 2017), (ii) there is very limited research support to underpin cumulative impact assessment (DUARTE et al., 2017), (iii) strategic environmental assessment has very limited practical effects (SÁNCHEZ, 2017), in particular in the Amazon (RITTER et al., 2017), (iv) there is limited capacity in State Environment departments (SÁNCHEZ, 2020), and (v) EIA has a weak influence on decision-making (RITTER et al., 2017). On top of such shortcomings, there is the prospect that current legislation could be weakened (FONSECA; GIBSON, 2020).

We emphasize the need to strengthen EIAs of mining projects to consider the cumulative impacts of forest loss. To cope with cumulative effects on forests, the assessment of future mining developments requires good practice assistance (ICMM, 2006; WORLD BANK, 2019a) and consideration of different scenarios to support spatial planning and interventions design (RUNGE et al., 2017). The pathway generally considered in the current EIAs practice is restricted to the direct loss of forest due to mine infrastructure construction, separated from ore transportation system and access roads. As such, the interaction between losses and landscape wide consequences are disregarded by current assessments (DIBO; NOBLE; SÁNCHEZ, 2018) because projects are assessed separately (NERI; DUPIN; SÁNCHEZ, 2016) and indirect and cumulative impacts are disregarded.

To advance recommendations for assessments and better-informed decision-making, we systematize three broad leverage points to improve cumulative impact assessment practice. Firstly, the terms of reference for an environmental impact study should specify guidelines to assess the cumulative impacts of projects and associated facilities on biodiversity features and present criteria to establish spatial and temporal boundaries based on ecological features of the affected valued component (DIBO; NOBLE; SÁNCHEZ, 2018). Tailor-made terms of reference are needed to provide clear guidance for assessing cumulative impacts (BORIONI; GALLARDO; SÁNCHEZ, 2017; VILARDO; LA ROVERE, 2018). Secondly, it is needed to build a comprehensive database and standardized protocols to collect and make data available for the purpose of cumulative impact assessment (NERI; DUPIN; SÁNCHEZ, 2016). Thirdly, considering the international practice, key recommendations for better integrating science and EIAs involves the use and development of robust methodologies, including modelling and scenarios proposal with retrospective and prospective investigative protocols (DUINKER et al., 2012; FOLEY et al., 2017).

2.5 Conclusions

Although the largest niobium mine in Brazil has reserves to operate for 200+ years considering the current production rates, Brazilian politicians are still voicing the willing to expand Nb mining to the Amazon, potentially affecting both, the market balance and the conservation of biodiversity and ecosystem services. The study of Seis Lagos Nb deposits illustrates the potential impacts of mining in a well-conserved region under pressure for exploitation, threatened by the weakening environmental legislation and poor enforcement within and surrounding protected areas. Despite the presence of an existing access road, there is a need to construct new infrastructure in case of Nb mining.

Considering the possible development of the two areas bearing known Nb deposits in the Brazilian Northwestern Amazon, sharing new transportation infrastructure, new roads would entail forest loss twice as large as the loss expected if such infrastructure is built separately, evidencing an impact minimization opportunity. The construction of roads plays an important role as driver of change leading to cumulative impacts on forests, being present in most pathways ending in forest loss. We thus alert to the risk of making ill-informed decisions without a comprehensive assessment of the severe adverse impacts that would arise from such mining development if the current approach to environmental permitting is applied, let alone in face of the possible weakening of current legislation.

These findings emphasize the need to strengthen the EIAs of mining projects to systematically consider the cumulative impacts of forest loss. We recommendations to better predict and assess the impacts of mining projects in Northwestern Amazon and ensure conservation as follows (i) specify guidelines for assessing cumulative impacts in carefully tailor-made term of reference; (ii) build a comprehensive database and standardize protocols to collect and make data available; (iii) develop robust methodologies, including modelling and scenarios proposal with retrospective and prospective investigative protocols to underpin the assessment of cumulative impacts.

Therefore, in the context of the existing drawbacks at the project level EIAs allied with to the political pressure for simplifying environmental permitting procedures, the current practice threatens the protection of biodiversity values, requiring a paradigm shift. Otherwise, the Amazon Nb should be kept in the ground.

2.6 Research limitations

This chapter seeks to analyze the forest extent to be impacted in each scenario of development in the regions of the Pico da Neblina National Park. The region is well conserved and almost no land use and cover changes were detected in the last years, which represents a hurdle to calibrate a spatially explicit land use model. However, the estimation of the other impacts - not only the extent of the forest to be impacted – can be further investigated to determine quantitatively the importance of each pathway.

3 Land use and land cover changes driving cumulative impacts in tropical mining regions

Objective

This chapter aims at identifying LULC changes leading to cumulative impacts in mining regions

Preface

This chapter classifies remote sensing products from the years 1997, 2007 and 2017 and analyses the spatial correlation to detect frontiers of great interest for cumulative impacts on native vegetation. Two mining regions were compared: one with mining as a key driver of change and other, with mining associated with cellulose industry. Pastureland, silviculture, and urban expansion were mapped in detail, revealing that silviculture growth impacted more forests than pastureland expansion when associated with mining and pulp mill activities. In contrast, in regions with gold and iron ore activities, the pastureland expansion was more relevant than all other drivers.

This chapter supports the thesis (Figure 1) on the: (i) understanding of the interplay between drivers of change in causing indirect impacts and cumulative impacts on native vegetation; (ii) identify areas of interest for cumulative changes over time and across the landscape; (iii) present implication for spatial and temporal boundaries in cumulative impact assessment.

Related publication

Siqueira-Gay, J.; Santos, D.; Nascimento Jr, W. R.; Souza-Filho, P.W.; Sánchez, L.E. Investigating changes driving cumulative impacts on native vegetation in the Brazilian Amazon. *In review.*

Abstract

Developing conservation strategies to mitigate cumulative impacts requires the understanding of historic drivers of land use and land cover change at the regional scale. By using a multisensory and multitemporal approach, we identified the major changes driving cumulative impacts on native vegetation in northeastern Amazon. Comparing two regions, one with mining as the key driver and another where mining is associated with other industrial activities (cellulose), we explore the land use and land cover historic dynamics and derive implications for the assessment of cumulative impacts. Transitions of forest cover to pastureland, silviculture, and urban expansion were mapped in detail over a 20-year period, revealing that silviculture growth cleared more forests than pastureland expansion when associated with pulp mill activities and kaolin mining. In contrast, in a region with gold and iron mining, pastureland expansion was more relevant, clearing mainly areas surrounding new roads. This research shows that the interplay of major mining and industrial investments can produce cumulative losses of native vegetation, depending on the associated industries and infrastructure required for the project development. Our findings emphasize that the definition of spatial and temporal boundaries for the assessment of cumulative impacts must consider different trends in impact accumulation and changes in their spatial distribution over time.

3.1 Introduction

LULC changes influence the functioning of the local, regional, and global systems (NOBRE et al., 2016). Underlying driving forces, such as industrial policies, market demand for commodities and demographic factors influence social processes regionally and globally (GEIST; LAMBIN, 2002), while proximate causes directly affect LULC at the local scale (GEIST; LAMBIN, 2002) by expanding agriculture (CURTIS et al., 2018; STEHFEST et al., 2019), infrastructure (LAURANCE; GOOSEM; LAURANCE, 2009) and mining (SONTER et al., 2013).

Although mining is an intensive land use (BRIDGE, 2004), when associated with infrastructure development, it can induce extensive LULC changes beyond its direct footprint (SONTER et al., 2017). In several regions, mining is developed alongside plantations, agriculture, cattle grazing, and urban development (BRAUN et al., 2015; HOTA; BEHERA, 2016; SONTER et al., 2014b). These different activities drive LULC changes that interact resulting in cumulative impacts (NERI; DUPIN; SÁNCHEZ, 2016; RAITER et al., 2014). Cumulative impacts are changes caused by multiple interactions among human activities and natural processes that accumulate across space and time (CCME, 2014) and can result from the sum of individual minor impacts (RAITER et al., 2014), leading to extensive consequences regionally, such as the progressive loss of habitat (RAITER et al., 2014; TREWEEK, 1999).

The worldwide expansion of mining over forests (WORLD BANK, 2019a) and sensible ecosystems (FERNANDES et al., 2018a) has been calling attention to impacts on biodiversity (SONTER; ALI; WATSON, 2018). In these regions, where several mines are spatially concentrated, the individual footprints may cause relatively minor direct loss on native vegetation however, the sum of clearing could entail major consequences to landscape (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020), hydroclimate balance (SOUZA-FILHO et al., 2016), and water quality (ABE et al., 2019). The direct, indirect and cumulative deforestation can affect vast areas, particularly in regions that present high biodiversity values and sensible ecosystems (KOBAYASHI; WATANDO; KAKIMOTO, 2014; MURGUÍA; BRINGEZU; SCHALDACH, 2016).

Cumulative Effects (or Impacts) Assessment (CEA) is a systematic procedure for identifying and evaluating the significance of the effects from multiple activities (BRODERICK; DURNING; SÁNCHEZ, 2018). CEA can be performed locally, analyzing projects individually (DIBO; NOBLE; SÁNCHEZ, 2018) or regionally, e.g. at watersheds level (DUBÉ et al., 2013) or any regional boundaries (NOBLE, 2008). The consideration of cumulative impacts remains uncertain in part of Environmental Impact Assessments (EIAs) of mining projects (SINGH et al., 2020), widely used in support to decision-making about new projects in these sensible regions. EIAs fail in addressing relevant aspects of biodiversity by analyzing projects in an individual basis and neglecting their combined effects in the landscape (BIASOTTO; KINDEL, 2018; GANNON, 2021; NOBLE; LIU; HACKETT, 2017; RITTER et al., 2017; SINGH et al., 2020; SLOOTWEG et al., 2010). In addition to policy failures that hinder the effectiveness of project-based EIA to deal with cumulative effects, there is a poor understanding of processes and interactions leading to cumulative impacts (DUINKER et al., 2012; SINGH et al., 2020), especially those resulting from temporal and spatial interactions affecting ecological processes, degrading ecosystems, and the services they provide (RAITER et al., 2014).

The challenges are higher in regions subject to fast LULC change, where evaluating the importance of each driver of change has the potential to inform comprehensive assessments of cumulative impacts of mining on forests and other ecosystems (SIQUEIRA-GAY; SÁNCHEZ, 2020; SONTER et al., 2013). A detailed spatial and temporal analysis of historic changes is required to inform about the past and current state of affected ecosystems (FAGIEWICZ, 2014; LECHNER et al., 2019; NUNES et al., 2016) in order to figure out plausible future changes and plan for adequate mitigation, including further management for the rehabilitation of mined land (GASTAUER et al., 2018; SÁNCHEZ; SILVA-SÁNCHEZ; NERI, 2014).

This research aims at identifying and quantifying the relevance of the LULC changes leading to cumulative impacts on native vegetation in the Amazon. We analyzed remote sensing products to map temporal changes and conducted spatial analysis for identifying trends LULC transitions of interest. By analyzing the impacts of LULC change in two regions in the Brazilian Amazon - one with predominant mining activities and another where mining is associated with industrial activities (cellulose) -, we describe the main drivers in both landscapes.

Our findings contribute to the understanding of drivers of change leading to cumulative impacts on native vegetation. By investigating the importance of the LULC changes, we discuss implications for cumulative impact assessment in landscapes with mining and other industrial developments. In the following sections, we present our methods, the main changes leading to cumulative impacts in both regions, and discuss their implications for cumulative impact assessment.

3.2 Materials and methods

The research was conducted in a highly biodiverse and conserved sector of the Brazilian Amazon (VENTER et al., 2016). Different mines are operating in the region surrounding Renca, a mining reserve where no commercial mining is allowed within its boundaries (Figure 8). Two neighboring mining areas are studied in detail: Serra do Navio and Jari. The former is known by its once important manganese mine, closed in 1998, and the urban settlement created in the 1950's to support its operation. In the 2000's, gold and iron ore mines were developed (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020). The region presents mainly small settlements, and small-scale grazing and agriculture, and no large industrial activity. In contrast, the *Jari Celulose,* is a large pulp plant, fed by homogeneous eucalyptus plantations that substituted the original rainforest in the late 1970's (MONTES et al., 2002). Taking opportunity of the infrastructure provided by the Jari project, both kaolin and a small bauxite mine were developed (OLIVEIRA, 2010). These two regions present significant interaction of activities related to mining projects in a biodiverse region of Amazon.

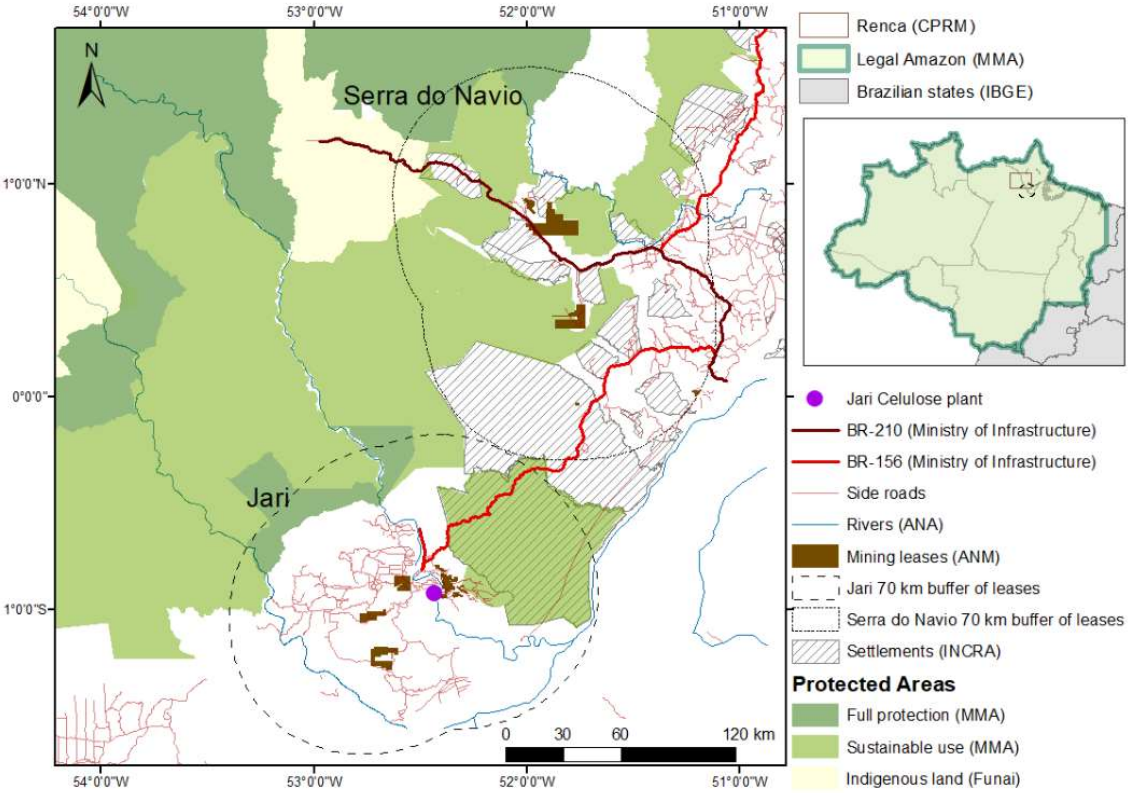


Figure 8. Study region, Protected Areas and mining leases

3.2.1 Digital image processing

To create the historical trajectory of LULC changes in the study region, the Landsat repository of images was consulted and images with less than 10% of clouds in the scene were downloaded (Table S1 in Appendix 2). Some regions in Amazon present high cloud cover (ARNESEN et al., 2013) and to support the classification we used the optical sensor as part of a combined approach for classification (NASCIMENTO et al., 2013). For this purpose, we used Scenes from Phased Array type Advanced Land Observing Satellite (ALOS) Phased L-band Synthetic Aperture Radar (PALSAR/PALSAR-2) sensors (Table S2 in Appendix 2). The Shuttle Radar Topography Mission (SRTM) (Table S3 in Appendix 2) with resolution 30m/1Arc was used as another layer to support the image classification.

The first step of processing used the software PCI Geomatica to perform atmospheric correction in the Landsat dataset, and apply the speckle filter to SAR data (Figure S1 in Appendix 2). In a second step, the images, elevation and optical products were mosaicked forming datasets for the years 1997, 2007 and 2017. The oldest Landsat image with less than 10% cloud cover available for the region was selected as reference (1997) while two others (2007, 2017) with 10 years interval made up for 20 years of analysis in total. Thirdly, we clipped with the buffers of Serra do Navio and Jari distancing 70 km (SONTER et al., 2017) from mining leases (Figure 8) to feature the area of influence to be analyzed in detail.

The Geographic Object-Based Image Analysis (GEOBIA) method features similar objects, dividing pixels with similar spectral reflectance into distinguishable entities (BLASCHKE et al., 2014; SOUZA-FILHO et al., 2018). This process involves segmenting using as parameters: (i) the scale, relating the object size to the mining mapping unit; (ii) the shape, to result in spectrally homogenous objects; (iii) compactness, fitting the object shape and its boundaries.

The multiresolution classification in the software eCognition was applied to identify the LULC classes: forest, pastureland and exposed soil, industrial, urban, silviculture, other native vegetation, and mining (Table 1). Membership functions (SOUZA-FILHO et al., 2018) were used for describing the fitting of values to be assigned in each class for both regions, Jari (Table S4 in Appendix 2) and Serra do Navio (Table S5 in Appendix 2).

Table 1. LULC classes description

LULC classes	Description
Forest	Native forests
Pastureland and exposed soil	Exposed soil, pasture, and recent deforested areas
Industrial	Facilities and infrastructure of industrial plants
Urban	Organized occupation featured by street design and high density of cleared areas
Silviculture	Forest plantation with homogenous tree cover and usually organized in terrain parcels
Other native vegetation	Vegetated flooded areas, riparian forests and open fields covered by savannas
Mining	Area of mineral extraction with exposed soil and mining facilities

To evaluate the accuracy of image classification, the procedure of accuracy assessment of the PCI Geomatica was performed to compare the classified map and the reference image. No field visit was conducted to collect ground truth points for accuracy assessment. Therefore, the accuracy of the classification was evaluated using the reference image only. The total of 600 points stratified by each LULC class were validated in each map. The confusion matrix and the accuracy statistics report including the overall Kappa index and Kappa index for each class were exported (Table S7 – Table S24 in Appendix 2).

3.2.2 Change detection

For the change detection, we vectorized the raster file exported from eCognition, dissolved each class and overlaid the resulted vector maps of 1997 with 2007 and maps of 2007 with 2017 (Figure S3 in Appendix 2). We used ArcGis 10.5 to handle vector maps and calculate the total area converted. Analyzing the resulting attribute table, we identified the main changes and areas that did not change between the years of analysis. Management plans of protected areas and planning documents of both regions were consulted to support the understanding of the socioeconomic drivers of change in each region (Table S24 in Appendix 2).

3.3 Main changes leading to cumulative impacts

3.3.1 Mining as a key driver of change

In the years of analysis 95% of forests remain intact and the 5% of deforestation results from pastureland, silviculture, mining and urban (Table 2 and Table 3). In the 20-year period, the total mining footprint increased in Serra do Navio due to gold and iron ore mines (OLIVEIRA, 2010). Mining expansion occurred at the expense of forest (25 km² in 1997-2007 and 34 km² in 2007-2017), increasing of more than 200% of mining footprint over forests between 1997-2007 (Table 2) and more than 100% between 2007-2017 (Table 3). More forests were converted to pastureland, silviculture and mining after 2007, while urban expansion over forests and other native vegetation occurred mainly during 1997-2007. The growth of Pedra Branca do Amapari is associated with gold mining (ICMBIO, 2009) while the Porto Grande, the largest town in the area, present port activities (OLIVEIRA, 2010).

The total of 25% of other native vegetation cover represented by the open fields in Amapá were converted to pastureland (23.2%), silviculture (1.8%) and urban during 1997-2007. In the period 2007-2017 more than 30% of other native vegetation was cleared mainly to pastureland (30%), silviculture (0.1%), urban (0.2%). Given the seasonality of silviculture harvesting, there is an important conversion mapped as from silviculture to pasture in both intervals of analysis. However, 42 times more forest was converted to pastureland than to mining (1,054 km² in 1997-2007 and 1,373 km² in 2007-2017). It is also possible to note pastureland transitioning to silviculture and pastureland expansion by clearing forests.

Table 2. Transitions of land use and land cover in Serra do Navio from 1997 (lines) to 2007 (columns) and percentage in relation to total area in 1997

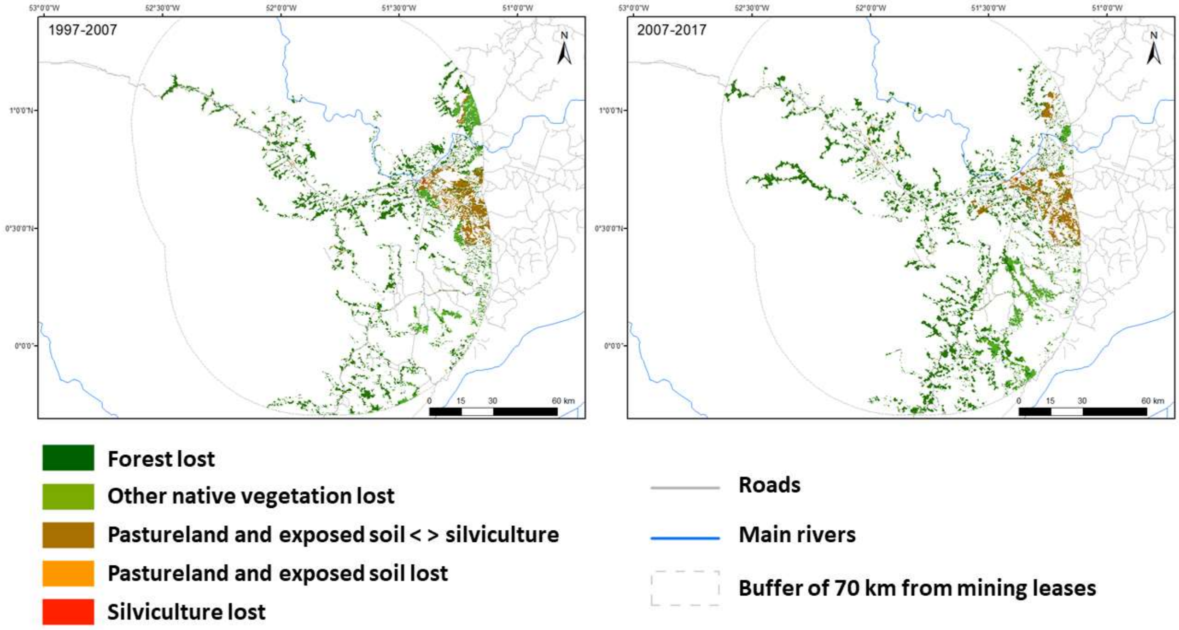
Land use and land cover classes in 1997	**Area (km²) converted from 1997 to 2007**					
	Forest	**Other native vegetation**	**Pastureland**	**Silviculture**	**Mining**	**Urban**
Forest	22,326 (95.3%)		1,054 (4.5%)	29 (0.1%)	25 (0.1%)	1 (0.0%)
Other native vegetation		859 (75.0%)	265 (23.2%)	20 (1.8%)		0.3 (0.0%)
Pastureland			1,058 (86.8%)	152 (12.5%)	3 (0.2%)	5 (0.4%)
Silviculture			148 (45.3%)	179 (54.7%)		
Mining					12 (100%)	
Urban						7 (70.8%)

Table 3. Transitions of land use and land cover in Serra do Navio from 2007 (lines) to 2017 (columns) and percentage in relation to total area in 2007

Land use and land cover classes in 2007	**Area (km²) converted from 2007 to 2017**					
	Forest	**Other native vegetation**	**Pastureland**	**Silviculture**	**Mining**	**Urban**
Forest	21,626 (93.7%)		1,373 (5.9%)	39 (0.2%)	34 (0.1%)	0.5 (0.0%)
Other native vegetation		699 (69.0%)	311 (30.7%)	0.6 (0.1%)		1.6 (0.2%)
Pastureland			1,581 (89.4%)	173 (9.8%)	4.2 (0.2%)	10 (0.6%)
Silviculture			118 (33.5%)	235 (66.5%)		
Mining					27 (100%)	
Urban						8.7 (62.4%)

During both years of analysis, the changes of forest cover in Serra do Navio occurred surrounding roads (Figure 9). During the years of 2007-2017 changes in forest cover were more prominent surrounding new roads in the west and intensified in the roads in the south. The dynamic of pastureland and exposed soil to silviculture was concentrated in the east together with the more densified road network (Figure 9).

Figure 9. Changes in Serra do Navio during 1997-2007 and 2007-2017



3.3.2 Mining associated with other industrial drivers

The total of 94% of forests remain in both intervals of analysis. Mining and the cellulose industry converted less than 1% of forests of the study area, evidencing the minor role of their footprint in contrast to the larger-scale impacts of the LULC driven by the associated activities. The silviculture and pasture to expansion cleared 1,021 km² (1997-2007) and 1,054 km² (2007-2017) (Table 4). Silviculture expansion was more significant in the period 1997- 2007 converting mainly forests (655 km²) and pastureland (378 km²), while in the more recent period 430 km² of forests and 181 km² of pastureland were cleared (Table 5).

In Jari, mining expansion was much smaller as compared to Serra do Navio. The most important mining development, a kaolin mine, took advantage of the presence of the pulp industrial project, itself the major driver of LULC change due to expansion of silviculture and further pasture expansion in surrounding plantations. Pastureland expansion is mainly associated to the industrial and infrastructure expansion surrounding Jari cellulose areas, as the region is far from the large agriculture frontier (SECRETARIA DE ESTADO DE MEIO AMBIENTE, 2010, 2011).

Table 4. Transitions of land use and land cover in Jari of from 1997 (lines) to 2007 (columns) and percentage in relation to total area in 1997

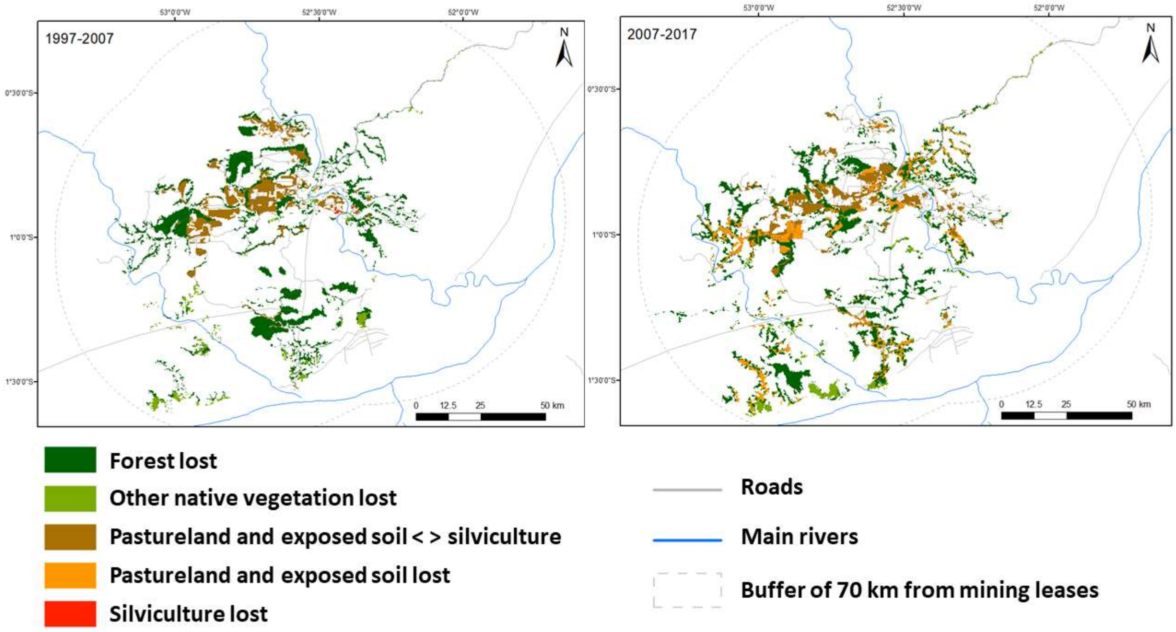
Land use and land cover classes in 1997	Area (km²) converted from 1997 to 2007						
	Forest	Other native vegetation	Pastureland	Silviculture	Mining	Cellulose industry	Urban
Forest	16,010 (94.0%)		366 (2.1%)	655 (3.8%)	0.9 (0.0%)	1.6 (0.0%)	0.7 (0.0%)
Other native vegetation		2,737 (95.8%)	99 (3.5%)	17.3 (0.6%)	0.009 (0.0%)	0.6 (0.0%)	2.6 (0.1%)
Pastureland			288 (43.1%)	378 (56.5%)	0.007 (0.0%)	0.4 (0.1%)	2.8 (0.4%)
Silviculture			88 (17.3%)	417 (81.9%)	1.2 (0.2%)	2.7 (0.5%)	
Mining					1.1 (100%)		
Cellulose industry						2.9 (100%)	
Urban							5.8 (100%)

Table 5. Transitions of land use and cover change in Jari from 2007 (lines) to 2017 (columns) and percentage in relation to total area in 2007

Land use and land cover classes in 2007	Area (km²) converted from 2007 to 2017						
	Forest	Other native vegetation	Pastureland	Silviculture	Mining	Cellulose industry	Urban
Forest	15,494 (93.6%)		624 (3.8%)	430 (2.6%)	0.13 (0.0%)	1 (0.0%)	1 (0.0%)
Other native vegetation		2,686 (94.5%)	147 (5.2%)	4 (0.1%)		0.01 (0.0%)	4 (0.2%)
Pastureland			414 (69.3%)	181 (30.3%)	0.01 (0.0%)	0.37 (0.1%)	2 (0.3%)
Silviculture			251 (22.8%)	850 (77.1%)	1 (0.1%)	0.49 (0.0%)	
Mining					3 (100%)		
Cellulose industry						8 (100%)	
Urban							12 (100%)

The most important loss of forests and other native vegetation in the Jari region occurs in the frontier surrounding roads and close to silviculture areas of *Jari Celulose* (Figure 10). The forest loss shared space with pasture-silviculture conversion and between the years of 2007 and 2017 and the dynamic of pastureland and exposed soil and silviculture more prominent in the region of *Jari Celulose*. Important transitions of forest loss occurred surrounding the road in the north during 1997-2007 while areas with forest and other native vegetation loss, during 1997-2007, are close to roads in the west of Jari, revealing a new frontier of deforestation in the region (Figure 10).

Figure 10. Changes in Jari during 1997-2007 and 2007-2017



3.4 Implications for cumulative impact assessment

CEA rationale is focused on valued environmental and social components that can be affected by a number of projects and other drivers of change (IFC, 2013). The definition of appropriate spatial and temporal scales is one of the most important challenges in CEA (DUINKER et al., 2012; FOLEY et al., 2017; JONES, 2016; SEITZ; WESTBROOK; NOBLE, 2011). Aiming at contributing to this debate in CEA, we discuss possible implications of our research on LULC to the establishment of both, spatial and temporal dimensions for the assessment of cumulative impacts on native vegetation.

By using this threshold of 70 km surrounding mining areas, we found that LULC changes in mining regions varied according to associated industry and infrastructure required. In the case of kaolin mining associated with pulp mill activities, silviculture cleared more forests than pastureland during one period of analysis. The gold and iron ore mines were served mainly by existing roads and the pastureland expansion surrounding these areas was responsible for the majority of forest conversion in both years of analysis. Therefore, when mining is associated with other projects, the LULC changes may be driven by industrial factors, for instance driving silviculture expansion and so that other transitions may not be significant. Moreover, in the case of kaolin silviculture and pastureland conversion dominated the landscape and occupation surrounding roads had less importance than in Serra do Navio sector. Therefore, CEA spatial boundaries must encompass frontiers of deforestation beyond the immediate area of the projects included in the assessment, considering the main past indirect changes related to each project (DUINKER et al., 2012) and other potential future developments leading to LULC in the region (ATLIN; GIBSON, 2017).

Besides the area to be impacted by each project, the spatial boundaries to analyze cumulative impacts on biodiversity have to be determined based on ecological thresholds to sustain ecological processes and functioning (DIBO; NOBLE; SÁNCHEZ, 2018; GANNON, 2021), considering the distribution of habitats and ecosystems to be affected and the relevance ecosystem-level processes (DIBO; NOBLE; SÁNCHEZ, 2018; FOLEY et al., 2017). In fragmented landscapes, the surrounding matrix may affect the degree of isolation of forests remnants, influencing the species richness and diversity they shelter (MANGUEIRA et al., 2021; RICKETTS, 2001). Therefore, each LULC change may have a different importance for biodiversity conservation, as in the case of pastureland (MANGUEIRA et al., 2021) and silviculture (VERSLUIJS et al., 2020), and this interplay of different LULC changes in the landscape must be considered when assessing the respective importance of each change in the magnitude of cumulative impacts on biodiversity.

In the study area analyzed, the relative importance of each driver remained the same in both periods of analysis, but their intensity changed, revealing different trends in the impact accumulation over time. The growth of pastureland over forests and other native vegetation in Serra do Navio and Jari occurred mainly during 2007-2017. In Serra do Navio, silviculture expanded over forests mainly during 2007-2017 (Table 3) but over other native vegetation mainly during 1997-2007 (In the years of analysis 95% of forests remain intact and the 5% of deforestation results from pastureland, silviculture, mining and urban (Table 2 and Table 3). In the 20-year period, the total mining footprint increased in Serra do Navio due to gold and iron ore mines (OLIVEIRA, 2010). Mining expansion occurred at the expense of forest (25 km² in 1997-2007 and 34 km² in 2007-2017), increasing of more than 200% of mining footprint over forests between 1997-2007 (Table 2) and more than 100% between 2007-2017 (Table 3). More forests were converted to pastureland, silviculture and mining after 2007, while urban expansion over forests and other native vegetation occurred mainly during 1997-2007. The growth of Pedra Branca do Amapari is associated with gold mining (ICMBIO, 2009) while the Porto Grande, the largest town in the area, present port activities (OLIVEIRA, 2010).

The total of 25% of other native vegetation cover represented by the open fields in Amapá were converted to pastureland (23.2%), silviculture (1.8%) and urban during 1997-2007. In the period 2007-2017 more than 30% of other native vegetation was cleared mainly to pastureland (30%), silviculture (0.1%), urban (0.2%). Given the seasonality of silviculture harvesting, there is an important conversion mapped as from silviculture to pasture in both intervals of analysis. However, 42 times more forest was converted to pastureland than to mining (1,054 km² in 1997-2007 and 1,373 km² in 2007-2017). It is also possible to note pastureland transitioning to silviculture and pastureland expansion by clearing forests.

). In contrast, in Jari, it expanded over forest and other native vegetation during 1997-2007, the same period of major growth of the cellulose industry (Table 4). In Serra do Navio, the villages expanded more during the period 1997-2007 while in Jari this expansion was observed during 2007-2017. Therefore, the definition of temporal boundaries in CEA of mining development's needs, as a minimum, to fully encompass the project lifecycle (ATLIN; GIBSON, 2017; SINGH et al., 2020), considering the long time frame from construction to closure and possible extension of the mine life (GASTAUER et al., 2018; SÁNCHEZ; SILVA-SÁNCHEZ; NERI, 2014) as well as past and reasonably foreseeable future activities (SCHULTZ, 2012), including changes in the rate of accumulation.

These different rates of change influence the interaction between impacts, intensifying or decreasing potential additive and synergistic effects (JONES, 2016). These changes feature potential feedbacks and influence possible impacts lags, both of fundamental importance for determining spatial and temporal boundaries in CEA (JONES, 2016; SEITZ; WESTBROOK; NOBLE, 2011). In Serra do Navio, pastureland and mining clear mainly forest and other native vegetation after 2007 (Table 3). Silviculture cleared mainly forests during 1997-2007 and other native vegetation after 2007 (In the years of analysis 95% of forests remain intact and the 5% of deforestation results from pastureland, silviculture, mining and urban (Table 2 and Table 3). In the 20-year period, the total mining footprint increased in Serra do Navio due to gold and iron ore mines (OLIVEIRA, 2010). Mining expansion occurred at the expense of forest (25 km² in 1997-2007 and 34 km² in 2007-2017), increasing of more than 200% of mining footprint over forests between 1997-2007 (Table 2) and more than 100% between 2007-2017 (Table 3). More forests were converted to pastureland, silviculture and mining after 2007, while urban expansion over forests and other native vegetation occurred mainly during 1997-2007. The growth of Pedra Branca do Amapari is associated with gold mining (ICMBIO, 2009) while the Porto Grande, the largest town in the area, present port activities (OLIVEIRA, 2010).

The total of 25% of other native vegetation cover represented by the open fields in Amapá were converted to pastureland (23.2%), silviculture (1.8%) and urban during 1997-2007. In the period 2007-2017 more than 30% of other native vegetation was cleared mainly to pastureland (30%), silviculture (0.1%), urban (0.2%). Given the seasonality of silviculture harvesting, there is an important conversion mapped as from silviculture to pasture in both intervals of analysis. However, 42 times more forest was converted to pastureland than to mining (1,054 km² in 1997-2007 and 1,373 km² in 2007-2017). It is also possible to note pastureland transitioning to silviculture and pastureland expansion by clearing forests.

), revealing changes in the rates of accumulation during the years of analysis and differently according to each native vegetation. In Jari, the industrial footprint expanded together with mining clearing forests in the period of 1997-2007 (Table 4). Similarly, silviculture expansion played an important role on converting forests and other native vegetation during 1997-2007. However, the pastureland expansion, which is related to plantations clearing dynamics, is more relevant in the period 2007-2017 (Table 5) revealing potential impact lag and feedbacks in the clearing of native vegetation for the cellulose industry.

In Serra do Navio, from 1997 to 2007 forest loss occurred mainly surrounding existing BR-210 and settlements (Figure 9), while during 2007-2017 native vegetation loss was concentrated mainly surrounding other roads in the south (Figure 9). In Jari, roads close to plantations present a high concentration of forest loss during 1997-2007, the same period of silviculture major expansion (Figure 10). Therefore, changes in spatial trends over the years of analysis reveal that cumulative impacts may affect differently regions across the landscape requiring the spatial prioritization of areas of interest, such as surrounding roads and plantations, when assessing the importance of cumulative impacts on native vegetation.

3.5 Conclusion

This research used remote sensing products and spatial analysis to investigate the importance of different drivers of land use and land cover change leading to cumulative impacts in two regions in the Brazilian Amazon, one where change is mostly driven by mining, and another where change is mostly steered by industrial activities. In the latter, the need to supply a pulp mill with raw material from exotic trees pushed for substituting plantations for native forests and for opening side roads which, in turn, induced deforestation. The infrastructure provided by the industry also facilitated kaolin and bauxite mining. In the former area, gold and iron mining expansion occurred at the expense of forest, but the indirect impacts of pastureland and silviculture expansion are considerably larger than the direct industrial footprint. Silviculture growth cleared more forests than pastureland expansion when mining is associated with the cellulose industry. Therefore, the drivers of change vary according to the type of commodity and infrastructure required for the development.

We contributed to the understanding of the spatial and temporal dimensions of cumulative impacts by emphasizing some implications for the assessments of future mining projects. Firstly, as native vegetation clearing may accumulate differently across the landscape over time, cumulative impact assessments must consider the interplay of changes when determining spatial boundaries, based on the associated infrastructure required for project implementation and operation as well as on the extent that could be affected by indirect changes. Secondly, the temporal rates of land use and land cover change have to be considered to understand impact accumulation and interaction. Changes in the clusters patterns over time show that cumulative impacts may affect differently regions across the landscape, requiring the spatial prioritization of areas of interest.

We sought to overcome challenges related to image classification in cloudy areas as well as to apply methods of spatial analysis to discriminate the importance of each driver of change in the landscape. However, our work presents some limitations regarding the temporal scale of analysis, noting the lack of cloudless images before 1997 and optical products, therefore we could not assess the whole impacts of roads in the region. Besides that, there is a limitation in the classification accuracy assessment due to the lack of ground truth data that entailed in the comparison of only the classified map with the reference image. The consideration of pastureland and exposed soil in the same class was suitable to analyzing the dynamics of native vegetation loss but for future developments, these classes should be considered in detail, especially considering their distinguish importance for biodiversity conservation. Similarly, a more detailed mapping of forest and secondary vegetation has the potential to be more informative about biodiversity loss and could be addressed by future research on the topic.

Our considerations for cumulative impact assessment are based only on historic changes in the landscape. In practice, future trends should be evaluated to comprehensively assess the cumulative impacts of potential projects as well. Further works could integrate participatory approaches and methods for assessing how these changes could affect different ecosystems and the services they provide.

3.6 Research limitations

This chapter classifies remote sensing products of 1997, 2007, 2017 but it does not cover the entire period the development in the region. The Serra do Navio started to be developed in the 1950's and the Perimetral Norte highway was constructed in the 1970's, requiring the analysis of different remote sensing products to map the entire trajectory of deforestation in the region. Images from 1970's were consulted but it was hard to find images with a satisfactory cloud cover. For the most recent images of 1990's with satisfactory cloud cover, the radar products helped in the classification.

4 Exploring mining impacts of forest loss and fragmentation

Objective

This chapter aims at quantifying the forest loss and fragmentation driven by mining and associated infrastructure.

Preface

This chapter brings the analysis of historic changes in the Renca region using public available LULC maps considering absolute changes from 1991 to 2014. The evaluation of the historic impacts focused on deforestation within protected areas and surrounding roads and the impacts in the landscape structure are investigated. Further implications for EIAs of future mining projects are discussed. More than 75% of total deforestation in Serra do Navio between 1991 and 2014 occurred surrounding roads. Forest loss affected a protected area of sustainable use and the landscape structure, including increased edge effects.

This chapter supports this thesis (Figure 1) by: (i) analyzing the relevance of deforestation surrounding roads; (ii) historic impacts within PAs; (iii) impacts in the landscape structure and fragmentation.

Related publication

Siqueira-Gay, J., Sonter, L.J., Sánchez, L.E., 2020. Exploring potential impacts of mining on forest loss and fragmentation within a biodiverse region of Brazil's northeastern Amazon. Resources Policy 67, 101662. https://doi.org/https://doi.org/10.1016/j.resourpol.2020.101662

Abstract

Mining is a significant driver of deforestation. Not only do mines clear native forests for mineral extraction, they also often establish new infrastructure, which indirectly facilitates new access to land and further clearing. Forest loss and fragmentation have serious effects on biodiversity, yet rarely are these cumulative impacts of mining studied at the regional scale. Here, we examine potential impacts of mining in a biodiverse region of the Brazilian Amazon. Renca is currently off limits to mining activities but was recently threatened with a move to permit mineral exploration. We analyzed historic forest loss and fragmentation within two mining regions neighboring Renca to explore what might happen if Renca were to be mined. We also investigated historic deforestation trajectories within Renca's protected areas, to determine how well conserved these forests are against current threats. We found that mining, and other infrastructure associated with mines (i.e. roads), caused significant forest loss and fragmentation within neighboring mining sectors, and that Renca's protected areas are not currently immune to forest loss. Permitting new mines within and surrounding Renca will place additional pressure on its biodiversity. If mineral development is to proceed, huge regulatory changes will be required to effectively manage negative impacts on forests and biodiversity. Environmental Impact Assessments for new mining projects must assess and mitigate the cumulative region-wide effects on forests, while existing protected areas must be strengthened to ensure they are not directly or indirectly compromised by mining activities.

4.1 Introduction

Mining is an extensive driver of land use change. Although mines occupy less than 1% of Earth's land surface (HOOKE; MARTÍN-DUQUE; PEDRAZA, 2012), mining and mineral processing can influence landscapes far beyond the site of mineral extraction. Mining projects can pioneer investments in pristine regions and, by establishing infrastructure and facilitating access to previously undisturbed land, can trigger further land use change and consequent impacts on biodiversity (SONTER; ALI; WATSON, 2018).

Understanding and managing the negative impacts of mining on native forests is a matter of global concern (WORLD BANK, 2019a). This is particularly true in the Amazon, where forest loss dynamics have changed (KALAMANDEEN et al., 2018) due to the weakening of environmental legislation and environmental agencies to enforce requirements (CARVALHO et al., 2019). Mining activities play an important role in Amazon forest loss (HIRONS, 2011), due to both small-scale artisanal mining (ASNER et al., 2013; DEZÉCACHE et al., 2017; LOBO et al., 2016; SWENSON et al., 2011) and large-scale mining projects (SOUZA-FILHO et al., 2016).

Impacts of mining on forests can extend far beyond the sites of mineral extraction. For example, previous studies about large-scale mining in the Brazilian Amazon have shown that indirect impacts outside of mining leases caused 9% of deforestation between 2005 and 2015 – representing an area 12 times larger than the mines themselves (SONTER et al., 2017). These findings emphasize the importance of assessing and managing cumulative impacts of mining on forests.

In addition to forest loss, mining also fragments forested landscapes. This may be particularly true when mines establish new infrastructure to process and transport extracted materials. Forest loss and fragmentation generate persistent, deleterious, and often unpredicted effects on biodiversity worldwide (HADDAD et al., 2015; LARREY-LASSALLE et al., 2018; PEDLOWSKI et al., 1997), yet despite this, few studies have investigated these cumulative impacts of mining. Hereafter, we understand cumulative impacts as “a change in the environment caused by multiple interactions among human activities and natural processes that accumulate across space and time” (CCME, 2014).

This chapter explores the cumulative impacts of mining on forest loss and fragmentation in a case study region of the northeastern Brazilian Amazon. We focus specifically on quantifying historic changes within two mining regions that neighbor Renca – a biodiverse mineral-rich region currently off limits to mining. We use remotely sensing imagery and historic land use maps to quantify forest loss and determine changes in landscape structure. By investigating the impacts of mining within neighboring mining regions, we explore the potential consequences of opening Renca up for future mining development.

4.2 The National Reserve of Copper and Associates (Renca)

4.2.1 The historical development of the region

A mining reserve of 47,000 km² destinated only to mineral research carried by the “Companhia de Pesquisa de Recursos Minerais” (CPRM). Although the purpose of Renca is unrelated to conservation, its establishment has prevented mining companies from expanding into the region, and thus passively contributed to conserving the region's forests and biodiversity. In eastern Amazon, Renca is located in one of the most intact and remote areas of the Brazilian Amazon, far for the arc of deforestation (CARVALHO et al., 2019) and new deforestation frontiers (ARAÚJO et al., 2019; KALAMANDEEN et al., 2018) (Figure 11).

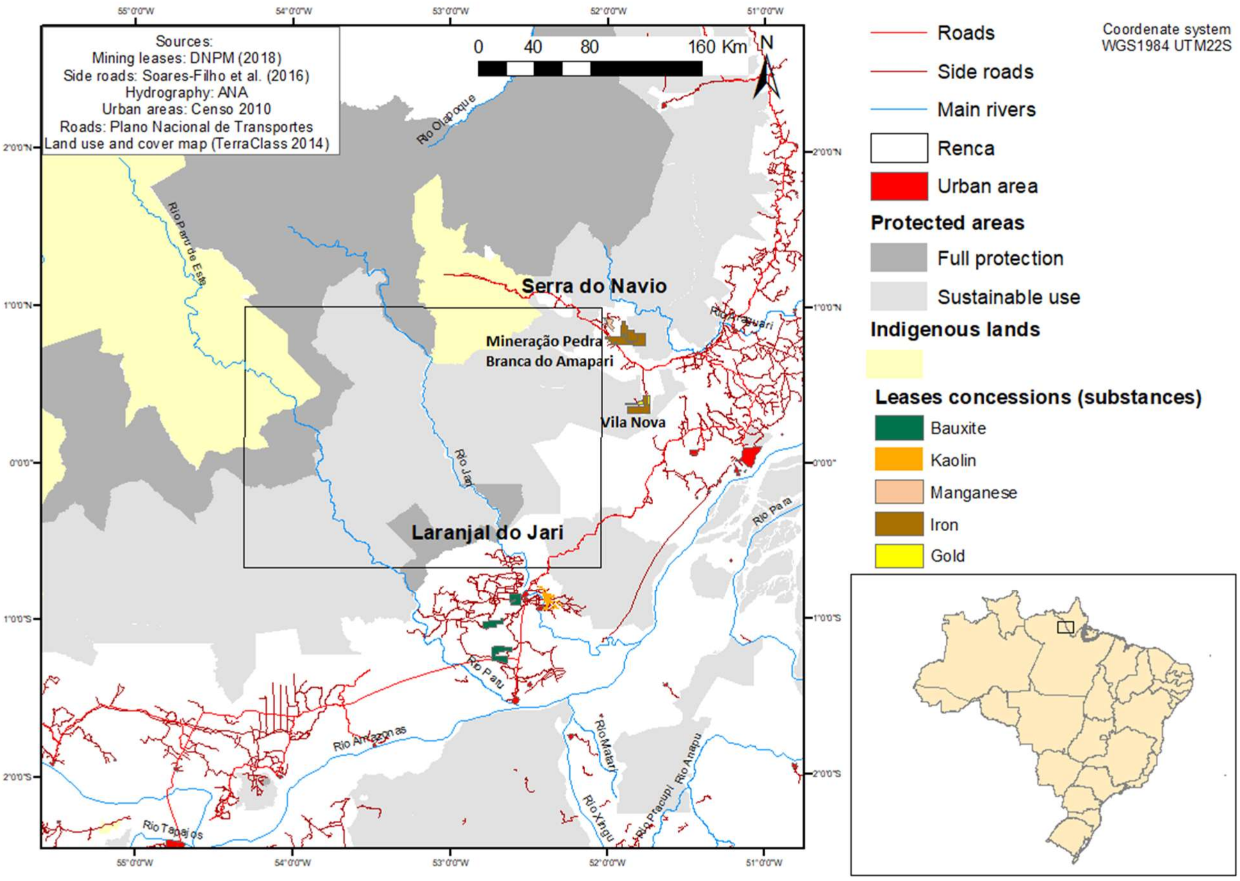


Figure 11. Location of Brazil's National Reserve of Copper and Associates (Renca) and its two neighbouring mining regions (Serra do Navio and Jari). The map also illustrates the mining leases that exist adjacent to Renca and the protected areas that currently exist within it.

A decree issued in August 22nd by the former Brazilian president Michel Temer (BRASIL, 2017a) suspended the previous decree of Renca creation in 1984 (BRASIL, 1984) (Figure 12). This act received attention of media and researchers due to the possible impacts caused by large scale mining and further occupation of the region. Given the pressure of the public opinion, the president decided to suspend the recently issued decree (BRASIL, 2017b), prohibiting large scale mining within Renca boundaries.

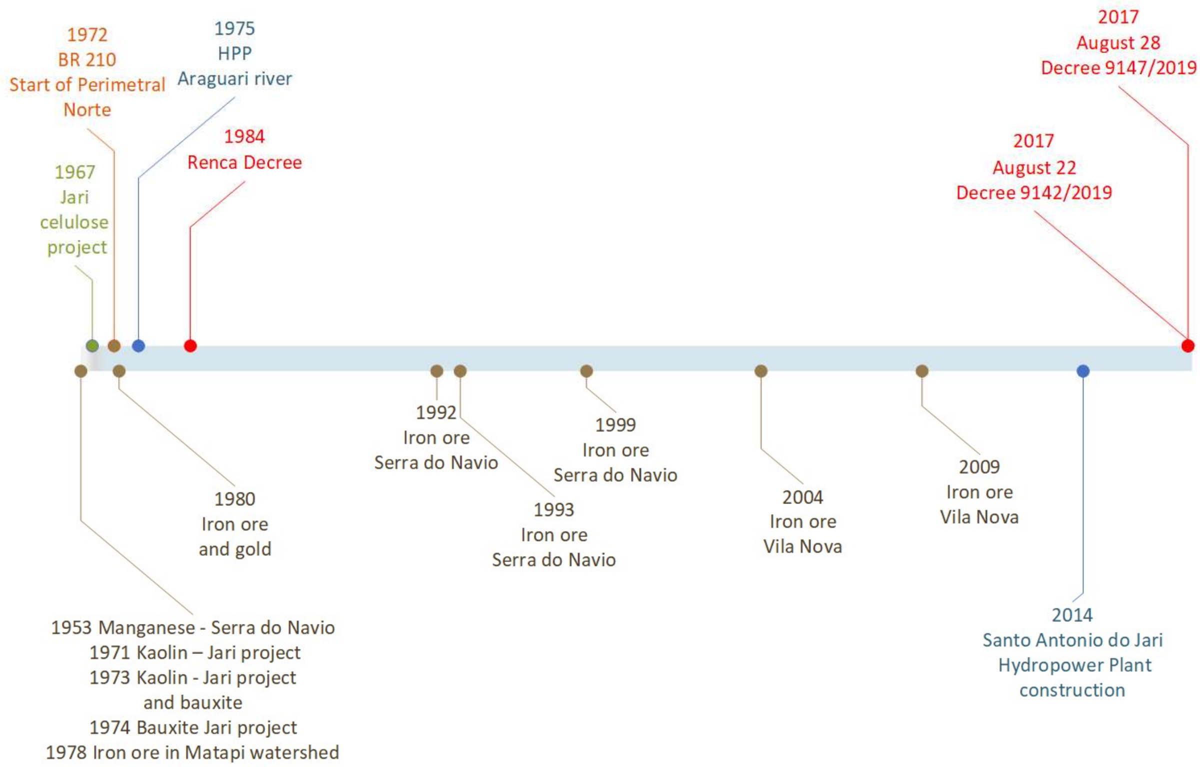


Figure 12. Timeline with the legislation and main projects developed in the region.

Currently, most of its territory is officially under mineral investigation and some industrial mining projects were development outside its borders (Figure 12). The creation of Renca came as an enforcement of the federal government to stop the advancing of international mining companies in the region. All mining concessions are under the Ministry of Mine and Energy authority and it is not possible to develop large scale mining projects. Renca contains valuable resources of copper, chromium, gold, iron, manganese, tantalum and tin (CPRM, 2001). Bordering Renca, there are two sectors of mining development, named here Serra do Navio and Jari (Figure 11).

Serra do Navio, northeast of Renca (Figure 11), was the site of initial mining development, with a manganese mine that commenced operation in 1957 (Table 6). Its installation required the construction of a 194 km long railway, a port in the Amazon river and two company towns, one in the mine site (Serra do Navio) and another adjacent to the port (Santana). The mine closed in 1997 due to depleted reserves, while the railway and the port remained in place and serviced iron ore development until the port broke up in March 2013. Gold mines in Pedra Branca do Amapari (Table 6) were first mined at an industrial venture in 1980, while iron ore was mined from the 1990s to 2013. The Tucano gold mine started operation in 2005 and received significant investments in recent years. In contrast, iron ore operations, initiated in 2007 (OLIVEIRA, 2010), are suspended due to the disruption in the Santana port and to judicial disputes involving companies and shareholders.

The current closed chromium mine initiated its operations at a small scale in 1988 and the exploited ore was trucked to the railway for delivery to customers (OLIVEIRA, 2010).

Table 6. Mining concessions in the two sectors surrounding Renca

Sector	Year of mining leases	Substance	Status	Associated infrastructure
Serra do Navio	1950	Manganese	Closed	Roads, village, railway, port
	1992; 1999; 2004	Iron ore	Care and maintenance	Roads
	1978	Clay	Closed	Roads
	1980; 2009	Iron ore	Care and maintenance	Roads, village, railway, port
	1975	Chromite	Closed	Roads
	1980	Gold	Operating	Roads
Jari	1971	Kaolin	Operating	Roads, village, port
	1974	Bauxite	Closed	Roads, village

The other mining sector is located to the south of Renca, near the mouth of the Jari river, where a kaolin mine commenced operation in 1976 (OLIVEIRA, 2010). It comprises an open pit, ore processing facilities and a fluvial port to ship minerals. It neighbors a paper pulp mill that was also developed in the early 1970s and is supplied by Eucalyptus planted in the area. A bauxite mine started operation in the 1970s and is now closed.

In addition to commercial mining, artisanal gold mining activities are a considerable driver of land use change and water pollution in the Renca region and surroundings. The Jari Ecological Station presented in the past artisanal gold mining activities and the impacts can be seen in the rivers nowadays (Figure 13). The garimpos are located mainly in Jari and Paru floodplains (SEMA , 2011, 2010) as well as in Pedra Branca do Amapari and Vila Nova (OLIVEIRA, 2010).

Figure 13. Areas with past artisanal gold mining actvities in the Jari Ecological Station.

The road system surrounding Renca (Table 7) comprises the unpaved BR-156, from Laranjal do Jari to Macapá and side roads connecting communities and rural properties to the urban centers (Figure 11). Laranjal do Jari, currently featuring about 50,000 inhabitants, was settled as a consequence of the construction of a large pulp mill owned by the company Jari Celulose. Besides the existing mine developments and roads, extraction of forest products (such as açaí berries, Brazil nuts, camu-camu) play a role in driving land use change and forest loss in the area (EMPRESA DE PESQUISAS ENERGÉTICAS, 2011). Agriculture and cattle ranching, important drivers of deforestation elsewhere in the Amazon, are less relevant in this region.

Table 7. Information about roads period of construction and current status

Road	Year of construction	Status
Perimetral Norte – BR 210 from Macapá to the Waiãpi Indigenous Land	1973	Unpaved
BR-156 from Laranjal do Jari to Macapá and BR-156 from Macapá to Oiapoque	1950s initial construction – 1985 partial pavement	Partially paved
BR156 from Oiapoque to French Guyana	2011 (operating since 2017)	Partially paved

4.2.2 Protected areas overlapping Renca

Renca territory comprises seven PAs, three with full protection, four aimed at sustainable use, and two indigenous lands (Figure 14). According to the National System of Conservation Units (SNUC), only four of the seven present the Environmental Management Plan. The Management Plan of *Parque Nacional Montanhas Tumucumaque* (ICMBIO, 2009) indicates the presence of an important cultural heritage in the indigenous lands (*Tiriyó, Wayana, Aparai* e Kaxuyana and Wajãpi). Given the mosaic of PAs and indigenous lands, the landscape is considered as homogenous, with dense rainforest (Figure 15). Some incursions can be seen close to Extractive Reserve (RESEX) of *Rio Cajari*, with urban settlements, reforestation and some pioneer formations (EPE, 2011).

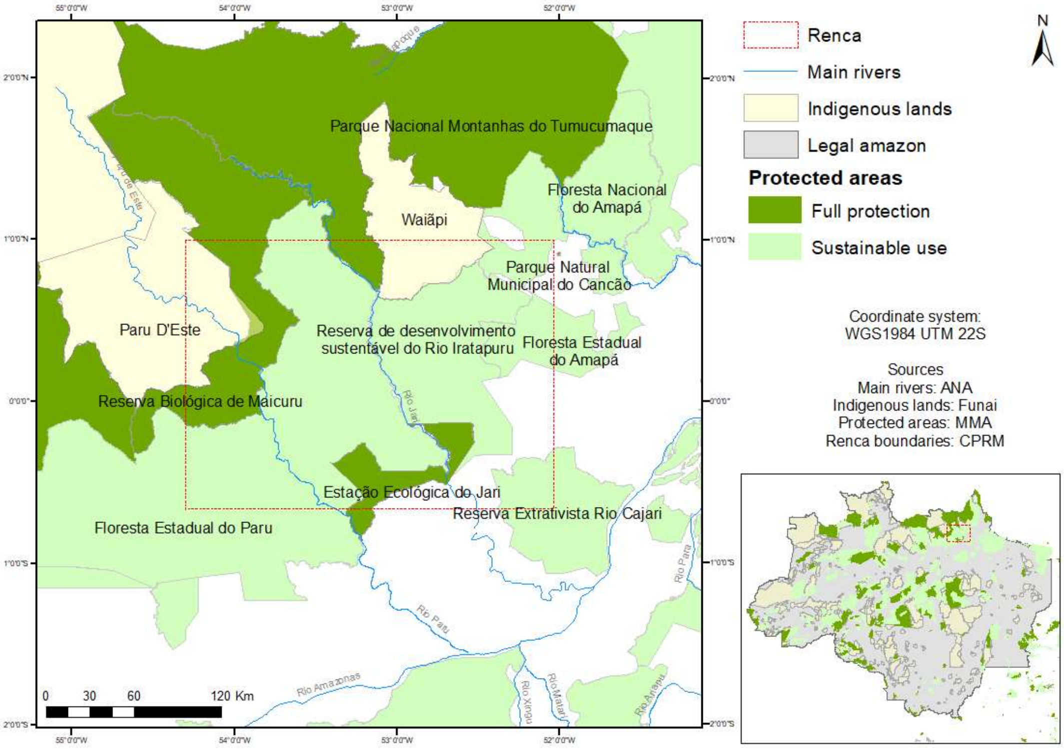


Figure 14. Renca protected areas.

Figure 15. Main entrance of Jari Ecological Station

Seven PAs and two indigenous lands are currently totally or partly inside Renca Under current legislation, no mining activity, including exploration is allowed inside full PAs and indigenous lands. Mining can be permitted in sustainable use PAs, provided it is in principle allowed in their individual management plan (Table 8). According to Federal Law 9985/2000, mining is not permitted in full protected units. Furthermore, according to the Federal Constitution it is not possible to legally mine in indigenous lands. In case Renca is extinguished, the current state of legislation allows mining, for example in the sustainable use units of *Paru* and *Jari* plains. This context emphasizes the importance of evaluating the potential effects of consolidated mining activities in the region.

Table 8. Protected areas in the Renca region

Type (1)	**Protected area / Indigenous Land**	**Year of creation**	**Area (km²)**	**Mining allowed?**
Full protection (1)	Jari Ecological Station	1982	2,309.9	No
	Tumucumaque National Park	2002	38,634.3	No
	Maicuru Biological Reserve	2006	11,747.0	No
Sustainable use (1)	Rio Cajari Extrativist Reserve	1990	5,316.4	No
	Rio Iratapuru Sustainable Develop. Reserve	1997	36,146.7	No
	Paru State Forest	2006	8,728.6	Yes (4)
	Amapá State Forest	2006	23,665.2	No
Indigenous lands (2)	Waiãpi (3)	1996	11969.1	No
	Rio Paru d'Este (3)	1997	6041.4	No

Notes: (1) Full protection and Sustainable use are categories of protection according to Federal Law 9985/2000 (2) Indigenous lands are not a category of protected area and are not regulated by Federal Law 9985/2000, but they contribute to biodiversity protection (3) the final step of official recognition of an indigenous land (called "homologation") has not been completed (4) Mining can be allowed in some areas according to the management plan.

Sources: http://www.funai.gov.br/ http://mapas.mma.gov.br/i3geo/datadownload.htm

4.3 Materials and methods

In this section we sequentially describe and justify the data sources and the methods of data analysis used in this research.

4.3.1 Data sources

The TerraClass project provides official maps of land use and cover of Brazilian Amazon (ALMEIDA et al., 2008), which have been previously used to quantify forest loss and fragmentation (LOBO et al., 2016; MILHEIRAS; MACE, 2019). These maps provide time series information at a 30 m spatial resolution (classified from Landsat images), suitable for studying large areas (BARROS et al., 2017; NEVES et al., 2017). Maps from the years 1991 and 2014 were selected to minimize cloud cover (the non-observed area corresponds to less than 0.05% of the total area). A description of the main land use and cover classes depicted by TerraClass is shown in Table 9.

Table 9. Description of the land use and cover classes of TerraClass maps (ALMEIDA et al., 2008)

Land use and land cover class	Description
Forest	Primary native forest
Other native vegetation	Areas in the Legal Amazon fringe featuring non forest native vegetation, such as mangroves
Secondary vegetation	Areas that were clear-cut and are at an advanced stage of regeneration featuring trees and shrubs.
Deforestation	Deforested areas covered by soil, shrubs, herbage and felled trees with no defined land use at this stage, defined as areas that were mapped by PRODES project as deforested in the year of analysis
Shrubby pasture	Pasture with herbage and coverage by species of grass between 50% and 80% associated to the presence of shrubby vegetation with coverage of 20% and 50%
Herbaceous pasture	Pasture with predominance of herbage and coverage between 90 and 100% by different species of grass
Agriculture	Extensive areas with predominance of annual crops, specially grains, highly technological such as certified seeds, enriched soil, chemicals, fertilizers, mechanization among other resources
Silviculture	Other forest types with economical interest with homogenous cover such as Pinus and Eucalyptus
Urban area	The area represents the concentration of population, forming small inhabited places, villages and cities with street design and high density of dwellings
Mining	Areas of mining extraction with bare soil and deforestation
Hydrography	Water bodies
Others	Areas not encompassed by other categories such as rocky outcrops or montaintops, river shores and sand banks, among others
Non-observed areas	Areas not possible to be interpreted due to clouds or cloud shade of the satellite overpass or recently burned areas

4.3.2 Data analysis methods

The analysis was conducted in two steps (Figure 3). First, we quantified historic forest loss between years 1991 and 2014. Second, we quantified the implications of these losses for forest fragmentation. All analyses were quantified using the Dinamica EGO™ modelling software (SOARES-FILHO; CERQUEIRA; PENNACHIN, 2002) and thematic maps were handled using ArcMap™ 10.7.1.

We quantified the total amount of forest loss within Renca's PAs to investigate potential indirect impacts of neighboring mines (Box A in Figure 17). We also quantified forest loss within the two mining sectors surrounding Renca (Serra do Navio and Jari; Figure 16). These regions were defined using 70 km buffers extending from the current mining leases (SONTER et al., 2017), which found mining causes deforestation up to 70 km from leases in Brazilian Amazon. We also quantified forest loss occurring close to roads crossing each mining sectors, Serra do Navio and Jari, using different distances from roads, as shown in Box B in Figure 17, in order to investigate the indirect deforestation driven by transportation infrastructure.

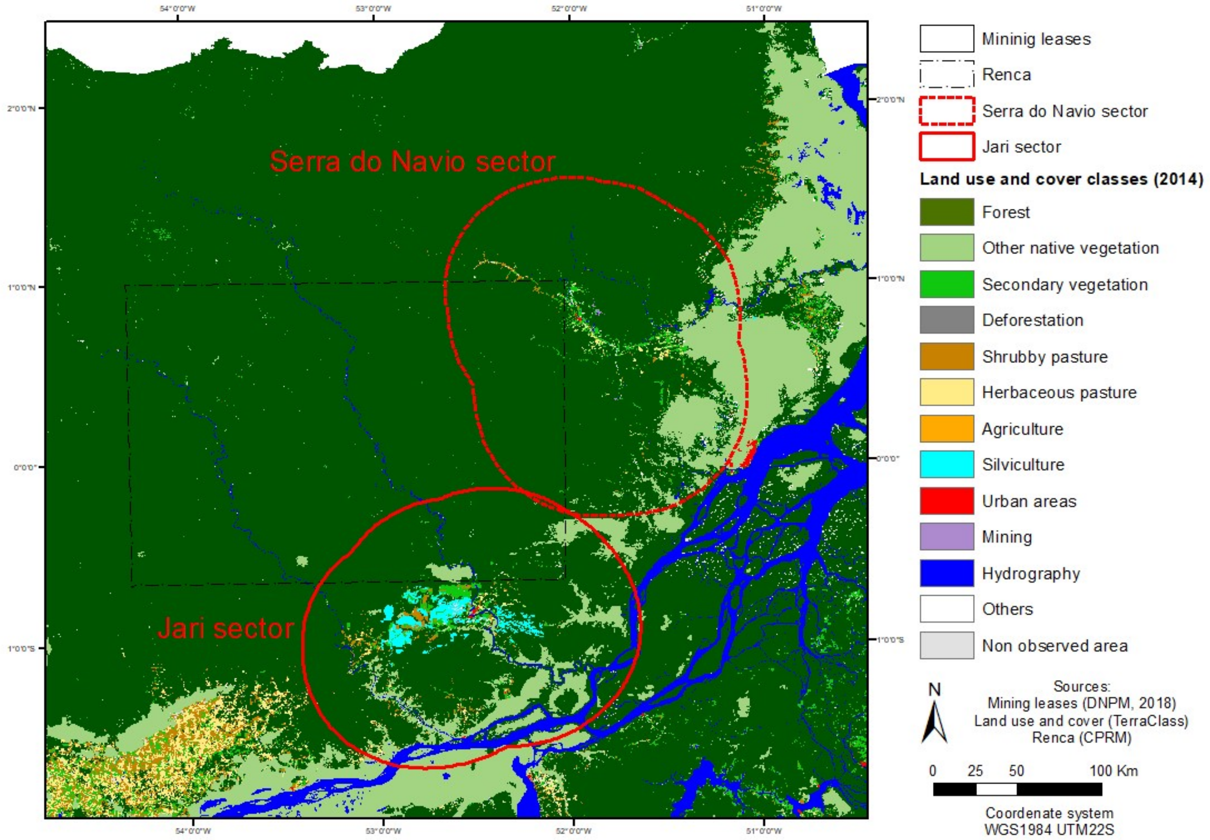
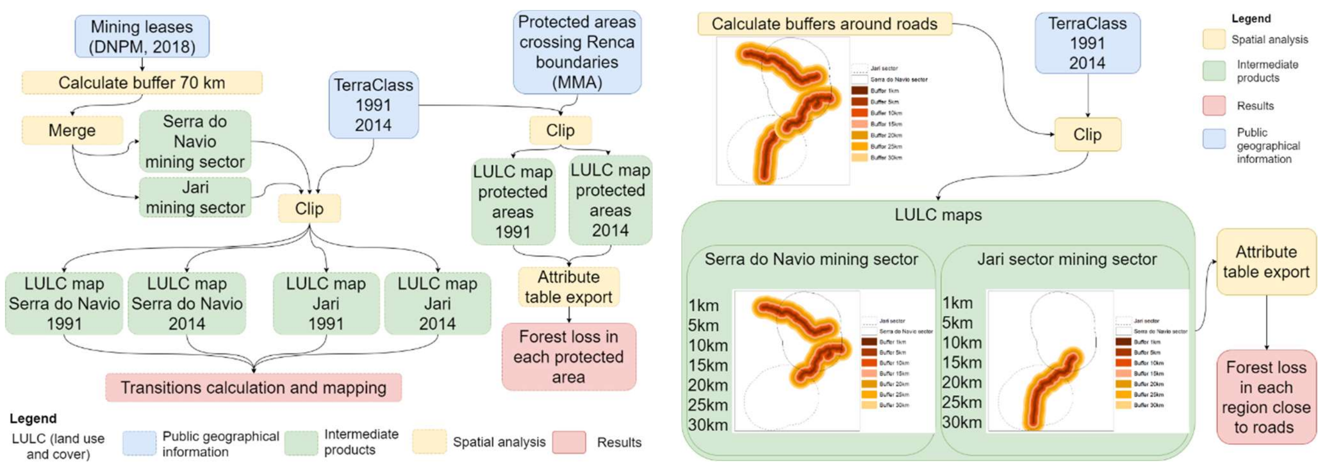


Figure 16. Land use and cover map from TerraClass and the two study sectors. Source: (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)



A B

Figure 17. Main steps of data analysis including geographical information used, spatial analysis and results (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)

The landscape metrics were applied in the Jari and Serra do Navio mining sectors to analyze the effects of mining activities in the landscape patterns. The metrics used to evaluate landscape changes in forest cover, and the rationale of metrics choices, are shown in Table 5. The analysis of landscape patterns in the Amazon is important for understanding not only ecological effects of forest loss, but also for considering anthropogenic activities and, consequently, to design better policies to manage deforestation (ROSA; GABRIEL; CARREIRAS, 2017) and ecological restoration.

Table 5. Landscape metrics used. Source: adapted from (MCGARIGAL; MARKS, 1995)

Metric	Rationale	Reference
Number of patches (NumP)	It supports one central hypothesis of landscape fragmentation, as larger is the NumP, there are more patches in the landscape and increased level of fragmentation	(CABRAL et al., 2018; COLSON; BOGAERT; CEULEMANS, 2011; MALAVIYA et al., 2010)
Mean patch size (MPS)	Broadly used to compare fragmented landscapes, the MPS is used as reference for the discussion of the minimum habitat requirement for some species. In this work, it is used to compare the patch sizes in both years of analysis and feature the structure forest cover throughout the landscape	(CABRAL et al., 2018; COLSON; BOGAERT; CEULEMANS, 2011; FAGIEWICZ, 2014; MALAVIYA et al., 2010)
Edge density (MPE)	It is a core area metric, which aim to inform about the patch's edges in relation to the total landscape area. The edges effects can influence different organisms and species, but in this present work it is used as another metric related to the degree of fragmentation in the forest cover	(COLSON; BOGAERT; CEULEMANS, 2011; FAGIEWICZ, 2014; ROSA; GABRIEL; CARREIRAS, 2017)
Mean nearest neighbor (MNN)	It measures the degree of patch isolation. Mostly used as a metric to evaluate fragmentation, it can inform about patches spatial distribution and landscape structure	(FAGIEWICZ, 2014)

4.4 Results

4.4.1 Forest loss in Renca's protected areas

We found forest loss occurred within many of Renca's PAs. Those with the largest percentage loss were the Maicuru Biological Reserve (0.37%), Paru State Forest (0.13%) and in the Rio Paru d'Este IL (0.10%) (Table 6). The Tumucumaque National Park, Iratapuru Sustainable Development Reserve and Waiãpi IL, located in more remote areas (Figure 18), present a lower percentage of forest loss in the period analyzed (Table 6).

Table 6. Forest loss inside protected areas in the Renca region according to TerraClass dataset. role in conservation.

Type	**Protected area**	**Area (km²)**	**Forest loss (km²) and percentage of forest loss in relation to the total PA area (1991-2014)**	**Proximity to mining leases**
Full protection	Jari Ecological Station	2,309.9	2.2 (0.09%)	Yes
	Tumucumaque National Park	38,634.3	15.1 (0.04%)	No
	Maicuru Biological Reserve	11,747.0	43.1 (0.37%)	No
Sustainable use	Rio Cajari Extrativist Reserve	5,316.4	1.1 (0.02%)	Yes
	Paru State Forest	36,146.7	46.2 (0.13%)	No
	Rio Iratapuru Sustainable Development Reserve	8,728.6	6.7 (0.08%)	No
	Amapá State Forest	23,665.2	90.6 (0.04%)	Yes
Indigenous lands	Rio Paru d'Este	11969.1	11.4 (0.10%)	No
	Waiãpi	6041.4	0.4 (0.01%)	No

Note: See Table 3 for details about protected areas and indigenous lands and Figure 3 for details about the method of calculations

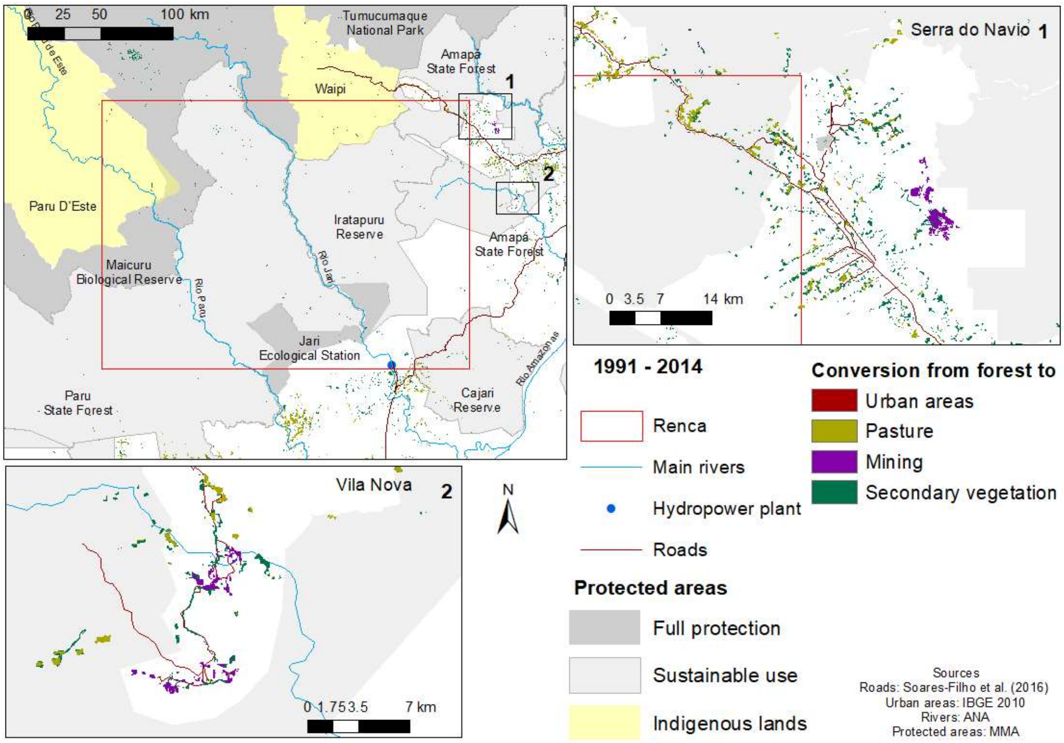


Figure 18. Forest loss predominantly occurred near to roads in the Renca region. Source: (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)

4.4.2 Forest loss inside mining sectors

Forests covered approximately 73% of the Jari sector and 85% of Serra do Navio sector in 2014 (Figure 19) and these regions lost 280 km² and 317 km² of forest cover since 1991, respectively. We found 15.5 km^2 of forest was directly cleared by mining activities in both mining sectors. Most of this area was lost in Serra do Navio sector, as there was no expansion of mining infrastructure in Jari, only expansion of silviculture and pasture were detected. As deforestation means the clear-cut area compared to the previous year, it is not mapped with a designated land use, being an intermediate stage in the conversion of forest to other land uses.

Most forest lost was converted to either secondary vegetation, other native vegetation or pasture (Figure 19). Conversion of forest to other native vegetation – an unlikely change -, such as conversion to mangrove or riparian forest (Table 4) could result from image classification due to changes in the seasonality of the water level. Forest conversion into mangroves may have been registered because the first map was based on an image taken during the dry season, while the second was taken during the flood season.

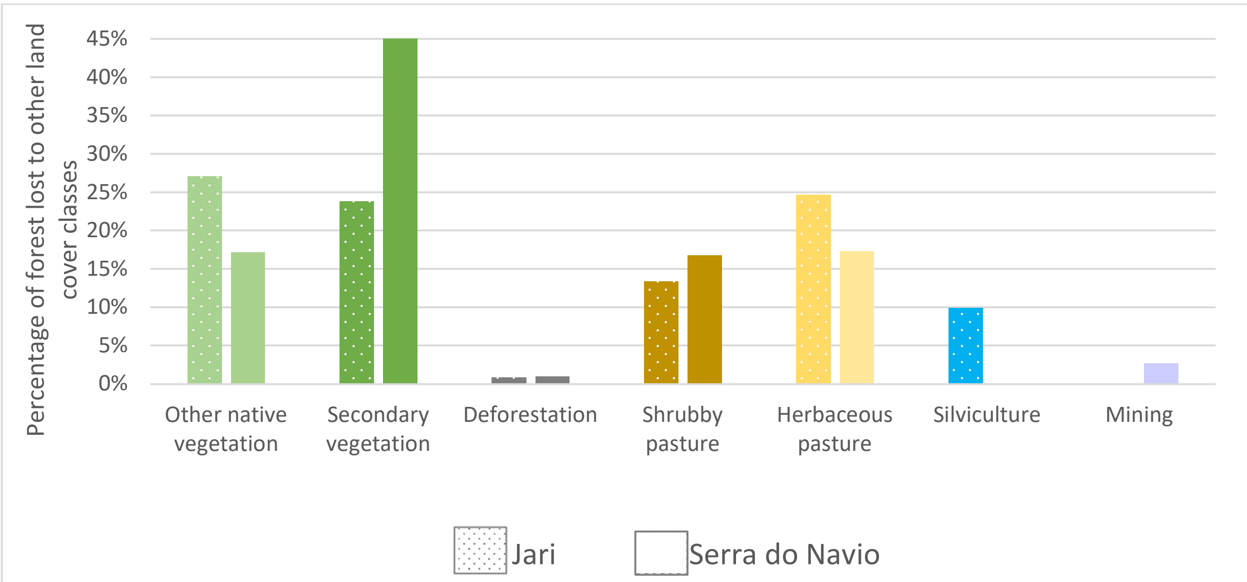


Figure 19. The main forest loss was to other native vegetation and secondary vegetation in both, Jari and Serra do Navio sectors. Source: (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)

4.4.3 Forest loss adjacent to roads

Roads are one of most important drivers of deforestation in Amazon (Laurance et al., 2009). Here, we quantified the forest loss that occurred within 30 km of roads (Soares-Filho et al., 2004) within each mining sector. The percentages shown in Figure 20 are related to the total forest loss in the period of 1991 to 2014, depicting how much of the total loss occur close to roads considering the distance of 1 km, 5 km, 10 km, 15 km, 20 km, 25 km and 30 km. We found 75% of forest loss occurred within 30 km of roads in Serra do Navio region and 37% in Jari. The remaining 25% of forest loss in Serra do Navio occur farther than 30km of roads, thus, beyond its assumed influence. In Jari, the major percentage of forest loss was not found close to roads (63% farther than 30km from the roads). Differences between the two regions can be explained by the occupation in the region occurred during the years of 1991 and 2014, which was not only related to roads but also other infrastructure built in the area, such as the Jari Celulose industrial area and silviculture expansion (Figure 20).

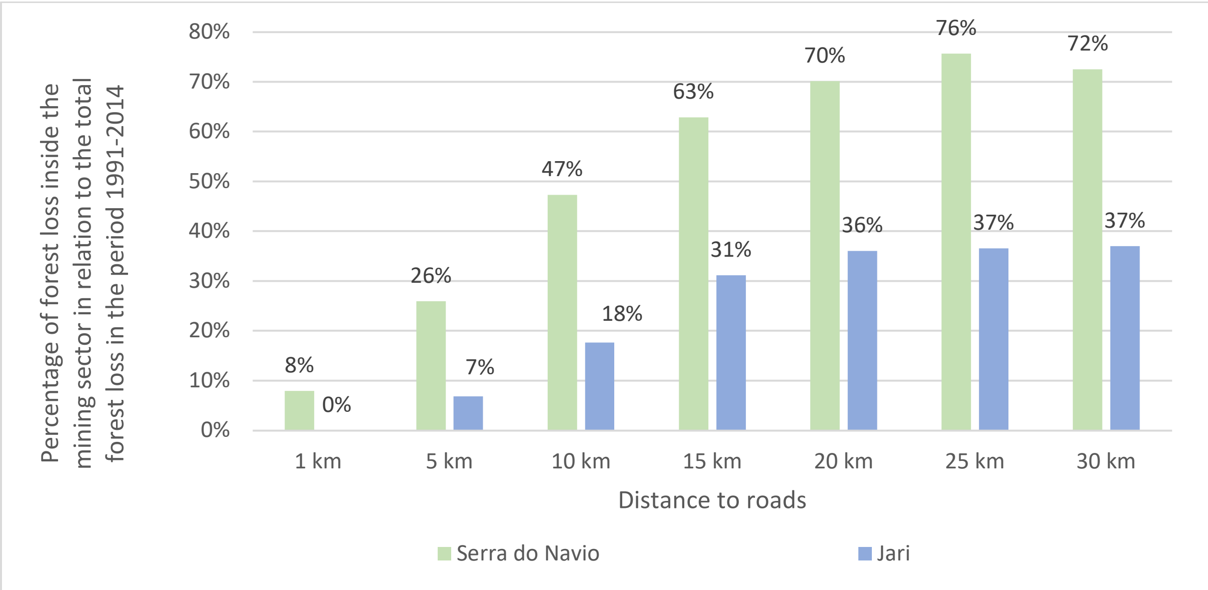


Figure 20. Forest loss around roads in both mining sectors. The larger percentage of loss was found in Serra do Navio sector, evidencing the importance of roads in land cover change. Source: (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)

The landscape by 1991 already presented fishbone patterns of fragmentation (Arima et al. 2016) caused mostly by the BR-210, the "Perimetral Norte" dirt road, which was constructed in the 1970's (B, Figure 21). In the region of iron ore expansion in Serra do Navio (C, Figure 21), forest loss is observed in small patches surroundings the side dirt road. In the Jari sector (A, Figure 21), it is possible to note the intensification of the fishbone pattern in the roads near the Eucalyptus plantation and roads of the Jari Celulose project, both to the East and West of the plantation (A, Figure 21).

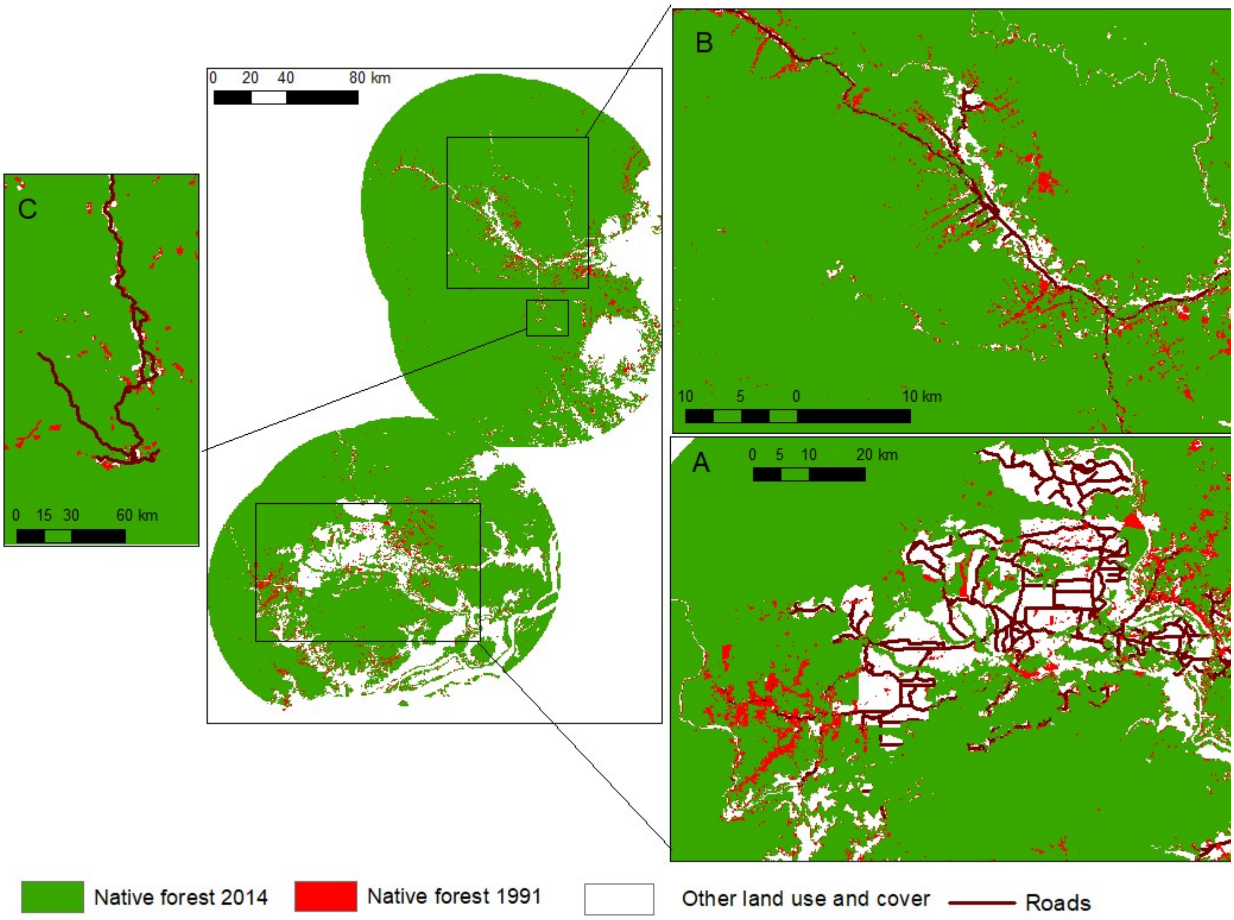


Figure 21. Forest loss in the two mining sectors for comparing the pattern generated by forest considering the two years of analysis. Source: (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)

4.4.4 Landscape structure analysis

For both mining sectors neighboring Renca, we found a decrease in the number of native forest patches (44% in Jari and 49% in Serra do Navio) and increased mean patch size between 1991 and 2014 (Table 5; Figure 22), the mean patch size doubled in Jari and was multiplied by 27 in Serra do Navio). We also found an increased mean distance between patches and considering the wider distribution of distance values analyzed, being larger the differences in Serra do Navio. We noted an increased edge density with a similar distribution in both mining sectors (Figure 22). These two metrics results demonstrate a change in both spatial distribution and shape of forest patches.

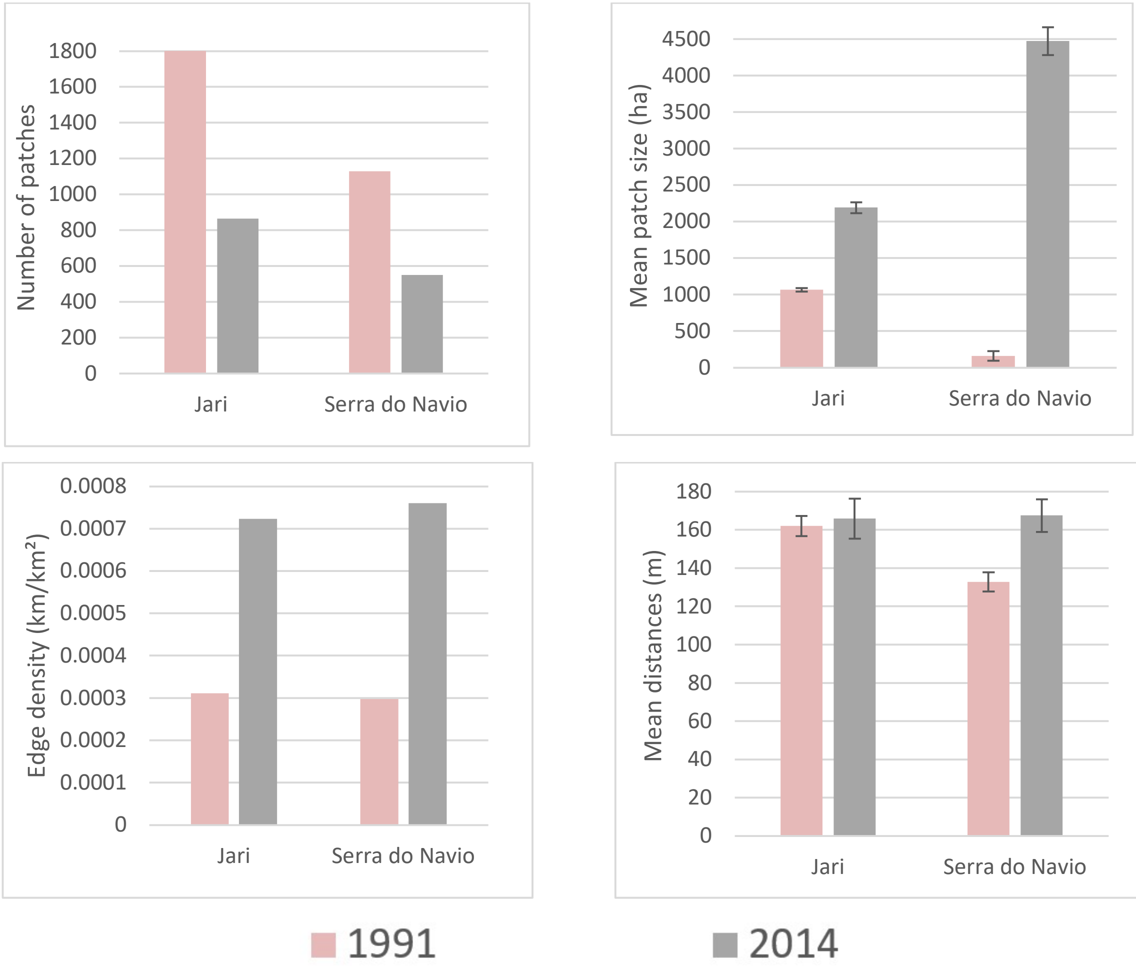


Figure 22. Results of changes in the landscape considering the forest class in the two mining sectors. Source: (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020)

4.5 Discussion

Mining the Amazon has negatively impacted native forests and their biodiversity (ASNER et al., 2013; DEZÉCACHE et al., 2017; SWENSON et al., 2011). Previous work has suggested these impacts extend great distances from mine sites (SONTER et al., 2017) and have potential to cause significant impacts on biodiversity by fragmenting landscapes. Here, we investigated potential impacts of mining within and surrounding the east Amazonian region of Renca. We found that mining and other infrastructure often associated with mines and necessary for mining development (i.e. roads) caused significant forest loss and fragmentation within Renca’s neighbouring mining sectors. We also found that Renca’s PAs are not currently immune to forest loss. Our results suggest that permitting new mines will place additional pressure on Renca’s biodiversity. Below we discuss these results, exploring differences between the Serra do Navio and Jari mining sectors and what these differences mean for predicting the impacts that may emerge within Renca. We then suggest some measures to manage and mitigate negative consequences if mineral development is to proceed.

The two sectors that neighbour Renca, Serra do Navio and Jari, had very different dynamics of landscape change considering the mainly forest converted to silviculture in Jari and more forest being degraded to shrubby pasture (Figure 19). In Jari, mining started after infrastructure was developed for the large tree plantation and pulp mill construction (1970's). Kaolin is used alongside cellulose for paper production, thus the existence of high-grade kaolin deposits very close to the industrial development prompted mining investment. The cellulose industry has control of the properties and thus indirectly control the human expansion over Jari river. In this case, the regional development promoted by the Jari Celulose changed considerably the landscape in the past, promoting the pasture and silviculture expansion. Between 1991 and 2014, forest loss occurred mainly along the Jari Celulose Eucalyptus plantation and side roads (Figure 21), evidencing the importance of the expansion of silviculture in the region (Figure 19).

In contrast, in Serra do Navio, mining pioneered occupation of the region and provided transportation and other infrastructure (1950's) that was subsequently used for other developments. Consequently, the areas deforested in Serra do Navio occurred mainly alongside roads (Figure 20), without competition with silviculture and urban areas. Mining expansion occurred mainly in Vila Nova (after 2000) and Pedra Branca do Amapari (leased in 1980 with expansion in the recent years (Figure 19). Although the new mines used the existing roads and railway to export production, more than 75% of forest loss occurred within 30 km of the region's main roads (Figure 20). Therefore, even considering a perspective of shared infrastructure, the induced forest loss can play an important role in terms of area of forest lost as well as in the landscape structure, intensifying the fishbone pattern.

The extensive forest loss caused by mining has potential to cause significant effects in the landscape patterns and structure (MALAVIYA et al., 2010). Previous studies in post-mining areas of opencast lignite mines indicated the increased level of fragmentation due to a higher number of patches, decreased average size of patches, increased edge effects and increased nearest-neighbor distance (FAGIEWICZ, 2014). The landscape structure changed to larger, more cohesive forest patches. Forest loss occurred mainly alongside the fishbone pattern of fragmentation caused by the previously constructed roads (Figure 21).

The trajectories of forest loss within these two mining sectors illustrate the importance of assessing and mitigating impacts of mining of forest loss in a regional scale. Therefore, if new mining projects are to go ahead, either within Renca or its neighboring mining regions, two policy actions may help to minimize negative impacts on biodiversity. Firstly, Environmental Impact

Assessments (EIA) for new projects in the region should address cumulative effects on native vegetation both inside mining leases and offsite. Secondly, it is necessary to strengthen the protected area policies and their enforcement in these regions.

World Bank, (2019a) states that current Environmental and Social Impact Assessments do not properly address all the effects of mining developments on native vegetation and natural habitats, offsite changes caused by roads, ore transportation infrastructure and human occupation. In particular, the Brazilian EIA process features several weaknesses that could hardly address those impacts, including poor consideration of cumulative effects (NERI; DUPIN; SÁNCHEZ, 2016) and poor use of baseline data for impact analysis and mitigation proposals (DUARTE et al., 2017). Although progressive improvement of EIAs of mining projects were documented in Brazil (LANDIM; SÁNCHEZ, 2012), the recent trend has been towards dwindling its scope (FONSECA; SÁNCHEZ; RIBEIRO, 2017). This research reinforces the need for EIA to address offsite and indirect impacts of mining in forest regions. Improving environmental legislation is out of question in Brazil, on the contrary, federal government and congressmen have been acting to weaken every single piece of legislation (ABESSA; FAMÁ; BURUAEM, 2019; PEREIRA et al., 2019). One possible way of strengthening EIA without changing legislation is through more attention to the scoping phase (BORIONI; GALLARDO; SÁNCHEZ, 2017), if carefully drafted terms of reference are written with clear and straightforward directions to the assessment of indirect and cumulative impacts, as found in EIA practice for offshore oil and gas activities (VILARDO; LA ROVERE, 2018).

PAs within the Amazon play an important role in regulating human occupation, affecting development of mining and other projects (PFAFF et al., 2015). In the 35-year period after Renca's creation, conservation policies changed significantly in the Brazilian Amazon (NAUGHTON-TREVES; HOLLAND, 2019), leading to the creation of more PAs. Seven PAs and two indigenous lands now overlap Renca (Table 3) and the Tumucumaque Key Biodiversity Area (LANGHAMMER; BUTCHART; BROOKS, 2018) lays near Renca's border. Under Brazil's legislation, there are two major classes of PAs: full protection, such as national parks and ecological stations, where mining is not allowed, and sustainable use, such as national and state forests as well as reserves, where mining is in some cases allowed, according to the management plan of each PA. In indigenous lands, which are regulated by different legislation, mining is currently not allowed, but proposals for amendment are under discussion in the National Congress.

In the Jari sector, PAs played a role in restraining deforestation and expansion of tree plantation, as the Jari Ecological Station blocked further industry expansion. In contrast, PAs in the Serra do Navio sector were created after 2000 and a different historical development in the region took place. Although historic mining developments, artisanal gold mining and infrastructure development have already impacted sustainable use PAs, their existence is a major formal barrier for large-scale mining development in Renca. If in practice they can hardly stop artisanal illegal mining, no major industrial mine can obtain an environmental license if mining is not allowed inside a PA.

To deal with these legal barriers to mining in Renca, the government can take two approaches. Firstly, PAs can be downgraded, downsized, or degazetted, and such moves have already been tried in the Brazilian Amazon (PACK et al., 2016), although they are legally controversial and would certainly be challenged in the Courts. Secondly, sustainable use PAs, where mining is currently not allowed, can have their management plans revised to allow exploration and mining. Revising management plans is a relatively easy and straightforward process, as those plans are, or at least should be, updated periodically. However, it is important to emphasize the hurdles in the enforcement and monitoring policies, being remoteness of Renca a contributing factor, limiting the access to control all borders and total PA extent. As 55% of the Renca area is covered by sustainable use PAs, changing their management plans is a likely path if government policy shifts towards fostering mining.

4.6 Conclusions

Loss of forests, including those inside PAs, and intensified forest fragmentation surrounding roads are evident from studying historical deforestation trajectories in a portion of northeastern Brazilian Amazon surrounding Renca – an area closed to mineral exploration since 1984.

If Renca would be opened to industrial mining, it is necessary to balance the short-terms economic gains with the long-term impacts on biodiversity. The management plans are a very important way to regulate the occupation, setting strict rules for PAs protection, as already seen, for example, in Carajás National Forest. Therefore, it is necessary to strengthen the enforcement of PAs legislation in the Renca, requiring the periodical update of management plans with the aim to avoid impacts of future large projects in the region.

Future research could evaluate the cumulative impact of combined new mining developments and associated infrastructure to be expanded in case of Renca be opened for commercial mining. The main contribution would be to test different scenarios considering projects configuration and plausible changes in PAs regulations to support decisions in the context of EIAs of future projects in the region.

4.7 Research limitations

Chapter 4 presents a specific analysis of forest loss and fragmentation during the years of 1991-2014. Fragmentation analysis did not capture all effects of road construction in breaking apart forests because some roads were constructed before the period analyzed. The use of publicly available products certainly reduces the time dedicated to developing the analysis but requires considerable work to completely understand the database preparation and its main constraints. The class description of TerraClass dataset was conceived for the entire Brazilian Amazon and some adjustments for the Renca region were required. For example, the class of “other native vegetation” is described as mangroves for the entire Amazon, but in the Renca region, it is represented by fields and grasslands – open areas with savannas.

5 The influence of protected areas in cumulative impacts of mining and associated infrastructure on forests

Objective

This chapter aims at analyzing the cumulative impacts on forests and biodiversity caused by facilitating mining and associated infrastructure inside PAs boundaries in different scenarios.

Preface

The possibility of Renca decree be suspended raises the discussion about potential future impacts that would be caused by large scale mining projects. Scenarios are proposed using a cellular automata model to simulate different scenarios of PADDD in the Renca region. This chapter contributes to the discussion of how PAs can influence on future developments in the region. Although the direct footprint due to clearing for mining expansion is smaller than the indirect deforestation of road expansion and human occupation, it affects highly biodiverse areas and fragments the landscape. The impacts of roads on forest loss and fragmentation are dependent on where are the PAs located in the landscape.

This chapter supports this thesis by: (i) evaluating the direct and indirect impacts of mining in different scenarios of PADDD; (ii) quantifying the relevance of direct and indirect mining deforestation in areas with the high interest in biodiversity conservation; (iii) quantifying the impacts of roads on fragmentation and highly biodiverse areas caused by PADDD. This chapter is organized as follows: introduction, context of PADDD in Renca, main results, conclusions and finally the methods with land use and cover database, modelling information and biodiversity analysis.

Related publication

Siqueira-Gay, J. Metzger, J.P.; Sánchez, L.E.; Sonter, L.J. Strategic planning needed to mitigate landscape-wide impacts of opening protected areas to mining. *In prep.*

Abstract

Global demand for minerals drives mining expansion and pressures protected areas to be downgraded, downsized, and degazetted (PADDD). For new projects, mining causes deforestation within leases and constructing additional linear infrastructure, extensively impacts forests and landscapes. To test the influence of PADDD on forests and biodiversity in response to facilitating mining and new roads to access mineral rich areas, we modelled land use changes in a region under pressure for economic development in the Brazilian Amazon, the National Reserve of Copper and Associates (Renca). We found that facilitating roads construction for reaching mineral deposits within protected areas would clear from 4,324 km² up to 7,809 km² of forests. Although mining direct deforestation for footprint expansion is up to 60 times smaller than indirect deforestation resulted from infrastructure development, areas with high biological importance would be affected by mining. The road network required to reach mineral deposits within protected areas not only would affect directly high biodiversity values but would increase fragmentation, depending on which areas remain protected across the landscape. Therefore, mining-driven PADDD decisions should be planned in an integrated and strategic way considering the access of multiple mining areas and optimizing their design to prevent perversely biodiversity loss.

5.1. Introduction

PAs creation and management are fundamental for global ecosystems conservation (ADAMS; IACONA; POSSINGHAM, 2019). However, the growing political pressure on PADDD for mining (NAUGHTON-TREVES; HOLLAND, 2019; RUDKE et al., 2020; SIQUEIRA-GAY et al., 2020a) and associated infrastructure (BARBER et al., 2014; EL BIZRI et al., 2016; FERREIRA et al., 2014a; MASCIA; PAILLER, 2011; SIQUEIRA-GAY; SÁNCHEZ, 2020) is threatening biodiversity (ADAMS; IACONA; POSSINGHAM, 2019; COOK et al., 2017; KRONER; KRITHIVASAN; MASCIA, 2016; MURGUÍA; BRINGEZU; SCHALDACH, 2016; PAIVA et al., 2020) and ecosystem services (SIQUEIRA-GAY et al., 2020a) conservation. The mining industry has played an important role in downgrading PAs by increasing human-related activities within its boundaries and associated infrastructure projects also downsize PAs limits to facilitate the construction of new developments (MASCIA et al., 2014).

Global demand for minerals drives mining expansion (SONTER et al., 2014a; SWENSON et al., 2011), requiring associated infrastructure for project implementation and operation (BEBBINGTON et al., 2018). For new developments, mining causes deforestation within leases, and indirectly clear forests by additional infrastructure, especially when operating in remote and untouched areas (BEBBINGTON et al., 2018; SONTER et al., 2017). New roads and ore transportation infrastructure fragment forests (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020) and intensify deforestation patterns (FERRANTE; FEARNSIDE, 2020), clearing more forests than the open pit area and mining project alone (SONTER et al., 2017). Noting the proximity of mining to biodiverse areas (ARMENDÁRIZ-VILLEGAS et al., 2015; KOBAYASHI; WATANDO; KAKIMOTO, 2014; MURGUÍA; BRINGEZU; SCHALDACH, 2016; SONTER et al., 2020), managing and conserving forests is of primary importance for promoting sustainable mining (BEBBINGTON et al., 2018).

The impacts of facilitating projects within PAs could be perceived locally (KRONER; KRITHIVASAN; MASCIA, 2016) and extent kilometers far away (SIQUEIRA-GAY et al., 2020a), such as clearing forests and increasing carbon emission (FORREST et al., 2015), adversely affecting biodiversity (PAIVA et al., 2020) and the cultural diversity they may shelter (FERREIRA, 2018; NAUGHTON-TREVES; HOLLAND, 2019). EIA responsible for assessing these impacts of new projects in remote and intact areas underestimate the area to be affected by roads (KILLEEN, 2007; VILELA et al., 2020), and mining (RITTER et al., 2017; WORLD BANK, 2019a), especially the cumulative impacts on biodiversity (WHITEHEAD; KUJALA; WINTLE, 2017). Consequently, fewer mitigation efforts and compensation measures are legally required for new projects in these regions, posing high risks for biodiversity conservation.

Decisions to open up PAs for development are either done in a piece-meal way (KRONER et al., 2019; NAUGHTON-TREVES; HOLLAND, 2019) or focus on easing restrictions across entire categories of PAs (MASCIA; PAILLER, 2011; SIQUEIRA-GAY et al., 2020a), failing in considering landscape dynamics (LÓPEZ et al., 2020) and the role of individual PAs in broader conservation initiatives (FERREIRA, 2018; LÓPEZ et al., 2020; MASCIA; PAILLER, 2011). This study aims at testing how facilitating mining and new access roads to mineral-rich areas into PAs would affect forests and biodiversity. By simulating land use changes, we investigated a biodiverse area under pressure for economic development in the Brazilian Amazon, the National Reserve of Copper and Associates (Renca).

5.2. Mounting pressure to PADDD in north-eastern Amazon

The Brazilian Amazon is the place of several PADDD proposals (NAUGHTON-TREVES; HOLLAND, 2019). In Renca, a mining reserve located in the north-eastern Amazon, no mining is allowed within its borders by decree and PAs currently cover 90% of the 47,000 km² of Renca (Figure S4 in Appendix 3). There are two indigenous lands, where no economic activities are allowed; three full protection areas, where only scientific research is permitted and; four sustainable use PAs, where activities such as mineral research, environmental educational project, and scientific expeditions are permitted.

In 2017, the Brazilian Federal government suspended the decree for allowing large scale mining projects to operate in the area (WWF, 2017b) but given the media and NGO's pressure, the president decided to re-establish the decree. If the decree would be suspended, changes in PAs regulations will be needed to permit mining expansion. According to the management plan of Floresta do Paru sustainable use area (SECRETARIA DE ESTADO DE MEIO AMBIENTE, 2010) and Reserva do Maicuru full protection area (SECRETARIA DE ESTADO DE MEIO AMBIENTE, 2011) large scale mining is already a threat for conservation. Moreover, there is a growing pressure over Brazilian indigenous lands by a recent bill proposal to open these areas for mining (SIQUEIRA-GAY et al., 2020a). Despite the awareness about the importance of the Renca region to biodiversity and ecosystem service in this region, the current government showed a strong willingness to open the region for mining.

We modelled four scenarios of PADDD in Renca to simulate the impacts of road network dedicated to permitting mining expansion (Figure 23). The general rate of deforestation in each scenario depends on the area not covered by PAs (Table S 19 in Appendix 3). In the Business as Usual (BAU) scenario, no PAs would be opened and the trend of past years was projected considering the existing road network. The Indigenous Lands (IL) scenario simulates the downgrading of indigenous lands (FERREIRA et al., 2014b; SIQUEIRA-GAY et al., 2020a) by proposing the construction of 575 km of roads to access 8 mineral deposits. The Sustainable Use (SU) scenario is based in proposals for downgrading Floresta Estadual do Paru (SECRETARIA DE ESTADO DE MEIO AMBIENTE, 2010; WWF, 2017b), permitting 752 km of new roads to access 170 mineral occurrences. The Sustainable Use and Indigenous Lands (SU & IL) scenario simulate the pressure over both types of PAs, permitting roads to access deposits in the remote Rio Paru D'Este indigenous land by crossing Floresta do Paru. The full development scenario (FullDev) presents the construction of roads to reach mineral deposits within all mineral rich PAs crossing Renca boundaries.

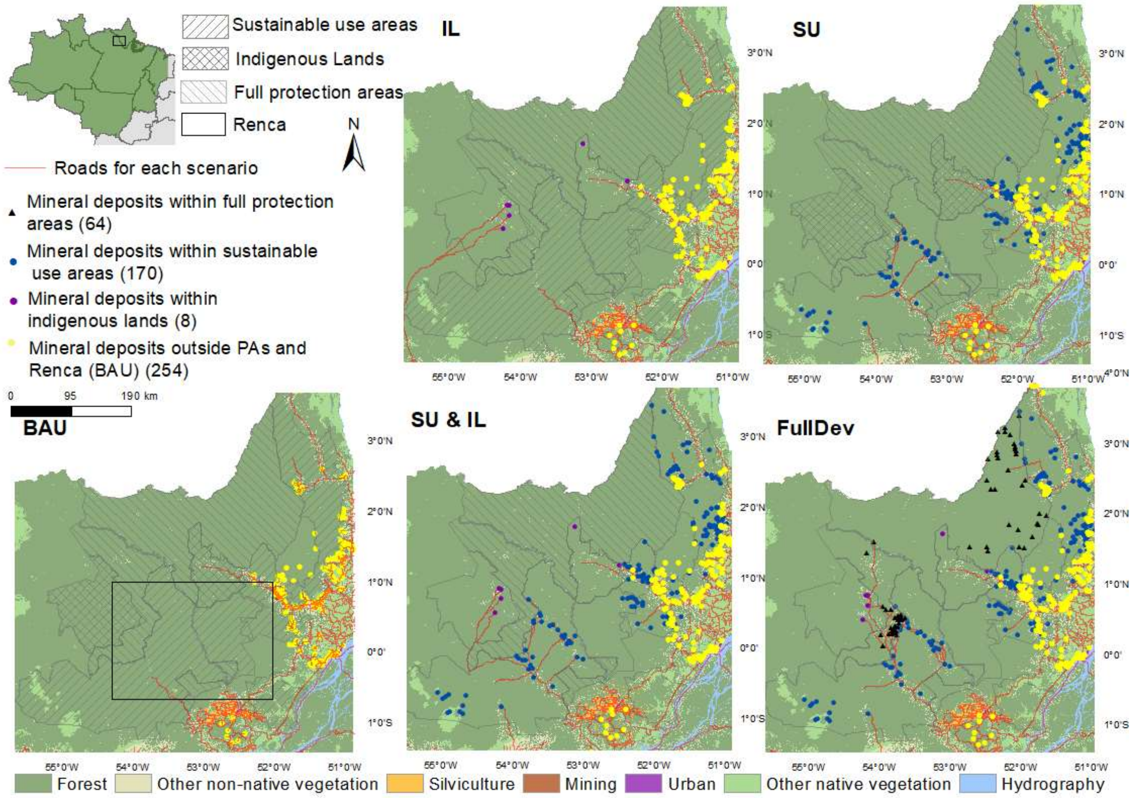


Figure 23. Scenarios of changes in PA regulations in the Renca region with different roads network required to access mineral occurrences

In the next sections, we present the chain of events that could be unleashed by mining-driven PADDD. We show that facilitating infrastructure in support of mining operations within PAs would cause direct and indirect deforestation in highly biodiverse areas and further consequences in the landscape.

Consequences of PADDD on deforestation and high biodiversity values

PADDD could cause from 4,324 km² to 7,809 km² of deforestation considering 30 years of development in the Renca region. While the indirect deforestation associated with pastureland expansion is mainly driven by the construction of new roads (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020; SOARES-FILHO et al., 2004, 2006), the direct deforestation for mining expansion is distributed according to the abundance of mineral deposits located in the weakened PAs. Indirect deforestation due to footprint expansion is from 40 to 60 times larger than direct clearing (Figure 24), depending on each type of PAs would be downgraded for mining and associated infrastructure. The large extent of direct and indirect impacts on forests in all scenarios depends on how many and in which PAs the road construction would be facilitated, revealing the important role of each PA in regulating broader-scale deforestation pressures. Therefore, specific efforts in conservation strategies to curb further deforestation (KIESECKER et al., 2010; SAENZ et al., 2013b) are needed to consider areas to remain protected to mitigate further deforestation pressures in the landscape.

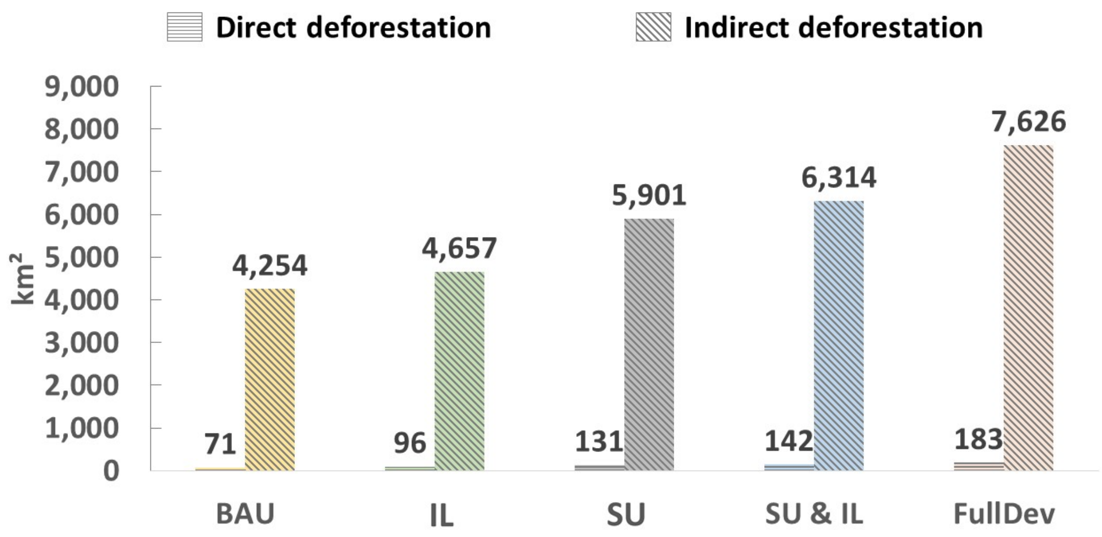


Figure 24. Differences between direct and indirect deforestation (km^2) in 30 years due to mining expansion

Mining poses a global pressure on biodiversity by its proximity to regions with high importance for conservation (KOBAYASHI; WATANDO; KAKIMOTO, 2014; MURGUÍA; BRINGEZU; SCHALDACH, 2016). Our scenarios revealed that mining direct clearing is smaller compared to indirect deforestation however, the sites affected shelter high biological importance (STRAND et al., 2018), with fundamental relevance for species richness and endemism (Figure 25). The mining direct clearing is driven by the existence of mineral deposits, therefore, the higher the spatial concentration of mineral occurrences, the higher is the deforestation caused by mining projects.

The uneven distribution of mineral occurrences influences the mining direct deforestation in each scenario. Expanding the existing mining areas in the BAU scenario would clear 61,5 km² of areas with high biological importance mainly driven by the high concentration of mineral deposits in biodiverse areas (Figure 25). Although facilitating projects within indigenous lands would not entail the highest rate of total deforestation (Figure 24), mining expansion over the 8 mineral deposits in the IL scenario would clear 75 km² of forest within highly biodiverse areas (Figure 25). In this case, the majority of mining deforestation is concentrated in this remote and biodiverse region in contrast to SU & IL scenario, where mining direct deforestation is scattered across the landscape in a region with less importance for biodiversity conservation (Figure 25).

These results emphasize the perverse impacts of mining on biodiversity showing that mining a remote and intact region would cause major consequences than developing mining across an entire floodplain. Protecting high biodiversity areas by keeping PAs protected is therefore of fundamental importance for avoiding the long-lasting and difficult to rehabilitate mining impacts on biodiversity (SONTER; ALI; WATSON, 2018).

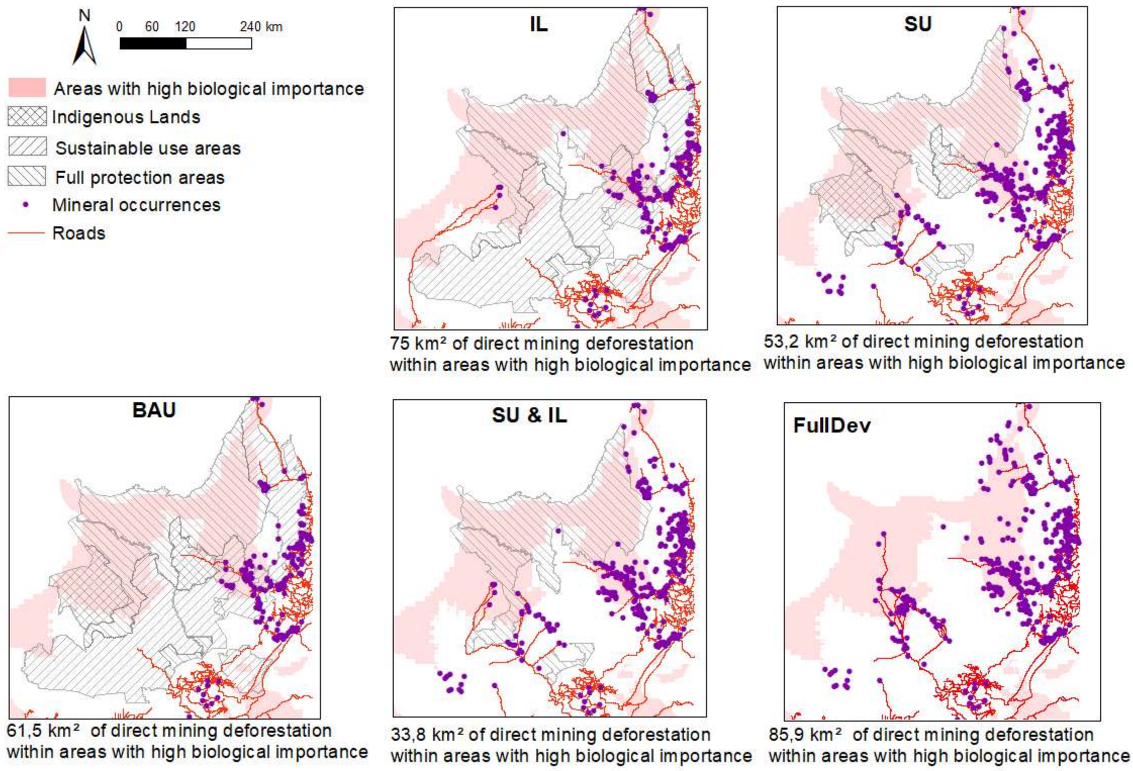


Figure 25. Areas to be cleared by direct mining footprint expansion with high biological importance

Consequences of roads induced deforestation on fragmentation and biodiversity loss

Deforestation affects kilometres far from the leases, especially considering the linear infrastructure for project development, causing landscape-wide effects. Fragmentation is considered an important cause of biodiversity losses worldwide, potentially harming species richness and diversity (FAHRIG, 2017). Although the area is well conserved and fragmentation effects are difficult to be detected in landscapes with high forest cover (PARDINI et al., 2010), in all scenarios analysed the increased deforestation would entail increased fragmentation (Figure 26). The relation of the number of patches and forest cover increased mainly in the FullDev and SU&IL scenarios both with longer roads (Figure 23). Protecting the majority of the landscape in the IL scenario led to less increased fragmentation in relation to other PADDD scenarios, revealing the importance of strategically protecting regions to curb further effects in the landscape.

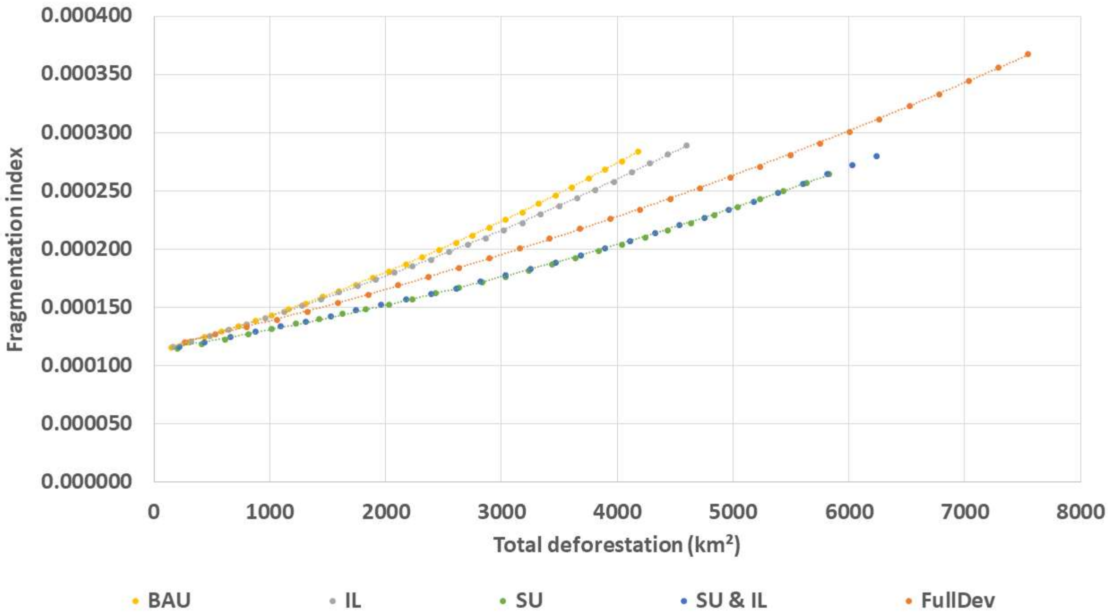


Figure 26. The increase of the fragmentation index compared to total deforestation in each scenario in 30 years of analysis

The deforestation impacts and fragmentation affects differently the PAs in each scenario. In the SU scenario, 679 km of new roads required to be constructed would lead to less fragmentation than allowing the construction of 598 km in the same PA in the FullDev scenario (Figure 27). In the case of IL scenario, the 237 km of new roads crossing this area would entail increased fragmentation than the 253 km resulted from facilitating the construction of roads in both SU and IL areas (Figure 27). These results show important trade-offs for biodiversity conservation emphasizing the importance of analysing the relation between roads extent, fragmentation, and deforestation. Despite the longer roads required, protecting one specific PA may prevent further effects in other regions under development.

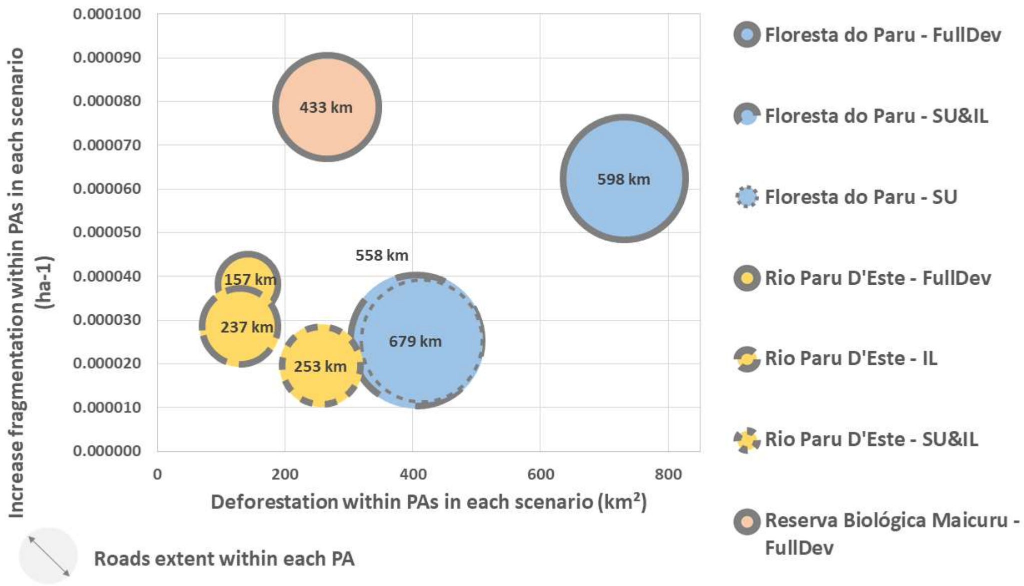


Figure 27. Increased fragmentation and deforestation effects in each PAs considering the four scenarios of development.

In addition to the extent of the roads, their design may affect different biodiversity values (FREITAS et al., 2012; RUNGE et al., 2017; VILELA et al., 2020). The buffer of 30 km from roads is the area to be affected by deforestation considering roads in Amazon (BARBER et al., 2014) and in the Renca region (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020). The BAU scenario showed 25,628 km² (19%) of areas with high biological importance up to 30 km of roads, revealing that despite the compactness of the road network, some important biodiversity values could be impacted and be even more affected by the proximity of more than one road (FREITAS et al., 2012). Although the roads of SU&IL scenario would impact more areas than in the IL scenario, the latter presents a larger extent of areas with high biological importance to be affected (38,168 km²), revealing an important trade-off related to road design in avoiding and minimizing impacts of deforestation.

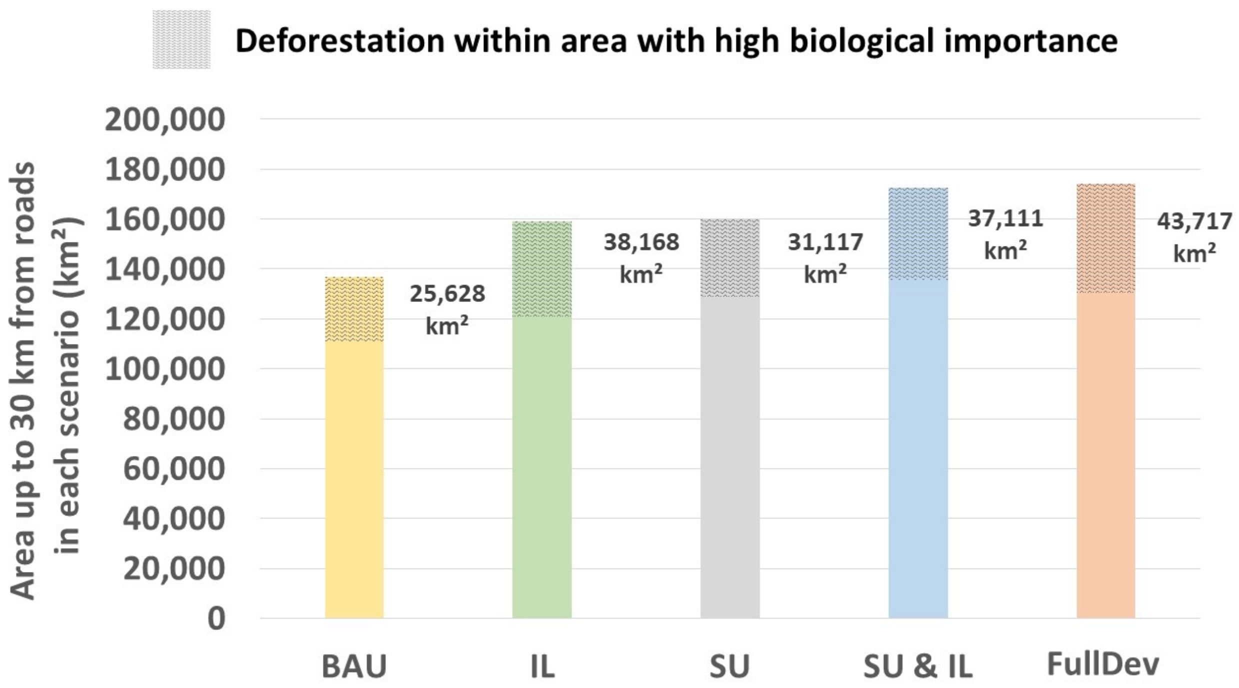


Figure 28. Area under influence roads showing increased loss of biodiversity values in IL and FullDev scenarios

5.3 Recommendations for conservation planning and mitigation requirements

Our results emphasize the relevance of impacts of mining and associated infrastructure on forests and biodiversity in the case of PADDD. All scenarios present increased biodiversity loss and fragmentation depending on the spatial distribution of remaining PAs and where are the mineral deposits to be developed. We showed that weakening one PA or one type of PA, particularly when located in remote places, would increase pressure for development elsewhere in the landscape.

PADDD proposals are threatening the conservation of unique biodiversity values worldwide (COOK et al., 2017; MASCIA et al., 2014; MASCIA; PAILLER, 2011). Here we show that the strategy of downgrading one PA may entail landscape-wide consequences, clearing high biodiversity values, and fragmenting habitats. Therefore, the proper consideration of areas that remain protected is fundamental for further conservation in these regions. Mining-driven PADDD decisions should be planned in an integrated and strategic way considering the access of multiple mining areas, minimizing road extension, and optimizing their design.

In practice, EIAs of mining projects do not properly consider the indirect and cumulative impacts on native vegetation (SINGH et al., 2020; WORLD BANK, 2019a). The magnitude of indirect deforestation and fragmentation caused by associated infrastructure show that EIAs of new mining and associated projects must consider comprehensive mitigation measures to curb these effects. Besides that, noting the relevance of direct clearing in highly biodiverse areas, proposals to mine remote and untouched areas must consider this importance when applying the mitigation hierarchy (ARLIDGE et al., 2018; IFC, 2012) to properly avoid and minimize and ultimately compensate the residual loss of biodiversity.

For future developments in Renca, conserving and protecting areas with high biological importance is of fundamental importance to avoid further deforestation and fragmentation pressures in the landscape. EIA of new mining projects, especially located in pristine regions, must assess and propose comprehensive mitigation measures for these impacts resulted from the new infrastructure needed to support the project implementation and operation.

5.4 Methods

The five scenarios simulated the consequences of mining expansion and associated infrastructure for the next 30 years (Table S 14 in Appendix 3). The main steps involved for the scenarios simulation and analysis are: (i) creation of database of LULC maps; (ii) calibration and validation of a spatial explicit land use model; (iii) design of roads based on least cost surface pathway; (iv) projection of 30 years of change using five different roads design; (v) analysis of further consequences for deforestation and biodiversity.

1.1.Land use and cover maps

The LULC maps used as reference was from the TerraClass project (ALMEIDA et al., 2008) due its extensive area cover by mapping LULC including mining, forests and non-native vegetation. The years of 2004, 2010 and 2014 were selected due to the low cloud cover (<1%) and the recent mining expansion. The original land use classes in dataset provided by INPE were: forest, secondary vegetation, herbaceous and shrubby pasture, areas recent deforested (deforestation), silviculture, urban, mining, other native vegetation, no forest and hydrography (Table S 4 in Appendix 3).

To model historic land use changes, the pre-processing procedure of land use maps involved: (i) resampling the original dataset with 30m cell size to 100m of cell size to increase model accuracy and decrease the calibration and simulation time (Table S 5 - Table S 13 in Appendix 3); (ii) the reclassification of the original 14 classes of TerraClass dataset into 7 classes to simplify the transitions for the model calibration (Table S 4 in Appendix 3); (iii) LULC classes and transitions adjustment for the purpose of simulating changes due to infrastructure expansion (Section 2 Appendix 3). The final land use and cover classes were: forest, other non-native vegetation (including secondary vegetation, herbaceous and shrubby pasture, deforestation), silviculture, urban, mining, other native vegetation (including the original other native vegetation and no forest) and hydrography (Table S 4 in Appendix 3). Mining land use encompasses only mining areas within official leases issued by the National Mining Agency until February 2020, therefore, no illegal mining (*garimpos*) expansion were considered in the model calibration and in the simulations.

1.2.Modelling land use changes

In the first step, the DINAMICA EGO (SOARES-FILHO; CERQUEIRA; PENNACHIN, 2002) was used to develop the spatial explicit land use change model based on the Weights of Evidence (WoE) algorithm (BOHAM-CARTER, 1994) (Figure S 5 - Figure S 10 in Appendix 3). The total of 6 transitions (Table S 19 in Appendix 3) were calibrated considering the maps from 2004 and 2010 and validated with the maps from 2010 and 2014: forest to other non-native vegetation, forest to silviculture, forest to mining, other non-native vegetation to silviculture, other non-native vegetation to mining and other non-native vegetation to urban. The model calibration was performed using 4 yearly time steps interactions (2010-2014) adjusting the WoE coefficients and ranges to allocate the spatial distribution of the land use changes (Figure S 5 - Figure S 10 in Appendix 3). For the validation, we used exponential decay function at multiple spatial resolutions implemented in DINAMICA EGO (SOARES-FILHO; RODRIGUES; FOLLADOR, 2013) and the calibrated model reached 50% of similarities with 17 ha and present significative improved performance compared to the null model (Figure S 11 – Figure S13 in Appendix 3).

The region is well conserved and the past rates analyzed are conservative face the possibility of opening up the area. Therefore, the simulated rates of land use change of each scenario are proportional to the size of the new area for development (Table S 18 in Appendix 3) depending on PAs category to be opened (Table S 19 in Appendix 3). To create the roads network required to access the mineral occurrences within PAs, a two-step approach involving the friction and cost surface was used with DINAMICA EGO (SOARES-FILHO et al., 2004). Firstly, a friction surface to represent the difficulty was proposed using slope, LULC and PAs cover information (Table S 15 in Appendix 3). Based on the friction raster, the mineral occurrences and the existing transportation infrastructure, the cost surface was created to estimate the least cost pathway for constructing a new road connecting the deposits to existing transportation infrastructure (Table S 16 in Appendix 3). As Renca has been little researched (SCARPELLI; HORIKAVA, 2018, 2017), when and if it is opened, the first investments will come and new deposits are likely to be discovered.

The scenarios were generated using: (i) explanatory variables including the new road network resulted from the cost surface analysis; (ii) the WoE of calibration to allocate changes in the landscape, including the distance to new roads; (iii) rates of land use changes to be simulated according to the area available for development in each scenario; (iv) real map of 2014 to be used as the most recent landscape to simulate future changes.

1.3.Biodiversity analysis

The map with biodiversity relevant areas (Figure S 21 in Appendix 3) for the Brazilian Amazon (STRAND et al., 2018) was elaborated using variables of species richness, area of endemism, phylogenetic endemism, and endemicity to create 8 categories of areas with high and low biological importance. For the analysis in this research, regions with high biological importance were used as a mask to analyse areas to be impacted by deforestation and roads.

To calculate the area under influence of deforestation surrounding roads, a buffer with 30 km from roads was used. Previous works show that area under influence of roads in Amazon could vary from 5.5 km for all roads and up to 32 km for highways (BARBER et al., 2014), while in Renca, the area of 30 km buffer encompasses most deforestation of the last years (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020).

For fragmentation analysis, we run 5 times each scenario and calculated the mean value of four landscape metrics to evaluate fragmentation: number of patches and edge density (CABRAL et al., 2018). The batch process of Fragstats (MCGARIGAL, 2015) was used to run the two metrics and class area for the forest class in each landscape of the 30 years of simulation. We calculated the mean value and standard deviation of each metric (Figure S 18 to Figure S 20 in Appendix 3) and used the number of patches and total forest extent in the fragmentation index (MONMONIER, 1974):

$$Fragmentation\ index = \frac{Number\ of\ patches - 1}{Total\ forest\ extent - 1}$$

5.5 Research limitations

This chapter classifies a spatially explicit model and analyses further impacts on deforestation. The historic changes in the area were projected by simulating landscapes for the next 30 years. Even considering the changes in the rates of development, in each transition, these rates were considered constant during the years of analysis. Large projects are not proposed for the region, but if it would be the case, these rates would change considerably as more infrastructure beyond that related to mining would be required.

6. Impacts of policy change to expand mining into indigenous lands

Objective

This chapter aims at evaluating the cumulative impacts on forests and ecosystem services derived of policy change for expanding mining into an entire category of PA.

Preface

The increased number of mining proposals within indigenous lands calls attention to the current pressure over these areas since 2018. In February 2020, the federal government proposed a bill to regulate mining and other activities in indigenous lands. With the sake of analyzing the impacts of policy change, the forest extent impacts and further consequences for ecosystem services loss were evaluated.

We found that 863,000 km² would be impacted, which corresponds to 20% more forest extent to be affected than the current scenario, causing the loss of USD 5 billion annually in the provision of food and raw materials, climate regulation, and GHG mitigation. Permitting mining in these areas not only would clear a large extent of forests but also cause the loss of valuable benefits for local, regional, and global communities. With this, we seek to enlarge the debate about the potential impacts of mining activities not only for indigenous communities but for the local and global population.

This chapter supports the thesis (Figure 1) by (i) quantifying the impacts on expanding mining over indigenous lands considering the entire Brazilian Amazon; (ii) quantify further losses of ecosystem services caused by forest to be impacted; (iii) discuss the implications to safeguard unique ecosystems and cultural values.

Related publication

Siqueira-Gay, J., Soares-Filho, B., Sánchez, L. E., Oviedo, A., & Sonter, L. J. (2020). Proposed legislation to mine Brazil’s Indigenous Lands will threaten Amazon forests and their valuable ecosystem services. One Earth, 3, 1–7. DOI: 10.1016/j.oneear.2020.08.008

Abstract

A recent proposal to regulate mining within indigenous lands threatens people and the unique ecosystems of Brazil's Legal Amazon. Here we show that this new policy could eventually affect more than 863,000 km² of tropical forests – 20% more than under current policies – assuming all known mineral deposits will be developed and impacts of mining on forests extend 70 km from lease boundaries. Not only are these forests home to some of the world's most culturally diverse communities, they also provide at least US$ 5 billion each year to the global economy, producing food, mitigating carbon emissions, and regulating climate for agriculture and energy production. It is unclear whether new mines within indigenous lands will be required to compensate for their direct and indirect environmental and social impacts but failing to do so will have considerable environmental and social consequences.

6.1. Introduction

On February 5th 2020, Brazil's President Bolsonaro signed a bill (PL 191/2020) that will permit mining inside indigenous lands – a unique category of PAs (SOARES-FILHO et al., 2010) covering 1.2 million km² (23%) of the Legal Amazon. Indigenous lands are home to 222 indigenous groups, with more than 644 thousand families living in traditional communities and speaking a combined 160 languages (INSTITUTO SOCIOAMBIENTAL, 2020a). The current political context is unfavorable to indigenous people (BEGOTTI; PERES, 2020; LIMA et al., 2020) and, if approved by Congress, the proposed policy changes have the potential to not only permanently transform the lives of indigenous communities (BEBBINGTON et al., 2018; FERRANTE; FEARSNIDE, 2020), but also negatively impact a large extent of biodiverse forests and the ecosystem services they provide[3].

Alongside Brazil's other PA categories, such as national parks and biological reserves, indigenous lands not only safeguard indigenous people and their traditional knowledge, but also protect ecosystems (BARAGWANATH; BAYI, 2020; PAIVA et al., 2020; SOARES-FILHO et al., 2010). There are 332 officially designated indigenous lands in the Brazilian Amazon, with another 92 in earlier stages of legal and administrative approval (See Note S1, Fig S1 of supplementary material of chapter 6). However, many of these areas are also known to contain valuable undeveloped mineral deposits (including a range of commodities, such as gold, copper and iron ore;

See Note S1, Table S1 in Appendix 4). Under current legislation, mining inside indigenous lands requires Congressional authorization – a Constitutional shield that has effectively deterred all industrial mining within these sites to date, albeit far less effective at deterring illegal small-scale mining activities.

Mining can affect forests through different pathways, either by directly clearing vegetation to establish open pits, mineral processing plants and ancillary installations (RUDKE et al., 2020) or indirectly due to the need to build infrastructure to access mine sites and transport minerals. Such infrastructure facilitates access to otherwise barely accessible land and can result in cumulative impacts from multiple mining operations and other surrounding land users (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020). In the Brazilian Amazon, mining has been found to indirectly affect forests up to 70 km from large-scale mining sites (SONTER et al., 2017). Indeed, this offsite deforestation was 12 times larger than the mines' local footprint between 2005 and 2015.

Forest loss also affects valuable ecosystem services (STRAND et al., 2018). Tropical forests provide benefits to society as a whole, for example, by storing carbon (FERREIRA et al., 2018) and regulating regional and global climate (PÜTZ et al., 2014); providing food and raw materials, such as nuts and rubber (STRAND et al., 2018), for both domestic use and export; securing freshwater quality and quantity (SIQUEIRA-GAY et al., 2020b); and providing recreational opportunities (SIQUEIRA-GAY et al., 2020b). All these ecosystem services, along with a unique socio-biodiversity (OECD, 2015) (i.e., the biological and cultural diversity sheltered by Amazonian ecosystems, as well as the products obtained by traditional extractives activities), may be affected by future policy changes that could unleash mineral exploration and extraction across the Brazilian Legal Amazon.

In this study, we assess the impacts of the proposed policy (ADELLE; WEILAND, 2012) by quantifying associated threats to forests and their ecosystems. We outline what is needed to safeguard indigenous rights, forest biodiversity and the services these places provide, for consideration by the National Congress should they vote on this bill later this year. In the following sections, we present our results and discuss foreseeable effects of the proposed policy change.

6.2. Methods

6.2.1. Data sources

Data on mining claims and proposals were accessed from the official spatial database of the Brazilian government's National Mining Agency (AMN): http://sigmine.dnpm.gov.br/webmap/. These data provide the current status (accessed on February 10th 2020) of all mineral claims and proposals covering the Brazilian territory. The data on mining deposits and occurrences were provided by the Brazilian Geological Survey (CPRM): http://geosgb.cprm.gov.br/. The PAs limits of sustainable use and full protection units were obtained from the Environmental Ministry website: http://mapas.mma.gov.br/i3geo/datadownload.htm. The indigenous lands boundaries are from the official database of the National Foundation of Indigenous People, containing the polygons, the current status, types, and ethnicity of each land: http://www.funai.gov.br/index.php/shape. The forest cover was provided by the National Institute for Space Research (INPE): http://terrabrasilis.dpi.inpe.br/. The spatial database of ecosystem services is from the study of Strand et al (2018)[13] available at https://csr.ufmg.br/amazones/. The illegal mining information is a data summary from different sources (such as studies and management plans) provided by Amazon geo-referenced socioenvironmental information network (RAISG): https://www.amazoniasocioambiental.org/en/.

6.2.2. Analysis of impacts on forests and ecosystem services

We calculated the total number of claims made each year between 2010 and 2019 and determined the proportion of these claims that occurred inside indigenous lands. To determine current threats, we overlaid indigenous lands with spatial data on legal mineral claims, illegal mining activities, forest cover, deforestation trajectories and ecosystem services.

We investigated two alternative scenarios for comparison to the current situation: (i) without policy implementation ("No policy"): developing all known mineral deposits and occurrences outside PAs and (ii) "Policy implementation": developing all mineral deposits and occurrences outside other PAs (conservation units of full protection and sustainable use) but permitting development inside indigenous lands. For both scenarios, we quantified the area of forests potentially influenced by mining – i.e. the forests that occur within a buffer surrounding each mineral occurrence using an upper limit of 70 km and a more conservative lower limit of 10 km (SONTER et al., 2017) (Fig 6). To estimate the forest extent potentially affected by mining under each scenario, we used data from 2018 of PRODES and overlaid this with our four scenario masks (Figure 29).

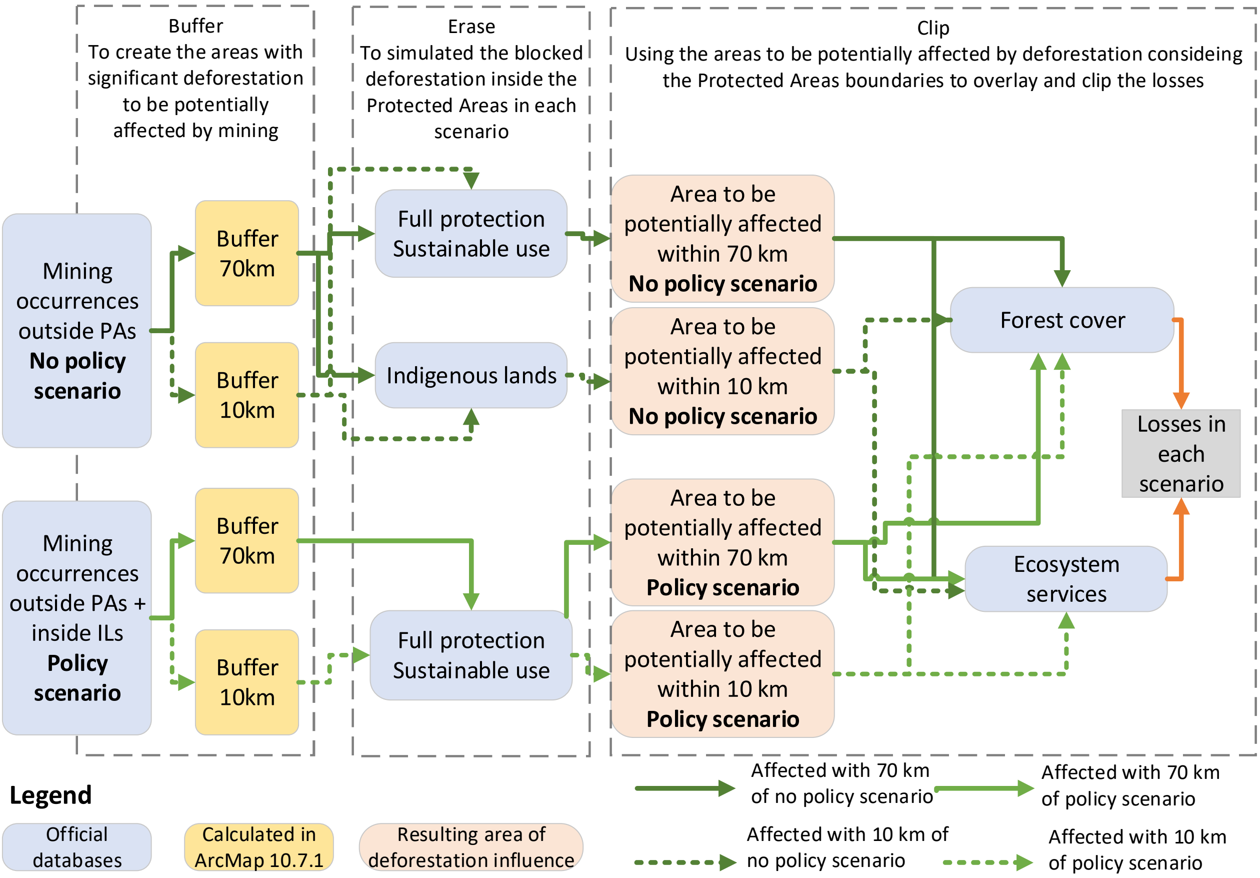


Figure 29. Main steps of data analysis considering the no policy and policy scenarios. Source: (SIQUEIRA-GAY et al., 2020a)

To estimate the impacts on ecosystem services, we used spatially explicit monetary valuations for four key services provided by Amazon forests: food production (Brazil nut), raw material provision (timber and rubber), GHG mitigation (reduction in CO_2 emissions), and climate regulation (rent losses to soybean, beef and hydroelectric production due to reduced rainfall) (STRAND et al., 2018). These ecosystem services maps were overlaid with our areas of forests affected by mining (for each the 10 km and 70 km buffers) to quantify the monetary value of ecosystem services provided by forests potentially influenced by the proposed policy.

6.3. Mining impacts on forests and ecosystem services

6.3.1. Current context of mining within indigenous lands

This is not the first time a policy change has seen mining interests threaten Brazil's protected areas – many previous bills have been presented (PL 1610/1996 (EL BIZRI et al., 2016); PL 3642/2012) (see Note S2 in Appendix 4). However, the exponential growth in mining proposals inside indigenous lands in 2018 (Figure 30) suggests that prospectors have anticipated and planned to exploit this opportunity in recent years.

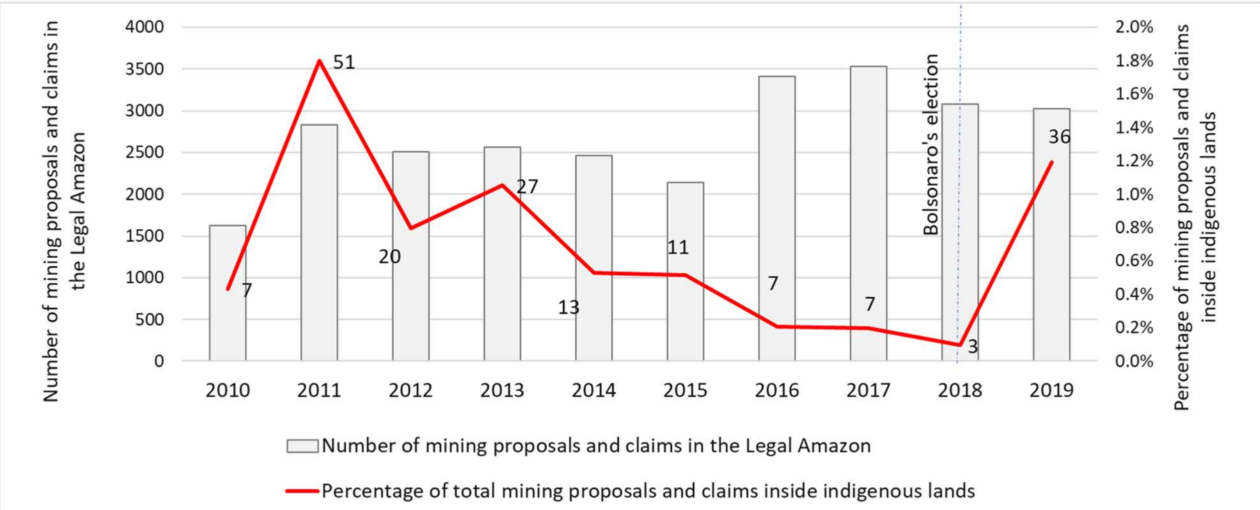


Figure 30. Mining proposals and claims registered with the National Mining Agency (Accessed in February 2020). The bars indicate the number of mining proposals and claims in the Legal Amazon per year and the red line indicates the number and the percentage of total proposals inside Indigenous Lands. Source: (SIQUEIRA-GAY et al., 2020a)

We found 115 indigenous lands (31%) contained at least one claim or mining proposal and most were already under application for exploration (the initial stage of mineral permit process) (Figure 31). In addition to legal mining activities, we found 148 indigenous lands (45%) already contain illegal mining activities (Figure 31), which may also eventually be influenced by an increase in legal operations. These activities may impact uncontacted and currently isolated groups living within these indigenous lands, requiring a special attention in future research.

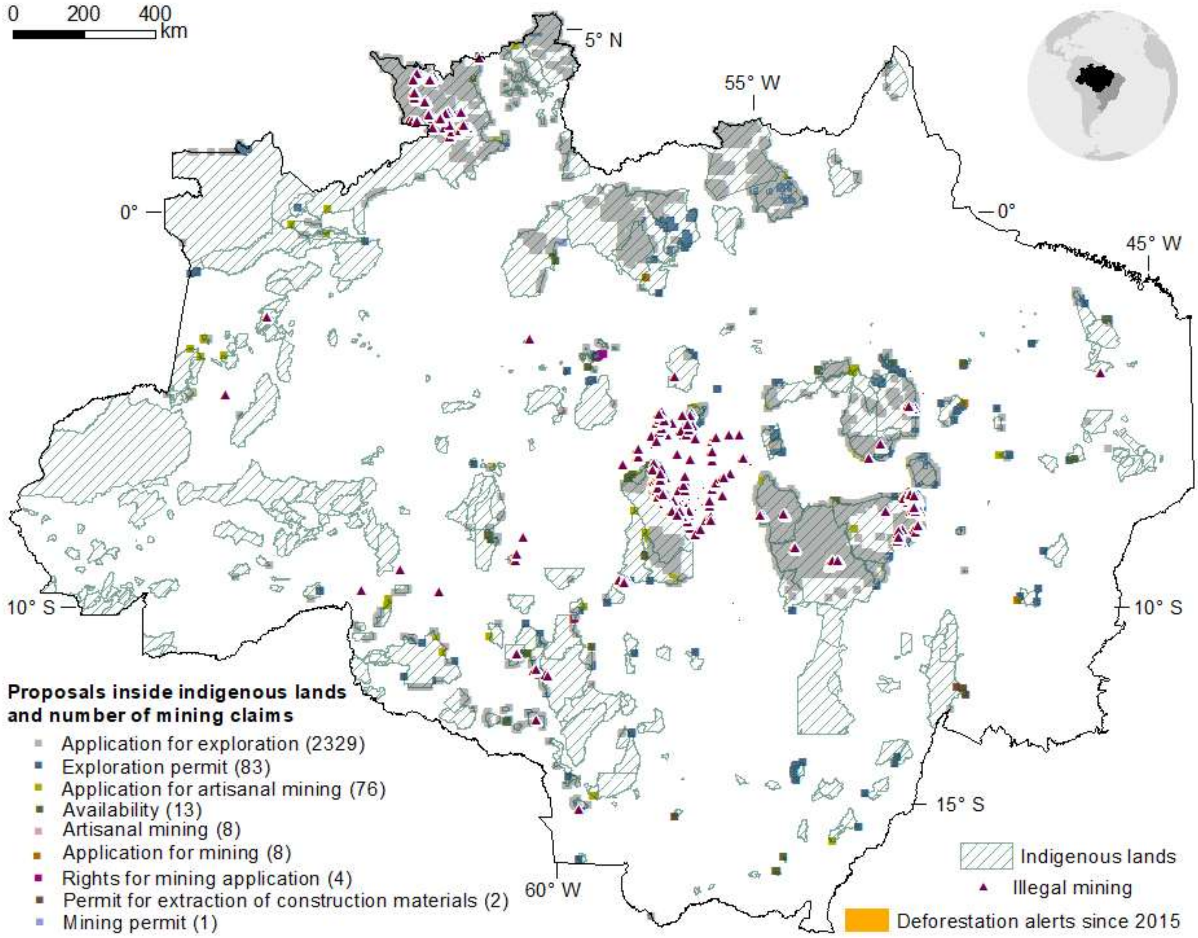


Figure 31. Current mining proposals and claims (accessed in February 2020) inside indigenous lands and the location of illegal mining (acessed in April 2020). Source: (SIQUEIRA-GAY et al., 2020a)

6.3.2. Impacts on forests and their ecosystem services

If all 4,600 known mineral deposits and known occurrences outside currently protected areas were to be developed, and assuming indirect impacts extent up to 70 km from mining sites, 698 thousand km^2 of forest may be affected by mining (Figure 32). However, the approval of the proposed policy (i.e. also permitting mining inside indigenous lands) could increase this area by 20% (up to 863 thousand km^2; Figure 33). Using a more conservative 10 km buffer to capture indirect effects reduces the total estimates of affected forests under our ‘no policy’ and ‘policy’ scenarios to 222 thousand km^2 and 182 thousand km², respectively; but this still represents a 22% increase in the area affected by the policy change.

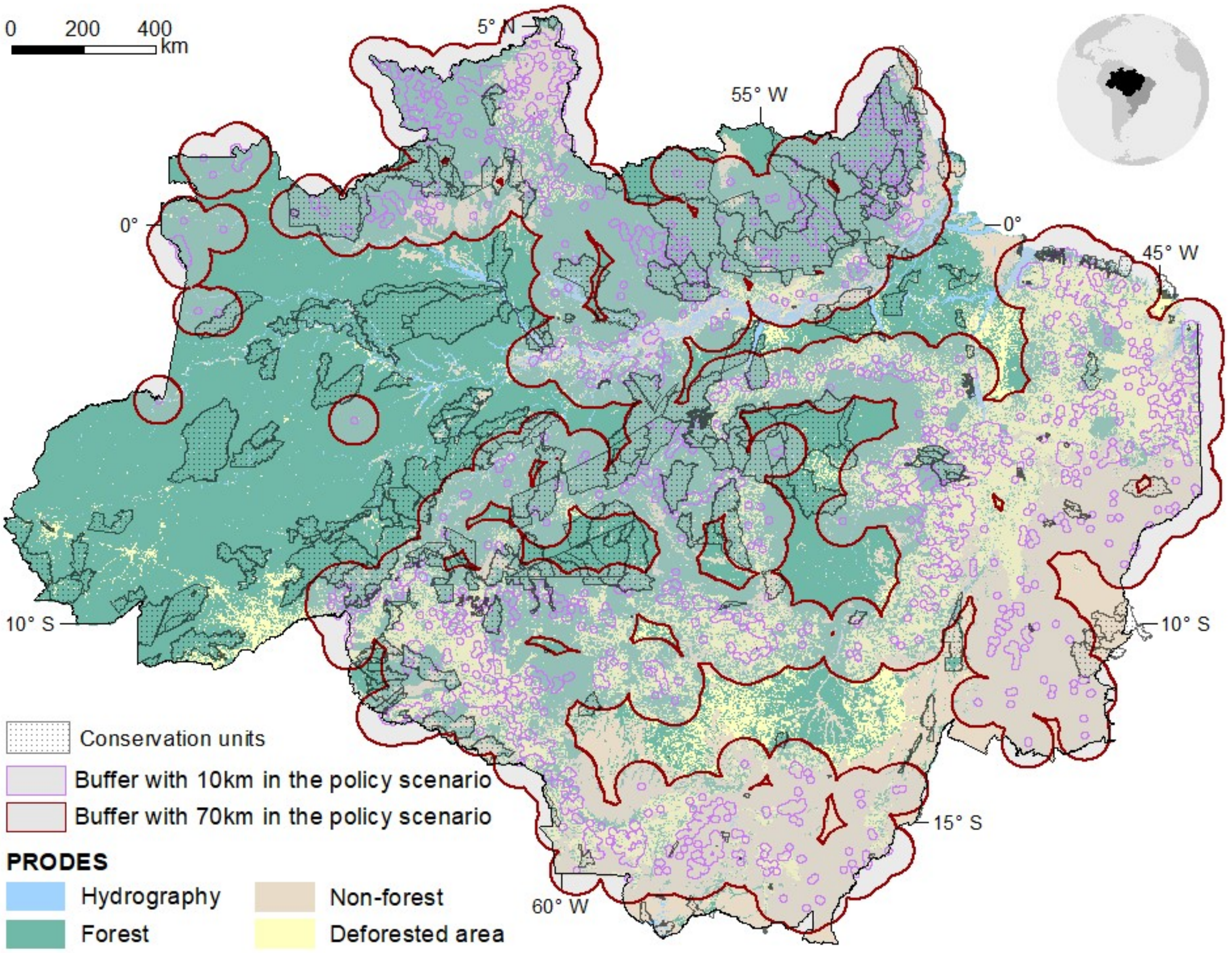


Figure 32. Forest cover and the extent of forests that could be affected in the policy scenario. Source: (SIQUEIRA-GAY et al., 2020a)

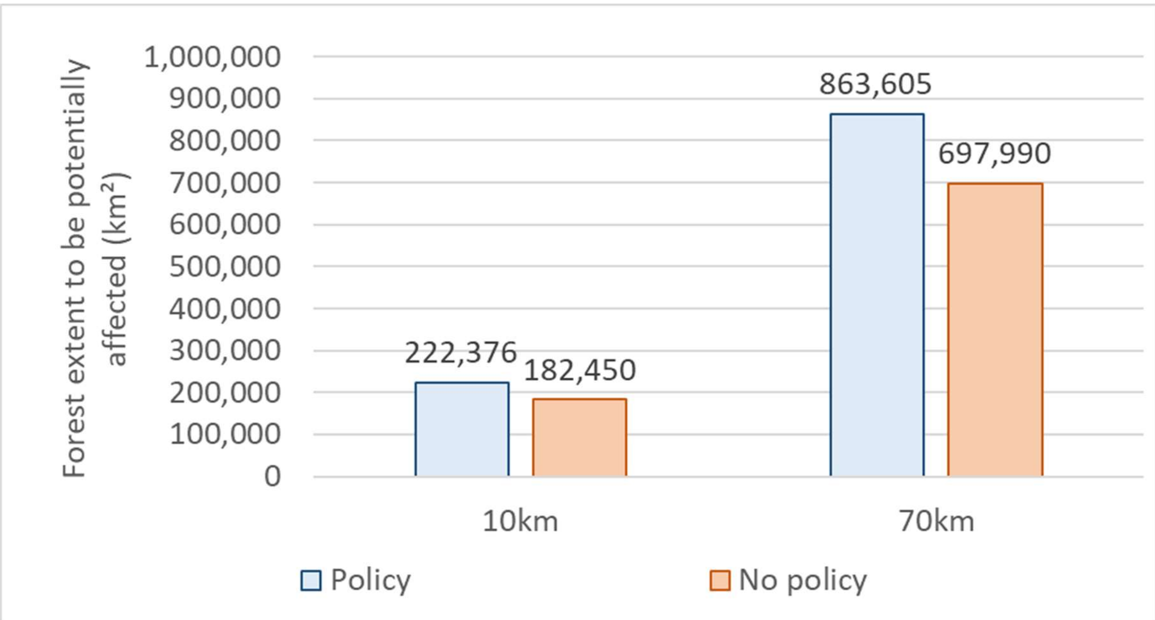


Figure 33. Consequences of mining all known mineral deposits on forest extent under two alternative scenarios – No policy and Policy implementation. Source: (SIQUEIRA-GAY et al., 2020a)

Considering the provision of only four ecosystem services (food production, raw materials provision, GHG mitigation and climate regulation), we estimated that affected forests provide > US $5 billion of value each year to the global economy (Figure 34). Our analysis reveals particularly large consequences for GHG mitigation reaching more than US $2.2 billion of losses. We found that raw materials provisions of rubber and timber would have considerable monetary losses (up to US $1.4 billion) in the scenario of policy implementation.

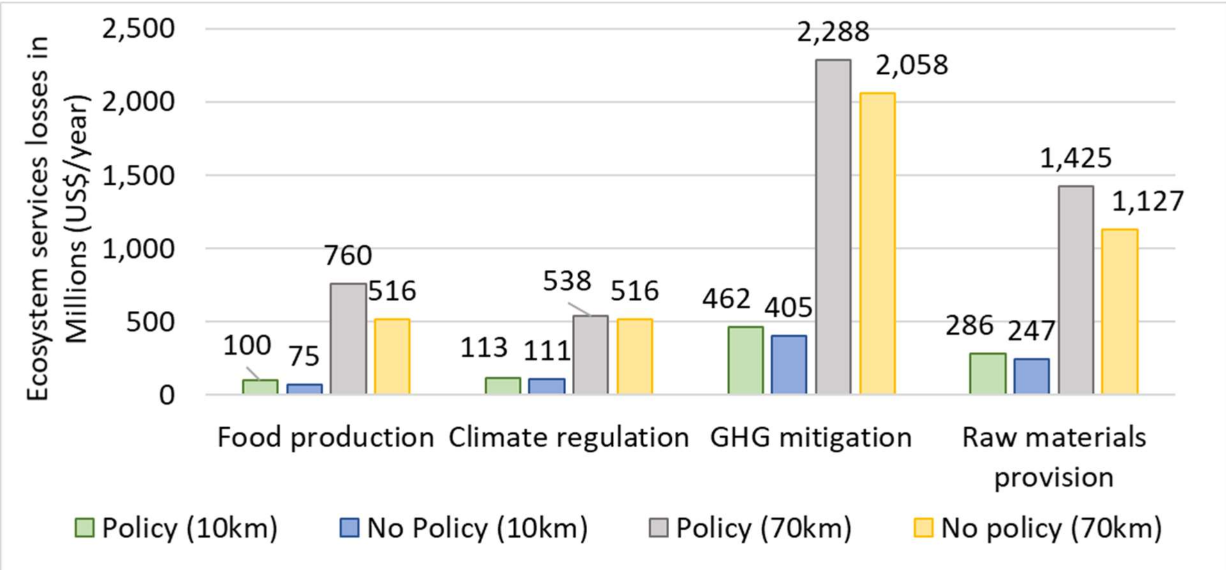


Figure 34. The monetary value of ecosystem services losses provided by potentially affected forests, assuming indirect impacts extend 10 and 70 km from mineral occurrences in both scenarios. Source: (SIQUEIRA-GAY et al., 2020a)

6.3.3. Other factors influencing the impacts

Two major factors need to be considered to fully appreciate the implications of the proposed policy on forests and ecosystem services. Firstly, the construction of transportation infrastructure and the emergence or growth of urban centers will add to the effects of mining on forests and ecosystems. Mine output (usually a concentrate, i.e. an enriched product for further metallurgical or industrial processing) requires bulk transportation to reach markets and in some cases construction of large mines requires a large labor force, hence attracting many migrants in search of jobs or opportunities. The size and impacts caused by infrastructure may change according to the commodity (See Note S1; Table S2 and Fig S3 in Appendix 4).

Secondly, most indigenous lands are located in remote areas, sheltering some of the world's most pristine ecosystems (SOARES-FILHO et al., 2010) and Brazil's socio-biodiversity (See Note S1; Table S3 in Appendix 4). Establishing even one new mine in these areas could trigger a cascade of further forest loss due to growth-inducing infrastructure (JOHNSON et al., 2019), and potentially send many yet-to-be discovered species and ecosystems extinct in addition to degrading traditional livelihoods.

In addition to the impacts of this proposed policy on forests and ecosystems, the presence of illegal gold miners within indigenous lands and their related activities trigger other impacts. These additional pressures include degradation and pollution of the environment with mercury(LOBO; COSTA; NOVO, 2015), and potentially more worrisome, transmission of diseases (SOUZA et al., 2019) such as the Covid-19, since indigenous communities are vulnerable and have poor access to health care (INSTITUTO SOCIOAMBIENTAL, 2020b). Given that the policy changes would increase the outside access to indigenous groups, a public health problem has potential to be intensified.

Moreover, there is currently a push to dismantle policies that protect the rights of indigenous groups (BEGOTTI; PERES, 2020; FERRANTE; GOMES; FEARNSIDE, 2020), as exemplified by recent government initiatives. Changes in the Ministry of Environment and National Indian Foundation-FUNAI's policies follow an extensive roadmap of setbacks (FERRANTE; GOMES; FEARNSIDE, 2020; LIMA et al., 2020): the emptying of the institution's functions (FERRANTE; FEARNSIDE, 2019) and budget (BEGOTTI; PERES, 2020; FERRANTE; GOMES; FEARNSIDE, 2020), granting of environmental authorizations with no indigenous consultation (FERRANTE; GOMES; FEARNSIDE, 2020), willingness to comply with requests for the extinction or reduction of PAs (BEGOTTI; PERES, 2020), and defending non-compliance of the law against illegal logging and mining (UNHRC, 2020). The faulty interpretation that the rights of indigenous people currently depend on the completion of the indigenous lands demarcation process confronts the Constitution and ignores the jurisprudence of the Supreme Court (LE TOURNEAU, 2015).

6.3.4. Management and mitigation requirements

The proposed bill does not contain any environmental or social safeguards and is silent about whether mining within indigenous lands will require EIAs. Considering the current regulatory status of mining claims in the Legal Amazon, less than 2% require a comprehensive EIAs for licensing (See Note 3; Table S4 and Table S5 in Appendix 4). Developing a mine in some claims may require only a simplified environmental assessment and licensing process (FONSECA; SÁNCHEZ; RIBEIRO, 2017), while other types of requirements are currently uncertain (See Note 3; Table S4 and Table S5 in Appendix 4) (MINISTÉRIO PÚBLICO FEDERAL, 2020).

While the proposed policy does suggest some financial compensation will be provided by companies to indigenous associations and leaderships for the use of the indigenous lands, there are no guarantees to ensure Free, Prior and Informed Consent (FPIC), as established by the UN Declaration on the Rights of Indigenous Peoples (HANNA et al., 2014). Under the terms of PL 191/2020, if the application for exploration is within a non-regulated indigenous land, it is not mandatory to consult the impacted communities. It is also unlikely that the compensation payment, calculated on the basis of net revenues and commodity type, will come even close to the value of ecosystem services lost due to mining (Figure 34).

We urge those involved in designing and approving this bill to seriously consider the impact it could have on ecosystems and people – not only indigenous people, but the community at large, as impacts will not be restrained to the boundaries of indigenous lands because, as many traditional communities could be displaced. If approved, at the very least, a mechanism for assessing and mitigating impacts must be established and compliant with best practice (IFC, 2012). All new mines must require a comprehensive EIA, including mitigation plans that comply with the mitigation hierarchy. Explicitly requiring FPIC would not only contribute to safeguard the rights of indigenous people but also benefit environmental protection and mitigation outcomes if linked to EIAs.

While Brazil decides on whether or not approve this bill, environmental NGOs can build awareness of these threats both in the country and internationally in order to protect such valuable environmental resources. Just like proposals to open Renca – a mineral rich biodiverse region – to mining were overturned (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020), the values of indigenous lands can too be protected from unchecked long-term damage.

6.4. Research limitations

This chapter quantifies the impacts on forests and ecosystem services resulting from changes in the policy of mining within indigenous lands. The monetary valuation of ecosystem services may reduce the value of nature to a simplistic number, however, in this case, served to enlarge the discussion by counterbalancing the financial compensation for affected indigenous people in the proposed bill. Only 4 ecosystem services were analyzed but many others still need to be evaluated. For example, cultural services are one of the most relevant ecosystem services to be impacted by this bill but were not evaluated in the analysis.

7. Perspectives on addressing cumulative impacts of mining on forests in the Brazilian Amazon

7.1. Major research contributions

The results of the research presented in the previous chapters are summarized here as three major research contributions:

Road construction plays an important role in leading to cumulative impacts on forests in mining regions

Mining drive cumulative impacts on forests by expanding its footprint and by requiring additional roads, transmission lines, and infrastructure for transporting ore. The construction of side roads, access roads, highways feature the main pathways leading to cumulative impacts in a mining region (Figure 7). The analysis of Serra do Navio, a region with mining as a key driver of change, revealed that the majority of deforestation occurred up to 30 km from roads (Figure 18).

Mining and associated infrastructure drive cumulative impacts on forests, extending far from the leases and fragmenting landscapes

The interplay LULC changes in mining regions leading to cumulative impacts on forests varied according to the associated industry and facilities required. Silviculture expansion was found to clear more forests than pastureland when mining is associated with pulp mill and cellulose processing. In contrast, in a region with gold and iron ore activities, the pastureland expansion was the more relevant change in clearing forests. Roads required to access mines and transport ore lead to a cumulative loss of forests causing further effects in the landscape structure.

Expand mining into PAs would cause extensive forest loss, affecting high biodiversity values and ecosystem services

Scenarios of facilitating road construction and mining inside PAs showed that highly biodiverse regions would be impacted and fragmented. Mining direct deforestation is from 40 to 60 times smaller compared to the indirect deforestation caused by constructing roads and infrastructure, depending mainly on the areas that remain protected. By examining the consequences of the recent proposal to allow mining within indigenous lands, the cumulative impacts on deforestation are estimated to extent kilometers far from mineral deposits, potentially affecting forests responsible to provide food and raw materials, and regulating the climate among other valuable ecosystem services. These evidences reinforce the importance of conserving areas with high biodiversity values and culturally rich to avoid further impacts on local, regional and global scales.

7.2. Relevance for Environmental Impact Assessments and conservation planning

7.2.1. The cumulative impacts of mining on forests

The cumulative interaction of actions and effects in mining landscapes relates to the fact that mining is spatially dependent on the existence of mineral deposits. The progressive cumulative loss of forest is considered as the classical nibbling cumulative impact (HEGMANN et al., 1999; TREWEEK, 1999). Forest clearing driven by infrastructure construction represents a persistent addition that interacts in an additive way, producing the cumulative loss (PETERSON; CHAN; PETERSON, 1987; TREWEEK, 1999). The cumulative forest loss could in turn be also a source for other impacts (NOBLE, 2015), especially effects in the landscape composition and structure. The cumulative forest loss affects biodiversity (Figure 7) by directly clearing biodiverse areas (BROOKS et al., 2002; COOPER, 2004; TREWEEK, 1999) and fragmenting habitats, reducing species richness and diversity. The provision of ecosystem services could be impacted by forest loss (Figure 7), affecting wildlife habitats (DOBSON et al., 2007), food production, erosion regulation, and recreation (COOPER, 2010). Fragmentation per se also affects relations of supply and demand of ecosystem services (MITCHELL et al., 2015; MITCHELL; BENNETT; GONZALEZ, 2014).

The induced deforestation caused by mining and associated infrastructure, such as new roads, urban expansion, and economic activities stimulated by mining is more significant than that caused by the intensive land use inside mining leases (SONTER et al., 2014b; SOUZA-FILHO et al., 2016). The importance of each driver of change could vary according to the commodities type. Pastureland and silviculture expansion could have a great relevance in clearing forests when associated to kaolin and pulp mill activities, while in regions with iron ore and gold mining, there was a higher relevance of pastureland expansion over all other drivers.

Landscapes dynamically altered by LULC changes can be fragmented by side roads, access roads, minerals transportation infrastructure construction, and other linear infrastructure required to support projects operation. Considering the transportation infrastructure in mining regions, Runge et al. (2017) state the importance of shared linear infrastructure (roads and railways) for conservation, evidencing the reduction of biodiversity loss caused by this type of arrangement. The majority of pathways leading to forest loss in a mining region in the Brazilian Amazon are related to road construction (Figure 7). The deforestation surrounding roads represented 75% of the total deforestation between 1991 and 2014 in Serra do Navio, revealing the importance of this type of structure in driving forest loss in remote mining regions (SIQUEIRA-GAY; SÁNCHEZ, 2020).

Historic mining activities associated with roads in Renca region entailed fragmentation, intensifying the previous fishbone pattern of deforestation (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020). These findings corroborate and provide additional evidence to previous studies that investigated the impacts of mining on forest landscapes. Malaviya et al. (2010) report that, besides forest loss, mining entails impacts on the landscape, even after the mining closure. Junker et al. (2015) discuss species distribution and fragmentation in different scenarios of conservation and emphasize the increased fragmentation due to mining and other projects in Liberia. Fagiewicz (2014) identified fragmentation as the main process in post-mining areas. Similarly, Malaviya et al., (2010) applied landscape metrics in a mining landscape and identified a greater degree of forest fragmentation. These evidences emphasize the importance of a comprehensive analysis of drivers of change at the regional level, assessing road effects in mining regions as a way to avoid and minimize impacts on biodiversity in further decisions of projects and land management in the region (SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020).

7.2.2. Policy tools for addressing cumulative impacts on forests

The spatial and temporal crowding of mining projects in the Brazilian Amazon (LOBO et al., 2018) calls for a proper impact assessment and rigorous implementation of the mitigation hierarchy (EKSTROM; BENNUN; MITCHELL, 2015). There are a plethora of policy tools, especially considering regional and territorial developments (UN ENVIRONMENT, 2018). Many formal processes and systematic analysis of environmental impacts are proposed: Strategic Environmental Assessment (GUNN; NOBLE, 2011), Regional Cumulative Impact Assessment (NOBLE, 2008), Watershed Cumulative Impact Assessment (DUBÉ et al., 2013), Integrated Environmental Assessment (GALLARDO et al., 2017), Territorial Impact Assessment (GOLOBIC; MAROT, 2011), among others. However, the diversity of environmental assessment did not entail a more efficient impact assessment, creating confusion between practitioners, developers, and the public (MORRISON-SAUNDERS et al., 2014).

In particular, CEA is increasingly necessary for contexts where multiple developments drive processes of land use change (NERI; DUPIN; SÁNCHEZ, 2016). CEA can be addressed at the projects level (DIBO; NOBLE; SÁNCHEZ, 2018; NOBLE; LIU; HACKETT, 2017), strategic level (GUNN; NOBLE, 2011), watershed level (GALLARDO et al., 2017), and regional level (GUNN; NOBLE, 2009). The current practice of project-based CEA in Brazil fails in evaluating and monitoring the cumulative impacts (NERI; DUPIN; SÁNCHEZ, 2016). The interaction between losses and landscape-wide consequences is disregarded by current assessments (WORLD BANK, 2019a) mainly when projects are assessed separately (NERI; DUPIN; SÁNCHEZ, 2016). The pathway generally considered in the current EIA practice is restricted to the direct loss of forest due to mining infrastructure construction, separated from ore transportation system and access roads (WORLD BANK, 2019a).

Noble (2008) emphasizes the lack of a framework to support good practice and the strong influence of project-based environmental assessment as the most important pitfalls in regional CEA in Canada. The case studied by Noble (2008) showed no assessment of projections of cumulative change and the author suggests a structured framework, such as Strategic Environmental Assessment, to properly address cumulative impacts. Similarly, Cooper (2004) emphasizes the importance of assessing the cumulative impacts of fragmentation at the strategic level to capture the entire landscape phenomenon. However, challenges regarding responsibilities and availability of information are considerable hurdles in strategic and regional assessments (NERI; DUPIN; SÁNCHEZ, 2016).

Despite all criticism of project-level CEA and difficulties of strategic-level CEA, careful scoping is always required as a good practice (IFC, 2013). As already identified by CEA practitioners (DIBO; NOBLE; SÁNCHEZ, 2018), terms of reference should specify the criteria for the determination of spatial and temporal boundaries based on ecological features of the affected valued component and methodological guidelines for assessing cumulative impact. Strengthening the scoping phase could also be a strategy to overcome the recent trend of weakening environmental licensing legislation (FONSECA; GIBSON, 2020; SIQUEIRA-GAY; SONTER; SÁNCHEZ, 2020) and avoid simplistic assessments. Dibo, Noble e Sánchez (2018) also identified a shared understanding among practitioners' perceptions about the proponent's responsibility in leading the CEA process and the government agencies in providing systematic and standardized databases and information. In this sense, Foley et al. (2017) also emphasize opportunities to bind the gap between science and practice in CEA by developing databases, tools, and models.

7.2.3. Protected areas as a conservation strategy influencing cumulative impacts on forests

Conservation planning creates PAs as a set-aside initiative to safeguard areas with high biological importance from large scale human activities. Especially in the Brazilian Amazon, their borders have been studied as barriers to deforestation (CABRAL et al., 2018; TESFAW et al., 2018) and fragmentation (CABRAL et al., 2018; MONTIBELLER et al., 2020; ROSA; GABRIEL; CARREIRAS, 2017), playing an important role in avoiding and minimizing cumulative impacts on forests. The creation of new PAs and their management (ADAMS; IACONA; POSSINGHAM, 2019) are of fundamental importance for the effectiveness of protecting and conserving biodiverse areas. The PAs management plans regulate the activities allowed in these areas, informing further decisions in the region.

Managing and mitigating cumulative impacts of mining on biodiversity and ecosystem services is an important goal for governments, conservation organizations, the mining industry (ICMM, 2006, 2010) and an urgent need in conservation science (RAITER et al., 2014; WHITEHEAD; KUJALA; WINTLE, 2017). However, there is a considerable gap between conservation science and EIA practice. From one side, conservation literature recognizes the importance of these impacts (HALPERN; FUJITA, 2013; MÖRTBERG; BALFORS; KNOL, 2007) and sets ambitious goals to conserve and restore ecosystems (CBD, 2020). On the other, there is criticism about the role played by EIA in halting impacts on biodiversity (BOND et al., 2021; GANNON, 2021).

As an initiative to be considered prior to the project's development, strategic planning for conservation allied to impact assessment demonstrated fruitful results when implemented in mining landscapes (KIESECKER et al., 2013; MÖRTBERG; BALFORS; KNOL, 2007). Conservation efforts considering the entire landscape and consequently the existing and potentially direct, indirect, and cumulative impacts caused by different projects in the region, could serve to help to support decisions about projects and their compensation schemes (SAENZ et al., 2013a).

The goal of minimizing the impacts on forests, including cumulative loss and fragmentation, through spatial arrangement of mining projects should be further discussed to generate outcomes for CEA practice (EKSTROM; BENNUN; MITCHELL, 2015) and conservation planning (KIESECKER et al., 2010, 2013; SAENZ et al., 2013b). Especially in the Brazilian Amazon, where offsetting impacts is dependent on companies' practice rather than legislation requirements, a strategic assessment considering priority areas for development at the strategic level (KIESECKER et al., 2010) could avoid negative consequences of piecemeal environmental licensing and better appraise cumulative impacts on forest, including fragmentation.

Although some strategic landscape planning initiatives require robust methodological aspects and interesting results, its implementation is challenging in regions where conflicts for forest areas, land tenure, and jurisdictions take place (WORLD BANK, 2019a). This emphasizes the importance of proper enforcement and strong governance to connect both, EIA and conservation planning. Stakeholder engagement and cross-sectoral collaborations are fundamental to guarantee the proper assessment of trade-offs involved in landscape planning and commitment with impacts on forest management (LEVIN-NALLY; WHITAKER; RACIONERO-GOMEZ, 2020; WORLD BANK, 2019a).

7.3. Recommendations for addressing cumulative impacts on forests in mining regions

Grounded on the previous discussion about the results of this thesis, the main recommendations for practice resulted from this research are synthesized in Figure 35.

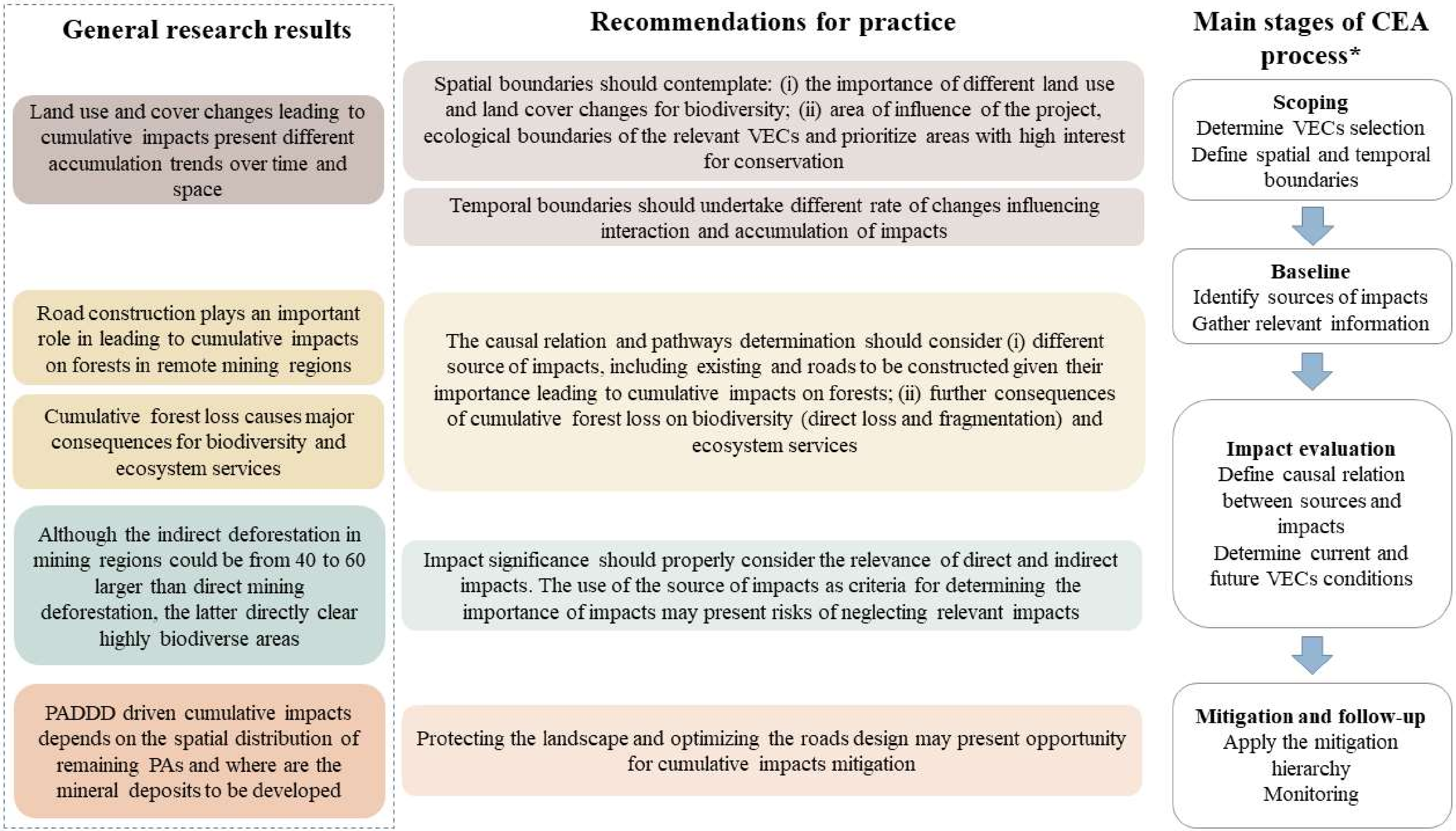


Figure 35. Framework showing the general research results and further recommendations for practice considering the main stages of CEA process. *Main stages of CEA process adapted from SÁNCHEZ (2020)

Land use and cover changes leading to cumulative impacts present different accumulation trends over time and affects different regions in the landscape

LULC depends mainly on the associated industry and infrastructure required for the development. These changes evolve with different rates of change, influencing in impacts interaction and accumulation. These aspects should be considered during the scoping phase when determining spatial and temporal boundaries. Spatial boundaries determination should contemplate the importance of different LULC changes for biodiversity considering different matrix effects in forest remnants. Besides that, the area of influence of the project should consider the ecological boundaries of the relevant VECs and avoid areas with high interest for conservation. In addition to this, temporal boundaries should undertake different rate of changes influencing interaction and accumulation of impacts

Roads play an important role in leading to cumulative impacts on forests in remote mining regions

The sources of impacts determination and causal relation definition are important parts of the CEA process (NOBLE, 2015; SÁNCHEZ, 2020). The case of mining development in pristine areas in the Amazon showed that roads play an important cause of cumulative impacts, featuring most of the pathways leading to cumulative forest loss. Analyzing historic changes, deforestation surrounding roads showed great importance in the regional dynamics, resulting in changes in the landscape structure. The determination of causal relations and pathways should properly consider the different sources of impacts, including existing and new roads to be constructed given their importance leading to cumulative impacts on forests.

Cumulative forest loss causes major consequences for biodiversity and ecosystem services

The determination of causality in impact evaluation includes featuring sources and all impacts related, including further secondary impacts (COOPER, 2004). The loss of forest surrounding roads would fragment the landscape, potentially causing biodiversity loss. The cumulative loss of forest also results in loss of ecosystem services when removing habitats essential for climate regulation, food, and raw materials production. In this context, the causal relation and further pathways should consider the entire chain of actions of impacts, including those related to biodiversity and ecosystem services loss.

Although the indirect deforestation in mining regions could be from 40 to 60 larger than direct mining deforestation, the latter directly clear highly biodiverse areas

Evaluate the impact significance is an important part of CEA process (IFC, 2013) and should properly consider the relevance of direct and indirect impacts. The use of the source of impacts as criteria for determining the importance of impacts may present risks of neglecting relevant impacts when associating less relevance for those indirect. The other way around is not valid as well, because a part of mining direct clearing occurs in highly biodiverse areas, revealing the great importance of these impacts for biodiversity conservation. The significance of both, direct and indirect impacts, should be evaluated to require proper mitigation strategies.

PADDD driven cumulative impacts depends on the spatial distribution of remaining PAs and where are the mineral deposits to be developed

IFC (2013) emphasizes, as part of the stage of cumulative impacts management, the importance of project design in cumulative impacts mitigation. Regional coordination between other projects and initiatives is required for the collaborative protection of regional and strategic areas to preserve biodiversity (IFC, 2013; KIESECKER et al., 2010). Our results emphasize the role played by protecting the landscape and optimizing the road design as an opportunity for avoiding and minimizing cumulative impacts on forests. Especially in CEA, where coordination between regional initiatives is required, we emphasize the need for binding the gap between conservation initiatives and impact assessments of future projects to curb extensive biodiversity and ecosystem services loss in mining regions.

8. Conclusions

8.1. General conclusions

Mining and associated infrastructure drive direct, indirect, and cumulative impacts on forests. Forests are directly affected by the construction and operation of the project and indirectly by third parties' projects, such as roads and other activities to support mining operation or induced by the operation of the mine and its transportation infrastructure. Both, direct and indirect impacts interact resulting in a cumulative loss of forests, which in turn is a source of effects for biodiversity by directly removing habitats and entailing fragmentation, and for ecosystem services by clearing important areas responsible to provide regulation, provision, and cultural ecosystem services.

The interplay among drivers of change in the landscape depends strongly on the type of commodities and infrastructure required for project development. A region with mining associated with pulp mill activities is highly influenced by silviculture expansion in contrast to another region with gold and iron ore activities, where pastureland expansion was more relevant surrounding roads. The results show the complexity of the interaction between cumulative impacts on forests. The results show the importance of maintaining PAs and protect areas with high biological importance in mining regions. Indirect deforestation would entail significant fragmentation depending on whether new roads and transportation are required and on the areas that remain protected in the landscape.

The systematic analysis of the type of the infrastructure associated with mining and other drivers of change at a regional and comprehensive scale is fundamental for proper assessment of impacts related to the project development. This research unfolded five main recommendations for addressing cumulative impacts on forests in mining regions: (i) spatial boundaries determination should tackle the dynamic of LULC change across the landscape; (ii) temporal boundaries should undertake rates of changes influencing interaction and accumulation of impacts; (iii) causal relation of impacts should take into account the importance of roads as source for impacts and further effects of forest loss on biodiversity and ecosystem services; (iv) the impact evaluation should properly consider the relevance of direct and indirect impacts; (v) protecting the landscape and optimizing roads design are opportunities for impacts mitigation. The integration of conservation efforts and EIA of mining projects is needed to ensure the conservation of forests and the rich benefits they provide.

8.2. Future research directions

This thesis advanced in the understanding of the problems involving cumulative impacts on forests in the Brazilian Amazon and further regulations of PAs to avoid and minimize these impacts. However, naturally, many more investigations could be done in this matter. Here, further research directions are presented.

Firstly, more can be done in investigating the impacts of artisanal and small-scale mining in forests and biodiversity. An initial investigation revealed that the rates of clearing can change among the two types of mining (for more information see Appendix 5). The comparison of impacts versus regulatory requirements of large scale and small scale is needed, especially considering that both could interact exacerbating forest impacts when both activities are in the same landscape (WORLD BANK, 2019a). Deforestation impacts of artisanal and small-scale mining are intrinsically dependent on the geology, type of mining and may occur in remote areas with high biological importance similarly to large-scale mining (WORLD BANK, 2019b). Further investigations of impacts on biodiversity, such as aspects related to affected ecological processes and forest resilience, are lacking to properly understand the impacts interactions and consequent management requirements.

Secondly, more can be done using participative methods for scenario design. Spatial explicit modeling using cellular automata is a proper tool for simulating future LULC changes, however, more can be done using traditional knowledge and participative approaches (SIQUEIRA-GAY et al., 2020b). Interviews, workshops, and focal groups are tools already explored by the literature, but there is a lack of usage of these approaches especially considering perceptions of traditional communities about mining projects and further communication of these in modeling and science.

References

ABE, C. A. et al. Modeling the effects of land cover change on sediment concentrations in a gold-mined Amazonian basin. Regional Environmental Change, v. 19, n. 6, p. 1801–1813, 2019.

ABESSA, D.; FAMÁ, A.; BURUAEM, L. The systematic dismantling of Brazilian environmental laws risks losses on all fronts. Nature Ecology and Evolution, v. 3, n. 4, p. 510–511, 2019.

ADAMS, V. M.; IACONA, G. D.; POSSINGHAM, H. P. Weighing the benefits of expanding protected areas versus managing existing ones. Nature Sustainability, v. 2, n. 5, p. 404–411, 2019.

ADELLE, C.; WEILAND, S. Policy assessment: The state of the art. Impact Assessment and Project Appraisal, v. 30, n. 1, p. 25–33, 2012.

AGENCIA NACIONAL DE MINERAÇÃO (ANM). Sumário mineral 2017. Volume 37. 2019

AGENCIA NACIONAL DE MINERAÇÃO (ANM). Processos minerários: Sistema de Informação Geográficas da Mineração (SIGMINE). Disponível em: http://sigmine.dnpm.gov.br/webmap/. Acesso em: fev 2020.

ALMEIDA, C. A. De et al. High spatial resolution land use and land cover mapping of the Brazilian Legal Amazon in 2008 using Landsat-5 / TM and MODIS data. Acta Amazonica, v. 46, n. 3, p. 291–302, 2008.

ALVES, A. R.; COUTINHO, A. dos R. Life cycle assessment of niobium: A mining and production case study in Brazil. Minerals Engineering, v. 132, n. September 2018, p. 275–283, 2019.

ANSELIN, L. Local indicators of spatial association—LISA. Geographical Analysis, v. 27, n. 2, p. 93–115, 1995.

ARAÚJO, M. L. S. de et al. Spatiotemporal dynamics of soybean crop in the Matopiba region, Brazil (1990–2015). Land Use Policy, v. 80, n. September 2018, p. 57–67, 2019.

ARLIDGE, W. N. S. et al. A Global Mitigation Hierarchy for Nature Conservation. BioScience, v. 68, n. 5, p. 336–347, 2018.

ARMENDÁRIZ-VILLEGAS, E. J. et al. Metal mining and natural protected areas in Mexico: Geographic overlaps and environmental implications. Environmental Science and Policy, v. 48, p. 9–19, 2015.

ARNESEN, A. S. et al. Monitoring flood extent in the lower Amazon River floodplain using ALOS/PALSAR ScanSAR images. Remote Sensing of Environment, v. 130, p. 51–61, 2013.

ASNER, G. P. et al. Elevated rates of gold mining in the Amazon revealed through high-resolution monitoring. Proceedings of the National Academy of Sciences, v. 110, n. 46, p. 18454–18459, 2013.

ASNER, G. P.; TUPAYACHI, R. Accelerated losses of protected forests from gold mining in the Peruvian Amazon. Environmental Research Letters, v. 12, p. 094004, 2016.

ATHAYDE, S. et al. Improving policies and instruments to address cumulative impacts of small hydropower in the Amazon. Energy Policy, v. 132, n. May, p. 265–271, 2019.

ATLIN, C.; GIBSON, R. Lasting regional gains from non-renewable resource extraction: The role of sustainability-based cumulative effects assessment and regional planning for mining development in Canada. The Extractive Industries and Society, v. 4, n. 1, p. 36–52, 2017.

BARAGWANATH, K.; BAYI, E. Collective property rights reduce deforestation in the Brazilian Amazon. Proceedings of the National Academy of Sciences, n. August, p. 1–8, 2020.

BARBER, C. P. et al. Roads, deforestation, and the mitigating effect of protected areas in the Amazon. Biological Conservation, v. 177, n. 2014, p. 203–209, 2014.

BARROS, M. N. R. et al. Validação dos dados TerraClass para o município de Paragominas. In: Anais do XVIII Simpósio Brasileiro de Sensoriamento Remoto -SBSR, Santos. 2017.

BEBBINGTON, A. J. et al. Resource extraction and infrastructure threaten forest cover and community rights. Proceedings of the National Academy of Sciences, v. 115, n. 52, p. 13164–13173, 2018.

BEGOTTI, R. A.; PERES, C. A. Rapidly escalating threats to the biodiversity and ethnocultural capital of Brazilian Indigenous Lands. Land Use Policy, v. 96, n. March, p. 104694, 2020.

BERNARD, E.; PENNA, L. A. O.; ARAÚJO, E. Evolution of Protected Areas in Brazil. Conservation Biology, v. 28, n. 4, p. 939–950, 2014.

BRITISH GEOLOGICAL SURVEY (BGS). Mineral profile Niobium-Tantalum. Minerals UK Centre for sustainable mineral development. 2011.

BIASOTTO, L. D.; KINDEL, A. Power lines and impacts on biodiversity: A systematic review. Environmental Impact Assessment Review, v. 71, n. November 2017, p. 110–119, 2018.

BLASCHKE, T. et al. Geographic Object-Based Image Analysis - towards a new paradigm. ISPRS Journal of Photogrammetry and Remote Sensing, v. 87, p. 180–191, 2014.

BOHAM-CARTER. Geographic Information Systems for Geoscientists: Modelling with GIS. In: Computer methods in the geosciences. New York: Elsevier, 1994. p. 402.

BOND, A. et al. Taking an environmental ethics perspective to understand what we should expect from EIA in terms of biodiversity protection. Environmental Impact Assessment Review, v. 86, n. May 2020, p. 106508, 2021.

BORIONI, R.; GALLARDO, A. L. F. C.; SÁNCHEZ, L. E. Advancing scoping practice in environmental impact assessment: an examination of the Brazilian federal system. Impact Assessment and Project Appraisal, v. 35, n. 3, p. 200–213, 2017.

BRAGAGNOLO, C. et al. Modelling Local Attitudes to Protected Areas in Developing Countries. Conservation and Society, v. 14, n. 3, p. 163–182, 2016.

BRASIL. Decreto No 89404 de 24 de fevereiro de 1984. Brasília, DF: Senado, 1984.

BRASIL. Decreto No 9.142, de 22 de agosto de 2017. Brasília, DF: Senado, 2017a.

BRASIL. Decreto No 9.147, de 28 de agosto de 2017. Brasília, DF: Senado, 2017b.

BRAUN, A. et al. Multi-method dynamical reconstruction of the ecological impact of copper mining on Chinese historical landscapes. Ecological Modelling, v. 303, p. 42–54, 2015.

BRIDGE, G. Contested terrain: mining and the environment. Annual Review of Environment and Resources, v. 29, p. 205–259, 2004.

BRISMAR, A. Attention to impact pathways in EISs of large dam projects. Environmental Impact Assessment Review, v. 24, n. 1, p. 59–87, 2004.

BRODERICK, M.; DURNING, B.; SÁNCHEZ, L. E. Cumulative effects. In: Methods of environmental and social impact. 4 Ed. ed. London: Routledge, 2018. p. 724.

BROOKS, T. M. et al. Habitat Loss and Extinction in the Hotspots of Biodiversity. v. 16, n. 4, p. 909–923, 2002.

CABRAL, A. I. R. et al. Deforestation pattern dynamics in protected areas of the Brazilian Legal Amazon using remote sensing data. Applied Geography, v. 100, n. October, p. 101–115, 2018.

CANADIAN COUNCIL OF MINISTERS OF THE ENVIRONMENT. Canada-wide definitions and principles for cumulative effects. 2014. Disponível em: <https://www.ccme.ca/files/Resources/enviro_assessment/CE Definitions and Principles 1.0 EN.pdf>. Acessado em: 08 mar 2021.

CARDOSO JÚNIOR, R. A. F.; MAGRINI, A.; DA HORA, A. F. Environmental licensing process of power transmission in Brazil update analysis: Case study of the Madeira transmission system. Energy Policy, v. 67, p. 281–289, 2014.

CARVALHO, W. D. et al. Deforestation control in the Brazilian Amazon: A conservation struggle being lost as agreements and regulations are subverted and bypassed. Perspectives in Ecology and Conservation, n. June, 2019.

CBD. Zero Draft of the Post-2020 Global Biodiversity Framework (2020). Disponível em: < https://www.cbd.int/article/zero-draft-update-august-2020>. Acesso em: 08 mar 2021.

COLSON, F.; BOGAERT, J.; CEULEMANS, R. Fragmentation in the Legal Amazon, Brazil: Can landscape metrics indicate agricultural policy differences? Ecological Indicators, v. 11, n. 5, p. 1467–1471, 2011.

COOK, C. N. et al. Quantifying the extent of protected-area downgrading, downsizing, and degazettement in Australia. Conservation Biology, v. 31, n. 5, p. 1039–1052, 2017.

COOPER, L. M. Guidelines for Cumulative Effects Assessment in SEA of Plans. EPMG Occasional Paper 04/LMC/CEA, Imperial College London. 2004.

COOPER, L. M. Network analysis in CEA, ecosystem services assessment and green space planning. Impact Assessment and Project Appraisal, v. 28, n. 4, p. 269–278, 2010.

CURTIS, P. G. et al. Classifying drivers of global forest loss. v. 361, n. 6407, p. 1108–1111, 2018.

DEZÉCACHE, C. et al. Gold-rush in a forested El Dorado: Deforestation leakages and the need for regional cooperation. Environmental Research Letters, v. 12, n. 3, 2017.

DIBO, A. P. A.; NOBLE, B. F.; SÁNCHEZ, L. E. Perspectives on driving changes in project-based cumulative effects assessment for biodiversity: lessons from the canadian experience. Environmental Management, v. 62, p. 929–941, 2018.

DOBSON, A. et al. Habitat Loss, Trophic Collapse, and the Decline of Ecosystem Services. Ecology, v. 87, n. 8, p. 1915–1924, 2007.

DUARTE, C. G. et al. Practitioners' perceptions of the Brazilian environmental impact assessment system: results from a survey. Impact Assessment and Project Appraisal, v. 5517, n. June, p. 1–17, 2017.

DUBÉ, M. G. et al. A framework for assessing cumulative effects in watersheds: An introduction to Canadian case studies. Integrated Environmental Assessment and Management, v. 9, n. 3, p. 363–369, 2013.

DUDLEY, N.; PETER, S.; SUE, S. Guidelines for applying protected area management categories including IUCN WCPA best practice guidance on recognizing protected areas and assigning management categories and governance types. Best Practice Protected Area Guidelines Series No. 21. 2013.

DUINKER, P. N. et al. Scientific dimensions of cumulative effects assessment: toward improvements in guidance for practice. Environmental Reviews, v. 52, n. October 2012, p. 121029052013006, 2012.

DURÁN, A. P.; RAUCH, J.; GASTON, K. J. Global spatial coincidence between protected areas and metal mining activities. Biological conservation, v. 160, p. 272–278, 2013.

EDWARDS, D. P.; LAURANCE, W. F. Preventing tropical mining disasters. Science, v. 350, p. 1482–1482, 2015.

EKSTROM, J.; BENNUN, L.; MITCHELL, R. A cross-sector guide for implementing the Mitigation Hierarchy. Cambridge: The Biodiversity Consultancy. 2015.

EL BIZRI, H. R. et al. Mining undermining Brazil's environment. Science, v. 353, n. 6296, p. 2–3, 2016.

EMPRESA DE PESQUISAS ENERGÉTICAS. Avaliação Ambiental Integrada da Bacia do rio Jari. Relatório. Hydros Engenharia, 2011.

EVANS, J. S.; KIESECKER, J. M. Shale gas, wind and water: Assessing the potential cumulative impacts of energy development on ecosystem services within the Marcellus play. PLoS ONE, v. 9, n. 2, p. e89210, 2014.

FAGIEWICZ, K. Spatial processes of landscape transformation in mining areas (Case study of opencast lignite mines in Morzysław, Niesłusz, Gosławice). Polish Journal of Environmental Studies, v. 23, n. 4, p. 1123–1136, 2014.

FEARNSIDE, P. M. Impacts of Brazil's Madeira River Dams: Unlearned lessons for hydroelectric development in Amazonia. Environmental Science & Policy, v. 38, p. 164–172, 2014.

FEARNSIDE, P. M. Brazilian politics threaten environmental policies. Science, v. 353, p. 746–748, 2016.

FERNANDES, G. W. et al. The deadly route to collapse and the uncertain fate of Brazilian rupestrian grasslands. Biodiversity and Conservation, v. 27, n. 10, p. 2587–2603, 2018a.

FERNANDES, G. W. et al. The deadly route to collapse and the uncertain fate of Brazilian rupestrian grasslands. Biodiversity and Conservation, v. 27, n. 10, 2018b.

FERRANTE, L.; FEARNSIDE, P. M. Brazil's new president and "ruralists" threaten Amazonia's environment, traditional peoples and the global climate. Environmental Conservation, v. 46, p. 261–263, 2019.

FERRANTE, L.; FEARNSIDE, P. M. The Amazon' s road to deforestation. Science, v. 20, n. 2019, p. 20–22, 2020.

FERRANTE, L.; FEARSNIDE, P. M. Brazil threatens Indigenous lands. Science, v. 368, n. 6490, p. 1–4, 2020.

FERRANTE, L.; GOMES, M.; FEARNSIDE, P. M. Amazonian indigenous peoples are threatened by Brazil's Highway BR-319. Land Use Policy, v. 94, n. October 2019, p. 104548, 2020.

FERREIRA, G. B. When the blanket is too short: Potential negative impacts of expanding indigenous land over a national park in a high priority area for conservation. Land Use Policy, v. 76, n. February, p. 359–364, 2018.

FERREIRA, J. et al. Brazil's environmental leadership at risk: Mining and dams threaten protected areas. Science, v. 346, n. 6210, p. 706–707, 2014a.

FERREIRA, J. et al. Brazil's environmental leadership at risk. Science, v. 346, p. 706–707, 2014b.

FERREIRA, J. et al. Carbon-focused conservation may fail to protect the most biodiverse tropical forests. Nature Climate Change, v. 8, n. 8, p. 744–749, 2018.

FOLEY, M. M. et al. The challenges and opportunities in cumulative effects assessment. Environmental Impact Assessment Review, v. 62, p. 122–134, 2017.

FONSECA, A.; GIBSON, R. B. Testing an ex-ante framework for the evaluation of impact assessment laws: Lessons from Canada and Brazil. Environmental Impact Assessment Review, v. 81, n. August 2019, p. 106355, 2020.

FONSECA, A.; SÁNCHEZ, L. E.; RIBEIRO, J. C. J. Reforming EIA systems: A critical review of proposals in Brazil. Environmental Impact Assessment Review, v. 62, p. 90–97, 2017.

FORREST, J. L. et al. Tropical deforestation and carbon emissions from Protected Area Downgrading, Downsizing, And Degazettement (PADDD). Conservation Letters, v. 8, n. 3, p. 153–161, 2015.

FRANKS, D. M.; BRERETON, D.; MORAN, C. J. The cumulative dimensions of impact in resource regions. Resources Policy, v. 38, n. 4, p. 640–647, 2013.

FREITAS, S. R. et al. A model of road effect using line integrals and a test of the performance of two new road indices using the distribution of small mammals in an Atlantic Forest landscape. Ecological Modelling, v. 247, p. 64–70, 2012.

GALLARDO, A. L. C. F. et al. A avaliação de impactos cumulativos no planejamento ambiental de hidrelétricas na bacia do rio Teles Pires (região amazônica). Desenvolvimento e Meio Ambiente, v. 43, p. 22–47, 2017.

GANNON, P. The time is now to improve the treatment of biodiversity in canadian environmental impact statements. Environmental Impact Assessment Review, v. 86, n. April 2020, p. 106504, 2021.

GASTAUER, M. et al. Mine land rehabilitation: Modern ecological approaches for more sustainable mining. Journal of Cleaner Production, v. 172, p. 1409–1422, 2018.

GEIST, H. J.; LAMBIN, E. F. Proximate causes and underlying driving forces of tropical deforestation. BioScience, v. 52, n. 2, p. 143, 2002.

GOLOBIC, M.; MAROT, N. Territorial impact assessment: Integrating territorial aspects in sectoral policies. Evaluation and Program Planning, v. 34, n. 3, p. 163–173, 2011.

GUNN, J. H.; NOBLE, B. F. A conceptual basis and methodological framework for regional strategic environmental assessment (R-SEA). Impact Assessment and Project Appraisal, v. 27, n. 4, p. 258–270, 2009.

GUNN, J.; NOBLE, B. F. Conceptual and methodological challenges to integrating SEA and cumulative effects assessment. Environmental Impact Assessment Review, v. 31, n. 2, p. 154–160, 2011.

HADDAD, N. M. et al. Habitat fragmentation and its lasting impact on Earth' s ecosystems. Applied Ecology, n. March, p. 1–10, 2015.

HALPERN, B. S.; FUJITA, R. Assumptions, challenges, and future directions in cumulative impact analysis. Ecosphere, v. 4, n. 10, p. 1–11, 2013.

HANNA, K.; NOBLE, B. F. Using a Delphi study to identify effectiveness criteria for environmental assessment. Impact Assessment and Project Appraisal, v. 33, n. 2, p. 116–125, 2015.

HANNA, P. et al. Improving the effectiveness of impact assessment pertaining to Indigenous peoples in the Brazilian environmental licensing procedure. Environmental Impact Assessment Review, v. 46, p. 58–67, 2014.

HEGMANN, G. et al. Cumulative Effects Assessment Practitioners Guide. Prepared by AXYS Environmental Consulting Ltd. and the CEA Working Group for the Canadian Environmental Assessment Agency, Hull, Quebec. 1999.

HIRONS, M. Managing artisanal and small-scale mining in forest areas: Perspectives from a poststructural political ecology. Geographical Journal, v. 177, n. 4, p. 347–356, 2011.

HOGAN, D. M. et al. Estimating the cumulative ecological effect of local scale landscape changes in South Florida. Environmental Management, v. 49, n. 2, p. 502–515, 2012.

HOOKE, R. B.; MARTÍN-DUQUE, J. F.; PEDRAZA, J. Land transformation by human: A review. GSA Today, v. 22, n. 12, p. 4–10, 2012.

HOTA, P.; BEHERA, B. Opencast coal mining and sustainable local livelihoods in Odisha, India. Mineral Economics, v. 29, n. 1, p. 1–13, 2016.

HUND, K.; SCHURE, J.; GOES, A. Van Der. Extractive industries in forest landscapes: options for synergy with REDD + and development of standards in the Democratic Republic of Congo. Resources Policy, v. 54, n. September, p. 97–108, 2017.

HYDE, J. L.; BOHLMAN, S. A.; VALLE, D. Transmission lines are an under-acknowledged conservation threat to the Brazilian Amazon. Biological Conservation, v. 228, p. 343–356, 2018.

ICMBIO. Plano de manejo Parque Nacional Montanhas do Tumucumaque. Macapá: Ministério do Meio Ambiente/Instituto Chico Mendes de Conservação da Biodiversidade. 2009.

ICMM. Good Practice Guidance for Mining and Biodiversity. London: International Council on Mining and Metals. 2006.

ICMM. Mining and Biodiversity: A collection of case studies. London: International Council on Mining and Metals. 2010.

IFC - INTERNATIONAL FINANCE CORPORATION. Overview of Performance Standards on Environmental and Social Sustainability. 2012.

IFC - INTERNATIONAL FINANCE CORPORATION. Good practice handbook: Cumulative impact assessment and management, guidance for the private sector in emerging markets. 2013.

INSTITUTO SOCIOAMBIENTAL. Povos indígenas no Brasil. Disponível em: <https://pib.socioambiental.org/pt/Página_principal>. Acesso em: 11 jun. 2020. 2020a.

INSTITUTO SOCIOAMBIENTAL. O impacto da pandemia na Terra Indígena Yanomami. São Paulo: Instituto Socioambiental. 2020b.

JOHNSON, C. J. et al. Growth-inducing infrastructure represents transformative yet ignored keystone environmental decisions. Conservation Letters, v. e12696, p. 1–7, 2019.

JONES, F. C. Cumulative effects assessment: theoretical underpinnings and big problems. Environmental Reviews, v. 24, n. 2, p. 187–204, 2016.

JUNKER, J. et al. Integrating wildlife conservation with conflicting economic land-use goals in a West African biodiversity hotspot. Basic and Applied Ecology, v. 16, n. 8, p. 690–702, 2015.

KALAMANDEEN, M. et al. Pervasive Rise of Small-scale Deforestation in Amazonia. Scientific Reports, v. 8, n. 1, p. 1–10, 2018.

KIESECKER, J. M. et al. Development by design: Blending landscapelevel planning with the mitigation hierarchy. Frontiers in Ecology and the Environment, v. 8, n. 5, p. 261–266, 2010.

KIESECKER, J. M. et al. Development by design: cooperative mitigation planning for Barrick Gold's Kanowna Belle operations in western Australia. Technical Report. The Nature Conservancy. 2013.

KILLEEN, T. A Perfect Storm in the Amazon Wilderness: Development and Conservation in the Context of the Initiative for the Integration of the Regional Infrastructure of South America (IIRSA). Arlington: Conservation International, 2007. v. 3

KOBAYASHI, H.; WATANDO, H.; KAKIMOTO, M. A global extent site-level analysis of land cover and protected area overlap with mining activities as an indicator of biodiversity pressure. Journal of Cleaner Production, v. 84, n. 1, p. 459–468, 2014.

KRONER, R. E. G. et al. The uncertain future of protected lands and waters. Science, v. 364, n. 6443, p. 881–886, 2019.

KRONER, R. E. G.; KRITHIVASAN, R.; MASCIA, M. B. Effects of protected area downsizing on habitat fragmentation in Yosemite National Park (USA), 1864 – 2014. Ecology and Society, v. 21, n. 3, 2016.

LANDIM, S. N. T.; SÁNCHEZ, L. E. The contents and scope of environmental impact statements: how do they evolve over time? Impact Assessment and Project Appraisal, v. 30, n. 4, p. 217–228, 2012.

LANGHAMMER, P. F.; BUTCHART, S. H. M.; BROOKS, T. M. Key Biodiversity Areas. Encyclopedia of the Anthropocene, v. 3, p. 341–345, 2018.

LARREY-LASSALLE, P. et al. A methodology to assess habitat fragmentation effects through regional indexes: Illustration with forest biodiversity hotspots. Ecological Indicators, v. 89, n. September 2017, p. 543–551, 2018.

LATRUBESSE, E. M. et al. Damming the rivers of the Amazon basin. Nature Publishing Group, v. 546, n. 7658, p. 363–369, 2017.

LAURANCE, W. F.; GOOSEM, M.; LAURANCE, S. G. W. Impacts of roads and linear clearings on tropical forests. Trends in Ecology and Evolution, v. 24, n. 12, p. 659–669, 2009.

LE TOURNEAU, F. M. The sustainability challenges of indigenous territories in Brazil's Amazonia. Current Opinion in Environmental Sustainability, v. 14, p. 213–220, 2015.

LECHNER, A. M. et al. Challenges of integrated modelling in mining regions to address social , environmental and economic impacts. Environmental Modelling and Software, v. 93, p. 268–281, 2017.

LECHNER, A. M. et al. Historical socio-environmental assessment of resource development footprints using remote sensing. Remote Sensing Applications: Society and Environment, v. 15, n. December 2018, p. 100236, 2019.

LEVIN-NALLY, E.; WHITAKER, S.; RACIONERO-GOMEZ, B. How to bring about forest-smart mining: strategic entry points. Technical Report. Levin Sources and Fauna & Flora International. 2020.

LIMA, J. M. G. Relatório técnico 20: Perfil da Mineração do Nióbio. Relatório técnico. Ministério de minas e energia e Banco Mundial. 2010.

LIMA, M. et al. The forests in the indigenous lands in Brazil in peril. Land Use Policy, v. 90, n. June 2019, p. 104258, 2020.

LINDENMAYER, D. B.; FISCHER, J. Habitat fragmentation and landscape change: an ecological and conservation synthesis. Washington DC: Island Press, 2006.

LOBO, F. D. L. et al. Distribution of Artisanal and Small-Scale Gold Mining in the Tapajós River Basin (Brazilian Amazon) over the Past 40 Years and Relationship with Water Siltation. Remote Sensing, v. 8, p. 579, 2016.

LOBO, F. D. L. et al. Mapping Mining Areas in the Brazilian Amazon Using MSI / Sentinel-2 Imagery (2017). Remote Sensing, v. 10, n. 1178, p. 1–14, 2018.

LOBO, F. L.; COSTA, M. P. F.; NOVO, E. M. L. M. Time-series analysis of Landsat-MSS/TM/OLI images over Amazonian waters impacted by gold mining activities. Remote Sensing of Environment, v. 157, p. 170–184, 2015.

LÓPEZ, S. et al. Landscape change in Southern Ecuador: An indicator-based and multi-temporal evaluation of land use and land cover in a mixed-use protected area. Ecological Indicators, v. 115, n. January 2019, p. 106357, 2020.

MACKAY, D. A. R.; SIMANDL, G. J. Geology, market and supply chain of niobium and tantalum—a review. Mineralium Deposita, v. 49, n. 8, p. 1025–1047, 2014.

MACKINNON, A. J.; DUINKER, P. N.; WALKER, T. R. The application of science in Environmental Impact Assessment. London: Routledge, 2018.

MALAVIYA, S. et al. Landscape approach for quantifying land use land cover change (1972-2006) and habitat diversity in a mining area in Central India (Bokaro, Jharkhand). Environmental Monitoring and Assessment, v. 170, n. 1–4, p. 215–229, 2010.

MANGUEIRA, J. R. S. A. et al. Plant diversity conservation in highly deforested landscapes of the Brazilian Atlantic Forest. Perspectives in Ecology and Conservation, 2021.

MARTINS, G. A. Metodologia da investigação científica para ciências sociais aplicadas. Secunda ed. São Paulo: Atlas, 2009.

MASCIA, M. B. et al. Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900-2010. Biological Conservation, v. 169, n. 2014, p. 355–361, 2014.

MASCIA, M. B.; PAILLER, S. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conservation Letters, v. 4, n. 1, p. 9–20, 2011.

MCGARIGAL, K. Fragstats Help. 2015.

MCGARIGAL, K.; MARKS, B. FRAGSTAT: Spatial pattern analysis program for quantifying landscape structure. Technical Report. United States Department of Agriculture, Pacific Northwest Research Station. 1995.

MILHEIRAS, S. G.; MACE, G. M. Assessing ecosystem service provision in a tropical region with high forest cover: Spatial overlap and the impact of land use change in Amapá Brazil. Ecological Indicators, v. 99, n. December 2018, p. 12–18, 2019.

MINISTÉRIO PÚBLICO FEDERAL. Mineração ilegal de ouro na Amazônia: marcos jurídicos e questões controversas. Brasília: MPF, 2020.

MITCHELL, M. G. E. et al. Reframing landscape fragmentation's effects on ecosystem services. Trends in Ecology and Evolution, v. 30, n. 4, p. 190–198, 2015.

MITCHELL, M. G. E.; BENNETT, E. M.; GONZALEZ, A. Forest fragments modulate the provision of multiple ecosystem services. Journal of Applied Ecology, v. 51, p. 909–918, 2014.

MONMONIER, M. S. Measures of pattern complexity for choroplethic maps. American Cartographer, v. 1, n. 2, p. 159–169, 1974.

MONTES, C. R. et al. Genesis, mineralogy and geochemistry of kaolin deposits of the Jari river, amapá state, Brazil. Clays and Clay Minerals, v. 50, n. 4, p. 494–503, 2002.

MONTIBELLER, B. et al. Increasing fragmentation of forest cover in Brazil's Legal Amazon from 2001 to 2017. Scientific Reports, v. 10, n. 1, p. 5803, 2020.

MORRISON-SAUNDERS, A. et al. Strengthening impact assessment: a call for integration and focus. Impact Assessment and Project Appraisal. v.32, n. 1, p. 2-8, 2014.

MÖRTBERG, U. M.; BALFORS, B.; KNOL, W. C. Landscape ecological assessment: A tool for integrating biodiversity issues in strategic environmental assessment and planning. Journal of Environmental Management, v. 82, n. 4, p. 457–470, 2007.

MURGUÍA, D. I.; BRINGEZU, S.; SCHALDACH, R. Global direct pressures on biodiversity by large-scale metal mining: Spatial distribution and implications for conservation. Journal of Environmental Management, v. 180, p. 409–420, 2016.

NASCIMENTO, W. R. et al. Mapping changes in the largest continuous Amazonian mangrove belt using object-based classification of multisensor satellite imagery. Estuarine, Coastal and Shelf Science, v. 117, p. 83–93, 2013.

NAUGHTON-TREVES, L.; HOLLAND, M. B. Losing ground in protected areas? Science, v. 364, n. 6443, p. 832–833, 2019.

NERI, A.; DUPIN, P.; SÁNCHEZ, L. E. A pressure-state-response approach to cumulative impact assessment. Journal of Cleaner Production, v. 126, p. 288–298, 2016.

NEVES, A. K. et al. TerraClass x MapBiomas : Comparative assessment of legend and mapping agreement analysis. In: Proceedings XVIII GEOINFO, December, Anais...2017.

NOBLE, B. Strategic approaches to regional cumulative effects assessmet: a case study of the Great Sand Hills, Canada. Impact Assessment and Project Appraisal, v. 26, n. 2, p. 78–90, 2008.

NOBLE, B. F. Cumulative Environmental Effects Assessment. In: Introduction to Environmental Impact Assessment. A guide to Principles and Practice. Third ed. [s.l.] Oxford University Press, 2015. p. 343.

NOBLE, B.; LIU, J.; HACKETT, P. The contribution of project environmental assessment to assessing and managing cumulative effects: Individually and collectively insignificant? Environmental Management, v. 59, n. 4, p. 531–545, 2017.

NOBRE, C. A. et al. Land-use and climate change risks in the Amazon and the need of a novel sustainable development paradigm. Proceedings of the National Academy of Sciences, v. 113, n. 39, p. 10759–10768, 2016.

NUNES, S. et al. Compensating for past deforestation: Assessing the legal forest surplus and deficit of the state of Pará, eastern Amazonia. Land Use Policy, v. 57, p. 749–758, 2016.

OECD (ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT). OECD Environmental Performance Reviews: Brazil 2015. 2015.

OLIVEIRA, M. J. de O. Diagnóstico do setor mineral do Estado do Amapá. Amapá: Instituto de Pesquisas Científicas e Tecnológicas do Estado do Amapá - IEPA. 2010.

PACK, S. M. et al. Protected Area Downgrading, Downsizing, and Degazettement (PADDD) in the Amazon. Biological Conservation, v. 197, p. 32–39, 2016.

PADILLA, A. J.; GHALAYINI, Z. T.; PAPP, J. F. Niobium. In: United States Geological Survey. Minerals Yearbook, 2019.

PAIVA, P. F. P. R. et al. Deforestation in protect areas in the Amazon: a threat to biodiversity. Biodiversity and Conservation, n. 29, p. 19–38, 2020.

PARDINI, P. F. P. R.; BUENO, A. A.; GARDNER, T. A.; PRADO, P. I.; METZGER, J. P. Beyond the fragmentation threshold hypothesis: Regime shifts in biodiversity across fragmented landscapes. PLoS ONE, v. 5, n. 10, 2010.

PARDINI, R.; NICHOLS, L.; PÜTTKER, T. Biodiversity Response to Habitat Loss and Fragmentation. Encyclopedia of the Anthropocene. v. 3, p. 229-249. 2018.

PEDLOWSKI, M. A. et al. Patterns and impacts of deforestation in Rondonia, Brazil. Landscape an urban planning, v. 38, p. 149–157, 1997.

PERDICOÚLIS, A.; GLASSON, J. How Clearly Is Causality Communicated in Eia? Journal of Environmental Assessment Policy and Management, v. 14, n. 03, p. 1250020, 2012.

PERDICOÚLIS, A.; PIPER, J. Network and system diagrams revisited: Satisfying CEA requirements for causality analysis. Environmental Impact Assessment Review, v. 28, n. 7, p. 455–468, 2008.

PERDICOÚLIS, A.; SÁNCHEZ, L. E.; JESUS, J. Registering cumulative effects in DCDs. Systems Planner, v. 28, n. 28, p. 1–3, 2014.

PEREIRA, E. J. de A. L. et al. Policy in Brazil (2016–2019) threaten conservation of the Amazon rainforest. Environmental Science and Policy, v. 100, n. April, p. 8–12, 2019.

PETERSON, E. B.; CHAN, Y.-H.; PETERSON, N. M. Cumulative effects assessment in Canada: An agenda for action and research. Ottawa: Supply and Services Canada, 1987.

PFAFF, A. et al. Protected areas' impacts on Brazilian Amazon deforestation: Examining conservation - Development interactions to inform planning. PLoS ONE, v. 10, n. 7, p. 1–17, 2015.

PHILLIPS, D. Bolsonaro declares "the Amazon is ours" and calls deforestation data "lies". The Guardian online. Disponível em:<https://www.theguardian.com/world/2019/jul/19/jair-bolsonaro-brazil-amazon-rainforest-deforestation>. Acesso em 19 jul 2019. 2019

PÜTZ, S. et al. Long-term carbon loss in fragmented Neotropical forests. Nature Communications, v. 5, n. 5037, p. 1–8, 2014.

RAITER, K. G. et al. Under the radar: Mitigating enigmatic ecological impacts. Trends in Ecology and Evolution, v. 29, n. 11, p. 635–644, 2014.

REDONDO-VEGA, J. M. et al. Changes in land use due to mining in the north-western mountains of Spain during the previous 50 years. Catena, v. 149, p. 844–856, 2017.

RICKETTS. The matrix matters: Effective isolation in fragmented landscapes. The American Naturalist, v. 158, n. 1, p. 87, 2001.

RITTER, C. D. et al. Environmental impact assessment in Brazilian Amazonia: Challenges and prospects to assess biodiversity. Biological Conservation, v. 206, p. 161–168, 2017.

RÖDIG, E. et al. Global patterns of tropical forest fragmentation. Nature, v. 554, n. 7693, p. 519–522, 2018.

ROSA, I. M. D.; GABRIEL, C.; CARREIRAS, J. M. B. Spatial and temporal dimensions of landscape fragmentation across the Brazilian Amazon. Regional Environmental Change, v. 17, n. 6, p. 1687–1699, 2017.

RUDKE, A. P. et al. Impact of mining activities on areas of environmental protection in the southwest of the Amazon: A GIS- and remote sensing-based assessment. Journal of Environmental Management, v. 263, n. March, p. 110392, 2020.

RUNGE, C. A. et al. Quantifying the conservation gains from shared access to linear infrastructure. Conservation Biology, v. 31, n. 6, p. 1428–1438, 2017.

SAENZ, S. et al. Development by design in Colombia: Making mitigation decisions consistent with conservation outcomes. PLoS ONE, v. 8, n. 12, 2013a.

SAENZ, S. et al. A framework for implementing and valuing biodiversity offsets in colombia: A landscape scale perspective. Sustainability (Switzerland), v. 5, n. 12, p. 4961–4987, 2013b.

SÁNCHEZ, L. E. Por que não avança a avaliação ambiental estratégica no Brasil? Estudos Avançados, v. 31, n. 89, p. 167–183, 2017.

SÁNCHEZ, L. E. Avaliação de Impacto Ambiental. Third ed. São Paulo: Oficina de textos, 2020.

SÁNCHEZ, L. E.; SILVA-SÁNCHEZ, S. S.; NERI, A. C. Guide for mine closure planning. Brasília: Instituto Brasileiro de Mineração.

SCHUELER, V.; KUEMMERLE, T.; SCHRÖDER, H. Impacts of surface gold mining on land use systems in Western Ghana. Ambio, v. 40, n. 5, p. 528–539, 2011.

SCHULTZ, C. A. The U.S. Forest Service's analysis of cumulative effects to wildlife: A study of legal standards, current practice, and ongoing challenges on a National Forest. Environmental Impact Assessment Review, v. 32, n. 1, p. 74–81, 2012.

SECRETARIA DE ESTADO DE MEIO AMBIENTE. Plano de manejo da floresta estadual do Paru. Belém: Secretaria de Estado de Meio Ambiente. 2010.

SECRETARIA DE ESTADO DE MEIO AMBIENTE. Plano de Manejo da Reserva Biológica Maicuru. Belém: Secretaria de Estado de Meio Ambiente. 2011.

SEITZ, N. E.; WESTBROOK, C. J.; NOBLE, B. F. Bringing science into river systems cumulative effects assessment practice. Environmental Impact Assessment Review, v. 31, n. 3, p. 172–179, 2011.

SIMANDL, G. J. et al. Economic geology models 2. Tantalum and niobium: Deposits, resources, exploration methods and market – A primer for geoscientists. Geoscience Canada, v. 45, n. 2, p. 85–96, 2018.

SINGH, G. G. et al. Scientific shortcomings in environmental impact statements internationally. People and Nature, v. 2, n. 2, p. 369–379, 2020.

SIQUEIRA-GAY, J. et al. Pathways to positive scenarios for the Amazon forest in Pará state, Brazil. Biota Neotropica, v. 20, p. e20190905., 2020a.

SIQUEIRA-GAY, J. et al. Proposed Legislation to Mine Brazil's Indigenous Lands Will Threaten Amazon Forests and Their Valuable Ecosystem Services. One Earth, v. 3, n. 3, p. 356–362, 2020b.

SIQUEIRA-GAY, J.; SÁNCHEZ, L. E. Keep the Amazon niobium in the ground. Environmental Science and Policy, v. 111, n. May, p. 1–6, 2020.

SIQUEIRA-GAY, J.; SONTER, L. J.; SÁNCHEZ, L. E. Exploring potential impacts of mining on forest loss and fragmentation within a biodiverse region of Brazil's northeastern Amazon. Resources Policy, v. 67, n. C, p. 101662, 2020.

SLOOTWEG, R. et al. Biodiversity in Environmental Assessment. New York: Cambridge University Press, 2010.

SOARES-FILHO, B. et al. Simulating the response of land-cover changes to road paving and governance along a major Amazon highway: The Santarém-Cuiabá corridor. Global Change Biology, v. 10, n. 5, p. 745–764, 2004.

SOARES-FILHO, B. et al. Role of Brazilian Amazon protected areas in climate change mitigation. Proceedings of the National Academy of Sciences of the United States of America, v. 107, n. 24, p. 10821–10826, 2010.

SOARES-FILHO, B.; RODRIGUES, H.; FOLLADOR, M. A hybrid analytical-heuristic method for calibrating land-use change models. Environmental Modelling and Software, v. 43, p. 80–87, 2013.

SOARES-FILHO, B. S. et al. Modelling conservation in the Amazon basin. Nature, v. 440, n. March, p. 3–6, 2006.

SOARES-FILHO, B. S.; CERQUEIRA, G. C.; PENNACHIN, C. L. DINAMICA - A stochastic cellular automata model designed to simulate the landscape dynamics in an Amazonian colonization frontier. Ecological Modelling, v. 154, n. 3, p. 217–235, 2002.

SONTER, L. J. et al. A land system science meta-analysis suggests we underestimate intensive land uses in land use change dynamics. Journal of Land Use Science, v. 10, n. 2, p. 191–204, 2013.

SONTER, L. J. et al. Processes of land use change in mining regions. Journal of Cleaner Production, v. 84, p. 494–501, 2014a.

SONTER, L. J. et al. Global demand for steel drives extensive land-use change in Brazil' s Iron Quadrangle. Global Environmental Change, v. 26, p. 63–72, 2014b.

SONTER, L. J. et al. Mining drives extensive deforestation in the Brazilian Amazon. Nature Communications, v. 8, n. 1, p. 1013, 2017.

SONTER, L. J. et al. Biodiversity offsets may miss opportunities to mitigate impacts on ecosystem services. Frontiers in Ecology and the Environment, v. 16, n. 3, p. 143–148, 2018.

SONTER, L. J. et al. Renewable energy production will exacerbate mining threats to biodiversity. Nature Communications, v. 11, n. 1, p. 6–11, 2020.

SONTER, L. J.; ALI, S. H.; WATSON, J. E. M. Mining and biodiversity: key issues and research needs in conservation science. Proceedings of Royal Society B, v. 285, n. 1892, p. 1–9, 2018.

SOUZA-FILHO, P. et al. A GEOBIA Approach for Multitemporal Land-Cover and Land-Use Change Analysis in a Tropical Watershed in the Southeastern Amazon. Remote Sensing, v. 10, n. 11, p. 1683, 2018.

SOUZA-FILHO, P. W. M. et al. Four decades of land-cover, land-use and hydroclimatology changes in the Itacaiúnas River watershed, southeastern Amazon. Journal of Environmental Management, v. 167, p. 175–184, 2016.

SOUZA-FILHO, P. W. M. et al. Mapping and quantification of ferruginous outcrop savannas in the Brazilian Amazon: A challenge for biodiversity conservation. PLoS ONE, p. 1–20, 2019.

SOUZA, P. F. et al. Spatial spread of malaria and economic frontier expansion in the Brazilian Amazon. PLoS ONE, v. 14, n. 6, p. 1–25, 2019.

SPALING, H.; SMIT, B. A conceptual model of cumulative environmental effects of agricultural land drainage. Agriculture, Ecosystems & Environment, v. 53, n. 2, p. 99–108, 1995.

STEHFEST, E. et al. Key determinants of global land-use projections. Nature Communications, v. 10, n. 1, p. 2166, 2019.

STRAND, J. et al. Spatially explicit valuation of the Brazilian Amazon Forest's Ecosystem Services. Nature Sustainability, v. 1, n. 11, p. 657–664, 2018.

SWENSON, J. J. et al. Gold mining in the peruvian amazon: Global prices, deforestation, and mercury imports. PLoS ONE, v. 6, n. 4, 2011.

TAKEHARA, L (Org.). Projeto Avaliação do Potencial de Terras Raras no Brasil: área Morro dos Seis Lagos, Noroeste do Amazonas. Brasília: CPRM, 2019.

TESFAW, A. T. et al. Land-use and land-cover change shape the sustainability and impacts of protected areas. PNAS, v. 115, n. 9, p. 2084–2089, 2018.

TOWNSEND, P. A. et al. Changes in the extent of surface mining and reclamation in the Central Appalachians detected using a 1976-2006 Landsat time series. Remote Sensing of Environment, v. 113, n. 1, p. 62–72, 2009.

TREWEEK, J. Ecological Impact Assessment. Oxford: Blackwell Publishing, 1999.

UN ENVIRONMENT. Assessing Environmental Impacts - A Global Review of Legislation. Nairobi: UN Environment Law Division and UN Environment World Conservation Monitoring Centre.

UNITED NATIONS HUMAN RIGHTS COUNCIL - UNHRC. Report Threats and Violations of Human Rights in Brazil: Isolated Indigenous Peoples Summary. Disponível em: <https://www.socioambiental.org/en/noticias-socioambientais/isa-report-at-the-un-denounces-high-risk-of-genocide-of-isolated-indigenous-peoples>. Acesso em: 08 mar 2021.

USGS (UNITED STATES GEOLOGICAL SURVEY). Mineral Commodity Summaries. 2020. https://doi.org/10.3133/mcs2020.

VENTER, O. et al. Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. Nature Communications, v. 7, p. 1–11, 2016.

VERSLUIJS, M. et al. Comparing the effects of even-aged thinning and selective felling on boreal forest birds. Forest Ecology and Management, v. 475, n. July, p. 118404, 2020.

VILARDO, C.; LA ROVERE, E. L. Multi-project environmental impact assessment: insights from offshore oil and gas development in Brazil. Impact Assessment and Project Appraisal, v. 36, n. 4, p. 358–370, 2018.

VILELA, T. et al. A better Amazon road network for people and the environment. Proceedings of the National Academy of Sciences of the United States of America, v. 117, n. 13, p. 7095–7102, 2020.

VILLÉN-PÉREZ, S. et al. Mining code changes undermine biodiversity conservation in Brazil. Environmental Conservation, v. 45, n. 1, p. 1–4, 2017.

VOEGELI, G.; HEDIGER, W.; ROMERIO, F. Sustainability assessment of hydropower: Using causal diagram to seize the importance of impact pathways. Environmental Impact Assessment Review, v. 77, n. October 2018, p. 69–84, 2019.

WALKER, R. T. et al. Avoiding Amazonian Catastrophes: Prospects for Conservation in the 21st Century. One Earth, v. 1, n. 2, p. 202–215, 2019.

WALKER, W. S. et al. The role of forest conversion, degradation, and disturbance in the carbon dynamics of Amazon indigenous territories and protected areas. Proceedings of the National Academy of Sciences, v. 117, n. 6, p. 201913321, 2020.

WHITEHEAD, A. L.; KUJALA, H.; WINTLE, B. A. Dealing with cumulative biodiversity bmpacts in strategic environmental assessment: A new frontier for conservation planning. Conservation Letters, v. 10, n. 2, p. 195–204, 2017a.

WHITEHEAD, A. L.; KUJALA, H.; WINTLE, B. A. Dealing with Cumulative Biodiversity Impacts in Strategic Environmental Assessment: A New Frontier for Conservation Planning. Conservation Letters, v. 10, n. 2, p. 195–204, 2017b.

WORLD BANK. Making mining “Forest-Smart” - Executive summary report. 2019a

WORLD BANK. Forest-Smart Mining: Artisanal & Small-Scale Mining in Forest Landscapes (ASM). 2019b.

WWF. Mineração e áreas protegidas: Cenário e perspectivas. 2017a

WWF. Renca: Situação legal dos direitos minerários da reserva nacional do cobre. 2017b.

Appendix 1. Supplementary information – Chapter 2

Keep the Amazon niobium in the ground

Juliana Siqueira-Gay[1*], Luis Enrique Sánchez[1]
[1]Escola Politécnica, University of São Paulo (USP), Av. Prof. Mello Moraes, 2373 Butantã, 05508-900, São Paulo, Brazil
*Corresponding author. Email: siq.juliana@gmail.com

1. Data analysis

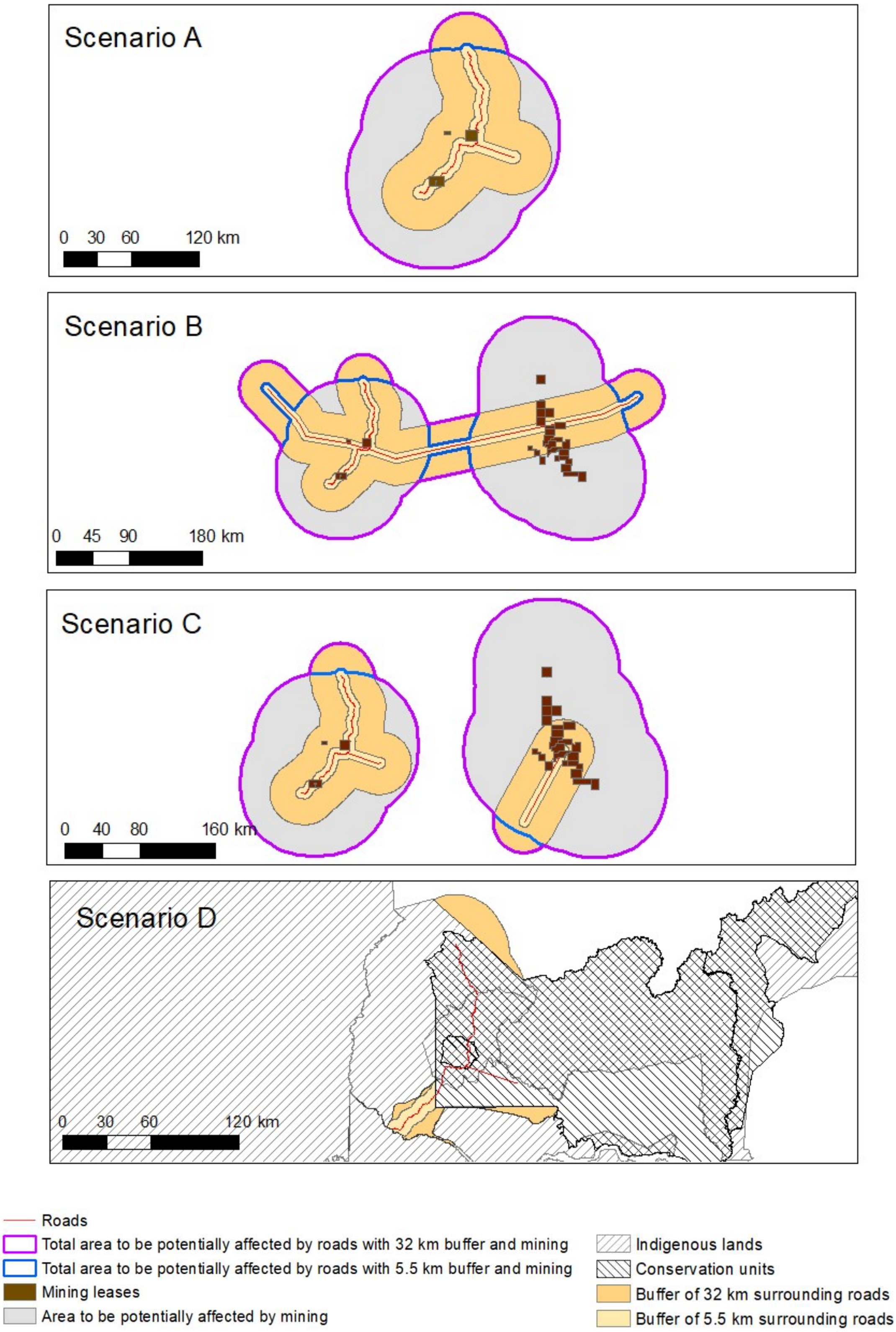
The geographical information used to propose scenarios and estimate the area to be affected is presented in Table S1.

Table S 1. Information used in the spatial data analysis

Data	Responsible	Description	Reference
Mining claims and proposals	AMN	Official spatial database of the National Mining Agency. It provides the current status (accessed on February 10th 2020) of all mineral claims and proposal covering the Brazilian territory. It contains information about the date of creation, substance and stage of the process	http://sigmine.dnpm.gov.br/webmap/
Conservation units – Sustainable use and full protection areas	MMA	Official spatial database of the Brazilian Conservation Units provided by the Ministry of the Environment	http://mapas.mma.gov.br/i3geo/datadownload.htm
Indigenous lands	Funai	Official database of the National Foundation of Indigenous People, containing the polygons, the current status, types and ethnicity of each land	http://www.funai.gov.br/index.php/shape
Roads	Ministry of Infrastructure	Roads planned and implemented (paved or not)	http://www.infraestrutura.gov.br/component/content/article/63-bit/5124-bitpublic.html#maprodo
Ports	Ministry of Infrastructure	Information about ports	http://www.infraestrutura.gov.br/component/content/article.html?id=5124
Waterways	Ministry of Transportation and Department of Transportation Infrastructure	Information about waterways classified according to the level of navigability	http://www.transportes.gov.br/conteudo/2822-base-de-dados-georreferenciados-pnlt-2010.html

For the scenarios of development of Nb deposits (A-C), a buffer of 5.5 km was calculated around the roads to feature the area to be affected by significant deforestation caused by all types of planned and unplanned roads in Amazon and a buffer of 32 km to feature the area to be affected by highways (BARBER et al., 2014) (Figure S1). To determine the area affected by significant deforestation caused by mining, a buffer of 70 km was calculated surrounding leases (SONTER et al., 2017). There is no PAs regulation in the scenarios A, B and C, therefore, the area to be affected is considered as the entire merged buffer of roads - both of 5.5 km and 32 km - and mining. For the scenario D, the buffers surrounding the existing roads was the reference (the same as in the scenario A with no new roads) and we removed the PAs boundaries to simulate the effects of blocking deforestation (PACK et al., 2016).

Figure S 1. Areas to be potentially affected in each scenario considering roads and mining leases



Appendix 2. Supplementary information – Chapter 3

This supplementary information presents:

1. Dataset information
2. Methods details
3. Parameters used for classification
4. Accuracy reports
5. Land use and land cover maps
6. Ranked transitions
7. Documents consulted

1. Dataset information

Table S1. Landsat images downloaded to compose the mosaic of satellite images

Path	Row	Date	Sensor
227	61	1993.10.20	TM/Landsat 5
		2006.07.04	TM/Landsat 5
		2017.07.18	OLI_TIRS /Landsat 8
226	61	1997.10.08	TM/Landsat 5
		2003.04.08	ETM/Landsat 7
		2018.08.15	OLI_TIRS /Landsat 8
226	60	1997.10.24	TM/Landsat 5
		2004.10.11	TM/Landsat 5
		2018.08.15	OLI_TIRS /Landsat 8
226	59	1997.10.08	TM/Landsat 5
226		2007.09.02	TM/Landsat 5
226		2018.08.15	OLI_TIRS /Landsat 8
225	60	1997.11.02	TM/Landsat 5
		2006.10.26	TM/Landsat 5
		2019.09.12	OLI_TIRS /Landsat 8

Table S2. SAR data used to compose the optical mosaics

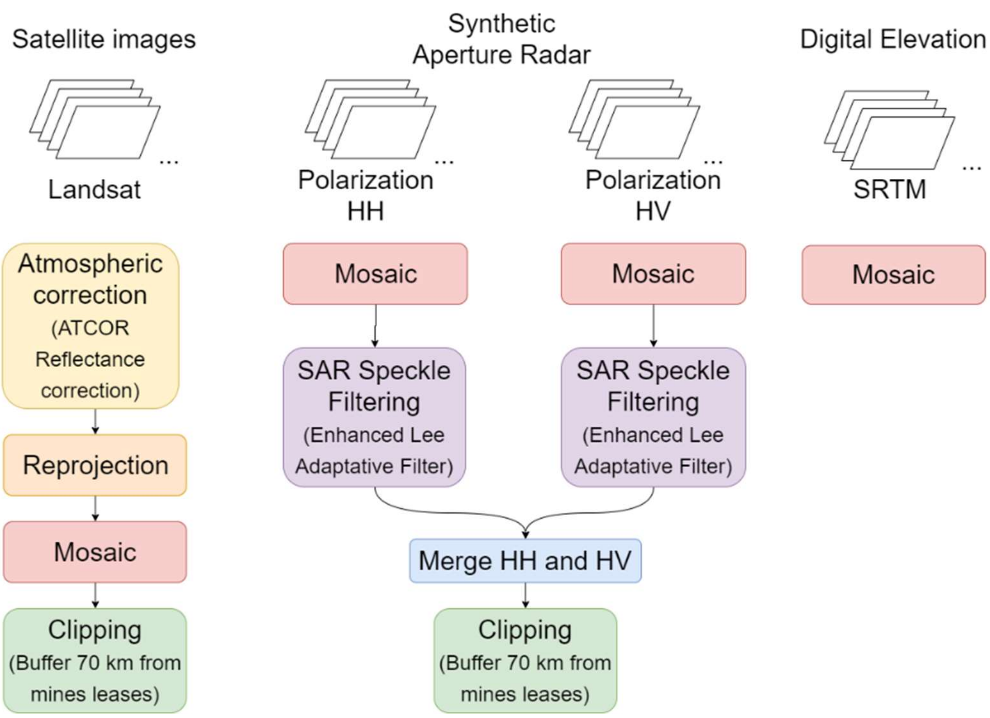
Date	Product	Grid
1996	JERS-1 HH Mosaic	N01W052 N01W053 N02W052 N02W053 N00W052 N00W052 N00W053 N00W054 S01W052 S01W053 S01W054
2007	PALSAR HH/HV mosaic	
2017	PALSAR-2 HH/HV mosaic	

Table S3. SRTM data download to compose to provide elevation information

Digital Elevation	Coordinates
SRTM 1 Arc-Second Global	N00W52
	N00W53
	N00W54
	N01W52
	N01W53
	N01W54
	S01W52
	S01W53
	S01W54
	S02W52
	S02W53
	S02W54

2. Methods details

Figure S 2. Main steps of digital image processing



3. Parameters used for classification

Table S4. Membership functions used for determining land use and land cover classes in Jari

	Jari 2017			Jari 2007			Jari 1997		
	Feature	**Limits**	**Function**	**Feature**	**Limits**	**Function**	**Feature**	**Limits**	**Function**
Forest	Mean HH	[7500]-[12000]	Min=0 Max=1	Mean HH	[5800]-[7650]	Min=0 Max=1	Mean HV	[3000]-[6000]	Min=0 Max=1
Hydrography	NDVI	[-0.26]–[0.06]	Min=0 Max=1	NDVI	[-0.108]-[0.24]	Min=0 Max=1	Mean B5	[0.2]-[8]	Min=0 Max=1
Other native vegetation	Mean SRTM	[1]-[21]	Min=0 Max=1	Mean SRTM	[-0.9]-[3.1]	Min=0 Max=1	NDVI	[0.2]-[0.5]	Min=0 Max=1
	Mean HH	[200]-[7000]	Min=0 Max=1						
Pastureland	Mean B5	[20]-[50]	Min=0 Max=1	B3	[9]-[20]	Min=0 Max=1	Mean B4	[9]-[40]	Min=0 Max=1
	Mean SRTM	[25]-[50]	Min=0 Max=1						

Table S5. Membership functions used for determining land use and land cover classes in Serra do Navio

	Serra do Navio 2017			Serra do Navio 2007			Serra do Navio 1997		
	Feature	**Limits**	**Function**	**Feature**	**Limits**	**Function**	**Feature**	**Limits**	**Function**
Forest	Mean HH	[6350]-[7300]	Min=0 Max=1	Mean HH	[6330]-[7400]	Min=0 Max=1	Mean HH	[8300]-[9200]	Min=0 Max=1
Hydrography	NDVI	[-0.25]–[0.31]	Min=0 Max=1	NDVI	[-0.3]-[0.45]	Min=0 Max=1	Mean B4	[7]-[16]	Min=0 Max=1
Other native vegetation	Mean SRTM	[1]-[75]	Min=0 Max=1	-	-	-	Mean SRTM	[0]-[65]	Min=0 Max=1

4. Accuracy reports

Table S6. Confusion matrix resulted from accuracy assessment of Serra do Navio 1997 land use and land cover map

Land use and land cover classes in Serra do Navio 1997		**Class-01**	**Class-02**	**Class-03**	**Class-04**	**Class-05**	**Class-06**	**Class-07**	**Totals**
		Forest	**Hydrography**	**Other native vegetation**	**Mining**	**Urban**	**Pastureland**	**Silviculture**	
Class-01	Forest	423	5	2	0	0	3	0	433
Class-02	Hydrography	0	17	1	0	0	3	0	21
Class-03	Other native vegetation	1	0	41	0	0	2	2	46
Class-04	Mining	0	0	0	18	0	0	0	18
Class-05	Urban	0	0	0	0	12	1	0	13
Class-06	Pastureland	3	1	0	0	0	44	0	48
Class-07	Silviculture	0	0	2	0	0	1	18	21
Totals		427	23	46	18	12	54	20	600

Table S7. Overall accuracy statistics of Serra do Navio 1997 land use and land cover map

Overall Accuracy	95.50%	**95% Confidence Interval**	(93.758%; 97.242%)
Overall Kappa Statistic	0.904	**Overall Kappa Variance**	0.063

Table S8. Accuracy for each class of Serra do Navio 2007 land use and land cover map

Land use and land cover classes in Serra do Navio 1997		**Producer's Accuracy**	**95% Confidence Interval**		**User's Accuracy**	**95% Confidence Interval**		**Kappa Statistic**
Class-01	Forest	99.06%	98.03%	100.09%	97.69%	96.16%	99.22%	0.9199
Class-02	Hydrography	73.91%	53.79%	94.03%	80.95%	61.78%	100.13%	0.8019
Class-03	Other native vegetation	89.13%	79.05%	99.21%	89.13%	79.05%	99.21%	0.8823
Class-04	Mining	100.00%	97.22%	102.78%	100.00%	97.22%	102.78%	1
Class-05	Urban	100.00%	95.83%	104.17%	92.31%	73.98%	110.64%	0.9215
Class-06	Pastureland	81.48%	70.20%	92.77%	91.67%	82.81%	100.53%	0.9084
Class-07	Silviculture	90.00%	74.35%	105.65%	85.71%	68.37%	103.06%	0.8522

Table S9. Confusion matrix resulted from accuracy assessment of Serra do Navio 2007 land use and land cover map

Land use and land cover classes in Serra do Navio 2007		Class-01	Class-02	Class-03	Class-04	Class-05	Class-06	Class-07	Totals
		Forest	Hydrography	Other native vegetation	Urban	Pastureland	Mining	Silviculture	
Class-01	Forest	409	6	4	0	4	0	0	423
Class-02	Hydrography	2	24	0	0	1	0	0	27
Class-03	Other native vegetation	2	0	33	0	0	0	0	35
Class-04	Urban	0	1	0	10	1	0	0	12
Class-05	Pastureland	3	0	0	0	56	0	1	60
Class-06	Mining	0	0	0	0	1	19	0	20
Class-07	Silviculture	0	0	0	0	1	0	22	23
Totals		416	31	37	10	64	19	23	600

Table S10. Overall accuracy statistics of Serra do Navio 2007 land use and land cover map

Overall Accuracy	95.50%	**95% Confidence Interval**	(93.76%; 97.24%)
Overall Kappa Statistic	0.908	**Overall Kappa Variance**	-1.714

Table S11. Accuracy for each class of Serra do Navio 2007 land use and land cover map

Land use and land cover classes in Serra do Navio 2007		Producer's Accuracy	95% Confidence Interval		User's Accuracy	95% Confidence Interval		Kappa Statistic
Class-01	Forest	98.32%	96.96%	99.67%	96.69%	94.87%	98.51%	0.8921
Class-02	Hydrography	77.42%	61.09%	93.75%	88.89%	75.18%	102.60%	0.8828
Class-03	Other native vegetation	89.19%	77.83%	100.55%	94.29%	85.17%	103.40%	0.9391
Class-04	Urban	100.00%	95.00%	105.00%	83.33%	58.08%	108.59%	0.8305
Class-05	Pastureland	87.50%	78.62%	96.38%	93.33%	86.19%	100.48%	0.9254
Class-06	Mining	100.00%	97.37%	102.63%	95.00%	82.95%	107.05%	0.9484
Class-07	Silviculture	95.65%	85.14%	106.16%	95.65%	85.14%	106.16%	0.9548

Table S12. Confusion matrix resulted from accuracy assessment of Serra do Navio 2017 land use and land cover map

Land use and land cover classes in Serra do Navio 2017		Class-01 Forest	Class-02 Hydrography	Class-03 Other native vegetation	Class-04 Mining	Class-05 Urban	Class-06 Pastureland	Class-07 Silviculture	Totals
Class-01	Forest	402	1	1	0	1	0	0	405
Class-02	Hydrography	2	18	0	0	1	0	0	21
Class-03	Other native vegetation	0	2	41	0	0	0	0	43
Class-04	Mining	1	0	0	8	0	0	0	9
Class-05	Urban	0	0	0	0	76	0	0	76
Class-06	Pastureland	0	0	0	1	0	23	0	24
Class-07	Silviculture	0	0	0	0	0	0	22	22
Totals		405	21	42	9	78	23	22	600

Table S13. Overall accuracy statistics of Serra do Navio 2017 land use and land cover map

Overall Accuracy	98.33%	**95% Confidence Interval**	(97.226%; 99.441%)
Overall Kappa Statistic	0.968	**Overall Kappa Variance**	-0.005

Table S14. Accuracy for each class of Serra do Navio 2017 land use and land cover map

Land use and land cover classes in Serra do Navio 2017		Producer's Accuracy	95% Confidence Interval		User's Accuracy	95% Confidence Interval		Kappa Statistic
Class-01	Forest	99.26%	98.30%	100.22%	99.26%	98.30%	100.22%	0.9772
Class-02	Hydrography	85.71%	68.37%	103.06%	85.71%	68.37%	103.06%	0.852
Class-03	Other native vegetation	97.62%	91.82%	103.42%	95.35%	87.89%	102.81%	0.95
Class-04	Mining	88.89%	62.80%	114.98%	88.89%	62.80%	114.98%	0.8872
Class-05	Urban	97.44%	93.29%	101.59%	100.00%	99.34%	100.66%	1
Class-06	Pastureland	100.00%	97.83%	102.17%	95.83%	85.76%	105.91%	0.9567
Class-07	Silviculture	100.00%	97.73%	102.27%	100.00%	97.73%	102.27%	1

Table S15. Confusion matrix resulted from accuracy assessment of Jari 1997 land use and land cover map

Land use and land cover classes in Jari 1997		Class-01 Forest	Class-02 Hydrography	Class-03 Other native vegetation	Class-04 Pastureland	Class-05 Urban	Class-06 Silviculture	Class-07 Cellulose industry	Class-08 Mining	Totals
Class-01	Forest	358	2	1	0	0	0	0	0	361
Class-02	Hydrography	0	38	0	0	0	0	0	0	38
Class-03	Other native vegetation	0	0	91	0	0	0	0	0	91
Class-04	Pastureland	1	0	1	30	0	0	0	0	32
Class-05	Urban	1	0	0	1	21	0	0	0	23
Class-06	Silviculture	0	0	0	3	0	21	0	0	24
Class-07	Cellulose industry	0	0	0	0	0	0	14	0	14
Class-08	Mining	0	0	0	0	0	0	0	17	17
Totals		360	40	93	34	21	21	14	17	600

Table S16. Overall accuracy statistics of Jari 1997 land use and land cover map

Overall Accuracy	98.33%	**95% Confidence Interval**	(97.226%; 99.441%)
Overall Kappa Statistic	0.972	**Overall Kappa Variance**	0

Table S17. Accuracy for each class of Jari 1997 land use and land cover map

Land use and land cover classes in Jari 1997		Producer's Accuracy	95% Confidence Interval		User's Accuracy	95% Confidence Interval		Kappa Statistic
Class-01	Forest	99.44%	98.54%	100.35%	99.17%	98.09%	100.24%	0.9792
Class-02	Hydrography	95.00%	87.00%	103.00%	100.00%	98.68%	101.32%	1
Class-03	Other native vegetation	97.85%	94.36%	101.34%	100.00%	99.45%	100.55%	1
Class-04	Pastureland	88.24%	75.94%	100.54%	93.75%	83.80%	103.70%	0.9337
Class-05	Urban	100.00%	97.62%	102.38%	91.30%	77.62%	104.99%	0.9099
Class-06	Silviculture	100.00%	97.62%	102.38%	87.50%	72.19%	102.82%	0.8705
Class-07	Cellulose industry	100.00%	96.43%	103.57%	100.00%	96.43%	103.57%	1
Class-08	Mining	100.00%	97.06%	102.94%	100.00%	97.06%	102.94%	1

Table S18. Confusion matrix resulted from accuracy assessment of Jari 2007 land use and land cover map

Land use and land cover classes in Jari 2007		**Class-01**	**Class-02**	**Class-03**	**Class-04**	**Class-05**	**Class-06**	**Class-07**	**Class-08**	**Totals**
		Forest	**Hydro**	**Pastureland**	**Other native vegetation**	**Mining**	**Industry**	**Urban**	**Silviculture**	
Class-01	Forest	332	5	1	1	0	0	0	4	343
Class-02	Hydro	1	44	0	0	0	0	0	0	45
Class-03	Pastureland	0	0	28	2	0	0	1	0	31
Class-04	Other native vegetation	5	0	2	77	0	0	0	0	84
Class-05	Mining	0	0	1	0	20	0	0	1	22
Class-06	Cellulose industry	0	0	0	0	0	16	0	0	16
Class-07	Urban	0	0	0	0	0	0	15	0	15
Class-08	Silviculture	2	0	3	0	0	0	0	39	44
Totals		340	49	35	80	20	16	16	44	600

Table S19. Overall accuracy statistics of Jari 2007 land use and land cover map

Overall Accuracy	95.17%	**95% Confidence Interval**	(93.37%; 96.966%)
Overall Kappa Statistic	0.925	**Overall Kappa Variance**	0

Table S20. Accuracy for each class of Jari 2007 land use and land cover map

Land use and land cover classes in Jari 2007		**Producer's Accuracy**	**95% Confidence Interval**		**User's Accuracy**	**95% Confidence Interval**		**Kappa Statistic**
Class-01	Forest	97.65%	95.89%	99.41%	96.79%	94.78%	98.80%	0.926
Class-02	Hydro	89.80%	80.30%	99.29%	97.78%	92.36%	103.20%	0.9758
Class-03	Pastureland	80.00%	65.32%	94.68%	90.32%	78.30%	102.34%	0.8972
Class-04	Other native vegetation	96.25%	91.46%	101.04%	91.67%	85.16%	98.17%	0.9038
Class-05	Mining	100.00%	97.50%	102.50%	90.91%	76.62%	105.20%	0.906
Class-06	Cellulose industry	100.00%	96.88%	103.13%	100.00%	96.88%	103.13%	1
Class-07	Urban	93.75%	78.76%	108.74%	100.00%	96.67%	103.33%	1
Class-08	Silviculture	88.64%	78.12%	99.15%	88.64%	78.12%	99.15%	0.8774

Table S21. Confusion matrix resulted from accuracy assessment of Jari 2017 land use and land cover map

Land use and land cover classes in Jari 2017		**Class-01**	**Class-02**	**Class-03**	**Class-04**	**Class-05**	**Class-06**	**Class-07**	**Class-08**	**Totals**
		Forest	**Hydrography**	**Urban**	**Other native vegetation**	**Mining**	**Pastureland**	**Cellulose industry**	**Silviculture**	
Class-01	Forest	322	2	0	6	0	4	0	1	335
Class-02	Hydrography	3	33	0	1	0	0	0	1	38
Class-03	Urban	0	0	13	0	1	2	0	0	16
Class-04	Other native vegetation	2	2	0	74	2	0	0	0	80
Class-05	Mining	1	0	0	0	16	1	0	5	23
Class-06	Pastureland	4	0	1	1	0	31	1	2	40
Class-07	Cellulose industry	0	0	0	0	0	1	22	1	24
Class-08	Silviculture	0	1	0	0	0	1	0	42	44
Totals		332	38	14	82	19	40	23	52	600

Table S22. Overall accuracy statistics of Jari 2017 land use and land cover map

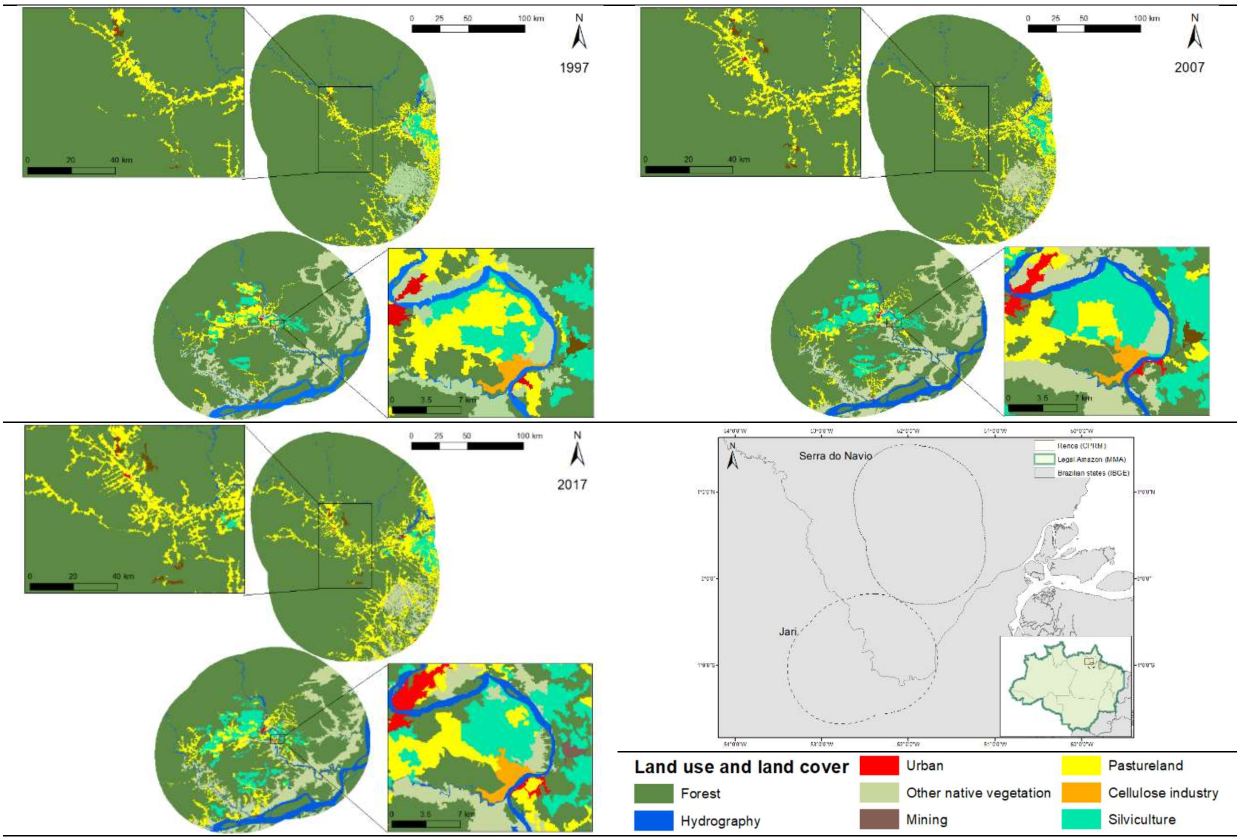
Overall Accuracy	92.17%	**95% Confidence Interval**	(89.93%; 94.40%)
Overall Kappa Statistic	0.88	**Overall Kappa Variance**	0

Table S23. Accuracy for each class of Jari 2017 land use and land cover map

Land use and land cover classes in Jari 2017		**Producer's Accuracy**	**95% Confidence Interval**		**User's Accuracy**	**95% Confidence Interval**		**Kappa Statistic**
Class-01	Forest	96.99%	95.00%	98.98%	96.12%	93.90%	98.34%	0.9131
Class-02	Hydrography	86.84%	74.78%	98.91%	86.84%	74.78%	98.91%	0.8595
Class-03	Urban	92.86%	75.80%	109.92%	81.25%	59.00%	103.50%	0.808
Class-04	Other native vegetation	90.24%	83.21%	97.28%	92.50%	86.10%	98.90%	0.9131
Class-05	Mining	84.21%	65.18%	103.24%	69.57%	48.59%	90.54%	0.6857
Class-06	Pastureland	77.50%	63.31%	91.69%	77.50%	63.31%	91.69%	0.7589
Class-07	Cellulose industry	95.65%	85.14%	106.16%	91.67%	78.53%	104.81%	0.9133
Class-08	Silviculture	80.77%	69.10%	92.44%	95.46%	88.16%	102.75%	0.9502

5. Land use and land cover maps

Figure S 3. Maps of Jari and Serra do Navio in the years 1997, 2007 and 2017



6. Documents consulted

Table S 2. Documents consulted

Documents	Source
Jari Integrated Environmental Assessment [Avaliação Ambiental Integrada do Jari]	Empresa de Pesquisas Energéticas (http://www.epe.gov.br/pt/publicacoes-dados-abertos/publicacoes/avaliacao-ambiental-integrada-aai)
Paru Management Plan [Plano de Manejo da Flota do Paru]	Imazon (https://imazon.org.br/cartilha-do-plano-de-manejo-da-flota-do-paru/)
Reserva Biológica Maicuru [Plano de Manejo da Reserva Biológica do Maicuru)	SEMA Pará
Parque Nacional do Tumucumaque [Plano de Manejo do Parque Nacional Montanhas do Tumucumaque]	Ministério do Meio Ambiente/ICMBio (http://www.icmbio.gov.br/portal/images/stories/imgs-unidades-coservacao/parna_montanhas-do-tumucumaque.pdf)
Diagnóstico do setor mineral do Estado do Amapá	Instituto de Pesquisas Científi cas e Tecnológicas do Estado do Amapá – IEPA (http://www.iepa.ap.gov.br/arquivopdf/diagnostico_mineral_amapa.pdf)

Appendix 3. Supplementary information – Chapter 5

The supplementary material content:

(1) Support documents consulted
(2) Land use and cover database corrections
(3) Modeling approach
(4) Weights of Evidence used for the model calibration
(5) Validation curves of calibrated and null models
(6) Scenarios parameters
(7) Fragmentation analysis

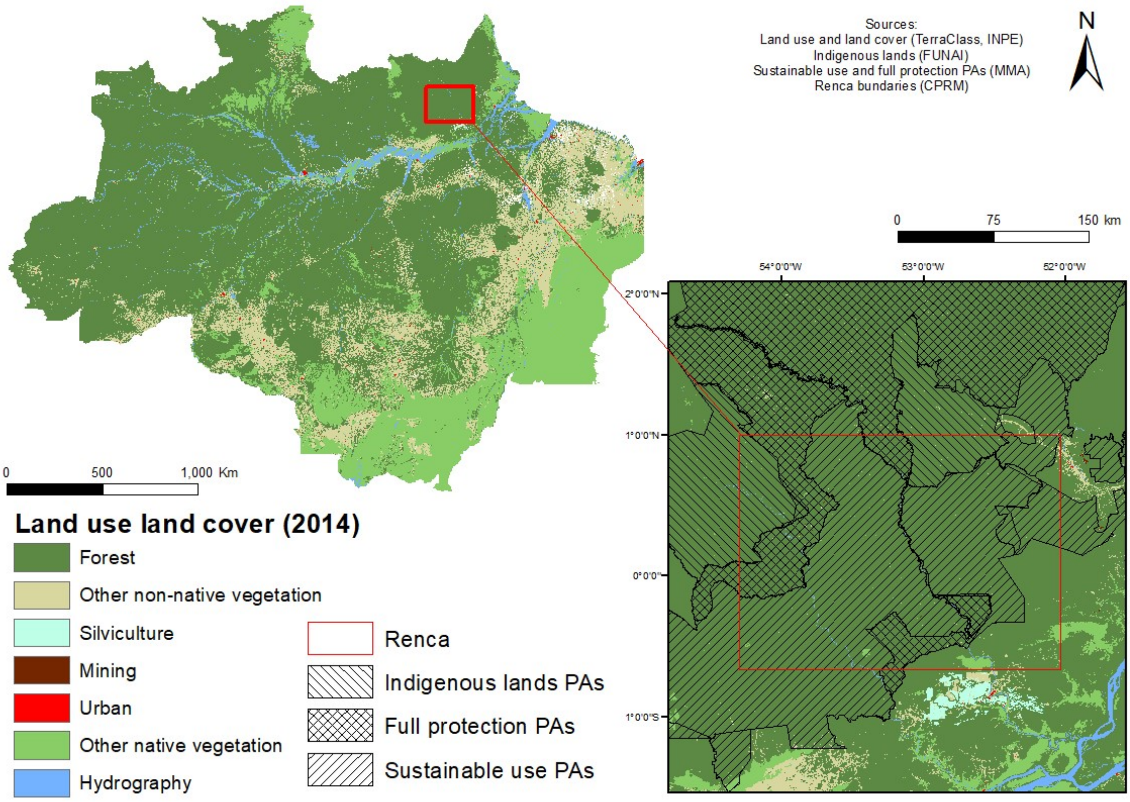
1. Support documents consulted

Table S 3. Documents consulted about planned projects in the region

Documents	Source	Description of future projects in the region
Jari Integrated Environmental Assessment [Avaliação Ambiental Integrada do Jari]	Empresa de Pesquisas Energéticas (**http://www.epe.gov.br/pt/publicacoes-dados-abertos/publicacoes/avaliacao-ambiental-integrada-aai**)	Transmission line in the development scenario of 2030 following the roads design.
MacroZEE Pará	Secretaria de Meio Ambiente e Sustentabilidade – Governo do Pará (**https://www.semas.pa.gov.br/diretorias/planejamento-ambiental/zee/**)	Area with low human pressure and protected by full protection and sustainable use PAs
Decenal Energy Plan [Plano Decenal de Energia]	Empresa de Pesquisas Energéticas (**http://www.epe.gov.br/pt/publicacoes-dados-abertos/publicacoes/plano-decenal-de-expansao-de-energia-pde**)	2029 – no relevant proposals for the region 2027 – no relevant proposals for the region 2026 – implementation of power station and transmission line from Laranjal do Jari to Jurupari
Paru Management Plan [Plano de Manejo da Flota do Paru]	Imazon (**https://imazon.org.br/cartilha-do-plano-de-manejo-da-flota-do-paru/**)	Zones with low – conservation and forest management - or moderate intervention – only extractives activities of non-timber products and mineral research

2. Land use and land cover database corrections

Figure S4. Renca surrounded by Protected Areas in a well conserved region in the Brazilian Amazon



2.1. Land use and cover dataset

Table S 4. Description of the land use and land cover classes

Land use and cover classes	Original classes from TerraClass	Description
Forest	Forest	Primary forest featured by dense and heterogeneous formation
Other non-native vegetation	Secondary vegetation, herbaceous and shrubby pasture, deforestation	Forest in second stage of regeneration and recovered deforested areas. Area cleared with cultivated pasture including areas recently deforested
Silviculture	Silviculture	Plantation of Eucalyptus or Pinus featured by homogenous formation
Urban	Urban	Areas with dense occupation and clear pattern of streets, residential and industrial use
Mining	Mining	Industrial mineral extraction
Other native vegetation	Other native vegetation and no forest	Areas featured by non-forest native vegetation, such as mangroves and flooded fields
Hydrography	Hydrography	Water bodies

2.2.Dataset preprocess

Table S 5. Changes in the land use and cover classes to correct transitions not important for the model calibration

Step	LULC TerraClass dataset	Change	Rationale
1º	Non observed	Pixels with non-observed (clouds) were assigned with value of pixels of previous LULC	Although the cloud cover was less than 1% in all years of analysis, the transitions involving non-observed were detected and they are not relevant for model calibration
2º	Hydrography	Pixels assigned with hydrography in at least one year (2004 or 2010 or 2014) was assigned as hydrography in all other years	A hydrography mask was made to correct transitions of other land uses and cover to hydrography and vice-versa
3º	Other native vegetation	Pixels classified with other native vegetation in at least one year (2004 or 2010 or 2014) was assigned as other native vegetation in all other years	The other native vegetation represents the vegetation cover in the Amazon fringe, including flooded areas and mangroves. As the focus was modeling mining and infrastructure expansion over mainly forests, the transitions with other native vegetation was not considered in the model calibration

Table S 6. Corrections in land use and land cover classes considering the expansion of mining, urban infrastructure, and forest plantation in the region

Step	LULC TerraClass dataset	Rationale
1º	Mining	No mines were closed in the years of 2004-2014, therefore, we considered that mining activities are expanding in the region
2º	Silviculture	The area of forest plantation expanded during the year of analysis due to the increased activities of the Jari project
3º	Urban	The new infrastructure, including new mining projects and silviculture expansion, induced human occupation and urban growth

Table S 7. Corrections to fix the transitions of interest in the TerraClass dataset

Transition in TerraClass dataset	Reclassified	Assumptions
2004 ONNV > 2010 Forest > 2014 ONNV	2004 ONNV > 2010 ONNV> 2014 ONNV	The area was always other non-native vegetation
2004 ONNV > 2010 Forest > 2014 Mining	2004 ONNV > 2010 Mining > 2014 Mining	The area was converted to mining already in 2010
2004 ONNV > 2010 Forest > 2014 Silviculture	2004 ONNV > 2010 Silviculture > 2014 Silviculture	The area was converted to silviculture already in 2010
2004 ONNV > 2010 Forest > 2014 Urban	2004 ONNV > 2010 Urban > 2014 Urban	The area was converted to urban already in 2010
2004 ONNV > 2010 Forest > 2014 Forest	2004 Forest > 2010 Forest > 2014 Forest	The pixel was considered permanently as forest
2004 Forest > 2010 ONNV > 2014 Forest	2004 Forest > 2010 Forest > 2014 Forest	The pixel was considered permanently as forest
2004 ONNV > 2010 ONNV > 2014 Forest	2004 ONNV > 2010 ONNV > 2014 ONNV	The pixel was considered permanently as other non-native vegetation

Table S 8. Number of cells and percentage of landscape considering each land use and cover before and after resampling the cells size in 2004

Land use and land cover	2004			
	Original (30m cells)		Resampled (100m cells)	
	Area (km²)	Percentage of landscape	Area (km²)	Percentage of landscape
Primary forest	224205	89.1%	224335	89.1%
Secondary vegetation	1303	0.5%	1306	0.5%
Silviculture	744	0.3%	743	0.3%
Shrubby pasture	744	0.3%	744	0.3%
Herbaceous pasture	339	0.1%	339	0.1%
Mining	6	0.002%	6	0.002%
Urban	80	0.032%	80	0.03%
Others	101	0.040%	101	0.04%
Non-observed	1002	0.4%	1002	0.4%
Deforestation	106	0.04%	105	0.04%
No forest	18646	7.4%	18659	7.4%
Hydrography	4302	1.7%	4303	1.7%

Table S 9. Number of cells and percentage of landscape considering each land use and cover before and after resampling the cells size in 2010

Land use and land cover	2010			
	Original (30m cells)		Resampled (100m cells)	
	Area (km²)	Percentage of landscape	Area (km²)	Percentage of landscape
Primary forest	222287	88.4%	222417	88.4%
Secondary vegetation	1752	0.7%	1753	0.7%
Silviculture	781	0.3%	781	0.3%
Shrubby pasture	1022	0.4%	1022	0.4%
Herbaceous pasture	496	0.2%	497	0.2%
Mining	19	0.01%	19	0.01%
Urban	94	0.04%	94	0.04%
Others	84	0.03%	83	0.03%
Non-observed	1960	0.8%	1960	0.8%
Deforestation	136	0.1%	136	0.1%
No forest	18646	7.4%	18659	7.4%
Hydrography	4302	1.7%	4303	1.7%

Table S 10. Number of cells and percentage of landscape considering each land use and cover before and after resampling the cells size in 2014

Land use and land cover	2014			
	Original (30m cells)		Resampled (100m cells)	
	Area (km²)	Percentage of landscape	Area (km²)	Percentage of landscape
Primary forest	222495	88.4%	222624	88.4%
Secondary vegetation	2040	0.8%	2041	0.8%
Silviculture	743	0.3%	743	0.3%
Shrubby pasture	1323	0.5%	1323	0.5%
Herbaceous pasture	1290	0.5%	1291	0.5%
Mining	31	0.01%	31	0.01%
Urban	103	0.04%	104	0.04%
Others	249	0.1%	248	0.1%
Non-observed	303	0.1%	303	0.1%
Deforestation	53	0.02%	54	0.02%
No forest	18646	7.4%	18659	7.4%
Hydrography	4302	1.7%	4303	1.7%

Table S 11. Area (km²) and percentage of landscape considering each land use and cover before and after the dataset preprocessing in 2004

Land use and land cover	2004			
	Before	%	After	%
Forest	224,335	89.2%	224599	89.2%
ONNV	2,494	1.0%	2840	1.1%
Silviculture	744	0.3%	837	0.3%
Mining	6	0.002%	6	0.002%
Urban	80	0.032%	83	0.03%
ONV	18,760	7.5%	19040	7.6%
Hydrography	4,303	1.7%	4319	1.7%
Non-observed	1,002	0.4	-	0%

Table S 12. Area (km²) and percentage of landscape considering each land use and cover before and after the dataset preprocessing in 2010

Land use and land cover	2010			
	Before	%	After	%
Forest	222,417	89.1%	223,372	88.7%
ONNV	3,408	1.4%	3,896	1.5%
Silviculture	781	0.3%	982	0.4%
Mining	19	0.01%	18	0.007%
Urban	94	0.04%	97	0.04%
ONV	18,742	7.5%	19,040	7.6%
Hydrography	4,303	1.7%	4,319	1.7%
Non-observed	1,960	0.8%	-	0%

Table S 13. Area (km²) and percentage of landscape considering each land use and cover before and after the dataset preprocessing in 2014

Land use and land cover	2014			
	%	Before	%	After
Forest	222,624	88.5%	222,789	88.5%
ONNV	4,709	1.9%	4,421	1.8%
Silviculture	743	0.3%	1,023	0.4%
Mining	31	0.01%	31	0.012%
Urban	104	0.04%	102	0.04%
ONV	18,907	7.5%	19,040	7.6%
Hydrography	4,303	1.7%	4,319	1.7%
Non-observed	303	0.01%	-	0%

3. Weights of Evidence used in the model calibration

Figure S 5. Weights of Evidence used to calibrate the transition of forest to other non-native vegetation

Forest to ONNV

1 – Indigenous lands; 2 – Sustainable use; 3 – Full protection; 4 - Unprotected

Figure S 6. Weights of Evidence used to calibrate the transition of forest to silviculture

Forest to silviculture

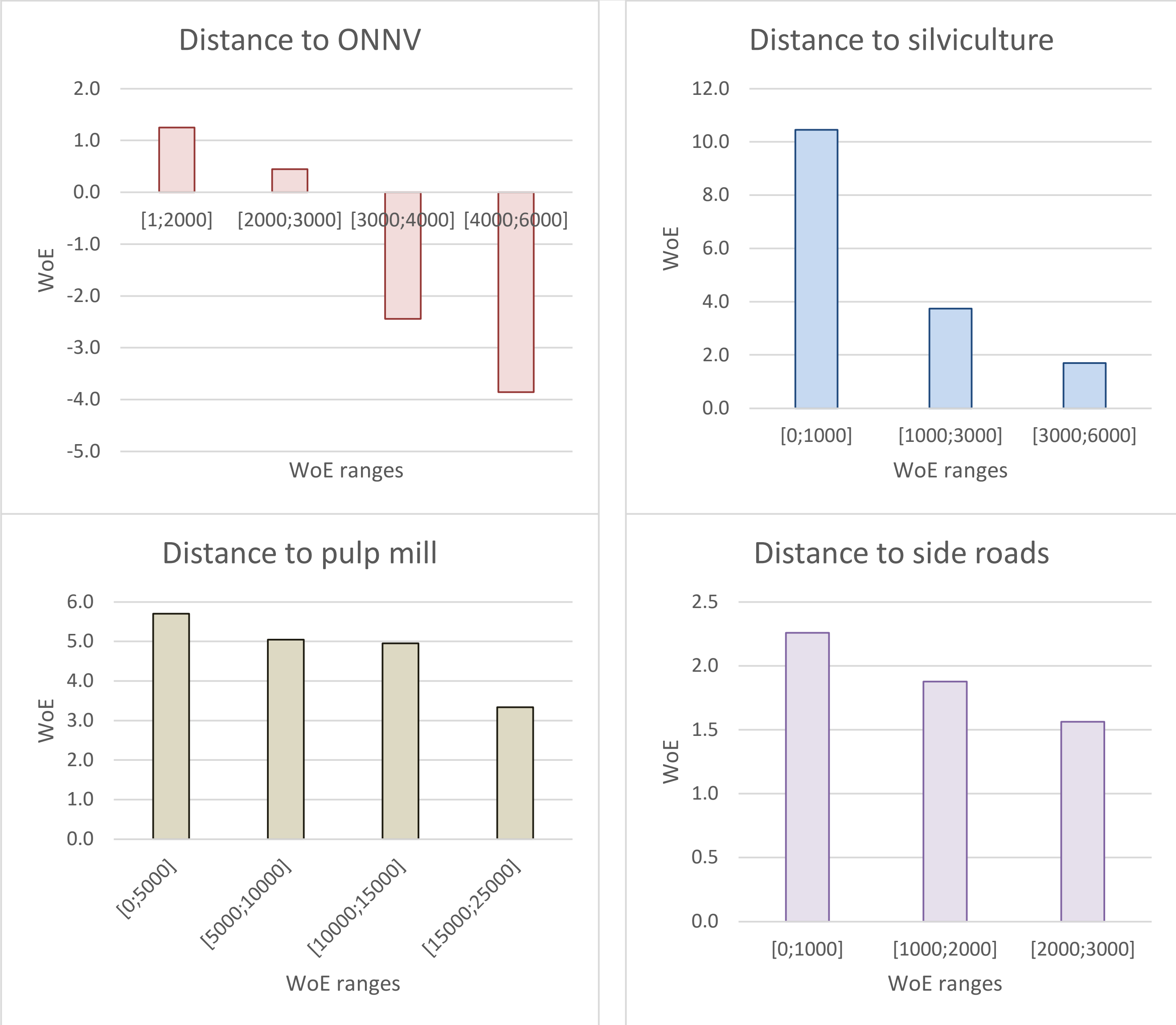


Figure S 7. Weights of Evidence used to calibrate the transition of forest to mining

Forest to mining

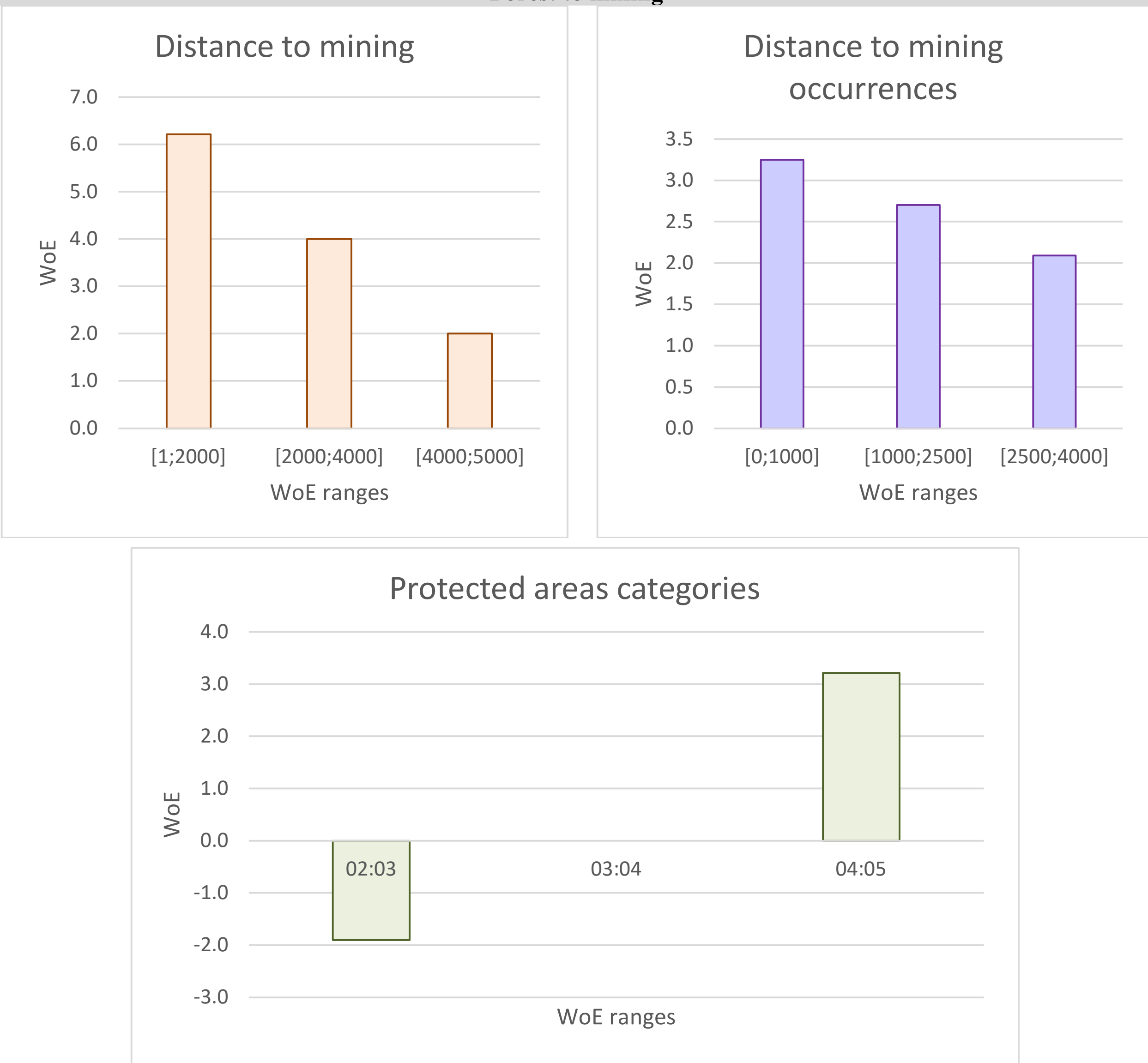


Figure S 8. Weights of Evidence used to calibrate the transition of other non-native vegetation to silviculture

ONNV to silviculture

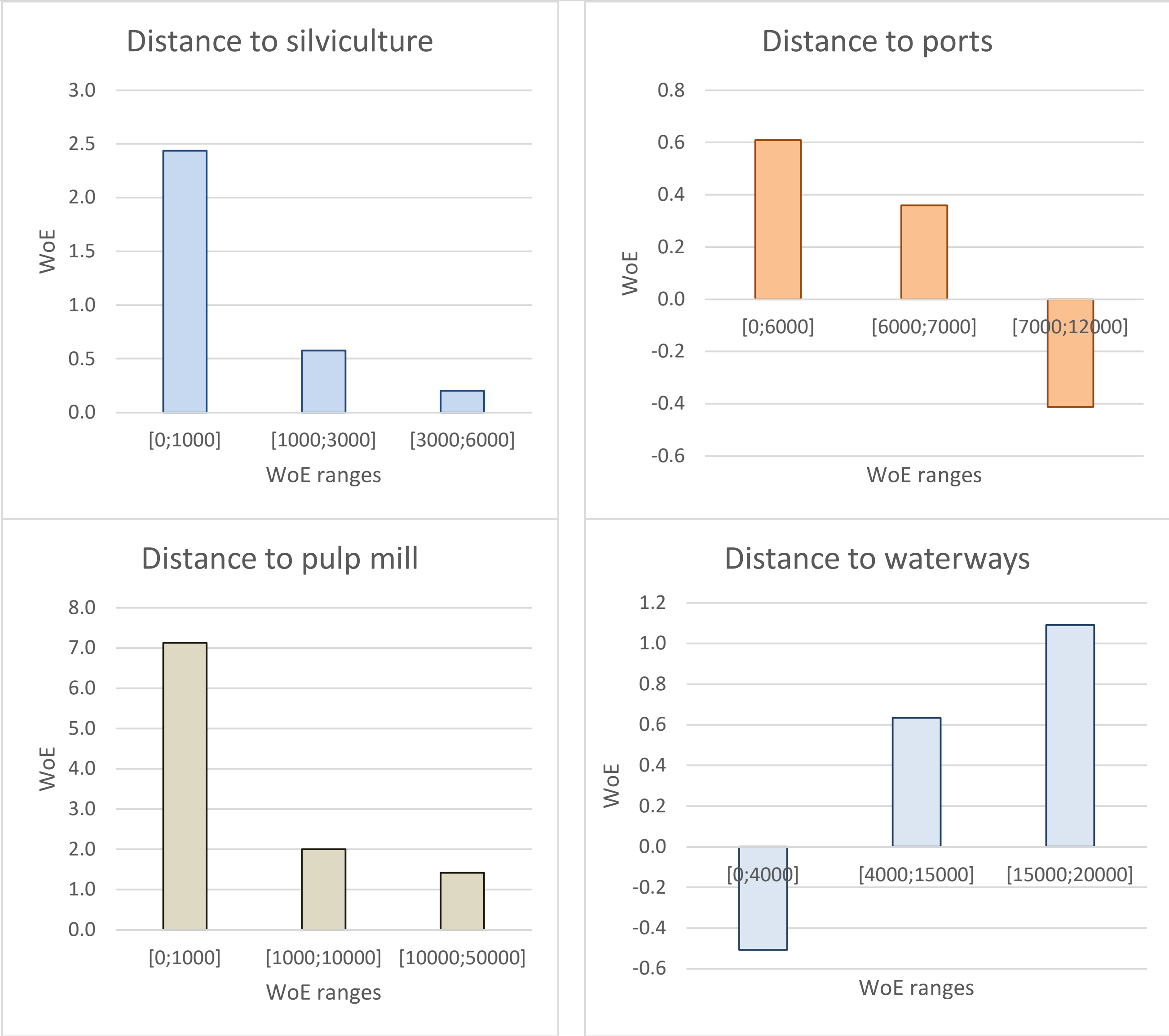


Figure S 9. Weights of Evidence used to calibrate the transition of other non-native vegetation to mining

ONNV to mining

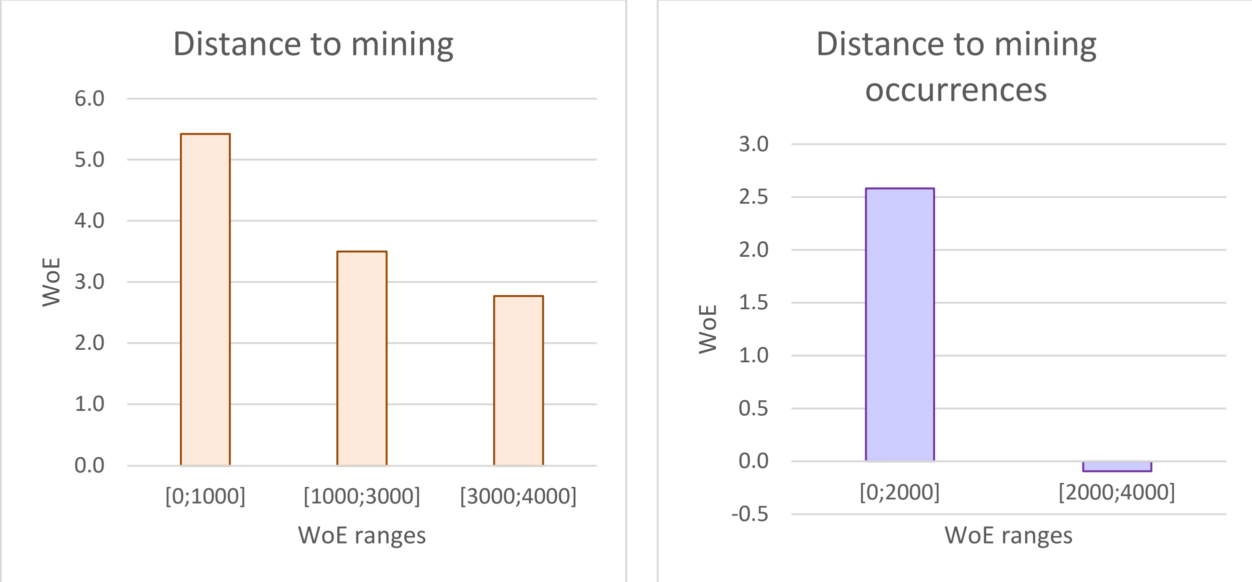


Figure S 10. Weights of Evidence used to calibrate the transition of other non-native vegetation to urban

ONNV to urban

4. Validation curves of the calibrated and null models

Figure S 11. Validation curve comparing the calibrated and null model for all transitions

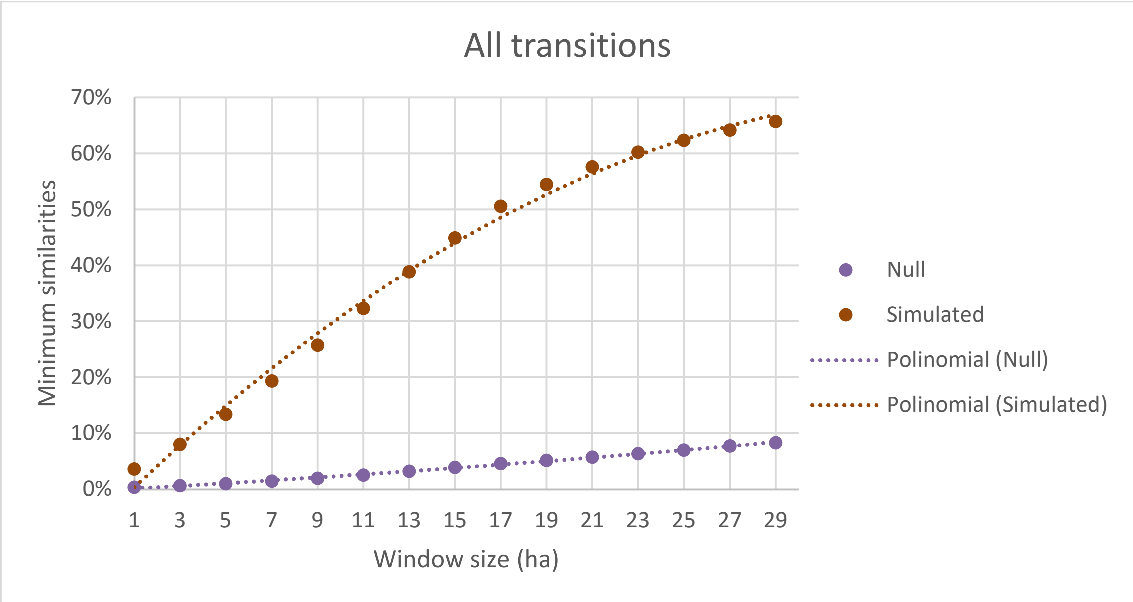


Figure S 12. Validation curve comparing the calibrated and null model for the transition of forest to other non-native vegetation

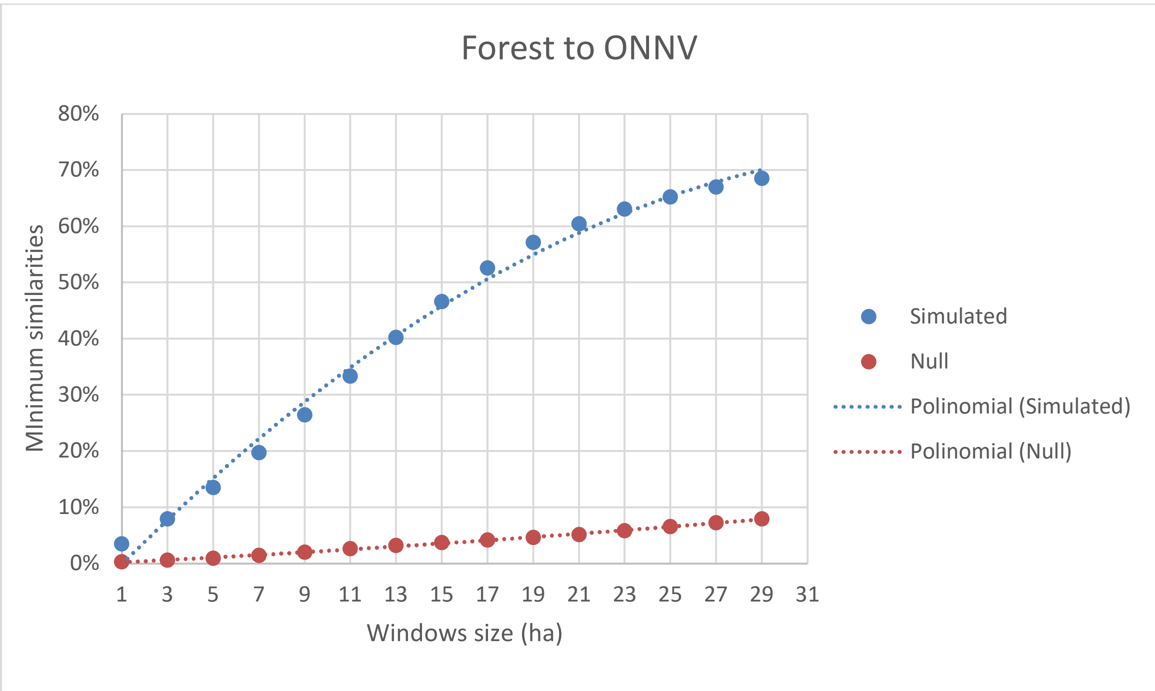


Figure S 13. Validation curve comparing the calibrated and null model for the transition of forest to silviculture

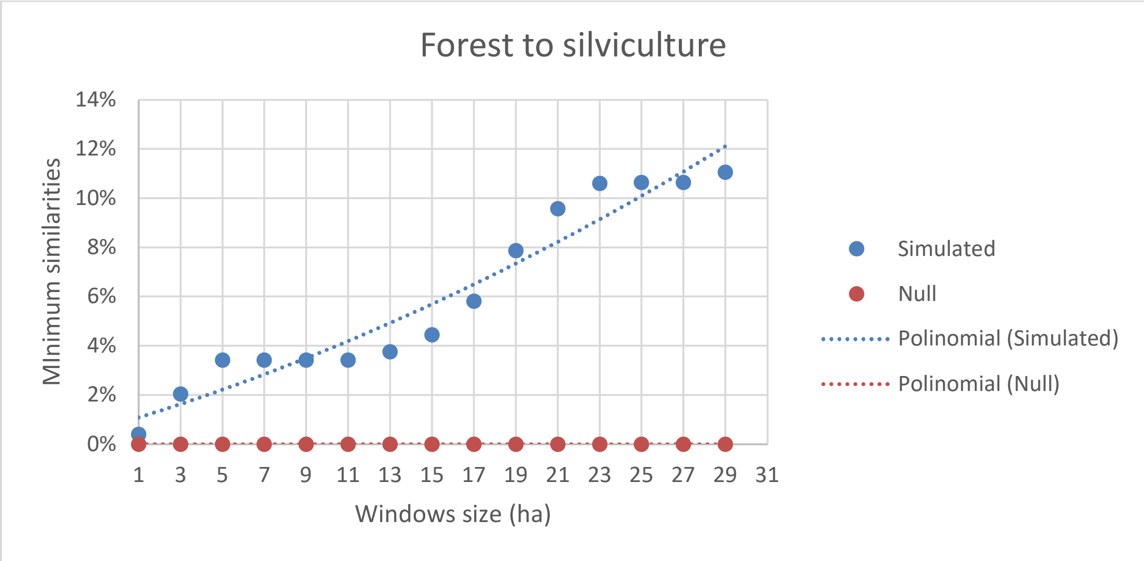


Figure S 14. Validation curve comparing the calibrated and null model for the transition of forest to mining

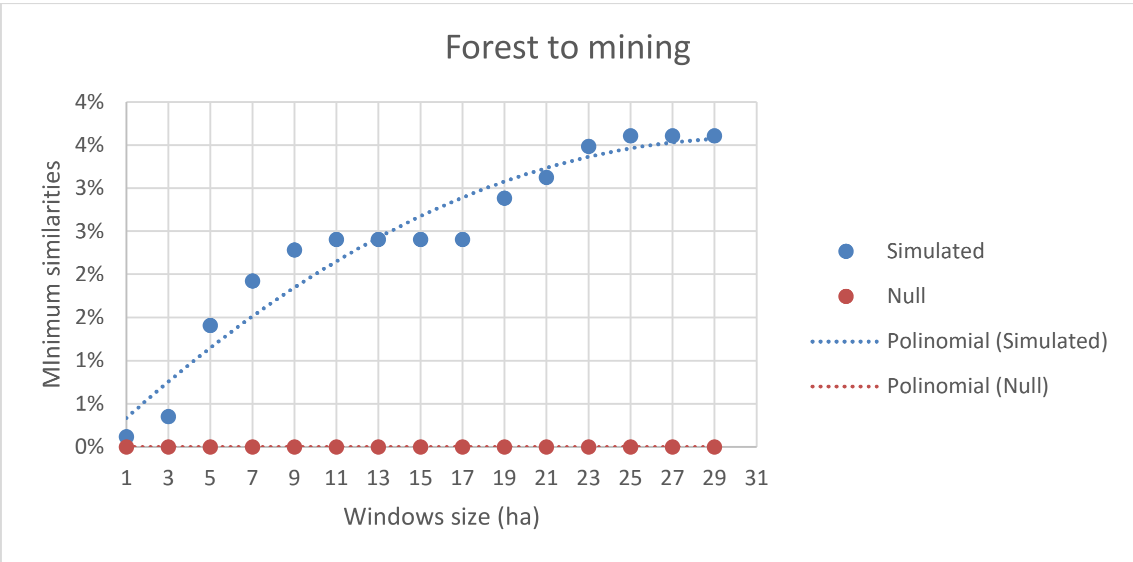


Figure S 15. Validation curve comparing the calibrated and null model for the transition of other non-native vegetation to silviculture

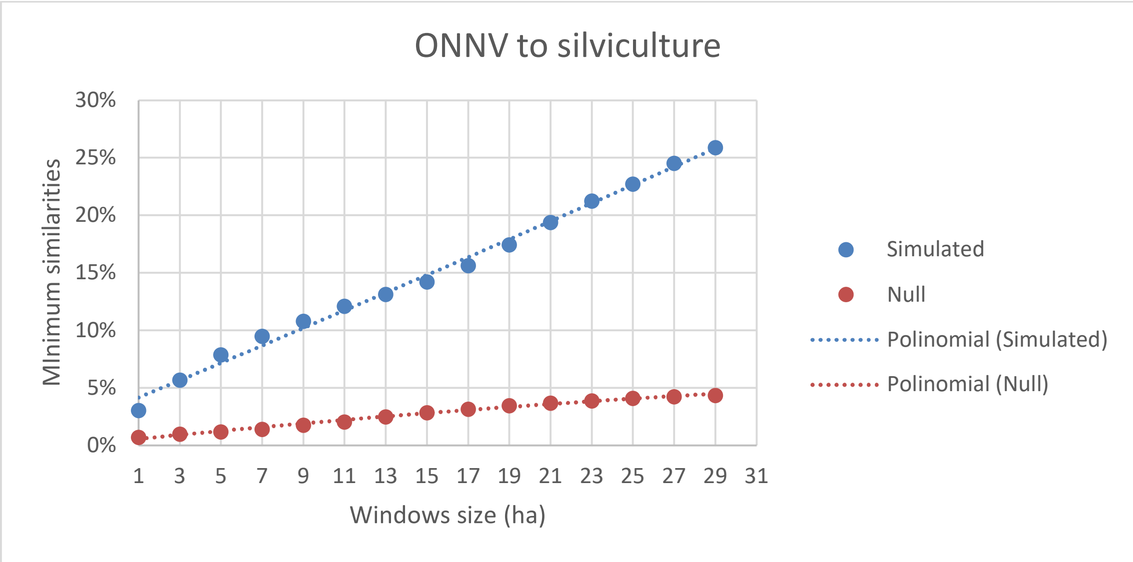


Figure S 16. Validation curve comparing the calibrated and null model for the transition of other non-native vegetation to mining

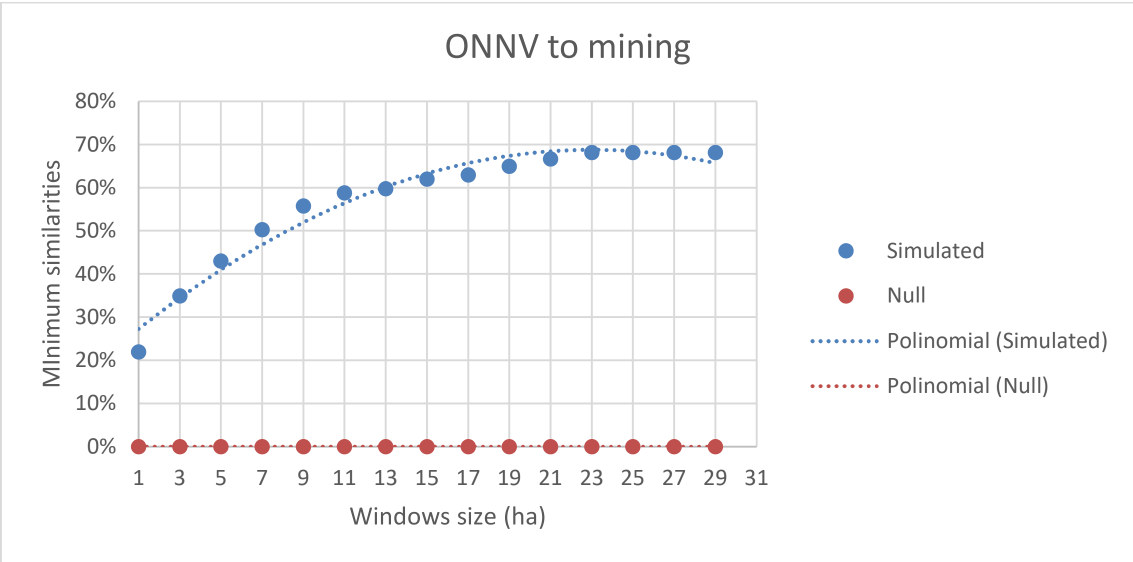
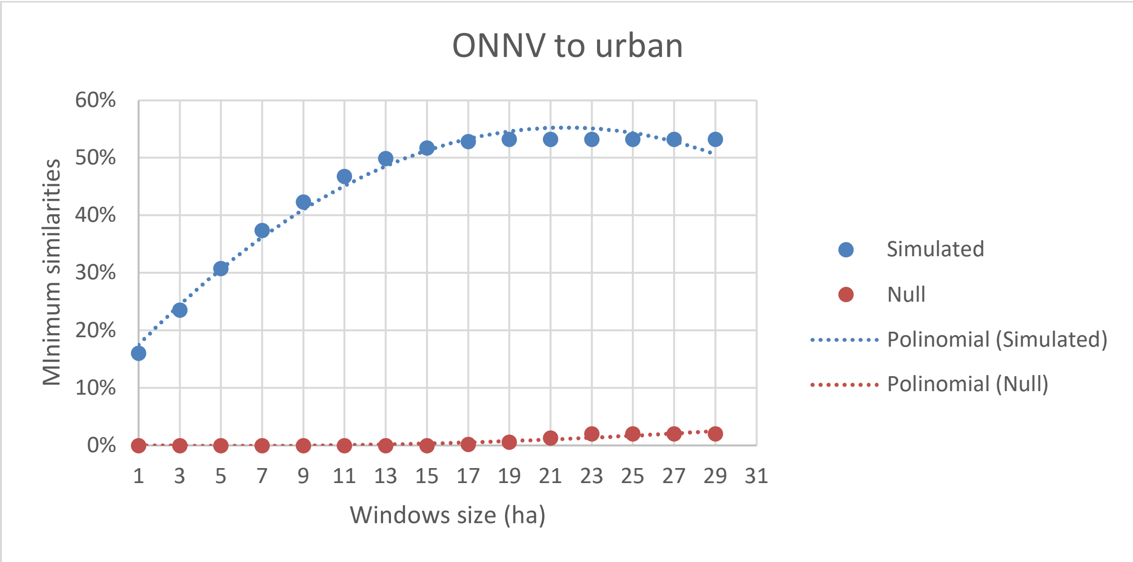


Figure S 17. Validation curve comparing the calibrated and null model for the transition of other non-native vegetation to urban



5. Scenarios parameters

Table S 14. Different PAs regulations explored in the scenario's proposal

Scenario		Description
BAU	Business as usual	No mining and transportation infrastructure are allowed inside all PAs.
W_SU	Weakening the protection in sustainable use PAs	Mining and transportation infrastructure are allowed inside sustainable use PAs
W_SU_IL	Weakening the protection in sustainable use and indigenous lands PAs	Mining and transportation infrastructure allowed inside sustainable use and indigenous lands PAs
W_IL	Weakening the protection in indigenous lands PAs	Mining and transportation infrastructure allowed inside indigenous lands PAs
Full_Dev	Weakening the protection of all protected areas, permitting the full development of mining occurrences in the region	Mining and transportation infrastructure allowed in all study region

Table S 15. Parameters used to create the friction surface

Variables	Weights	Rationale
Land use and land cover	Forest=5 ONNV=1 Silviculture=3 Mining=8 Urban=10 Other native vegetation=3 Hydrography=10	Lowest value – basic cost - to areas already cleared The native vegetation, such as forest and others, present an intermediate level of difficulty due need for clearing and in some cases license to supress native forest The highest value to mining, virtually barrier, and hydrography due to necessary to build bridges and large infrastructure
Slope (in degrees)	<1 =1 <5 =2 <10 =3 <15 =4 <20 =5 >=20 = 100	Greater difficulty in building roads with increasing slope

Table S 16.Values assigned to friction variable to feature the difficulty to roads construction considering each scenario

Variables	BAU	Weakening sustainable use PAs	Weakening sustainable use and indigenous lands PAs	Weakening Indigenous lands PAs	Full development
Sustainable use PAs	8	1	1	8	1
Full protection PAs	1000	1000	1000	1000	1
Indigenous lands PAs	1000	1000	1	1	1

Table S 17. Protected areas extent and percentage of the total landscape

Protected area	Area (km²)	Percentage of total landscape
Indigenous lands	18,040	6%
Sustainable use	70,369	22%
Full protection	52,072	17%
Under no protection	174,452	55%

Table S 18. Increased area available for development in each scenario used to calculate the increased rates of change

Scenario	Free area available for development (km²)	Increased percentage in relation to BAU scenario
Business as usual	174,452	-
Weakening sustainable use PAs	244,821	40%
Weakening sustainable use and indigenous lands PAs	262,861	50%
Weakening indigenous lands	192,492	10%
Full development	314,932	81%

Table S 19. Rates used to calibrate and simulate the scenarios

Transitions	Rates in the calibration		Rates for the scenarios				
	2004-2010	2010-2014	BAU	Weakening sustainable use PAs	Weakening sustainable use and indigenous lands PAs	Weakening indigenous lands PAs	Full development
Forest>ONNV	0.000899	0.000641	0.000641	0.0008968	0.000961	0.000705	0.001166
Forest>Silviculture	0.000008	0.000003	0.000003	0.0000046	0.000005	0.000004	0.000006
Forest>Mining	0.000007	0.000009	0.000009	0.0000130	0.000014	0.000010	0.000017
ONNV>Silviculture	0.006811	0.002289	0.002289	0.0032039	0.003433	0.002517	0.004165
ONNV>Mining	0.000109	0.000247	0.000247	0.0003465	0.000371	0.000272	0.000450
ONNV>Urban	0.000840	0.000338	0.000338	0.0004737	0.000508	0.000372	0.000616

6. Fragmentation analysis

Figure S 18. Number of patches

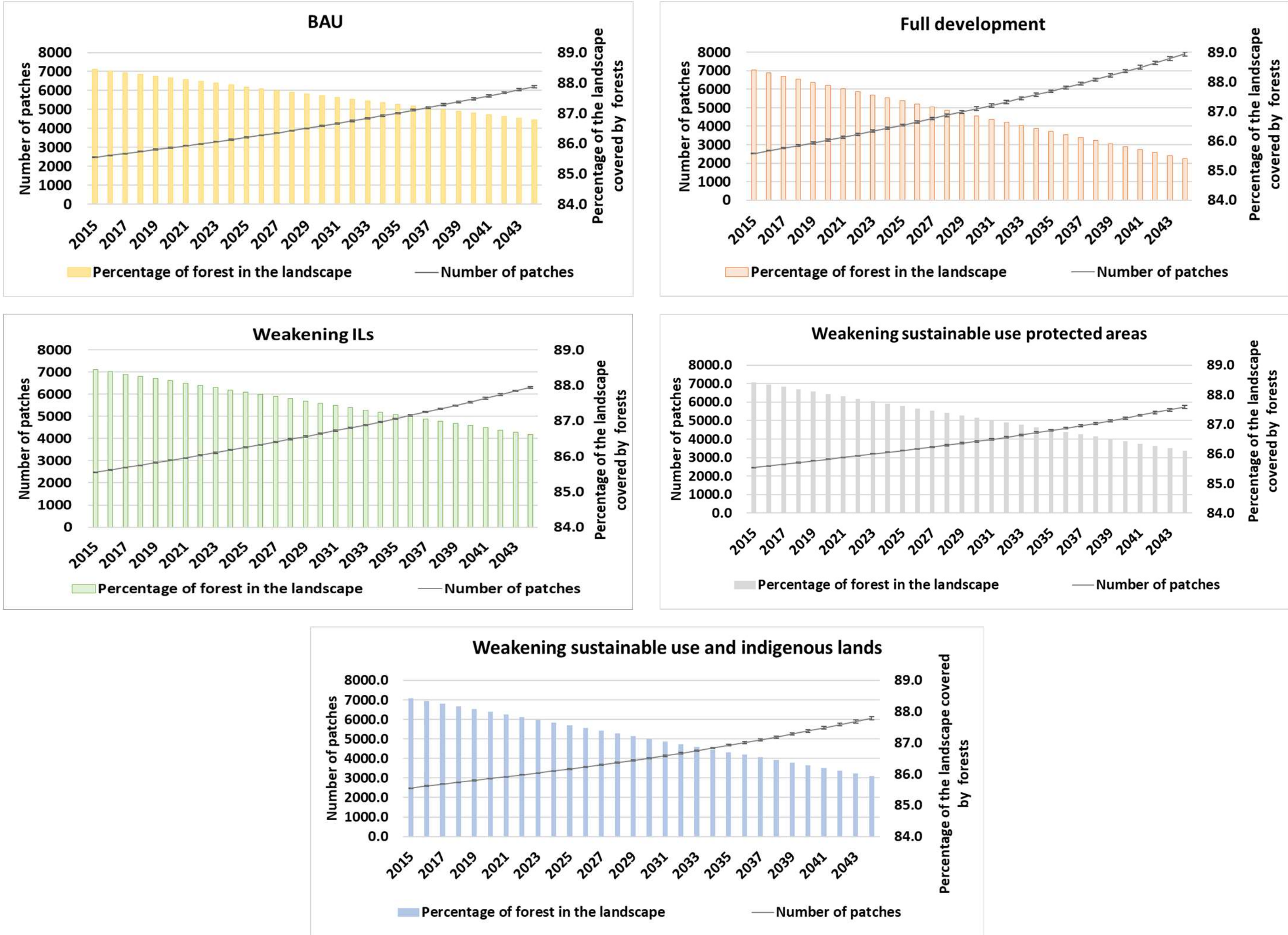


Figure S 19. Edge density

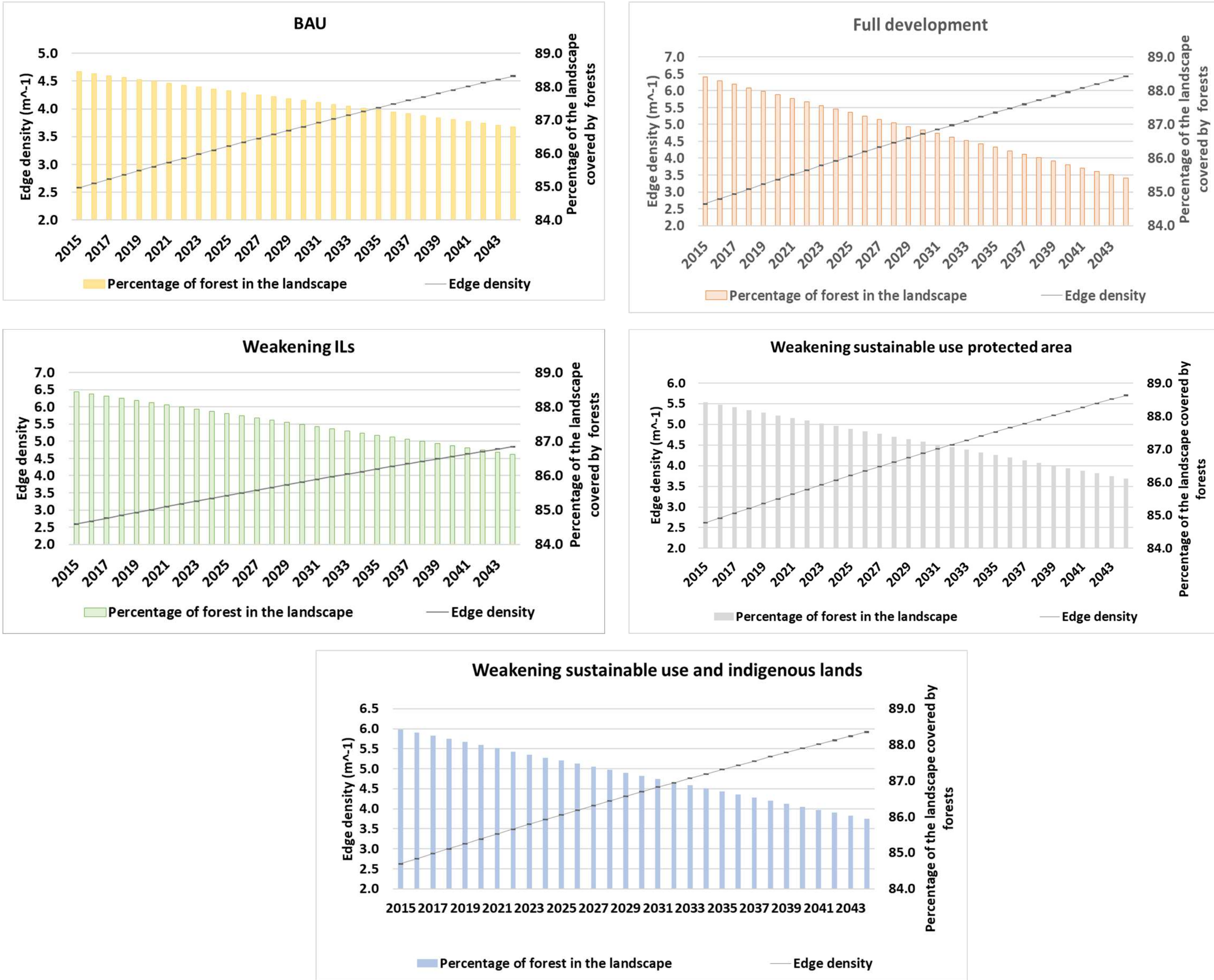


Figure S 20. Fragmentation index

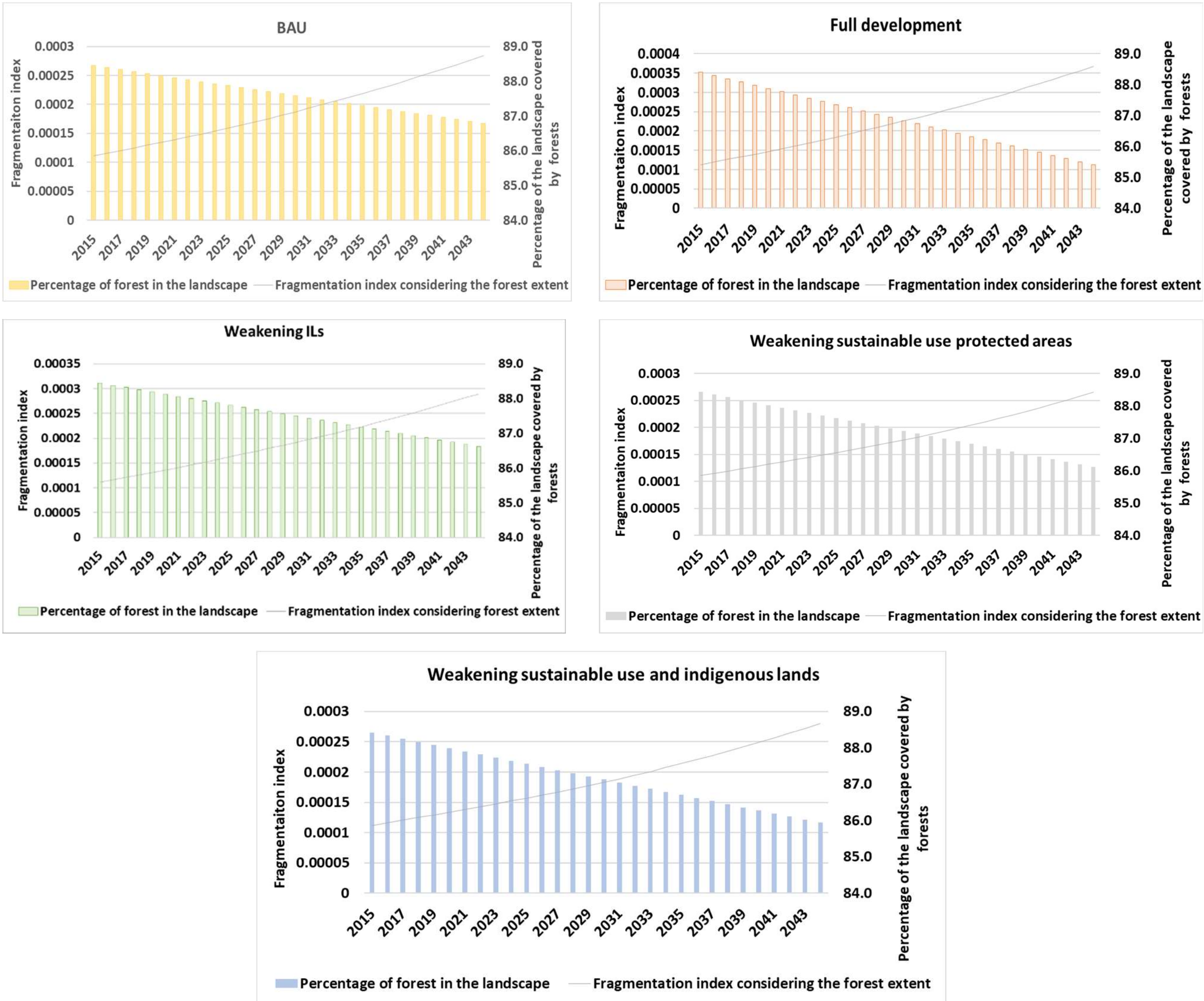
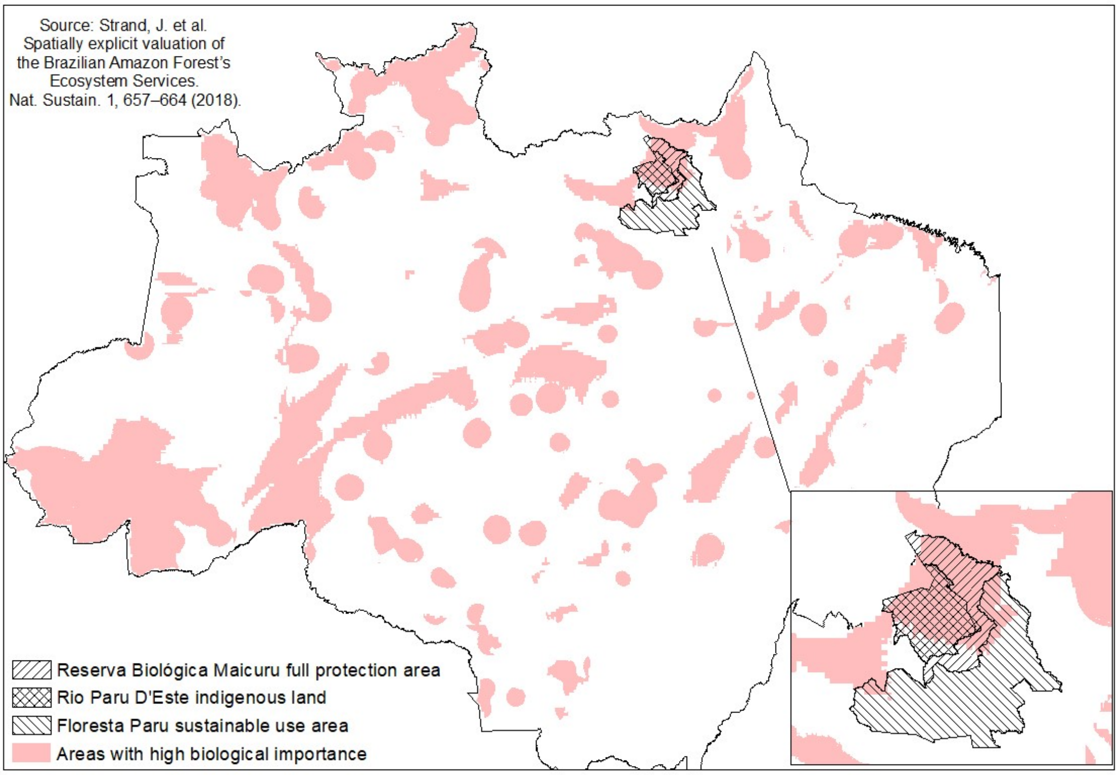


Figure S 21. Areas with high biological importance from Strand et al (2018)



Appendix 4. Supplementary information – Chapter 6

Notes S1: Detailed information about mineral occurrences and the status of Indigenous Lands in the Legal Amazon

Figure S 22. Spatial distribution and status of ILs in the Legal Amazon in February 2020.

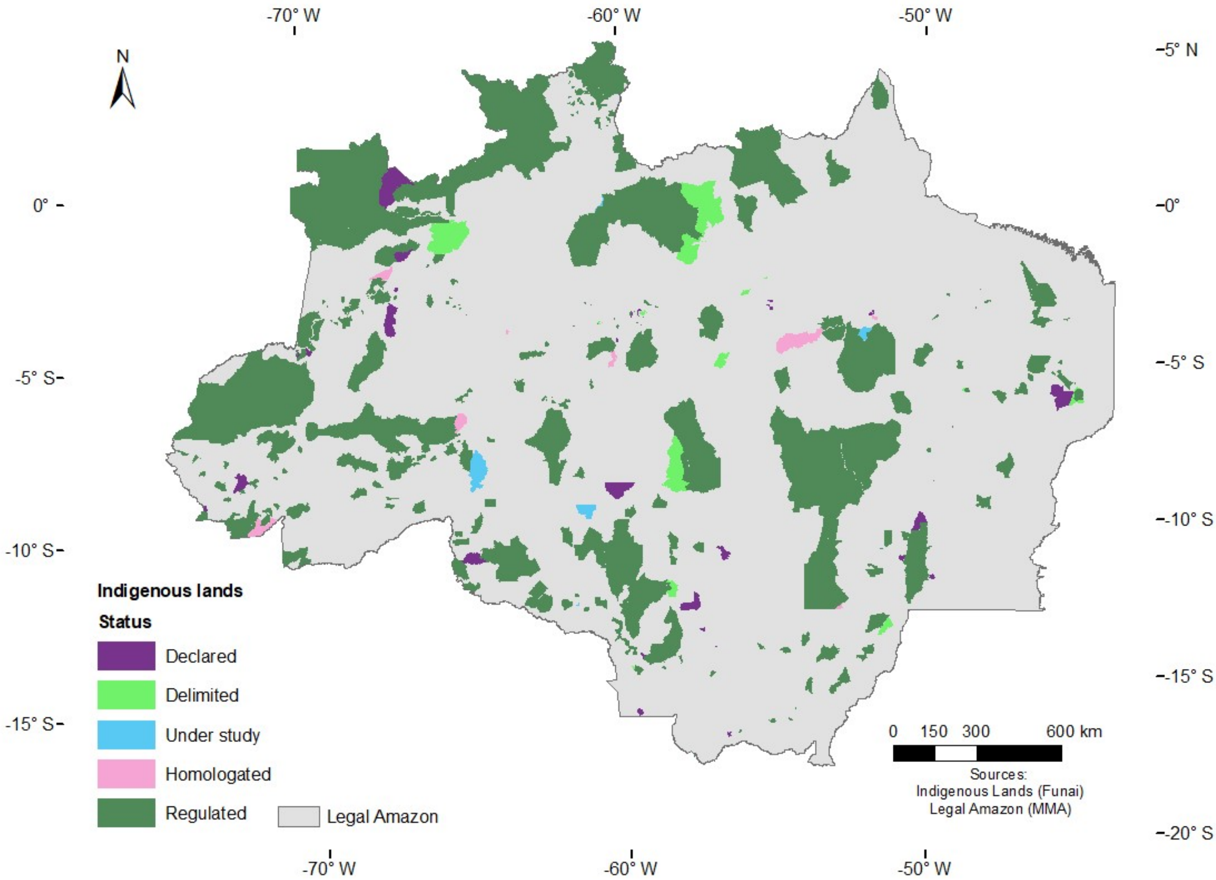


Table S 20. Minerals claims and proposals within Indigenous Lands (ILs) and other protected areas (PAs) in the Legal Amazon by substance (All records – from 1944 until February 2020)

Substance (1)	**Legal Amazon**	**Inside IL**	**Inside other PAs**
Aluminum	73	21%	62%
Sand	3970	0%	11%
Bauxite	840	1%	23%
Lead	70	64%	26%
Copper	201	42%	36%
Diamond	898	2%	11%
Iron	36	14%	47%
Aluminum ore	159	3%	46%
Copper ore	2345	2%	23%
Tin ore	297	8%	21%
Iron ore	1148	2%	25%
Manganese ore	7	1%	6%
Nickel ore	184	2%	5%
Gold ore	15625	3%	38%
Niobium	35	66%	20%
Gold	9612	10%	60%
Silver	65	26%	3%

Note: (1) denomination of substances according to the ANM database

Figure S 23. Mineral claims and proposal and Protected Areas (February 2020)

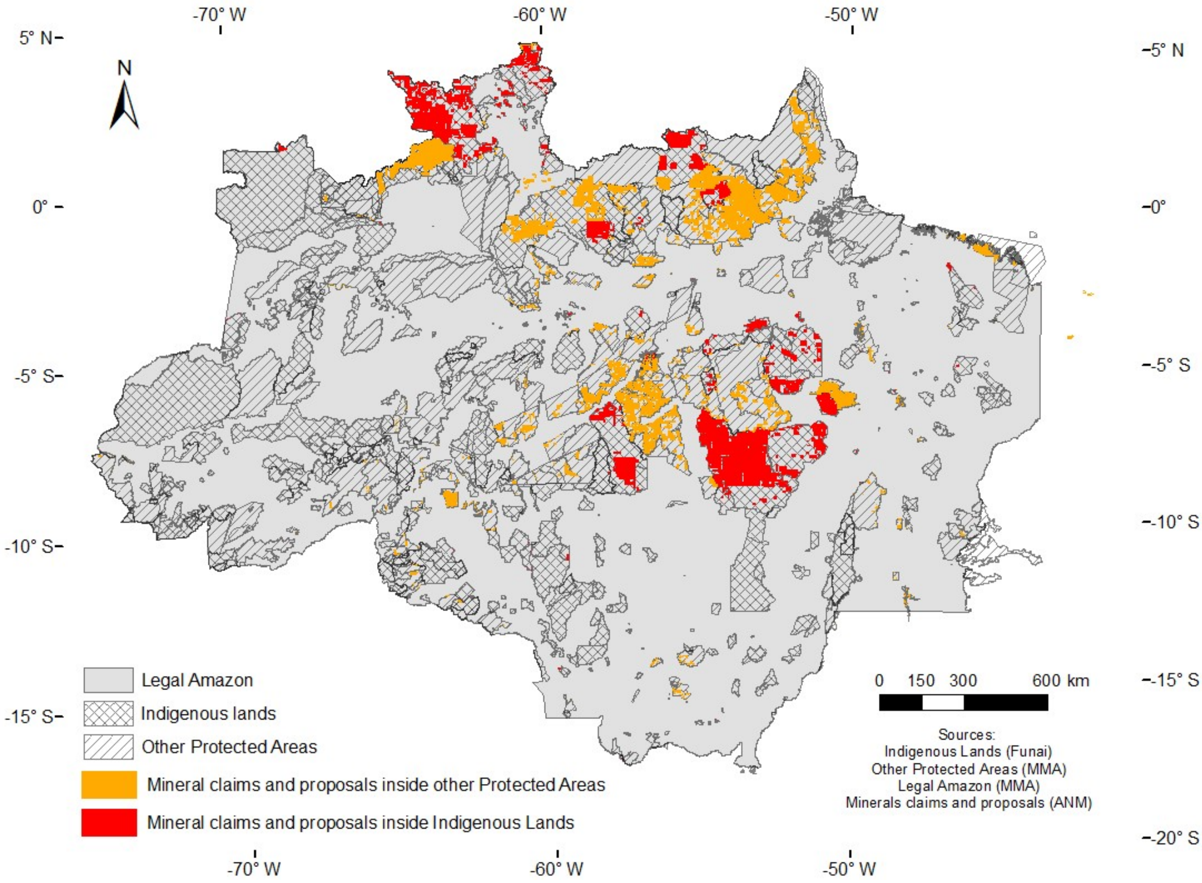


Figure S 24. Mineral occurrences by substance and indigenous lands (February, 2020)

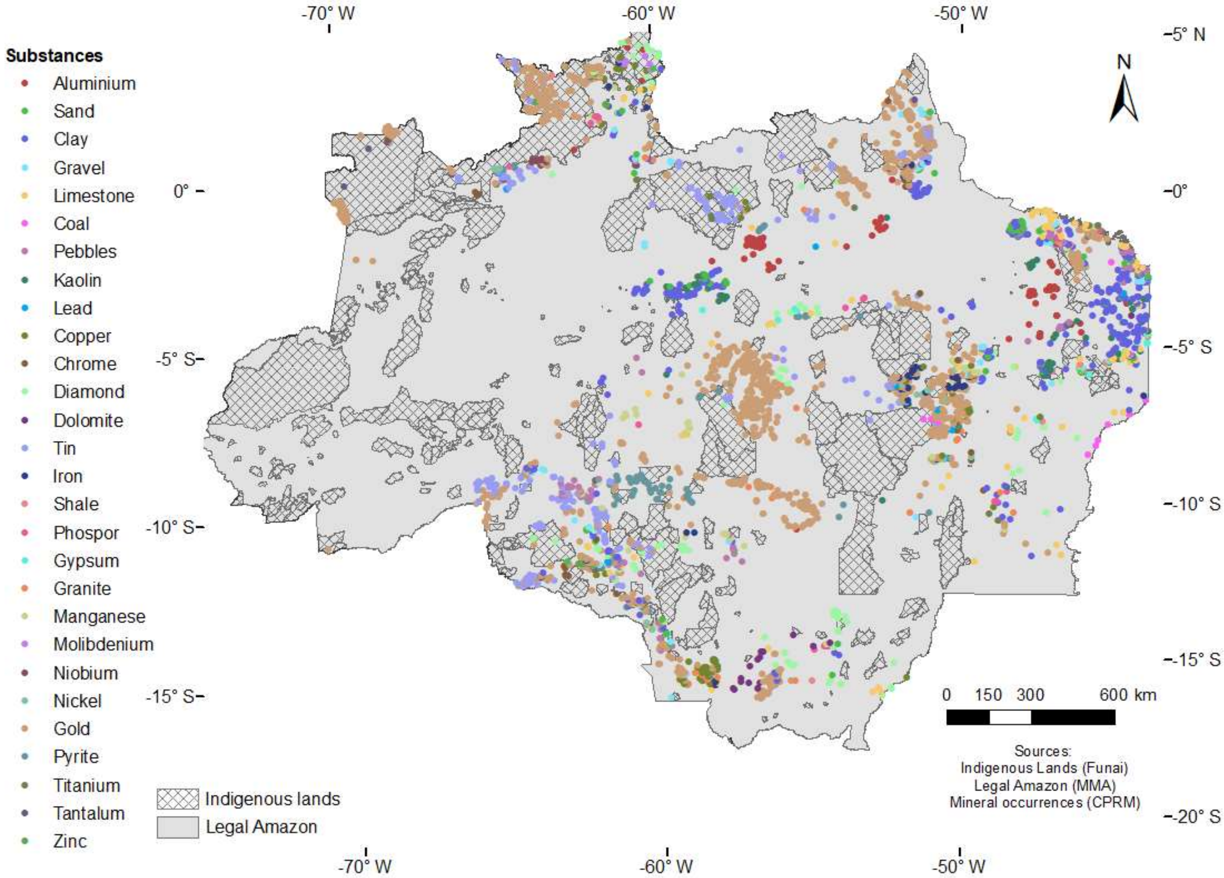


Table S2. Minerals occurrences within Legal Amazon and Indigenous Lands, and those included in each of our analysis scenarios (Policy and BAU)

Substance	**Number of minerals occurrences and percentage of the number of occurrences in relation to the total number in each area and scenarios**			
	Legal Amazon	**Inside IL**	**Policy Scenario**	**BAU scenario**
Gold	1954 (33.43%)	346 (52.97%)	1859 (34.06%)	1513 (31.65%)
Clay	503 (9.03%)	2 (0.42%)	432 (8.26%)	429 (9.25%)
Tin	325 (5.50%)	98 (17%)	311 (5.66%)	213 (4.22%)
Sandstone	151 (2.63%)	1 (0.25%)	148 (2.81%)	147 (3.13%)
Diamond	135 (2.52%)	25 (4.2%)	128 (2.60%)	103 (2.4%)
Sand	117 (2.26%)	3 (0.88%)	111 (2.36%)	108 (2.55%)
Copper	93 (1.59%)	15 (2.66%)	93 (1.75%)	78 (1.64%)

Table S3. Cleared area inside each PA category

Protected Area category	**Area cleared since 2000 (km²)**
Sustainable use	27,822 (4.35%)
Full protection	2,683 (0.57%)
Indigenous lands	7,054 (0.73%)

Notes S2: Detail of bill proposals mentioned in the manuscript

PL 1610/96 available at:

www.camara.gov.br/proposicoesWeb/fichadetramitacao?idProposicao=16969

PL3682/2012 available at:

www.camara.gov.br/proposicoesWeb/fichadetramitacao?idProposicao=541161

PL191/2020 available at:

https://www.camara.leg.br/propostas-legislativas/2236765

Notes S3: information about the Environmental Impact Assessment requirements for the minerals claims and proposals

According to Brazilian legislation, mineral resources belong to the federal government and any claim for exploration or mining requires a valid title issued by the National Mining Agency. There are different kinds of titles depending on the stage, from exploration to mining, and on the type of minerals. Additionally, there is one particular authorization that entitles “artisanal” mining (*permissão de lavra garimpeira*).

The competent authority for environmental review is an environmental agency (a State agency in most cases), that issues an environmental license upon approval of required environmental studies. Although requirements vary across states, no environmental license is required for exploration, while to start a mine, an environmental impact assessment must be filed and approved. However, stone quarries and sand pits may be licensed on the basis of a simplified procedure and corresponding simplified environmental assessment, as applied also to artisanal mining.

The simplified environmental licensing requires only simplified reports for issuing the permit for the project and extraction development. The complete environmental licensing is a three-stage process that requires an Environmental Impact Study and mitigation plan for issuing the environmental license.

In Table S4, as an exploration claim moves to a mining claim, environmental licensing becomes necessary and the category “uncertain” was assigned to all claims in the initial stages, because it is unknown whether or not they will evolve to a mine proposal. In such cases, either a simplified or a complete (i.e. an environmental impact assessment) review will be necessary as a requirement for issuing an environmental license.

Table S 21. Type of Environmental Impact Assessment required for each stage of mining proposals and claims.

Types of exploration and mining claims	Environmental impact assessment for environmental licensing
Application for exploration (*Requerimento de pesquisa*)	Uncertain
Exploration permit (*Autorização de pesquisa*)	Uncertain
Application for mining (*Requerimento de lavra*)	Uncertain
Application for artisanal mining (*Requerimento de lavra garimpeira*)	Simplified
Application for extraction of construction materials (*Requerimento de registro de extração*)	Simplified
Artisanal mining (*Permissão de lavra garimpeira*)	Simplified
Permit for extraction of construction materials (*Registro de extração*)	Simplified
Mining permit (*Concessão de lavra*)	Complete

Table S 22. Number of processes under each category of Environmental Impact Assessment considering each PAs category, outside PAs and all Amazon

Protected area category	Under simplified environmental licensing	Uncertain	Require a complete Environmental and Social Impact Assessment
Strictly protected	291 (47.9%)	307 (50.6%)	9 (1.5%)
Sustainable use	9929 (64.2%)	5373 (34.7%)	171 (1.1%)
Indigenous lands	89 (3.5%)	2420 (96.4%)	1 (0.04%)
Outside all PAs	12158 (40.8%)	16994 (57.0%)	645 (2.2%)
All Amazon	22467 (46.4%)	25094 (51.9%)	826 (1.7%)

Appendix 5. The outbreak of illegal gold mining in the Brazilian Amazon boosts deforestation

The outbreak of illegal gold mining in the Brazilian Amazon boosts deforestation

Authors: Juliana Siqueira-Gay[1]*, Luis E. Sanchez[1]

Affiliations:

[1] Escola Politécnica, University of São Paulo, São Paulo, Brazil

*Correspondence to: siq.juliana@gmail.com

Abstract

Increased prices and political pressure are boosting illegal gold mining in the Brazilian Amazon, threatening forests, Indigenous people, and conservation of biodiversity in protected areas. The rate of illegal mining deforestation increased more than 90% from 2017 to 2020, reaching 101,8 km² annually in 2020 compared to 52,9 km² annually in 2017. In that period, illegal mining deforestation rate grew more than the rate of clearing within mining leases. While formal mining is required to comply with environmental regulations, most small-scale or artisanal mining and especially illegal mining areas are abandoned after reserves are exhausted, without proper rehabilitation. Deforestation due to illegal mining is likely to increase in the next years, calling for coordination between local and regional policies as well as for strengthening and expanding international mechanisms to increase traceability of minerals supply chains with certification schemes to help to curb illegal mining.

Keywords: artisanal and small-scale mining; *garimpo*; environmental impacts; forest degradation

Illegal gold mining has expanded significantly in the Amazon in the last decade (Asner et al. 2013; Alvarez-Berríos and Aide 2015; Asner and Tupayachi 2016), up to 18% annually in some regions (SWENSON et al., 2011). This expansion is driven by the increase in gold prices in international markets (Alvarez-Berríos and Aide, 2015; USGS, 2020). In Brazil specifically, since 2018, it is bolstered by political support from the government of President Bolsonaro (SOLOMON, 2020).

The occupation of illegal miners, mostly in search of gold, is encroaching on protected areas (ESPEJO et al., 2018) and threatening Indigenous peoples' well-being (CIMI 2018; Souza et al. 2019; Calvimontes et al. 2020; Siqueira-Gay et al. 2020a). Illegal gold mining pollutes water with heavy metals (ABE et al., 2019; LOBO et al., 2016), affecting aquatic (SWENSON et al., 2011) and terrestrial ecosystems (ASAMOAH et al., 2017; DEZÉCACHE et al., 2017; SCHUELER; KUEMMERLE; SCHRÖDER, 2011), leading to negative impacts on human health (CASTILHOS et al., 2015). In addition to local consequences, effects such as water contamination and increased sedimentation can be traced far beyond the mining sites, threatening communities and ecosystems hundreds of kilometers away (Alvarez-Berríos and Aide 2015; Sánchez-Cuervo et al. 2020). These small scale illegal activities are claimed to cause minor direct adverse impacts on forests (WORLD BANK, 2019b), however they can lead to extensive environmental impacts when associated with other drivers of change, such as large infrastructure development (SÁNCHEZ-CUERVO et al., 2020). Combined with broader infrastructural interventions, illegal gold mining can cumulatively degrade forests and reduce their area over time (ASNER; TUPAYACHI, 2016).

Two satellite monitoring systems are currently used to map the entire Brazilian Amazon (Assis et al. 2019). PRODES provides the annual extent of forest cover and deforestation rates (INPE, 2017), and DETER-B (DINIZ et al., 2015) issues daily warnings of deforestation to support direct field inspections and control (Assis et al. 2019). Data from DETER-B show that the annual rate of clearing due to illegal mining activities, i.e., the sum of all warnings in a year, increased more than 90% from 2017 to 2020, reaching 101,8 km² in 2020 compared to 52,9 km² in 2017, the year before Bolsonaro's terms started (Fig 1). In contrast, by overlaying deforestation data from PRODES to mining leases from the National Mining Agency (ANM, 2020) (Fig 1), the same trend is not observed inside areas with lease permits (Table S1-S4).

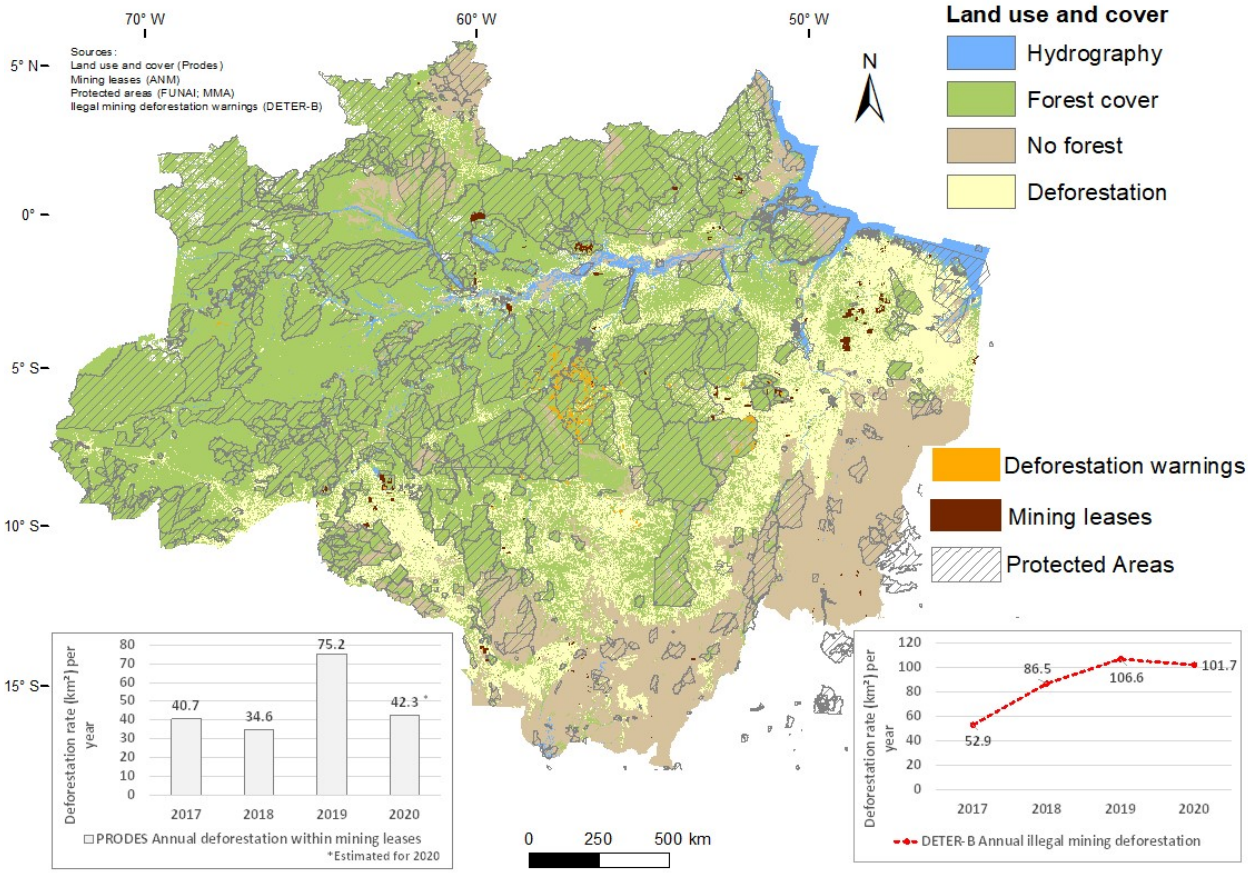


Fig 1. Deforestation in the Brazilian Legal Amazon, industrial mining leases, and deforestation warnings due to illegal mining. In the graph, the grey bars are the annual deforestation rate within mining leases and the red line features the sum of deforestation warnings per year due to illegal mining.

The types of gold mining in Brazil are twofold: (i) industrial mining, which are required to have a lease permit; and (ii) small-scale (seldom artisanal) mining, known as *garimpos,* where simplified forms of exploration, extraction, processing, and transportation are used (OECD, 2016) (SOUSA et al., 2011). Several *garimpos* use heavy machinery to extract gold and are not considered artisanal but small scale mining (CALVIMONTES et al., 2020). Most artisanal and small-scale mining of gold in Brazil is illegal (SOUSA et al., 2011), but obtaining a *garimpo* lease is possible. *Garimpos* are not required to comprehensively assess and mitigate their social and environmental impacts (MINISTÉRIO PÚBLICO FEDERAL, 2020).

Garimpos not only clear but degrade forests (ESPEJO et al., 2018), severely limiting the recovery of ecosystems after disturbance (KALAMANDEEN et al., 2020), and particularly degrading riparian vegetation. Clearing forest and other native vegetation for industrial mining requires legal authorization and the corresponding adherence to specific terms and conditions, including the commitment to rehabilitate degraded areas. However, deforestation in illegal gold mines are not held to similar conditions and the mined areas are usually abandoned after the reserves are exhausted, without any rehabilitation (SOUSA et al., 2011).

There is growing knowledge and experience around restoring Amazon forests (DA CRUZ et al., 2020), including after large-scale mining (GASTAUER et al., 2019). Post-mining restoration experiences date back to the 1990s (PARROTTA; KNOWLES, 2001), but the success of these interventions are highly dependent on how the rehabilitation process is managed (NERI; SÁNCHEZ, 2010). Key ingredients for successful restoration, such as contextual knowledge, adequate financial resources, and long-time frames necessary to monitor a restoration trajectory (DA CRUZ et al., 2020; KALAMANDEEN et al., 2020) are lacking in areas where illegal gold mining occurs.

In order to curb extensive illegal mining deforestation, recommendations are twofold. First, it is critical to track the source of minerals to distinguish gold mined illegally from legal and formalized gold mining and this can be done through certification schemes. Second, strengthening national and local policies to tackle illegal deforestation is needed. Investigations by the Federal Police and inquiries conducted by the Federal Prosecutor Office found that a significant amount of illegally mined gold is falsely declared as coming from legal sources – i.e. legalized *garimpos* – in order to be sold to authorized buyers (SOLOMON, 2020). Initiatives such as the Chain of Custody Standard of the Responsible Jewellery Council (RJC, 2019), the Minerals Due Diligence of the Responsible Minerals Initiative (RMI, 2020), and Fairmined Standard for Gold (ALLIANCE FOR RESPONSIBLE MINING FOUNDATION, 2014) have been developed to assist buyers of gold and other precious metals to verify their suppliers' compliance with environmental and social standards for responsible mining (WORLD BANK, 2019b) and have potential to contribute to minimize gold trade originated from illegal *garimpos*.

Although these schemes have limitations (SIPPL, 2020; SOVACOOL et al., 2020), certification that is informed by regular audits and inspections could encourage international gold buyers to join international investors in supporting fair enforcement of Brazilian environmental legislation, as a way to curb illegal mining supply chain (WORLD BANK, 2019b). Promisingly, there is growing awareness around this. On April 23 2020, an open letter signed by 29 financial institutions holding more than US$ 3.7 trillion total assets (HARRIS, 2020) warned that "the escalating deforestation in recent years, combined with reports of a dismantling of environmental and human rights policies and enforcement agencies, are creating widespread uncertainty about the conditions for investing in or providing financial services to Brazil" (Storebrand Asset Management et al. 2020).

Since 2004, public policies to tackle deforestation in the Amazon were strengthened by the Action Plan to Prevention and Control of Deforestation in the Legal Amazon (PPCDAm) responsible for the creation of protected areas and regulatory instruments (SILVA JUNIOR et al., 2020; SIQUEIRA-GAY et al., 2020b). However, since 2018 (Silva Junior et al. 2020), trust in the government's willingness to curb deforestation has drastically decreased, given the roll-back of environmental protection legislation in Brazil (ABESSA; FAMÁ; BURUAEM, 2019; FERRANTE; FEARNSIDE, 2019) as well as the continuous erosion of most regulations promoted by the Ministry of the Environment (PEREIRA et al., 2019). In contrast to present moves towards weakening environmental regulations in Brazil, previous initiatives to tackle mining-driven deforestation in the Amazon have demonstrated the importance of local and regional cooperation to support and implement national policies (DEZÉCACHE et al., 2017).

Increasing gold prices (Cherringtion et al. 2020) contribute to the prospect of rapidly increasing forest degradation by illegal gold miners in the coming years. Restraining deforestation is unlikely to happen without sustained pressure on investors and consumers for strengthening and expanding international mechanisms to increase traceability of mineral supply chains. Likewise, national support for increasing cooperation between federal and local policies, and instruments is needed to restrain the growth of illegal mining and consequent deforestation in the Brazilian Amazon.

Acknowledgements

This research is supported by the São Paulo Research Foundation (grant 2018/12475-7). This study was financed in part by the Coordenação de Aperfeiçoamento de Pessoal de Nível Superior - Brasil (CAPES) - Finance Code 001. The authors thank the editors and the reviewer for the comments in the manuscript and INPE experts for clarifying details of satellite monitoring programs.

References

Abe CA, Lobo FL, Novo EML, Costa, M, Yonas, D (2019) Modeling the effects of land cover change on sediment concentrations in a gold-mined Amazonian basin. Reg Environ Chang 19:1801–1813. https://doi.org/10.1007/s10113-019-01513-8

Abessa D, Famá A, Buruaem L (2019) The systematic dismantling of Brazilian environmental laws risks losses on all fronts. Nat Ecol Evol 3:510–511. https://doi.org/10.1038/s41559-019-0855-9

Alliance for Responsible Mining Foundation. (2014) Fairmined standard for gold from artisanal and small-scale mining, including associated precious metals. https://fairmined.org. Accessed September 2020.

Alvarez-Berríos NL, Aide TM (2015) Global demand for gold is another threat for tropical forests. Environ Res Lett 10:014006. https://doi.org/10.1088/1748-9326/10/2/029501

Agência Nacional de Mineração (ANM) (2020) Processos minerários: Sistema de Informação Geográficas da Mineração (SIGMINE). Available at: http://sigmine.dnpm.gov.br/. Accessed January 2021

Asamoah E, Zhang L, Liu G, Owusu-Prempeh N, Rukundo E (2017) Estimating the "Forgone" ESVs for small-scale gold mining using historical image data. Sustainability 9:1976. https://doi.org/10.3390/su9111976

Asner GP, Llactayo W, Tupayachi R, Luna ER (2013) Elevated rates of gold mining in the Amazon revealed through high-resolution monitoring. Proc Natl Acad Sci 110:18454–18459. https://doi.org/10.1073/pnas.1318271110

Asner GP, Tupayachi R (2016) Accelerated losses of protected forests from gold mining in the Peruvian Amazon. Environ Res Lett 12:094004. https://doi.org/https://doi.org/10.1088/1748-9326/aa7dab

Assis LFFG, Ferreira KR, Vinhas L, Maurano L, Almeida C, et al (2019) TerraBrasilis: A spatial data analytics infrastructure for Large-Scale Thematic Mapping. ISPRS Int J Geo-Information 8:513. https://doi.org/10.3390/ijgi8110513

Calvimontes J, Massaro L, Araujo CHX, Moraes RR, Mello J, et al (2020) Small-scale gold mining and the COVID-19 pandemic: Conflict and cooperation in the Brazilian Amazon. Extr Ind Soc 1–4. https://doi.org/10.1016/j.exis.2020.08.013

Castilhos Z, Rodrigues-Filho S, Cesar R, Rodrigues AP, Villas-Bôas R, et al (2015) Human exposure and risk assessment associated with mercury contamination in artisanal gold mining areas in the Brazilian Amazon. Environ Sci Pollut Res 22:11255–11264. https://doi.org/10.1007/s11356-015-4340-y

Cherringtion B, Todorova M, Falcidia L, Tejerina A (2020) Gold report 2020. World Gold Council. https://global-business-reports-2.preview.foleon.com/ie/gbr-gold-report-2020/gold-2020-cover. Accessed Accessed September 2020.

Conselho Indigenista Missionário - CIMI (2018) Violência contra os povos indígenas do Brasil - Dados 2018. 156 p. https://cimi.org.br/2019/09/a-maior-violencia-contra-os-povos-indigenas-e-a-apropriacao-e-destruicao-de-seus-territorios-aponta-relatorio-do-cimi/. Accessed December 2020.

da Cruz DC, Benayas JMR, Ferreira GC, Santos SR, Schwartz, et al (2020) An overview of forest loss and restoration in the Brazilian Amazon. New For. https://doi.org/10.1007/s11056-020-09777-3

Dezécache C, Faure E, Gond V, Salles JM, Vieilledent G, et al (2017) Gold-rush in a forested El Dorado: Deforestation leakages and the need for regional cooperation. Environ Res Lett 12: 034013. https://doi.org/10.1088/1748-9326/aa6082

Diniz CG, Souza AADA, Santos DC, Dias MC, Luz NC, et al (2015) DETER-B: The new amazon near real-time deforestation detection system. IEEE J Sel Top Appl Earth Obs Remote Sens 8:3619–3628. https://doi.org/10.1109/JSTARS.2015.2437075

Espejo JC, Messinger M, Román F, Ascorra C, Fernandez LE et al (2018) Deforestation and forest degradation due to gold mining in the Peruvian Amazon: a 34-year perspective. Remote Sens 10:1903. https://doi.org/10.20944/preprints201811.0113.v1

Ferrante L, Fearnside PM (2019) Brazil's new president and "ruralists" threaten Amazonia's environment, traditional peoples and the global climate. Environ Conserv 46:261–263. https://doi.org/10.1017/S0376892919000213

Gastauer M, Souza Filho PWM, Ramos SJ, Caldeira CF, Silva JR et al (2019) Mine land rehabilitation in Brazil: Goals and techniques in the context of legal requirements. Ambio 48:74–88. https://doi.org/10.1007/s13280-018-1053-8

Harris B (2020) Investors warn Brazil to stop Amazon destruction. Financ. Times. https//www.ft.com/content/ad1d7176-ce6c-4a9b-9bbc-cbdb6691084f. Accessed September 2020.

Instituto Nacional de Pesquisas, Espaciais (INPE) (2017) Projeto PRODES Digital: Mapeamento do desmatamento da Amazônia com Imagens de Satélite. http://terrabrasilis.dpi.inpe.br/. Accessed January 2021

Kalamandeen M, Gloor E, Johnson I, Agard S, Katow M, et al (2020) Limited biomass recovery from gold mining in Amazonian forests. J Appl Ecol 1–11. https://doi.org/10.1111/1365-2664.13669

Lobo FDL, Costa M, Novo EMLM, Telmer K (2016) Distribution of artisanal and small-scale gold mining in the Tapajós River Basin (Brazilian Amazon) over the past 40 years and relationship with water Siltation. Remote Sens 8:579. https://doi.org/10.3390/rs8070579

Ministério Público Federal (2020) Mineração ilegal de ouro na Amazônia: marcos jurídicos e questões controversas. MPF, Brasília

Neri AC, Sánchez LE (2010) A procedure to evaluate environmental rehabilitation in limestone quarries. J Environ Manage 91:2225–2237. https://doi.org/10.1016/j.jenvman.2010.06.005

OECD (2016) OECD Due Diligence Guidance for Responsible Supply Chains of Minerals from Conflict-Affected and High-Risk Areas. Available at: https://www.oecd.org/daf/inv/mne/OECD-Due-Diligence-Guidance-Minerals-Edition3.pdf

Parrotta JA, Knowles OH (2001) Restoring tropical forests on lands mined for bauxite: Examples from the Brazilian Amazon. Ecol Eng 17:219–239. https://doi.org/10.1016/S0925-8574(00)00141-5

Pereira EJ de AL, Ferreira PJS, Ribeiro LC de S, Terciane SC, Pereira HBB (2019) Policy in Brazil (2016–2019) threaten conservation of the Amazon rainforest. Environ Sci Policy 100:8–12. https://doi.org/10.1016/j.envsci.2019.06.001

Responsible Jewellery Council (RJC) (2019) Chain of custody standard. https://www.responsiblejewellery.com/wp-content/uploads/RJC-CoC-2017-V2-Standard-2.pdf. Accessed September 2020

Responsible Minerals Initiative (RMI) (2020) Minerals Due Diligence. http://www.responsiblemineralsinitiative.org/minerals-due-diligence/. Accessed September 2020

Sánchez-Cuervo AM, de Lima LS, Dallmeier F, Garate P, Bravo A, et al (2020) Twenty years of land cover change in the southeastern Peruvian Amazon: implications for biodiversity conservation. Reg Environ Chang 20:1. https://doi.org/10.1007/s10113-020-01603-y

Schueler V, Kuemmerle T, Schröder H (2011) Impacts of surface gold mining on land use systems in Western Ghana. Ambio 40:528–539. https://doi.org/10.1007/s13280-011-0141-9

Silva Junior CHL, Pessôa ACM, Carvalho NS, Reis JBC, Anderson L. et al (2020) The Brazilian Amazon deforestation rate in 2020 is the greatest of the decade. Nat Ecol Evol 0–1. https://doi.org/10.1038/s41559-020-01368-x

Sippl K (2020) Southern Responses to Fair Trade Gold: Cooperation, Complaint, Competition, Supplementation. Ecol Econ 169:106377. https://doi.org/10.1016/j.ecolecon.2019.106377

Siqueira-Gay J, Soares-Filho B, Sánchez LE, Oviedo A, Sonter LJ (2020a) Proposed legislation to mine Brazil's Indigenous Lands will threaten Amazon forests and their valuable ecosystem services. One Earth 3:1–7. https://doi.org/10.1016/j.oneear.2020.08.008

Siqueira-Gay J, Yanai AM, Lessmann J, Borja D, Canova M, et al (2020b) Pathways to positive scenarios for the Amazon forest in Pará state, Brazil. Biota Neotrop 20:e20190905. https://doi.org/10.1590/1676-0611-BN-2019-0905

Solomon M (2020) A nova corrida do ouro na Amazônia. Instituto Escolhas. http://www.escolhas.org/wp-content/uploads/2020/05/TD_04_GARIMPO_A-NOVA-CORRIDA-DO-OURO-NA-AMAZONIA_maio_2020.pdf. Accessed September 2020

Sousa R, Veiga M, Van Zyl D, Telmer K, Spiegel S, et al (2011) Policies and regulations for Brazil's artisanal gold mining sector: Analysis and recommendations. J Clean Prod 19:742–750. https://doi.org/10.1016/j.jclepro.2010.12.001

Souza PF, Xavier DR, Mutis MCS, da Mota JC, Peiter PC, et al (2019) Spatial spread of malaria and economic frontier expansion in the Brazilian Amazon. PLoS One 14:1–25. https://doi.org/10.1371/journal.pone.0217615

Sovacool BBK, Ali SH, Bazilian M, Radley B, Nemery B, et al (2020) Policy coordination is needed for global supply chains. Science (80-) 367:30–33. https://doi.org/10.1126/science.aaz6003

Storebrand Asset Management, KLP, Gjensidige, Sparebank 1 Forsikring, MP Pension, et al (2020) Open Letter from financial institutions to halt deforestation in Brazil. https://www.actiam.com/49e60f/siteassets/4_verantwoord/documenten/en/open-letter-brazilian-embassy-202006.pdf. Accessed September 2020

Swenson JJ, Carter CE, Domec JC, Delgado CI (2011) Gold mining in the peruvian amazon: Global prices, deforestation, and mercury imports. PLoS One 6:4. https://doi.org/10.1371/journal.pone.0018875

United States Geological Survey (USGS) (2020) Mineral Commodity Summaries: Gold. https://www.usgs.gov/centers/nmic/gold-statistics-and-information. Accessed September 2020

World Bank (2019) Forest-Smart Mining: Artisanal & small-scale mining in forest landscapes (ASM). Washington DC: World Bank

Supplementary material

The outbreak of illegal gold mining in the Brazilian Amazon boosts deforestation

Authors: Juliana Siqueira-Gay[1]*, Luis E. Sánchez[1]

Affiliations:

[1] Escola Politécnica, University of São Paulo, São Paulo, Brazil

*Correspondence to: siq.juliana@gmail.com

Data from mining leases (last access 21 January 2021): http://sigmine.dnpm.gov.br/webmap/

Data from deforestation (last access 19 January 2021): http://terrabrasilis.dpi.inpe.br/

1. Information about illegal mining deforestation

Table S 23. Detailed information about Illegal mining deforestation according to DETER-B database

Year	Total area cleared by illegal mining per year (km²)	Number of warnings per year
2017	52,85	465
2018	86,51	901
2019	106,59	1,080
2020	101,75	1,131

2. Information about mining leases

According to National Mining Agency (SIGMINE) database, the last mining leases in the Brazilian Amazon were issued on 2017. Therefore, for the analysis of deforestation inside leases, the same polygons of mining concessions were considered in the analysis

Table S 24. Detailed information about mining leases in the Brazilian Amazon according to the National Mining Agency database

Year	Number of new leases in each year
Before 2008	1211
2008	15
2009	16
2010	22
2011	10
2012	8

2013	14
2014	2
2017	4
2017-2020	None
Total	**1302**

3. Information about deforestation rates in the Brazilian Amazon

Table S 25. Detailed information about the yearly deforestation in the Brazilian Amazon according to PRODES database

Year	**Annual deforestation rates in the Brazilian Amazon (km²/year)**
2010	7,000
2011	6,418
2012	4,571
2013	5,891
2014	5,012
2015	6,207
2016	7,893
2017	6,947
2018	7,536
2019	10,129
2020	11,088

4. **Information about deforestation within leases**

Table S 26. Detailed information about deforestation within mining leases

Year	**Annual deforestation rates within mining leases (km²/year)**
2010	68,14
2011	39,96
2012	48,88
2013	32,16
2014	36,33

2015	43,44
2016	48,18
2017	40,78
2018	35,34
2019	76,34
2020	42,70*

*The total deforestation of 2020 is a preliminary estimation for the yearly deforestation