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The environmental trade-offs of mining in a biodiversity hotspot

Devenish, Katie

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The environmental trade-offs of mining in a biodiversity hotspot

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Katie Devenish
School of Natural Sciences

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Abstract

Mining supplies critical minerals, supports millions of livelihoods, and can help fuel economic development, particularly in low- and middle-income countries. Yet, these contributions can entail substantial trade-offs for biodiversity and ecosystems, through habitat loss, deforestation, and pollution. These impacts are particularly concerning when mining occurs in places which are also hotspots for biodiversity, such as Madagascar. In this thesis, I explore the challenges and opportunities for reconciling mining and biodiversity conservation in Madagascar. Using spatial data analysis and counterfactual methods for impact evaluation I evaluate the real and potential impacts of mining on the forests and, by proxy, biodiversity of Madagascar, and the effectiveness of policies to mitigate that impact.

First, I evaluate the effectiveness of a key policy mechanism for mitigating the impacts of infrastructure development on biodiversity: biodiversity offsetting. Using statistical methods for counterfactual impact evaluation (statistical matching, difference-in-differences, and fixed effects panel regressions), I show that Madagascar's largest mine, Ambatovy, has likely prevented enough deforestation within its four biodiversity offsets to compensate for the 2,000 hectares of forest cleared at the mine site. As such, the mine has likely achieved No Net Loss of forest. This shows that biodiversity offsetting can contribute towards mitigating the impact of mining, but it requires considerable investment and there are important caveats.

Next, I switch focus from industrial to artisanal and small-scale mining (ASM). ASM is widespread across Madagascar and has occurred within several protected areas. Using geological data to define and map the potential distribution of primary ruby, sapphire, and emerald deposits, I find that 11-14% of the most important area for biodiversity in Madagascar (defined using four measures) could host primary gem deposits and be impacted by gem mining in future. However, this approach also revealed a vast area of potentially prospective land outside these areas where decentralised, community managed zones for ASM could be established, minimising environmental trade-offs.

Finally, I evaluate what can happen when an artisanal mining rush involving tens of thousands of people occurs within a protected area: focussing on the 2016 sapphire rush at Bemainty in the eastern rainforests of Madagascar. Using the synthetic control approach for impact evaluation and drawing on additional field data (from a lemur census and semi-structured interviews) I find that the mining rush did not significantly increase deforestation or degradation, relative to counterfactual forest loss from other causes. Field data indicate that lemur populations appeared to remain healthy three years after the rush. These results emphasize the heterogeneity of impacts of ASM and the need for more robust, case-study evaluations to ensure policy responses are evidence-based and appropriate.

Overall, I find that while there is potential for biodiversity to be impacted by artisanal gem mining across Madagascar, the impacts of ASM are highly heterogenous and, in some cases, may be lower than alternative livelihood activities driving land cover change. Furthermore, there are opportunities for policies, including pragmatic alternative approaches to licensing and zonation, to mitigate the environmental trade-offs of mining in Madagascar, but these require effective governance.

Declaration

I hereby declare that this thesis is the results of my own investigations, except where otherwise stated. All other sources are acknowledged by bibliographic references. This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree unless, as agreed by the University, for approved dual awards.

I confirm that I am submitting this work with the agreement of my Supervisor(s).'

______Katie Devenish____________________________

'Yr wyf drwy hyn yn datgan mai canlyniad fy ymchwil fy hun yw'r thesis hwn, ac eithrio lle nodir yn wahanol. Caiff ffynonellau eraill eu cydnabod gan droednodiadau yn rhoi cyfeiriadau eglur. Nid yw sylwedd y gwaith hwn wedi cael ei dderbyn o'r blaen ar gyfer unrhyw radd, ac nid yw'n cael ei gyflwyno ar yr un pryd mewn ymgeisiaeth am unrhyw radd oni bai ei fod, fel y cytunwyd gan y Brifysgol, am gymwysterau deuol cymeradwy. Rwy'n cadarnhau fy mod yn cyflwyno'r gwaith hwn gyda chytundeb fy Ngoruchwyliwr (Goruchwylwyr

Author Contributions

Chapter 1:

I wrote this chapter which was reviewed by Simon Willcock, Julia Jones, and Kathryn Goodenough.

Chapter 2:

I conceived and designed the study with Julia Jones and Simon Willcock. Sébastien Desbureaux contributed to the design of the statistical analysis. I compiled the data, wrote the code, and performed the main statistical analyses. Sébastien Desbureaux wrote and ran the robustness checks. I wrote the paper which was reviewed by all authors plus Kathryn Goodenough.

Chapter 3:

I conceived and designed the study with Kathryn Goodenough, Julia Jones, and Simon Willcock. Kathryn Goodenough selected the geological units to map the area of gem potential. Harifidy Rakoto Ratsimba advised on the biological data. I collected and edited the data, and conducted all analyses. I wrote the paper which was reviewed by all authors.

Chapter 4:

I conceived and designed this study with Julia Jones and Simon Willcock. The field data comes from a survey in 2019 designed by Julia Jones, Rio Heriniaina and Kim Reuters. Rio Heriniaina conducted the lemur survey and interviews and collated the data. I collated the spatial data and performed all statistical analyses. I wrote the paper, which was reviewed by Simon Willcock, Julia Jones, Rio Heriniaina, and Sarobidy Rakotonarivo.

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I wrote this chapter which was reviewed by Simon Willcock, Julia Jones, and Kathryn Goodenough.

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Contents

List of Abbreviations

AMEA	Authorisation for Artisanal Mining
ANK	Ankerana (a biodiversity offset)
ASGM	Artisanal and small-scale gold mining
ASM	Artisanal and small-scale mining
BA	Before-After
BCMM	Bureau de Cadastre Minier de Madagascar
CAR	Central African Republic
CAZ	Coridor Ankeniheny-Zahamena
CFAM	Coridor Forestiere Analamay-Mantadia (a biodiversity offset)
CI	Control - Intervention
CZ	Conservation Zone (a biodiversity offset)
DiD	Difference-in-differences regressions
DRC	Democratic Republic of Congo
GDP	Gross Domestic Product
GFC	Global Forest Change data
IFC	International Finance Corporation
KBA	Key Biodiversity Area
MSPE	Mean Square Prediction Error
NGO	Non-governmental organisation
NNL	No Net Loss

OLS	Ordinary Least Squares
PA	Protected Area
PGRM	Projet de Gouvernance des Ressources Minérales
PREA	Permis Réservé aux Exploitants Artisanaux
PRSM	Projet de Reforme du Secteur Minier
PSM	Propensity Score Matching
QMM	QIT Minerals Madagascar
REDD+	Reduced Emissions from Deforestation and Degradation
SDGs	Sustainable Development Goals
TMF	Tropical Moist Forests Dataset
TTF	Torotorofotsy (a biodiversity offset)
UN	United Nations
USD	US Dollar

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Chapter 1: Introduction

Mining provides the material foundations upon which society is built. Global consumption and demand for metals and minerals is enormous - and growing. Consider an average 110 g mobile phone. One phone can contain 53 different metals, requiring between 5 kg and 139 kg of ore to be mined (90% of which is for gold, platinum, palladium, and Rare Earth Elements); 7.5 billion smartphones were sold between 2012 and 2017, consuming over 366,000 tonnes of metals (Bookhagen *et al.*, 2020). As global consumption grows, technology continues to proliferate, and the world turns to electrification to mitigate the climate crisis, demand for the metals required for electric circuitry and low-carbon technology (including nickel, cobalt, lithium, copper, chromite) is expected to increase significantly (Hund *et al.*, 2020; IEA, 2021; Calvo and Valero, 2022). Current production levels will be insufficient to meet demand for some metals, meaning for society to reach global emissions reductions targets, mining will likely need to expand (Hund *et al.*, 2020; IEA, 2021; Zeng *et al.*, 2022).

Yet, mining can entail significant environmental costs (Sovacool *et al.*, 2020), trading-off biodiversity, ecosystem services, and even climate stability (metal mining accounted for at least 8% of global greenhouse gas emissions in 2019; IPCC, 2022). In this thesis, I explore these environmental trade-offs, using the case study of Madagascar, a country rich in both minerals and biodiversity. In this chapter I outline the rationale for this thesis and seek to answer the following questions: 1) Why mining? 2) Why Madagascar? And 3) What is the focus of my research? In doing so, I review the current state of knowledge on the environmental trade-offs of mining, reveal key knowledge gaps, and identify appropriate methodology to fill those knowledge gaps. Throughout this thesis I frame mining in relation to the Sustainable Development Goals (SDGs), considering how the industry fits within the global blueprint for a more sustainable future. This framing encourages consideration of both synergies and trade-offs between mining and sustainable development. It also serves as a useful reminder that the negative impacts should be assessed in relation to the benefits and compared to viable alternatives.

Mining and the Sustainable Development Goals

Before exploring these synergies and trade-offs we must first define mining and distinguish the two main forms, as this influences the volume and distribution of costs and benefits.

Differentiating Industrial and Artisanal and small-scale Mining

Mining involves the extraction and primary processing of minerals, metals, aggregates, or fossil fuels from the Earth's surface. It is a highly diverse activity, employing an array of methods and operating across a range of scales, modes of organisation, and technology use. The form of mining depends on the geological characteristics of the deposit (including the target mineral, ore grade, depth, scale, and location) and level of capitalisation.

Here, I differentiate mining into two broad categories according to scale and formal organisation: industrial (elsewhere termed large-scale mining), and artisanal and small-scale mining (ASM).

Industrial mining refers to medium to large-scale operations which are capital-intensive, highly-mechanized, often high-tech, and which possess the required permits and licenses to operate legally. For example, the Ambatovy nickel and cobalt mine in Madagascar covers more than 2,000 hectares, uses advanced technology in a high-tech processing facility, and has an annual production capacity of 60,000 tonnes of nickel and 5,600 tonnes of cobalt (Von Hase *et al.*, 2014; Ambatovy, 2022). There are an estimated 6,000 industrial mines in operation worldwide affecting an estimated 57,000km^2 of global land (Maus *et al.*, 2020).

Definitions of artisanal and small-scale mining vary, but it is generally considered a labour-intensive form of mining characterised by limited use of machinery (Hilson and McQuilken, 2014). Low capital requirements mean ASM is easily accessible (Yakovleva, 2007; Hilson, 2009; Funoh *et al.*, 2017). ASM is extensive, occurring across Asia, Latin America, and Sub-Saharan Africa, and exploits a variety of minerals (IGF, 2017). In Madagascar an estimated 500,000 people are involved in ASM for gold, ruby and sapphire (World Bank, 2010). However, up to 90% of global ASM is illegal as miners do not possess the required titles or permits to mine legally (World Bank, 2020). As such, ASM is mostly confined to the informal economy (IGF, 2017).

Different minerals and deposits are suited to different types of mining. Artisanal and small-scale mining can exploit deposits which are either uneconomical or ill-suited to industrial mining (Dondeyne *et al.*, 2009; Gandiwa and Gandiwa, 2012; Klein, 2020; Nopeia *et al.*, 2022). Such deposits may be low-grade (i.e. only a small fraction of the ore consists of the desired mineral), low-value (e.g. mica), small, or highly-dispersed. ASM also exploits high-value deposits which are amenable to industrial mining, but there is no industrial interest. Although individual ASM operations produces far less than an industrial mine, collectively, the sheer number of miners involved means that ASM makes significant contributions to mineral supply. Globally, ASM supplies an estimated 20% of global gold, 80% of sapphires, 25% of tin, and 20% of tantalum (considered a critical raw material for its applications in electronics; IGF, 2017; Bookhagen *et al.*, 2020). In some countries, ASM is the major contributor to exports of certain minerals.

Both industrial mining and ASM encompass an enormous heterogeneity of forms, and the boundaries can be blurred. In some places (e.g., the Amazon, Ghana and Indonesia) artisanal and small-scale mining has become increasingly capitalised and mechanised, and can involve the use of heavy machinery and high-powered water pumps (Espejo *et al.*, 2018; Lahiri-Dutt, 2018b; Owusu, Bansah and Mensah, 2019; Siqueira-Gay and Sánchez, 2021). While the individual scale of such operations remains small, where there is an aggregation of such semi-mechanised ASM the scale and impacts can resemble, or even exceed, those from industrial mining (Lobo *et al.*, 2016; De Haan, Dales and McQuilken, 2020; Barenblitt *et al.*, 2021; Siqueira-Gay and Sánchez, 2021). At the other end of the ASM spectrum, artisanal mining is performed with only manual labour, simple tools (e.g. a pick, bucket, and a simple hand-made sieve) and no machinery (Figure 1; Tilghman, Baker and Deleon, 2007).

The following sections evaluate the benefits and trade-offs of mining for people, economies, and the environment in relation to the Sustainable Development Goals. When assessing the balance of impacts it is important to consider the end-uses and functions of mined products. Some mined products are necessary to meet our energy, infrastructural, and development needs (e.g. copper and nickel). However, others (such as gemstones) are luxury, non-essential products destined for affluent consumers.

Figure 1: An artisanal sapphire miner excavating material from a ~9 metres deep pit dug by hand in Ilakaka, Madagascar. Photo credit: Author.

Mining's positive contributions to the Sustainable Development Goals

Mining makes positive contributions to almost all the SDGs, the set of aspirational global goals for more sustainable future, particularly those relating to economic and social development (No Poverty, Zero Hunger, Good Health and Well-being, Quality Education, Decent Work and Economic Growth; Figure 2). Mining can promote infrastructure development, industrialisation, and responsible consumption and production. Furthermore, mined products are essential for the growth of low-carbon energy, technology and sustainable cities (Sonesson, Davidson and Sachs, 2016; De Haan, Dales and McQuilken, 2020; Hilson and Maconachie, 2020). Mining, particularly ASM, can also contribute towards reducing economic and gender inequalities (Yakovleva, 2007; Lawson, 2018). Both ASM and industrial mining make important contributions to socio-economic development, but often through different channels, and to different beneficiaries.

Figure 2: The 17 UN Sustainable Development Goals. Source: UN

Most of the economic contributions to society from industrial mining are disbursed to governments (national, regional, or local), through the payment of taxes, royalties, and dividends (ICMM, 2016). This can provide a much-needed boost to government budgets and enable investment in socio-economic development (ICMM, 2016), although this may not always reach communities directly affected by the mining activity. The economic contributions of industrial mining are particularly significant in low- and middle-income countries where mineral production and rents comprise a large proportion of GDP and total exports (Ericsson and Löf, 2019). For example, mineral rents comprise over 20% of national GDP in Suriname, Mauritania and Mongolia, and 10% - 20% in Guyana, Democratic Republic of Congo (DRC), Liberia and Papua New Guinea (Ericsson and Löf, 2019). Industrial mining also drives improvements in infrastructure, facilitating investments in roads, railways, power networks, and ports (Canavesio, 2014; ICMM, 2016). Industrial mining benefits individuals through direct employment (although there are no robust global estimates of employment in industrial mining, it is typically between 1-2% of the national workforce; ICMM, 2016), the provision of training, and local procurement of goods and services (Sonesson, Davidson and Sachs, 2016). This stimulates businesses, boosts local economies, and generates substantial indirect employment and tax revenue (ICMM, 2016). For example, in 2022 Ambatovy spent $344 million on local procurement, employed over 4,000 people (of whom 13% are women) and contributed $46 million to the Malagasy government in taxes and royalties (Ambatovy, 2022). Yet the number of people who directly benefit from industrial mining is still much smaller than those supported by ASM, which constitutes an estimated 90% of the global mining workforce (Villegas *et al.,* 2012).

ASM directly involves 45 million people in 80 countries and supports an estimated 135 million more in downstream industries (World Bank, 2020). The economic benefits from ASM accrue mostly to the individual miners, traders, and those involved in the attendant service industry. As ASM operates mostly outside of the legal economy products are often traded informally without paying taxes or royalties, meaning little revenue is captured by the state (IGF, 2017).

ASM is generally considered a poverty-driven activity and has rapidly expanded over the past three decades (from ~6 million miners in 1993; IGF, 2017) in response to poverty, economic crises, increasing mineral prices, growing unemployment, and the declining viability of agriculture due to structural adjustment, market liberalisation, and climate change (Hilson and Garforth, 2012; Hilson and McQuilken, 2014; Ncube-Phiri *et al.*, 2015; Arthur *et al.*, 2016; Dezécache *et al.*, 2017; Bansah, Arthur-Holmes and Assan, 2023). Engaging in ASM generally provides enough income to meet basic needs while additional income (i.e., from larger finds) can fund healthcare, education, and enable investments in land, housing, livestock, or business development to strengthen future livelihoods (Werthmann, 2009; Hilson and Garforth, 2012; Villegas, Turay and Sarmu, 2013; Arthur *et al.*, 2016; Funoh *et al.*, 2017; Lawson, 2018). Although income from ASM can still be precariously low and inconsistent (Tilghman, Baker and Deleon, 2007; Cartier, 2009; Lahiri-Dutt, 2018b), it often exceeds income from alternative livelihoods, such as subsistence agriculture (Schure *et al.*, 2011; Hilson and Garforth, 2012; Villegas *et al.*, 2012; Macháček, 2019; Baffour-Kyei *et al.*, 2021). For some people ASM is their sole income-generating activity, while for others it is a complementary activity undertaken alongside agriculture or other non-farm livelihood activities (Banchirigah and Hilson, 2010; Kamlongera, 2011; Hilson, 2016). For farmers who engage in ASM seasonally or part-time, supplementary income from ASM has helped to fill the gap in household income left by diminishing agricultural returns, buffer shocks, and enabled the purchase of expensive fertilisers or pesticides (Kamlongera, 2011; Hilson and Garforth, 2012; Stoudmann *et al.*, 2021). Contrary to fears that ASM weakens agricultural economies, evidence has shown that ASM has helped to sustain smallholder agriculture in the face of increasingly challenging market conditions (Hilson and Garforth, 2012). ASM can also stimulate agricultural production by increasing local demand (Maconaghie and Binns, 2007).

ASM provides an independent means of making money for women and consequently can support financial autonomy and emancipation, and enhance gender equality (Yakovleva, 2007; Lahiri-Dutt, 2018a; De Haan, Dales and McQuilken, 2020; Lawson and Lahiri-Dutt, 2020). Research from Madagascar and Burkina Faso has shown that women engage in ASM, despite the risks (discussed below), to gain autonomy, raise funds to invest in land or business development, and provide a better life for their children (Werthmann, 2009; Lawson, 2018).

ASM can drive in-migration, urban expansion, and stimulate the local economy and business development (Hilson, 2016; Canavesio and Pardieu, 2019). It also facilitates (often private) investment in infrastructure, transport, communications, clean water, and sanitation (De Haan, Dales and McQuilken, 2020). For example, the discovery of sapphires at Ilakaka in Southern Madagascar in 1998 triggered a massive influx of prospective miners (termed a mining rush) which transformed a tiny hamlet into a bustling trading town of 100,000 people (Canavesio and Pardieu, 2019). Population growth associated with mining brought access to electricity, and later internet, improved transport connections and stimulated an attendant service industry of cooks, porters, traders, restaurants, hotels, shops, taxis and bars (Canavesio, 2014).

Negative socio-economic impacts of mining

However, mining (both industrial and ASM) can also bring concerning socio-economic costs which can undermine the sector's positive contributions to the SDGs, and hinder progress towards poverty alleviation, education, health, gender equality, and security goals.

Social injustices permeate throughout the global mining sector. Economic and supply chain inequalities create power asymmetries between producers (i.e. miners), who labour under poor working conditions while receiving only a tiny proportion of the product's final value, and consumers. These inequities are particularly striking in the context of ASM, where miners face the serious risks outlined below to supply luxury, high-end products such as gold and gemstones to affluent consumers.

Mining is a dangerous occupation and miners are exposed to a range of physical and chemical health hazards (ILO, 1999). Some of the main health risks from mining include: injury or death from mine collapse, explosions or flooding; asphyxiation in poorly ventilated underground spaces; accidents while using equipment; respiratory illnesses from dust inhalation (e.g. silicosis); and exposure to mercury or cyanide (discussed in the following section; ILO, 1999; Donoghue, 2004; Gibb and O'Leary, 2014; Smith *et al.*, 2016; Nkuba, Muhanzi and Zahinda, 2022; Adomako and Hausermann, 2023). These risks are exacerbated by poor health and safety practices, particularly in ASM (Arthur *et al.*, 2016; Bansah, Yalley and Dumakor-Dupey, 2016; Smith *et al.*, 2016). In Ghana, there were 622 deaths associated with ASM reported between 2007 and 2020 (and this is likely an underestimate as deaths from ASM are often under-reported; Stemn, Amoh and Joe-Asare, 2021). Furthermore, poor sanitation at mine sites and informal settlements, and a lack of clean water increase the risk of disease (Perkins, 2016; Smith *et al.*, 2016).

The influx of people and money associated with mining can cause a range of social issues. It can cause inflation (potentially undermining food security), increase local crime, drug use, alcoholism, prostitution, and spread sexually-transmitted diseases, including HIV (ILO, 1999; Colchester, La Rose and James, 2002; Walsh, 2003; Bansah *et* *al.*, 2018; Stoudmann *et al.*, 2021). At an ASM site in Mali the HIV rate among the sampled population (N = 224) was 8%, far above the national average of 1.5% (Sagaon-Teyssier *et al.*, 2017). Furthermore, the study showed that the risks of contracting HIV was greatest for female non-sex workers. Women in ASM risk sexual assault, stigmatisation, and exclusion (Colchester, La Rose and James, 2002; Werthmann, 2009; Lawson and Lahiri-Dutt, 2020). Gender inequality remains pervasive; there is often a strict division of labour, women are usually confined to less profitable roles or locations, and are more vulnerable to being cheated (Yakovleva, 2007; Werthmann, 2009; Kelly, 2014; Lawson and Lahiri-Dutt, 2020).

Child labour in ASM is widespread and considered one of the 'Worst Forms of Child Labour' by international organisations (ILO, 1999), prompting commitments to eliminate it (Hilson, 2010). However, such blanket approaches tend to overlook the complex and nuanced interactions between child labour, poverty, and education (De Haan, Dales and McQuilken, 2020), and the precarious reality of life for the poorest and most marginalised, which means families may have little choice but to send, or bring, their children to work. Working in ASM can interrupt or prevent schooling (four in ten children in the Tarkwa mining region of Ghana miss school daily to engage in ASM; Bansah *et al.,* 2018), and expose children to exploitation, abuse, and the health risks outlined above (ILO, 1999; Hentschel, Priester and Hruschka, 2002; Potter and Lupilya, 2016; Schwartz, Lee and Darrah, 2021). However, research has shown that child labour in ASM can also enable education, as income is used to pay school fees which may otherwise be unaffordable (Hilson, 2010; Potter and Lupilya, 2016). Child labour can also make essential contributions to household income, helping families to meet their basic needs (Hilson, 2010; Thorsen, 2012; Darko, 2014; Potter and Lupilya, 2016).

The informality of ASM marginalises miners and leaves them vulnerable to being cheated or exploited by mine owners, sponsors, buyers, and corrupt officials seeking bribes (Duffy, 2007; Cartier, 2009; Kelly, 2014; Verbrugge, 2015; Stoudmann *et al.*, 2021). Informality and lack of regulation can also attract organised crime. In Colombia, the DRC, and the African Great Lakes region criminal organisations and armed groups have turned to small-scale mining to fund their activities, bringing conflict and insecurity, threatening miners and local communities (Global Witness, 2009; Rettberg and Ortiz-Riomalo, 2016). Armed groups can take control of mines, use forced labour, and extort payments from miners (Global Witness, 2009; Kelly, 2014; Rettberg and Ortiz-Riomalo, 2016). The 'lootability' of mineral resources such as gold, diamonds or gemstones (i.e. small size, easy to mine, transport, and trade covertly) has played a key role in fuelling and financing large-scale armed conflicts in many African countries, the most well-known being the role of 'blood diamonds' in prolonging Sierra Leone's brutal civil war (Le Billion, 2001; Ross, 2003; Collier and Hoeffler, 2004; Silberfein, 2004; Maconachie and Binns, 2007). Mineral wealth can also promote corruption and economic mismanagement, leading to the underselling of mining assets and unfavourable contracts with foreign mining companies (Geenen, 2012), and potentially causing political instability. This has spawned the narrative that in countries with weak governance, mineral wealth can constitute a *resource curse,* slowing or even reversing development (Collier and Hoeffler, 1998, 2004; Ross, 1999). This has certainly been the case in certain places at certain times, but is not necessarily guaranteed or fixed (Mehlum, Moene and Torvik, 2006; Maconachie and Binns, 2007; Badeeb, Lean and Clark, 2017). For example, Maconaghie and Binns (2007) describe how diamonds have helped to fuel post-conflict reconstruction and local economic development in Sierra Leone, supported by schemes (such as the Kimberly Process and the Peace Diamond Alliance) aimed at increasing transparency and reducing smuggling.

Overall, these risks mean that for some unlucky miners, ASM can become a poverty trap. Ill-health and accidents can prevent people from working, while meagre earnings or debt can keep miners trapped, potentially in exploitative labour arrangements (Cartier, 2009). These impacts can even span generations: for example, through the effects of mercury on children's development or poor access to education. Yet, despite these risks poverty, a lack of viable alternatives, and the potential for a better life continue to draw people into ASM.

The environmental trade-offs of mining

Mining can also involve serious environmental trade-offs. These trade-offs are particularly concerning given the substantial overlap between mineral wealth and biodiversity (Pascal *et al.*, 2008; Edwards *et al.*, 2014; Sonter, Ali and Watson, 2018). Deposits of some intensively mined metals (e.g. silver, bauxite and nickel) are disproportionately concentrated in areas of high biodiversity (Murguía, Bringezu and Schaldach, 2016; Sonter, Ali and Watson, 2018; World Bank, 2019). Mining also occurs in, or near, many Protected Areas and Key Biodiversity Areas (Durán, Rauch and Gaston, 2013; Sonter *et al.*, 2020a). Globally, 10% of land potentially impacted by industrial mining (i.e., within 10km of a planned or existing mine) occurs within Protected Areas, 6% within Key Biodiversity Areas and 15% with Remaining Wilderness Areas (Sonter *et al.*, 2020a).

Both ASM and industrial mining produce similar environmental impacts, although at a very different scale. The environmental impacts of industrial mining are relatively well-documented and global databases on industrial mining locations (e.g., the SNL Minerals and Mining database) facilitate case-study and global evaluations (e.g., Giljum *et al.*, 2022). This evidence has been synthesised by Sonter *et al* (2018) and Farjana *et al* (2019). Therefore, here I briefly outline the general impacts of mining (including some statistics on industrial mining) before digging more deeply into the literature on the environmental impacts of ASM. Despite the global extent and economic importance of ASM there is limited quantitative evidence on its environmental impacts, particularly on land cover and biodiversity. This lack of evidence is particularly concerning given the expansion of ASM in some of the most biodiverse regions on earth (Villegas *et al.*, 2012).

Overview of the environmental impacts of mining

Impacts can be categorised as direct impacts, which result from on-site mineral extraction and processing activities, and indirect impacts, which stem from off-site associated infrastructure development and the knock-on effects of mining on land and resource use by other actors, or within the supply chain (Edwards *et al.*, 2014). Indirect impacts can potentially be extensive (Sonter *et al.*, 2017; Giljum *et al.*, 2022).

Mining directly impacts biodiversity and ecosystem services through land cover change, hydrological impacts, and pollution (Northey *et al.*, 2016; Sonter, Ali and Watson, 2018; Farjana *et al.*, 2019; Werner, Bebbington and Gregory, 2019). Land clearance for mining can cause deforestation, and the loss and fragmentation of natural habitat, leading to biodiversity loss (Sonter *et al.*, 2014; Alvarez-Berríos and Mitchell Aide, 2015; Obodai *et al.*, 2019; Giljum *et al.*, 2022). Between 2000 and 2019 industrial mining caused an estimated 326,400 hectares of deforestation worldwide (Giljum *et al.*, 2022). Although mining is a minor contributor to global deforestation, accounting for 7% compared to 73% from agriculture (Hosonuma *et al.*, 2012), in some places, mining is the greatest driver of forest loss (eg., Suriname and Guyana; Alvarez-Berríos and Mitchell Aide, 2015; Giljum *et al.*, 2022).

Mining can alter hydrological regimes through water abstraction, river diversion, and increased erosion and sedimentation (Northey *et al.*, 2016; Flatley and Markham, 2021; Kinyondo and Huggins, 2021). This can deplete groundwater and reduce water flow, quality, and downstream water availability (Northey *et al.*, 2016, 2018).

Waste is a key source of environmental risk from industrial mining (Islam and Murakami, 2021). The volume of waste and risks generated depends on the type of mineral (which affects the ore grade), the mining method used (e.g. open pit vs. underground mining), and the degree of processing and remediation (Dold, 2008; Lèbre, Corder and Golev, 2017). Waste is stored in rock dumps and tailings dams (for liquified waste) which, if not appropriately constructed and maintained, risk collapse (Azam and Li, 2010; Kossoff *et al.*, 2014). Some of the worst mining disasters in the past 100 years have been caused by the collapse of tailings dams (Santamarina, Torres-Cruz and Bachus, 2019; Islam and Murakami, 2021). Dam collapses release a mudslide of contaminated sediment which can destroy land, settlements, and claim many lives (eg., the 2019 collapse of a large tailings dam near Brumadinho, Brazil killed over 250 people; Kossoff *et al.*, 2014; Santamarina, Torres-Cruz and Bachus, 2019; Silva Rotta *et al.*, 2020).

Poorly-managed mines and tailings facilities are also vulnerable to acid mine drainage and heavy metal leaching. Acid mine drainage occurs when sulphides within tailings and abandoned mines are exposed to air and oxidise to form sulphuric acid which, if not appropriately contained, can be leached from the tailings by rainwater (Johnson and Hallberg, 2005; Dold, 2008). Water drainage through mines and tailings can also leach heavy metals and erode contaminated sediment (Wang *et al.*, 2019). This acidic and/or metalliferous run-off contaminates soils, groundwater, and downstream river systems with heavy metals, alters pH and reduces water quality (Luís *et al.*, 2009; Li *et al.*, 2014; Wang *et al.*, 2019; Macklin *et al.*, 2023). This increases the risk of contact or ingestion of heavy metals by humans and wildlife, with potentially harmful health effects (Zhao *et al.*, 2012; Li *et al.*, 2014; Affandi and Ishak, 2019). Hazardous pollutants from mining include lead, mercury, chromium, arsenic, and cadmium, which are toxic and can be carcinogenic in high enough concentrations (Nriagu, 1988). Overall, an estimated 23 million people worldwide are exposed to potentially harmful levels of heavy metals from mining-related pollution (Macklin *et al.*, 2023).

However, the direct environmental impacts of industrial mining vary significantly depending on the target mineral, local environment, and the geological characteristics of the deposit. The latter determines the mining methods, scale, water requirements, amount of land clearance, and waste risks (Sonter, Ali and Watson, 2018).

Mining can also generate indirect environmental impacts through local population growth, urban expansion, attendant infrastructure development, and supply chain activities (Edwards *et al.*, 2014; Sonter *et al.*, 2017; Giljum *et al.*, 2022). Mining development can attract large numbers of people searching for employment opportunities (Nyame, Andrew Grant and Yakovleva, 2009; Sonter *et al.*, 2017). This in-migration can increase local demand for land and resources, leading to land conversion for agriculture and urban expansion (Mwitwa *et al.*, 2012; Bebbington *et al.*, 2018; Obodai *et al.*, 2019). The construction of new roads, railways, or settlements to serve mining can open up remote frontiers, facilitating the access and expansion of other extractive resource uses (e.g., agriculture, logging or hunting; Mwitwa *et al.*, 2012; Bebbington *et al.*, 2018; Siqueira-Gay, Sonter and Sánchez, 2020; Kinyondo and Huggins, 2021). These factors generate indirect and cumulative impacts which are potentially greater, wider-reaching, and longer-lasting than the direct impacts (Sonter *et al.*, 2017; Seki *et al.*, 2022). For example, Sonter *et al* (2017) found that deforestation within a

70km buffer zone around mining leases in Brazil was 12 times greater than deforestation *within* the leases, and significantly higher than in similar non-mining areas. In Tanzania, Seki *et al* (2022) found that tree and butterfly diversity, tree density, and carbon storage was actually higher inside the mining lease than outside, and decreased with growing distance from mine and increasing proximity to the mining town. This suggests that the indirect impacts from associated urban expansion had a greater impact on local biodiversity and forest cover than the direct impact of the mine itself.

Evidence on the environmental impacts of ASM

Evidence on the environmental impacts of ASM is limited and tends to be biased towards certain impacts (mercury pollution), and certain regions (Amazonia and Ghana). In contrast to industrial mining there are no global (and very few national) spatial databases on ASM occurrences. This impedes evaluations and contributes to the concerning lack of evidence. Much of the evidence which does exist consists of descriptive accounts from site visits, or qualitative evidence from interviews (e.g. Cook and Healy, 2012; Gandiwa and Gandiwa, 2012; Macháček, 2019; Kinyondo and Huggins, 2021; Achina-Obeng and Aram, 2022; Denison Mundi, 2022). These accounts have recorded impacts (direct and indirect) including deforestation, habitat loss, heavy metal pollution, hunting, river sedimentation, and reduced water quality and flow.

Much of the literature focusses on the impacts of mercury pollution from artisanal and small-scale gold mining (ASGM). This is not surprising, given the serious consequences of mercury pollution for human and ecosystem health (Boening, 2000; Gibb and O'Leary, 2014), and the fact that ASGM is the largest contributor of global anthropogenic mercury emissions (UN Environment, 2019). In 2015, ASGM released an estimated 1,220 tonnes of mercury into the environment (UN Environment, 2019). Mercury is used in ASGM to amalgamate fine gold particles. The amalgam is then burned to isolate the gold, releasing mercury into the atmosphere (Hinton, Veiga and Veiga, 2003). Much of this condenses into soils and enters river systems, potentially far from the initial source point (Ouboter *et al.*, 2012). Mercury can also spill directly into rivers at mine sites

(Diringer *et al.*, 2015). Once in rivers, mercury is transformed into organic methyl-mercury which is absorbed by micro-organisms and enters the food chain, where it biomagnifies and accumulates within higher predators such as birds and large fish (Boening, 2000; Ouboter *et al.*, 2012). Mercury contamination has been shown to reduce fitness and reproductive success, and increase mortality in fish (Sandheinrich and Wiener, 2011) and birds (Sierra-Marquez *et al.*, 2018). Mercury spreads into humans who inhale the vapours released from burning or eat contaminated food, particularly fish (Gibb and O'Leary, 2014; Wyatt *et al.*, 2017; UN Environment, 2019). This affects the health of mine workers, people living near processing facilities (Yard et al, 2012), and non-mining communities for hundreds of kilometres downstream (Diringer *et al.*, 2015; Wyatt *et al.*, 2017). Mercury is toxic to the nervous system, can cause neurological illnesses, developmental problems in children (Grandjean *et al.*, 1999), kidney problems (Yard et al, 2012), and a range of other chronic health issues (Gibb and O'Leary, 2014).

Other studies have documented the physical impacts of ASM on local hydrology, which affects water quality, flow, and downstream water availability (Rajaee *et al.*, 2015; Lobo *et al.*, 2016; Bansah *et al.*, 2018). ASM, particularly of alluvial deposits, is often conducted in or near waterways (Snapir, Simms and Waine, 2017; Obodai *et al.*, 2019). Excavation disturbs sediment on the riverbed, banks and surrounding landscape, increasing erosion and sediment load (Macháček, 2019). Panning releases extracted sediment directly into waterways which can cause siltation and increase water turbidity (Figure 3; Rajaee *et al.*, 2015; Bansah *et al.*, 2018; Kinyondo and Huggins, 2021). This can decrease photosynthesis, deplete oxygen levels and kill aquatic biodiversity (Mol and Ouboter, 2004; Villegas *et al.*, 2012; Rajaee *et al.*, 2015; Lobo *et al.*, 2016; Affandi and Ishak, 2019). High sediment loads can also lead to the sedimentation of rivers and reservoirs, raising water levels and reducing capacity (Ncube-Phiri *et al.*, 2015; Funoh *et al.*, 2017). In Zimbabwe an estimated 172,000 kg of sediment is moved by artisanal gold miners daily, most of which is washed into rivers. This has caused sedimentation of the Umzingwane Dam, reducing capacity and water supply, while the high turbidity has increased the cost of water treatment (Ncube-Phiri *et al.*, 2015). Declining water quality and availability from ASM has been documented elsewhere in Zimbabwe (Gandiwa and Gandiwa, 2012), Rwanda (Macháček, 2020), Ghana (Rajaee *et al.*, 2015; Arthur *et al.*, 2016; Achina-Obeng and Aram, 2022), Cameroon (Funoh *et al.*, 2017), DRC (Schure *et al.*, 2011; Nkuba, Muhanzi and Zahinda, 2022), Brazil (Lobo *et al.*, 2016), Suriname (Mol and Ouboter, 2004), and Madagascar (Cook and Healy, 2012).

Figure 3: Sapphire miners at Bemainty, Madagascar sieving extracted sediments in a highly turbid stream diverted from the main channel. Photo credit: Rosey Perkins

Studies have documented indirect impacts of ASM from local population growth, miners' natural resource use, and improved accessibility (Villegas *et al.*, 2012; Funoh *et al.*, 2017; Kinyondo and Huggins, 2021). Miners harvest wood for fuel and constructing shelters and mineshaft supports, resulting in deforestation (Schure *et al.*, 2011; Funoh *et al.*, 2017; Macháček, 2020; Nkuba, Muhanzi and Zahinda, 2022). Evidence from Central Africa and Madagascar shows that miners may also engage in other forms of natural resource use (e.g. hunting or charcoal production) to supplement income, particularly as the volume of finds decreases (Cook and Healy, 2012; Funoh *et al.*, 2017; Spira *et al.*, 2019; Zhu and Klein, 2022). In the Sangha Tri-National Landscape (which crosses Cameroon and the Central African Republic [CAR]), Schure *et al* (2011) found that 21% of interviewed Cameroonian miners and 28% of CAR miners harvested forest products, including bushmeat, for subsistence and sale. Elsewhere, the expansion of ASM into remote areas has facilitated access for poachers (in Cameroon poachers use ASM camps as a base to hunt protected species for the wildlife trade) and illegal timber harvesting (Funoh *et al.*, 2017; Denison Mundi, 2022). In eastern DRC hunting associated with ASM was linked to population declines and the local disappearance of large mammals, including elephants, chimpanzees, and gorillas (Spira *et al.*, 2019).

Deforestation and natural habitat loss from ASM has been reported by many of the above studies but only a handful have attempted to quantify the extent of ASM-induced land cover changes. These studies are concentrated mostly in Amazonia (Swenson *et al.*, 2011; Asner *et al.*, 2013; Alvarez-Berríos and Mitchell Aide, 2015; Lobo *et al.*, 2016; Asner and Tupayachi, 2017; Espejo *et al.*, 2018; Guyana Forestry Commission, 2023) and Ghana (Snapir, Simms and Waine, 2017; Obodai *et al.*, 2019; Gallwey *et al.*, 2020; Barenblitt *et al.*, 2021; Baddianaah, Baatuuwie and Adongo, 2022), where ASM is particularly extensive. These studies can be split into two categories: 1) primary analyses of land cover change based on classification of satellite imagery, 2) those using secondary forest loss data to quantify deforestation in mining regions. First, I present several regional and national-scale studies which estimate ASM-related deforestation based on primary classification of satellite imagery. Then, I review a comparative study which uses the second approach.

In the Amazon, Alvarez-Berríos and Mitchell Aide (2015) evaluated forest change at 1,600 potential gold mining sites (identified from secondary data sources and visual analysis of satellite imagery) by classifying a time-series of satellite imagery. Between 2001 and 2013, they estimate 168,000 hectares of forest was lost at these sites, concentrated within four major hotspots (the Guianas [41%], Madre de Dios, Peru [28%]; Tapajos- Xingu forests, Brazil [11%] and Antioquia, Colombia [9%]). However, these estimates include impacts from industrial mining and the low resolution of the satellite images (250m) was unable to capture clearance for very small-scale artisanal mining (Alvarez-Berríos and Mitchell Aide, 2015). In a subsequent study, Espejo *et al* (2018) estimate that ASM caused nearly 100,000 ha of deforestation in Madre de Dios alone between 1984 and 2017 – 53% of which occurred since 2011. They find that smaller-scale, less-mechanised mining began to increase and expand into new areas from 2000, accounting for the majority of new mining development and deforestation (Asner and Tupayachi, 2017; Espejo *et al.*, 2018). The government of Guyana reports annual deforestation from ASM as part of its REDD+ Monitoring and Reporting obligations. Mining (which is predominantly ASM) is the largest contributor to deforestation in Guyana and in 2022 caused 5,262 ha of forest loss (81% of the national total), releasing an estimated 5.5 million tonnes of CO_2 (Guyana Forestry Commission, 2023). To my knowledge there are no national-scale estimates of land cover change from ASM in Ghana, but there are some regional estimates. In south-western Ghana, Barenblitt *et al* (2021) estimated 42,400 ha of forest was lost to ASM between 2005 and 2019 (compared to 5,000 ha from industrial mining), including 700 ha of deforestation within Protected Areas. In the decade from 2007-2017, which encompasses the global peak in gold prices (Dezécache *et al.*, 2017; Owusu, Bansah and Mensah, 2019), ASM quadrupled in extent in the study region.

In Ghana and Amazonia, ASM is extensive, impactful, and consequently high-profile. However, there is a concerning lack of quantitative evidence on the land cover impacts of ASM from elsewhere, where mining may be less extensive but could have potentially severe localised impacts. The most comprehensive analysis of the impacts of ASM on forests was conducted by the World Bank (2019a) as part of the Forest Smart Mining project. This study quantifies deforestation (using the Global Forest Change data) within a 5km buffer zone around 21 ASM sites in 12 different countries. They find that the rate of forest loss around these sites was highly variable, ranging from 0.1% to 46% between 2000 and 2016. However, there are serious caveats to this analysis, which influence the interpretation of these results. First, it uses a raw global dataset which may not be well suited to national contexts (Galiatsatos *et al.*, 2020; Kinnebrew *et al.*, 2022). Second, the deforestation rate is calculated over the period 2000 – 2016 without considering the start date of mining, meaning some deforestation could have occurred before mining began. Third, it does not attempt to isolate the deforestation caused mining by controlling for other drivers of forest loss. This means that the 122 ha of forest loss around Noyod mine in Mongolia (the site with 46% forest loss) is not necessarily attributable to the mining (which was also very small-scale – covering only 7 ha), but could have been caused by other unrelated factors (i.e. fire, timber harvesting, clearance for agriculture).

Counterfactual methods for evaluating the impact of mining

None of the existing quantitative evaluations of the forest impacts of ASM consider what would have happened in their study area in the absence of mining. Perhaps in Ghana if there had been no gold, faced with economic hardship and declining returns from agriculture (Hilson and Garforth, 2012) farmers may have been forced to clear forest to gain more land for farming, or engage in illegal timber harvesting, potentially causing an equivalent amount of deforestation to that which occurred under ASM. While this scenario may seem difficult to imagine, given how entrenched ASM is in Ghana, the point is that without ASM, people would have pursued alternative livelihoods which have their own environmental footprint. To fully evaluate the impact of something (i.e., a policy, intervention, or event), one must compare to what would have happened without it, i.e., the counterfactual scenario (Ferraro and Hanauer, 2014; Baylis *et al.*, 2016; Pressey *et al.*, 2021). In the case of mining in Ghana, this indicates how much additional deforestation was caused by mining, relative to how much *would have happened* from other causes in the absence of mining. This approach also helps to control for the influence of other drivers of change.

The impact of an event or intervention (e.g., an artisanal mining rush in a protected area) on outcomes (e.g., deforestation) in the exposed unit can be formally expressed as:

$$X_{1,t} = Y_{1t}^{I} - Y_{1t}^{N}$$ Equation 1 (adapted from Abadie *et al,* 2021)

Where Y_{1t}^{I} represents the observed outcome in the treated unit (*1*) after being exposed to an intervention (I) in time period *t*, and Y_{1t}^{N} is the potential outcome in the same unit *1* in time period *t* with no exposure to the intervention (N). The causal effect of an intervention ($X_{1,t}$) is the difference between the observed outcome with the intervention Y_{1t}^{I}, and the unobserved counterfactual outcome without it Y_{1t}^{N} (Stuart, 2010; Ferraro and Hanauer, 2014; Abadie, 2021). As this counterfactual is unobserved (as it didn't happen) it must be estimated (Ferraro and Hanauer, 2014). Fortunately, there are various statistical methods available for doing so (Blackman, 2013; Schleicher *et al.*, 2019).

In Before-After analyses, trends in pre-intervention outcomes ($Y^N_{1,0-t}$) are extrapolated into the post-intervention period (*t*) to estimate counterfactual outcomes (Blackman, 2013; Wauchope *et al.*, 2021). While this has the benefit of using the same unit (1) for comparison, it assumes that in the absence of the intervention, outcome trends would have remained stable (Wauchope *et al.*, 2021). This assumes that there are no external shocks and little change in the factors influencing outcomes (Blackman, 2013). As such, this method fails to account for the temporal heterogeneity prevalent in real-world applications (Christie *et al.*, 2019).

Control-Intervention analyses compare outcomes in the treated unit to those in a control unit (0) not subject to the intervention ($Y^N_{0,t}$). This avoids temporal bias, as outcomes are compared over the same post-intervention period *t.* However, it can induce spatial biases, as interventions are not randomly assigned in the landscape (Joppa and Pfaff, 2009; Blackman, 2013; Larsen, Meng and Kendall, 2019). Factors influencing the allocation of the intervention (i.e., selection to treatment) are likely to be correlated (directly or indirectly through other factors) with the outcome of interest (Stuart, 2010; Ferraro and Hanauer, 2014). For example, protected areas are disproportionately located in places which are steep, high, and far from cities or roads (Joppa and Pfaff, 2009). These factors reduce the likelihood of deforestation. Comparing outcomes in a protected area to a control unit (e.g., a buffer zone outside the protected area) which is flatter and more accessible will be biased, as any differences in outcomes could be attributable to these factors and not the effects of the protected area (Joppa and Pfaff, 2009; Blackman, 2013). Therefore, these differences confound estimates of the impact of protection (Ferraro and Hanauer, 2014). For example, in a case study of Costa Rican protected areas, Andam *et al,* (2008) showed that failure to control for differences in observable characteristics, which influence both outcomes and selection to treatment (i.e. slope, elevation, distance to road etc.), inflated estimates of protected area effectiveness by 65%. To control for such confounding factors and isolate the causal effect of an intervention, differences in observable characteristics between treated and control units must be minimised.

Statistical matching is one approach used to minimise these differences by identifying a set of control units that are as similar as possible to the treated unit(s) in a range of covariates hypothesized to influence the outcome of interest, and selection to treatment (Stuart, 2010; Ferraro and Hanauer, 2014). As matched treated and control units have a similar probability of outcomes (e.g., deforestation) under baseline conditions, yet differ in exposure to the intervention, the matched control units can be used to estimate counterfactual outcomes in the treated units (Schleicher *et al.*, 2019). Similarity can be measured using propensity scores, Mahalanobis distance, or exact matching, and a matching algorithm selects the closest match(es) to maximise similarity across the whole sample (Stuart, 2010; Ho *et al.*, 2011). However, while matching will reduce differences in observable characteristics, it will not completely eliminate them (Schleicher *et al.,* 2019). It also cannot control for unobserved differences which may confound outcomes.

Matching is often used as a pre-processing step to produce an appropriate control sample in non-experimental applications where the intervention is not randomly assigned. Matched data is then usually input into regressions to control for any remaining observed differences, and compare outcomes between treated and matched control units (Stuart, 2010; Schleicher *et al.*, 2019).

Difference-in-differences regressions

When panel data is available difference-in-differences regressions are often used to estimate the impact of interventions applied to a single unit (Angrist and Pischke, 2009; Larsen, Meng and Kendall, 2019). Difference-in-differences (DiD) regressions combine Before-After and Control-Intervention analyses (it is termed BACI in ecology), comparing the change in average outcomes before-after the intervention between treated and control units (Angrist and Pischke, 2009; Blackman, 2013; Wauchope *et al.*, 2021). DiD assumes that, in the absence of the intervention, the treated unit would have experienced the same average change in outcomes over the before-after period $\left(\Delta\overline{Y}_1^N\right)$ as the control unit ($\Delta\overline{Y}_0^N$), and this represents the counterfactual (Ryan *et al.*, 2019; Figure 4). Therefore, the difference between this estimated counterfactual and observed outcomes represents the impact of the intervention (black line in Figure 4). This assumption is usually contingent on the presence of parallel trends in outcomes between treated and matched control units *before* the intervention (Angrist and Pischke, 2009; Ryan *et al.*, 2019). This is an identifying assumption behind DiD and must be demonstrated for DiD to be considered a valid approach (although recent work is exploring extensions of DID which can cope with non-parallel trends; Ryan *et al.*, 2019; Rambachan and Roth, 2023). Furthermore, a lack of parallel trends suggests that there are some unobserved differences between the two groups which are causing outcomes to diverge, meaning the matched control units likely do not represent a good counterfactual (Fick *et al.*, 2021).

The difference-in-differences regression takes the form:

$$Y_{i,t} = BA_t + CI_i + (BA \times CI)_{i,t} + \in_{i,t},$$

where BA and CI are binary variables indicating whether the observation occurred before or after the intervention, in the control or intervention (i.e., treated) sample. The coefficient of BA X CI represents the average effect of the intervention on outcomes ($Y_{i,t}$). $\in_{i,t}$ represents the unobserved error.

Overall, DiD regressions are appropriate for assessing the impact of an intervention on a single unit, when the data meet the condition of parallel trends, and when outcomes are not considered to exhibit temporal autocorrelation and fluctuate around mean values (such as deforestation rates; Angrist and Pischke, 2009; Fick *et al.*, 2021; Wauchope *et al.*, 2021).

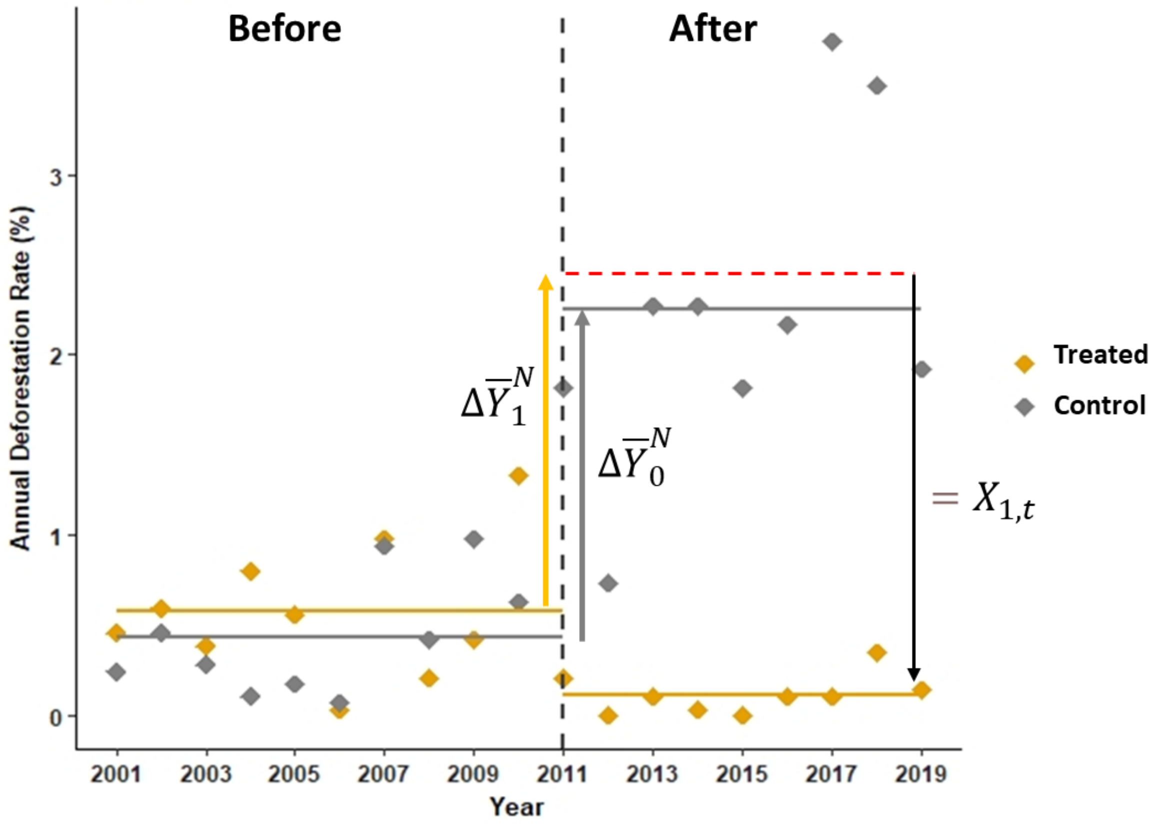

Figure 4: Diagram illustrating the theory behind difference-in-differences regressions using a plot from Chapter 2. Points show the annual deforestation rate in a sample of treated (yellow) and matched control (grey) pixels. The average annual deforestation rate over the before (2001-2010) and after (2011-2019) periods is shown by the solid yellow (treated) and grey (control) lines. The dashed black line indicates the year of the intervention in 2011. DiD analyses assume that in the absence of the intervention, the treated unit would have experienced the same change in average outcomes ($\Delta\overline{Y}_1^N$) over the before-after period as the control unit $\Delta\overline{Y}_0^N$, and this is the counterfactual (shown by the red line). The solid black line represents the estimated causal effect.

Matching plus DiD controls for the influence of observed confounding variables but it cannot control for unobserved confounders (Jones and Lewis, 2015). Where there are unobserved factors correlated with treatment assignment which also influence outcomes, estimates of impact can be biased (as the error term is correlated with the treatment indicator, violating the statistical assumption of endogeneity; Andam *et al.*, 2008; Ferraro and Hanauer, 2014; Larsen *et al.,* 2019). Rosenbaum bounds can be used to test the sensitivity of results to unobserved bias by estimating how strong the effect of omitted variables would need to be change the significance of results (Rosenbaum, 2007; Schleicher *et al.*, 2019). If results can change with only a small amount of unobserved bias, researchers may need to widen their choice of covariates or consider an alternative approach.

Fixed-effects panel regressions

Fixed effects panel regressions are a common approach used in impact evaluations where there are multiple treated units (e.g., Jones *et al.*, 2017; Carlson *et al.*, 2018; Oldekop *et al.*, 2019; Sills *et al.*, 2020), which can also help to reduce unobserved bias (Jones and Lewis, 2015). Fixed effects panel regressions exploit cross-sectional and temporal variability in panel data (repeated observations of selected units over time) to control for time-invariant unobserved bias through time-demeaning (Blackman, 2013; Jones and Lewis, 2015; Wooldridge, 2015). For each unit *i*, this involves taking an average of Equation 3 over all time periods *t* and subtracting from the original Equation 3. This differences out time-invariant components (including observed and unobserved covariates), but still enables consistent estimation of the treatment effect (Jones and Lewis, 2015). As such, fixed effects panel regressions provide a more accurate estimate of treatment effect when unobserved bias is present than other methods (Jones and Lewis, 2015). For example, out of several approaches tested, Ferraro and Miranda (2017) found that matching plus fixed effects panel regression was able to most closely approximate the results of treatment effect obtained from an experimental study. However, fixed effects panel regressions cannot reduce the influence of unobserved time-varying covariates (e.g., price volatility).

Following Jones and Lewis (2015), the fixed effect panel regression takes the following form:

$$Y_{it} = a + \beta 1 X_{i,t} + \beta 2\, C_{i,t} + \beta 3\, Y_t + a_i + \varepsilon_{i,t} \qquad \text{Equation 3}$$

Where Y_{it}= is the outcome at location *i* in year *t*, X_{it} represents a vector of observed covariates, C_{it} is a binary variable indicating treatment status, Y_t are time fixed effects, and a_i and $\varepsilon_{i,t}$ represent time-invariant and time-variant unobservable factors respectively. β2 is the coefficient of interest, representing the effect of treatment on the outcome, controlling for observed covariates and time-invariant unobserved bias.

Fixed effects panel regressions are typically used in cases where there are multiple treated units with different treatment timings and where unobserved time-invariant bias is present or suspected (Ferraro and Miranda, 2017; Cunningham, 2021). However, the validity of using fixed effects panel regressions in these contexts has been questioned. Fixed effects panel regressions can involve inappropriate pair-wise comparisons (i.e., use of early treated groups as controls for late-treated groups) that can potentially bias the estimated treatment effect, particularly when the effects of the intervention vary over time (Goodman-Bacon, 2021; de Chaisemartin and D'Haultfoeuille, 2023). However, Ferraro and Simorangkir (2020) show that in a real-world conservation application the weights assigned to these comparisons are small (as they form a minority of comparisons) and therefore unlikely to bias results.

Synthetic control

However, what about cases where an intervention is applied to a single large, aggregate unit (e.g., a country or state), or a unit for which appropriate controls might be few and far between (e.g., a rare type of ecosystem; Fick *et al.*, 2021)? In these cases, where traditional statistical matching may struggle to find appropriate matches, an alternative approach, based on construction of a synthetic control may be a better option (Abadie and Gardeazabal, 2003; Abadie, Diamond and Hainmueller, 2010). The synthetic control is a weighted combination of real-world control units, weighted such that pre-intervention characteristics of the synthetic control reproduce, as closely as possible, those in the treated unit (Abadie and Gardeazabal, 2003; Sills *et al.*, 2015; Abadie, 2021; Fick *et al.*, 2021). The characteristics include observed covariates hypothesised to influence selection to treatment and the outcome of interest, and average pre-intervention outcomes (Abadie, Diamond and Hainmueller, 2010). Including pre-intervention outcomes helps to control for the influence of unobserved factors, as a similar pattern of pre-intervention outcomes indicates the synthetic control has a similar response to unobserved factors (i.e. external political shocks) as the treated unit (Sills *et al.*, 2015). Covariates are weighted based on their importance as predictors of outcomes. This ensures that control units are weighted to maximise similarity in the most important predictors, and, as a result, produces the synthetic control that most closely reproduces treated outcomes in the pre-intervention period (Abadie, 2021). Weighting of control units is designed to be sparse, meaning only a few control units are selected to comprise the synthetic control, allowing the researcher to easily sense-check the selection (Abadie, 2021).

Constructing a synthetic control with similar characteristics and pre-intervention outcomes to the treated unit helps control for the influence of observed and unobserved confounders, meaning post-intervention outcomes in the synthetic control represent an appropriate counterfactual for the treated unit. Overall, the synthetic control is a transparent, data-driven approach to the selection of a comparison unit, which is particularly useful when appropriate controls are limited.

Relevant applications of counterfactual methods

These counterfactual methods are not new. They have been used to evaluate the impacts of programmes in economics, policy, and health for decades (e.g., Card, 1990; Card and Krueger, 1994; Abadie and Gardeazabal, 2003; Wolfers, 2006). In conservation science counterfactual methods were first used in the late 2000s (e.g. Andam *et al.*, 2008) and have become increasingly popular since then (Blackman, 2013). This has enabled and encouraged more (and better) evaluations of policies, programmes, and interventions, helping to improve understanding of what works, and what doesn't, in conservation (e.g., Jones and Lewis, 2015; Carlson *et al.*, 2018; Geldmann *et al.*, 2019; Oldekop *et al.*, 2019; West *et al.*, 2020).

Counterfactual methods can also be applied to mining, to help us better understand the direct and indirect impacts of mining relative to alternative land uses. However, to date, there are only two known studies using these methods to evaluate the environmental impacts of mining, and both were focussed on industrial mining (Sonter *et al.*, 2017; Morley *et al.*, 2022). Morely *et al.,* (2022) use statistical matching and regression models to evaluate the direct and indirect impacts of industrial mining on deforestation in Zambia. They find that deforestation inside, and within a 25km radius, of 22 active mining leases was not significantly higher than counterfactual deforestation, estimated using matched control pixels from exploration mining leases. In contrast, in the Iron

Quadrangle in Brazil Sonter *et al.,* (2017) show that mining significantly increases deforestation up to 70km around mining leases, using matched control pixels located further than 100km from a mine to estimate counterfactual outcomes. Furthermore, they find that indirect deforestation in the buffer zone exceeds deforestation within the mining lease by up to 12 times.

Why Madagascar is a good case study to investigate the environmental trade-offs of mining

Madagascar is extremely rich in both minerals and biodiversity (Pezzotta, 2001; Yager, 2019; Richard, 2022). Yet despite this wealth of natural resources Madagascar is one of the poorest countries in the world. Over 45% of the population live in severe multidimensional poverty (OPHI, 2022), 42% of children under five are chronically malnourished (WFP, 2023), and on average people are poorer now than they were in 1960 (average per capita income has declined by 45%; Figure 5; World Bank, 2023).

Mining has expanded over the last 30 years and makes important contributions to economic development and poverty alleviation (Sarrasin, 2006; Tilghman, Baker and Deleon, 2007; Klein, 2020). As such, the government is committed to expanding the industrial mining sector to help drive national development (Klein, 2020; EDBM, 2021).

However, overlap between the island's mineral and biological wealth means that the positive economic contributions of mining may entail substantial trade-offs with the island's globally important and threatened biodiversity (Cardiff and Andriamanalina, 2009; Jones *et al.*, 2019b). Conversely, efforts to conserve species by forgoing or preventing mining, could bring considerable economic opportunity costs and may provoke land use conflicts (Canavésio, 2010; Cook and Healy, 2012; Vuola, 2022). As such, Madagascar presents an important and timely case-study with which to evaluate the environmental trade-offs of mining.

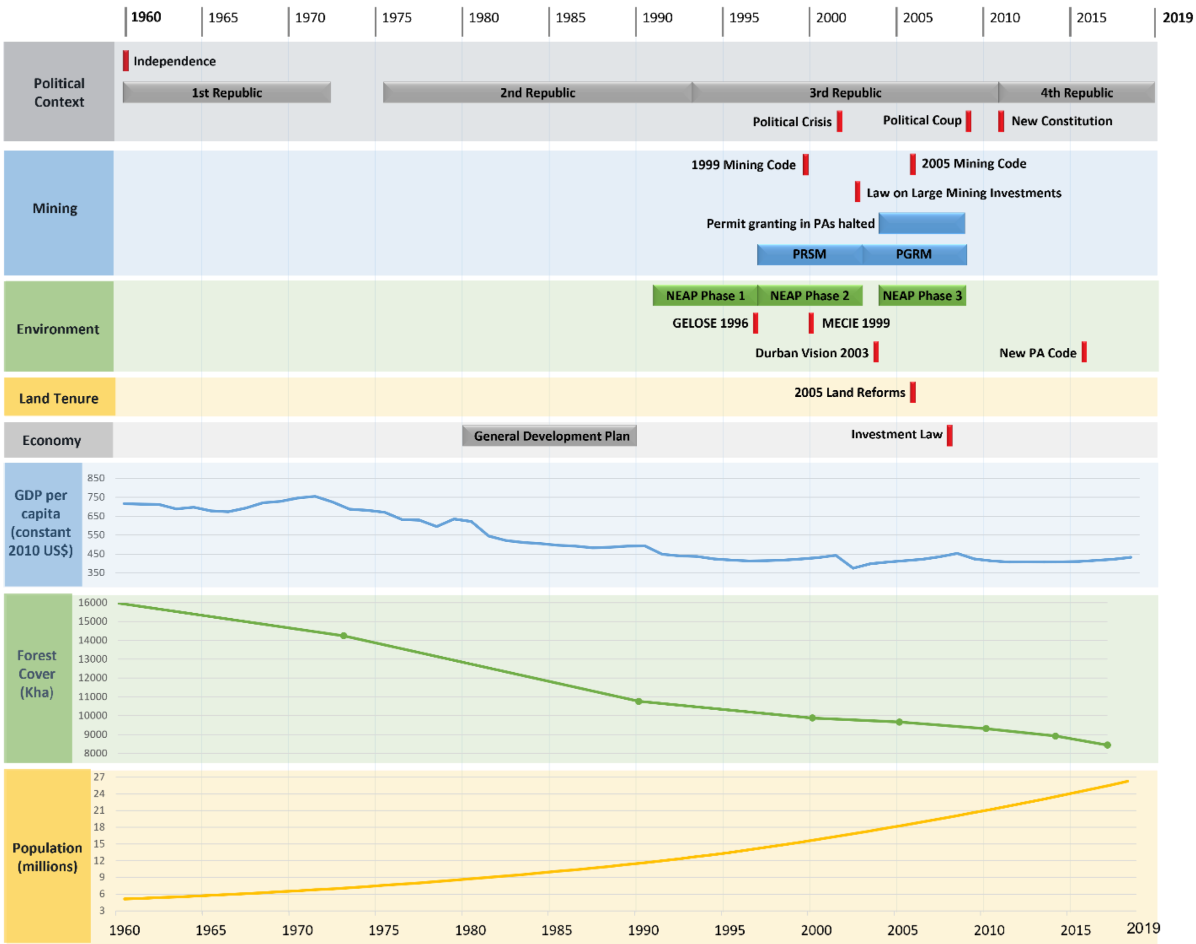

Figure 5: Timeline of key events affecting mining and environmental governance in Madagascar. PRSM and PGRM refer to the World Bank's Mining Sector Reform Project and subsequent Mineral Resources Governance Project. NEAP = National Environmental Action Plans. GELOSE = legislation allowing community-based management of renewable resources. The MECIE decree outlines the social and environmental obligations for large-scale investments. The Investment Law permits foreign-owned companies registered in Madagascar to buy land for the first time (Ferguson *et al.*, 2014; Huff, 2016). The Durban Vision refers to a 2003 commitment to treble the extent of Protected Areas. Data sources: Vieilledent et al (2018) for forest cover, and World Bank (2021) for population and GDP per capita.

Biodiversity

Madagascar's isolation and varied geography produced exceptional biodiversity, much of which is found nowhere elsewhere on earth (Antonelli *et al.*, 2022; Goodman, 2022; Richard, 2022). An estimated 82% of vascular plants, 56% of birds, 95% of mammals, 98% of reptiles, and all native amphibians are endemic to the island (Antonelli *et al.*, 2022). However, much of this globally important biodiversity is under threat, principally from habitat loss and overexploitation (e.g., hunting, timber and resource harvesting; Jones *et al.*, 2019a; Ralimanana *et al.*, 2022). The majority of the island's biodiversity depends on forests (Goodman, 2022), but forests are rapidly disappearing (Ralimanana *et al.*, 2022). Between 1953 and 2014, Madagascar lost an estimated 44% of its natural forests (Vieilledent *et al.*, 2018). By 2017, there were an estimated 8,445,890 ha of forest remaining, covering 14% of the island. These remaining forests have become increasingly fragmented; nearly half of Madagascar's forest area was within 100m of a forest edge in 2014 (Vieilledent *et al.*, 2018). While colonial-era logging and cash cropping played a large role in historical forest loss (Jarosz, 1993), contemporary deforestation is driven predominantly by subsistence agriculture (although in some places, cash cropping also plays an important role; Vieilledent *et al.*, 2020; Ralimanana *et al.*, 2022; Jones, Rakotonarivo and Razafimanahaka, 2022). Biodiversity is also threatened by invasive species, such as the toxic Asian Common Toad (*Duttaphrynus melanostictus*) which was unintentionally released from a shipping container (linked to the Ambatovy mine) to the port of Toamasina (McClelland *et al.*, 2015; Marshall *et al.*, 2018). Habitat loss, fragmentation, overexploitation, and invasive species have reduced native species abundance and distributions, increasing the risk of extinction (Ralimanana *et al.*, 2022).

The combination of exceptional but highly threatened biodiversity has resulted in Madagascar being considered one of the world's hottest biodiversity hotspots and a major focus of global conservation efforts (Myers *et al.*, 2000; Brooks *et al.*, 2006). These efforts have mostly concentrated on conserving the remaining forest habitat, predominantly through the establishment of Protected Areas (although there are also programmes aimed at non-forest habitats and individual species; Gardner *et al.*, 2018; Jones, Rakotonarivo and Razafimanahaka, 2022). There are currently 114 Protected Areas in Madagascar (Rebioma, 2017) with varying degrees of management and effectiveness (Eklund *et al.*, 2019), from so-called 'orphan parks' which effectively only exist on paper and lack a management authority (Goodman, 2022), to highly-visited and relatively well-resourced Protected Areas (Goodman, Raherilalao and Wohlauser, 2018).

Minerals and metals

Madagascar hosts a variety of economic mineral deposits, widely distributed across the island. This includes the metals gold, chromite, aluminium (in bauxite), copper, nickel, cobalt, titanium, Rare Earth Elements and iron ore; industrial minerals such as graphite, zircon, ilmenite, and mica; precious and semi-precious gemstones including ruby, sapphire, emerald, garnet and tourmaline; and the decorative stones labradorite and quartz (Pezzotta, 2001; Cook and Healy, 2012; Yager, 2019).

This mineral wealth is a product of Madagascar's long and dynamic geological history, tied to the formation and subsequent break-up of the supercontinent of Gondwana (Supplementary Methods 2). Episodes of continental convergence and rifting led to intense reworking of the crust and produced the volcanism and metamorphic conditions necessary for the formation of many of Madagascar's mineral deposits (Pezzotta, 2001; Rakotondrazafy *et al.*, 2008).

Mining in Madagascar

Despite this wealth of resources, the formal mining sector in Madagascar is small and relatively under-developed, compared to its potential and its neighbours in Sub-saharan Africa (World Bank, 2010; Faure, Rakotomalala and Pelon, 2015). There are only three large-scale industrial mines operating in the country; 1) the state-owned and operated Kroama chromite mine in Betsiboka (Ernst Young, 2019; Yager, 2019); 2) the Ambatovy nickel and cobalt mine near Moramanga, which is a joint venture between the Japanese Sumitomo Corporation and the South Korean state-owned mining corporation KOMIR (Ambatovy, 2022) and; 3) the QMM ilmenite mine near Fort Dauphin of which 80% is owned by Rio Tinto and the other 20% by the Malagasy government (Rio Tinto, 2020). There are several controversial large projects in development, including the Base Toliara ilmenite mine near Toliara in south-west Madagascar which has been halted several times due to local opposition (Huff, 2016; Vyawahare, 2019), and the Tantalus Rare Earths mine in north-west Madagascar (Ernst Young, 2019; Yager, 2019; Caramel, 2023). There are also several medium-scale operations exploiting graphite and construction materials (Ernst Young, 2019), and a range of smaller mines for which there is little publicly-available data. Currently, the formal mining sector contributes approximately 4.2% of Madagascar's GDP (World Bank Group, 2019).

In contrast, the ASM sector in Madagascar is extensive and growing. An estimated 500,000 people are directly involved in ASM and a further 2.5 million more work in related industries (World Bank, 2010; Hilson, 2016). Most artisanal and small-scale miners in Madagascar target gold and high-value ruby and sapphire deposits (Cartier, 2009). A smaller number mine for emerald, tourmaline, garnet, quartz, mica, and labradorite (Cook and Healy, 2012). Artisanal gold mining has been ongoing for centuries in Madagascar (Klein, 2020), but prior to the 1990s artisanal gem mining was very limited (Cartier, 2009; Cook and Healy, 2012). That all changed with the discovery of a high-value sapphire deposit at Andranondambo in the far south of the island in 1992 (Schwarz, Petsch and Kanis, 1996; Tilghman, Baker and Deleon, 2007). Word of this discovery quickly spread, attracting a wave of in-migration from across the island to form Madagascar's first gem rush. At its peak an estimated 10,000- 25,000 people were mining at Andranondambo (Canavesio and Pardieu, 2019). Subsequent discoveries of rubies and sapphires followed; in the far North next to Ankarana Special Reserve in 1996, and in the hamlet of Ilakaka in the south in 1998. The Ilakaka discovery triggered the largest, most transformative, and longest-lived mining rush to date in Madagascar (Canavesio, 2009; Canavesio and Pardieu, 2019). This was sustained by the size of the deposit, which extends for hundreds of square kilometres within the alluvial sediments of the Isalo group (Rakotondrazafy *et al.*, 2008; Pardieu, 2013; Giuliani *et al.*, 2020). As deposits close to the town became depleted, new discoveries further away maintained the supply of sapphires through the town and the gem-based economy (Pardieu, 2013). Twenty-five years later Ilakaka remains the hub of the Malagasy gem trade (Figure 6; Lawson and Lahiri-Dutt, 2020). Discoveries and subsequent rushes continued for the next two decades, driven by the network of experienced, mobile miners formed during the early rushes (Canavesio and Pardieu, 2019). Notable examples are the rushes in the eastern rainforests at Moramanga (2005), Didy (2012), and Bemainty (2016), and the demantoid garnet rush in the mangroves of north-west Madagascar in 2009 (Pezzotta, Adamo and Diella, 2011; Cook and Healy, 2012; Canavesio and Pardieu, 2019).

Figure 6: A sapphire trader inspecting a stone in Ilakaka, Madagascar. Photo credit: Author.

In Madagascar, ASM employs very basic methods and is reliant on manual labour. Miners use picks, shovels, or spikes to dig pits and shafts, and hand-made sieves to filter extracted sediment (Figure 1: Tilghman, Baker and Deleon, 2007). Some miners may have access to water pumps (provided by a sponsor or rented) to remove water from pits or wash extracted sediment (Cook and Healy, 2012; Perkins, 2017). Very few operations use heavy machinery. Mercury or cyanide is not currently widely used in gold mining (Canavésio, 2010; Klein, 2022b). Miners work independently, in family or kinship groups, or in groups organised by a sponsor or mine-owner under various arrangements (Cartier, 2009; Lawson, 2018). The latter may provide miners with food and equipment in return for first refusal on finds, pay miners a daily wage, or a percentage of earnings, although such arrangements can be exploitative (Tilghman, Baker and Deleon, 2007; Klein, 2020). Most ASM is illegal, as miners do not possess the permits required to mine legally. As such, most trade passes through informal channels until the point of export, severely limiting the amount of revenue captured by the state (Cook and Healy, 2012).

As demonstrated elsewhere, ASM in Madagascar is predominantly poverty-driven. People participate in artisanal mining, despite the risks and dangerous working conditions, to try to alleviate daily poverty and hardship, and because there are few other options (Tilghman, Baker and Deleon, 2007; Cartier, 2009; Lawson, 2018; Stoudmann *et al.*, 2021). Declining returns from agriculture, particularly in Southern Madagascar hard-hit by insecurity and recurrent drought, have pushed many into ASM (Lawson and Lahiri-Dutt, 2020; Stoudmann *et al.*, 2021). Yet income from ASM is highly variable, depending on the productivity of the deposit (which depends on the geological characteristics) and luck (Cartier, 2009). Gemstone mining is risky and more transient, but can potentially yield life-changing sums, while gold mining tends to be a more reliable, long-term activity (Cartier, 2009; Klein, 2000). Both are often performed alongside agriculture as a complementary income-generating activity (Stoudmann *et al.*, 2021). In Betsiaka, a survey by the German Development Agency (GIZ) found that alluvial gold miners can earn $2.30 - $7.87 per day, higher than alternative rural livelihoods (Klein, 2020). For artisanal gem mining income can be highly inconsistent (Lawson and Lahiri-Dutt, 2020; Stoudmann *et al.*, 2021), as emphasized by Alvine, a female sapphire miner working near Ilakaka: *"Sometimes I find something, sometimes I don't. Some days I earn 10–20,000 Ar a day* [$2.23 - $4.50] *but often I earn nothing. I use everything I earn on my daily expenses"* (Lawson, 2018, p.180).

Cartier (2009) describe how unlucky ruby and sapphire miners can leave the sector poorer than when they arrived or, unable to afford to return home, become trapped at the mine site and further impoverished. 61% of miners interviewed in two villages in north-east Madagascar reported their well-being had declined since participating in ASM, citing the physical health risks, inconsistency of income, and vulnerability to exploitation (Stoudmann *et al.*, 2021). Yet, at the other end of the spectrum, life-changing discoveries do happen, and continue to fuel dreams of riches. Lawson (2018) and Stoudmann et al (2021) tell of respondents who have found valuable stones which have enabled them to buy livestock, housing, solar panels, pay for their children's education, or invest in business development. However, the most substantial livelihood contributions from artisanal gem mining comes from the small and low-quality stones which comprise the majority of finds. These are the stones that enable daily survival, prevent hunger, and keeping miners going until they find a larger-stone, or not (Cartier, 2009; Walsh, 2012).

The development of the policy framework governing mining in Madagascar

Mining in Madagascar is principally governed by the Mining Code. Like in many other African nations, the legislative and policy framework governing mining is a product of the Structural Adjustment-induced economic and regulatory reforms of the 1980s, 1990s, and early 2000s (Campbell, 2003; Sarrasin, 2006).

In the 1980s the Malagasy economy was on the brink of collapse after a decade of economic mismanagement by socialist leader Didier Ratsiraka (Gow, 1997; Sarrasin, 2006). Faced with escalating poverty, unemployment, and social unrest Ratsiraka requested emergency loans from the World Bank. In return Madagascar was forced to implement reforms to promote the World Bank's vision for development (outlined in the General Development Plan; Figure 6), which was to be achieved through export-oriented capitalism and foreign investment (Campbell, 2003; Sarrasin, 2006; Huff, 2016). Throughout the 1980s and 1990s, compelled by the conditions attached to loans and the need for continued financial assistance, Madagascar, like many other African nations, liberalised and deregulated the economy, privatised state-run enterprises, decentralised, and redefined the role of the state (Campbell, 2003; Huff, 2016). Land reforms focused on privatization, promoting the legal titling of customary land, and eliminating state ownership of untitled land (Andrianirina Ratsialonana *et al.*, 2011; Ferguson *et al.*, 2014). These measures aimed to facilitate land markets and access for private businesses, including foreign investors (Sarrasin, 2006; Huff, 2016). Reforms also encompassed environmental governance, leading to the National Environmental Action Plans of the 1990s (Pollini, 2011).

The World Bank recognised that Madagascar's mineral wealth could help fuel development, and expansion and formalisation of the mining sector became a key aspect of the development strategy (Sarassin, 2009). From 1998 the World Bank's Mining Sector Reform Project (PRSM) aimed to create a favourable legal and fiscal environment to attract foreign investment and facilitate the development of industrial mining (Sarrasin, 2006; Ferguson *et al.*, 2014). This involved a reduction in the role of the state from primary operator (under Ratsiraka's socialist regime all industry was nationalised), to regulator and facilitator, although previous structural adjustment reforms had weakened the capacity of the state to effectively regulate mining (Sarrasin, 2006, 2009; Cook and Healy, 2012; Ferguson *et al.*, 2014). The PRSM culminated in the formulation of a new Mining Code, which was adopted in 1999. The new legislation was highly favourable to foreign investors; it streamlined regulations, limited fiscal contributions to 2% of the value earnt at first sale and permitted investors to transfer funds out of the country and hold in foreign accounts (*Code Minier*, 1999; Sarrasin, 2006; Huff, 2016). It also provided the legal basis for the creation of the Bureau du Cadastre Minier de Madagascar (BCMM), a government agency responsible for the issuance and monitoring of mining permits (Huff, 2016).

The Mining Code has undergone two reforms since then. First, in 2005 and again in 2023 (the updated version was promulgated on 3rd August 2023 and will likely be implemented next year [2024]; *Code Minier*, 2023). The updated Mining Code outlines legal requirements for mining (including permitting, land rights, environmental and social requirements), taxation, penalties for non-compliance, and institutional responsibilities. Below, I briefly outline some of the main prescriptions governing mining in this Code (*Code Minier*, 2023).

All mineral resources in Madagascar are the property of the State. Permits confer on holders (individuals, co-operatives, or companies) the right to legally mine these resources within a certain area, subject to the conditions outlined in the Code and payment of a fee (miners must also obtain approval from holders of surface land rights). Madagascar's land area is divided into a grid of mining squares each measuring 625m^2 (this was reduced from 2.5km in the 1999 Mining Code to better suit to the scale of ASM), which forms the geographical basis for permit allocation. There are three mining permits available which are issued by the BCMM and approved by the Ministry of Mines:

- *Permis de Recherche* (Research Permit) – These are valid for five years, renewable twice, and grants holders the exclusive right to prospecting and research for the named minerals within the permit area. A research permit can cover up to 5,000km^2.

- *Permis d'Exploitation* (Exploitation Permit) – These are valid for 25 years, renewable once for an extra 15 years, and give holders the exclusive right to mine within the permit area, which covers up to 500km^2.
- *Permis Réservé aux Exploitants Artisanaux* (Permit reserved for artisanal miners; PREA) – These permits are only available to Malagasy nationals who mine using traditional techniques (i.e. manual labour and only light machinery). Permits are valid for eight years, renewable twice for four years each time, and give holders exclusive rights to mine in the permit area, covering a minimum area of 625m^2 up to 50km^2.

All permit holders have the right to use wood and water within the permit area and construct infrastructure.

There are also two other authorisations available to artisanal miners. Co-operatives of artisanal miners can request an *Authorisation for Artisanal Mining* (AMEA). This grants members the exclusive right to mine within a corridor designated by the Ministry of Mines, or within the mining square of another permit holder in agreement with the permit holder. Authorisation is valid for six months, renewable once, and is restricted to four mining squares. This also applies to artisanal gold miners exploiting primary deposits.

Specific regulations apply to artisanal gold miners exploiting secondary alluvial deposits. This is permitted freely within active riverbeds, and within specific corridors (designated by the Commune authorities) for miners is possession of a gold card. Gold cards are restricted to Malagasy nationals. They are valid for one year and confer the right to mine within gold mining corridors anywhere in the commune. Mining without a permit, authorisation, or gold card is illegal.

The Mining Code incorporates regulations on environmental and social protection. Mining is illegal within Protected Areas (unless the government decides to degazette the Protected Area) and is considered a serious offense. Anyone caught mining in Protected Areas can be given a prison sentence of up to five years or fined up to 50 million Ariary (currently ~ £9000). To obtain a mining permit applicants must submit an Environmental

Impact Assessment (for Exploitation Permits) or an Environmental Engagement Plan (for Research Permits, PREA and AMEA) which is approved by the Ministry of Mines. For projects over 1,000 ha, these fall under the remit of the MECIE decree (see below; Ernst and Young, 2019). Permit holders must take necessary measures to minimise and restore environmental damage from mining activities, including reforesting cleared land. The use of mercury, cyanide and other chemicals in artisanal gold mining is prohibited.

The Mining Code also regulates trade and taxation. A *laissez-passer* permit is required to transport a mined product outside the permit area, sell, and, crucially, export. Gold traders require a Collectors Card to buy and sell gold within the Commune, or *Comptoir d'Or* status to operate nationally. All sales must be formally registered. A 5% tax is levied on product sales, of which 2% is shared between regional and commune authorities and 3% goes to the central government budget. An additional one-off contribution to a Social and Community Investment Fund is due upon issuance of the permit.

The changes within the new Mining Code approved in 2023 strengthen the position of the Malagasy state, increase the rents captured from mining, and provide more suitable permitting and trade regulations for ASM, reducing (but not eliminating) barriers to obtaining a permit to mine legally (L'Express de Madagascar, 2023; NEWSMADA, 2023). Key changes compared to the previous version include a 50% reduction in the size and duration of Research and Exploitation permits, an increase in the tax rate from 2% to 5%, mandated contributions to the Social and Community Investment Fund, the creation of the *Authorisation for Artisanal Mining*, and provisions allowing the designation of artisanal mining corridors (*Code Minier*, 1999; *Code Minier*, 2005; *Code Minier*, 2023).

Other relevant legislation includes the Law on Large-scale Investments, and the MECIE degree. The Law on Large-Scale Mining Investments (Law No. 2001-031 modified by Law No. 2005-022) was also a product of the PRSM and provides an even more favourable tax, customs, and legal regime for investments over USD$25 million (Cook and Healy, 2012; Huff, 2016). It also provides a stability clause and exempts eligible investments from increases in the corporate tax rate (Ernst Young, 2019). So far, only Ambatovy has exceeded the investment threshold to qualify for this special regime (Ernst Young, 2019;

which should mean it is exempt from the increased tax rate outlined in the updated mining code).

The *Mise en Compatibilité des Investissements avec l'Environnement* (MECIE) decree (No. 99-954 of 15 December 1999 amended by Decree No. 2004-167 of 3 February 2004) outlines the environmental and social obligations for large projects over 1,000 hectares, including mines. Project proponents are required to conduct a community consultation, complete a detailed Environmental and Social Impact Assessment, and submit a management plan outlining how the negative social and environmental impacts of the project will be reduced and mitigated (Andrianirina Ratsialonana *et al.*, 2011; Huff, 2016). These documents are assessed by l'Office Nationale pour l'Environnement (ONE) and if approved, an environmental permit is granted (Sarrasin, 2006).

Previous research on mining in Madagascar

Most of the relevant research on mining in Madagascar has come from the fields of anthropology, political ecology, geography, and gemmology. For industrial mining, studies have interrogated discourses of sustainability used by mining companies (Seagle, 2012), revealed negative environmental and social impacts of the QMM mine which contradict these discourses (Huff, 2016; Huff and Orengo, 2020), and explored biodiversity offsetting as means for mitigating the environmental impacts of mining (Virah-Sawmy, Ebeling and Taplin, 2014; Bidaud, Hrabanski and Meral, 2015; Bidaud *et al.*, 2017; Bidaud, Schreckenberg and Jones, 2018). For ASM, research has explored the development of the sector and mining rushes (Canavesio, 2009; Canavesio and Pardieu, 2019; Klein, 2020); practices, cultures, and social impacts (Walsh, 2003, 2004, 2012; Tilghman, Baker and Deleon, 2007; Canavesio, 2009, 2014; Cartier, 2009); the experiences of women (Canavésio, 2011; Lawson, 2018; Lawson and Lahiri-Dutt, 2020); and examined informal systems of governance (Klein, 2022b, 2022a; Vuola, 2022). Several studies describe the environmental impacts of ASM at case study sites based on site visits and interviews (Tilghman, Baker and Deleon, 2007; Canavesio, 2009; Baker-Médard, 2012).

The most comprehensive research into the environmental impacts of ASM in Madagascar was conducted by Cook and Healy (2012). This study was part of a global project aiming to assess the impacts of ASM in Protected Areas and Critical Ecosystems and identify feasible, sustainable solutions to mitigating conflict between ASM and conservation (Villegas *et al.,* 2012). Based on site visits and key informant interviews, Cook and Healy (2012) describe the conditions and positive and negative impacts of ASM at 11 sites across Madagascar. They find that the environmental impacts varied between sites depending on the number of miners, the location and size of the deposit, mining methods used, and the ecological characteristics of the site (i.e., land cover type, species endemism). ASM has had a concerning environmental impact at several sites; causing substantial deforestation within Ankarana Special Reserve in Northern Madagascar (where mining within caves has also threatened the fragile cave ecosystems and endemic biodiversity), Zombitse-Vohibasia National Park in the south-west, and Ranomafana National Park in the east, where artisanal gold mining has destroyed rare wetland *pandanus* forest and increased conflict and insecurity (Cook and Healy, 2012; Cabeza *et al.*, 2019). However, this evidence is purely descriptive: Cook and Healy do not attempt to quantify mining-related deforestation. To date, there is only one study (the World Bank Forest Smart Mining study outlined above) which quantifies deforestation around ASM sites in Madagascar (World Bank, 2019). But this does not attempt to isolate the causal impact of ASM. Prior to this thesis there were no robust evaluations of the environmental impacts of ASM in Madagascar using counterfactual methods.

This constitutes a concerning evidence gap, given the extensiveness and economic importance of ASM, continuing new discoveries, and increasing encroachment of ASM into Protected Areas. It also means that policy responses to the challenges of ASM are not informed by robust evidence, and therefore may not be appropriate or effective. For industrial mining, quantitative evidence on environmental impacts is mostly restricted to the Environmental Impact Assessments of the Ambatovy and QMM mines. Both these companies have taken steps to mitigate their substantial environmental impacts following international standards (Business and Biodiversity Offsets

Programme (BBOP), 2012; International Finance Corporation, 2012), but there is no robust evidence on whether these measures have worked.

The focus of this thesis

In this thesis, I aim to tackle these knowledge gaps, focussing on the impacts of mining on the forests and biodiversity of Madagascar. Using spatial data and counterfactual methods for impact evaluation, I explore the real and potential impact of mining on the island's forests and biodiversity, and the effectiveness of measures to mitigate that impact. I try to take a balanced and pragmatic view throughout, considering the environmental impacts as trade-offs relative to the positive contributions, and situating the impacts of mining in the context of alternative land uses. Building on this evidence and the literature, I explore challenges and opportunities for reconciling mining and biodiversity conservation in Madagascar, in a way which minimises the environmental trade-offs and enhances the development benefits. The remainder of this thesis is structured as follows:

- In Chapter 2 I evaluate the effectiveness of a key policy mechanism for mitigating the environmental impact of industrial mining – biodiversity offsetting. Using counterfactual methods I evaluate whether the Ambatovy mine has managed to compensate for forest cleared at the mine site by effectively conserving an equivalent amount of forest within its four biodiversity offsets. These results have important implications beyond the case study for global efforts to mitigate the impacts of mining.
- In Chapter 3 I switch focus to ASM to explore the extent and location of potential trade-offs between artisanal gem mining and biodiversity conservation in Madagascar. To do so I use geological data to identify and map the potential distribution of primary ruby, sapphire and emerald deposits across the island and quantify the spatial overlap with areas of conservation significance (defined using four measures). This reveals areas of potential future conflict between

mining and conservation but also opportunities for policies to promote and support ASM while minimising impacts to biodiversity.

- In Chapter 4 I investigate the environmental outcomes when ASM occurs at scale within a protected forest, focussing on the 2016 sapphire rush at Bemainty in eastern Madagascar. Combining the synthetic control method for impact evaluation and field data from interviews and lemur surveys I evaluate the impact of the mining rush on the surrounding forests.
- In the final chapter I examine dominant narratives around mining and their influence on mainstream policy responses to the challenges of mining. I consider how my research challenges or reinforces these discourses, and consider alternative, pragmatic solutions to minimising the environmental trade-offs of mining.

Chapter 2: On track to achieve No Net Loss of forest at Madagascar's biggest mine

Photo: The author with members of Ambatovy's biodiversity and ecosystem services team. Photo taken on 16th June 2023 after giving a presentation and discussing the results of this study.

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Abstract

Meeting the Sustainable Development Goals requires reconciling development with biodiversity conservation. Governments and lenders increasingly call for major industrial developments to offset unavoidable biodiversity loss, but there are few robust evaluations of whether offset interventions ensure No Net Loss (NNL) of biodiversity. We focus on the biodiversity offsets associated with the high-profile Ambatovy mine in Madagascar and evaluate their effectiveness at delivering NNL of forest. As part of their efforts to mitigate biodiversity loss, Ambatovy compensate for forest clearance at the mine site by slowing deforestation driven by small-scale agriculture elsewhere. Using a range of methods, including extensive robustness checks exploring 116 alternative model specifications, we show that the offsets are on track to avert as much deforestation as was caused by the mine. This encouraging result shows that biodiversity offsetting can contribute towards mitigating environmental damage from a major industrial development, even within a weak state, but there remain important caveats with broad application. Our approach could serve as a template to facilitate other evaluations and so build a stronger evidence-base of the effectiveness of No Net Loss interventions.

Introduction

The UN Sustainable Development Goals underline the importance of economic growth and infrastructure development in alleviating poverty, while at the same time emphasising that halting biodiversity loss is vital for global prosperity (Blicharska et al., 2019; Thacker et al., 2019). Policies aimed at delivering No Net Loss (NNL) of biodiversity, in theory, allow development to proceed whilst avoiding environmental damage (Bull et al., 2013; Maron et al., 2018). NNL depends on implementation of the mitigation hierarchy: damage to biodiversity resulting from development must first be avoided, minimised, and restored (McKenney and Kiesecker, 2010), and any residual biodiversity loss offset through equivalent gains elsewhere (Quétier and Lavorel, 2011). One hundred and one countries either mandate some form of biodiversity compensation or support voluntary measures (zu Ermgassen et al., 2019a). In countries with less established environmental governance, lender requirements, such as the International Finance Corporation performance standards, are important drivers of NNL commitments (IFC, 2012; Bidaud, Schreckenberg and Jones, 2018). Over 12,000 biodiversity offsets exist worldwide (Bull and Strange, 2018), yet evaluations of their effectiveness are rare, and most do not use robust methods (zu Ermgassen et al., 2019b).

Offsets generate gains in biodiversity by creating or restoring habitat, or protecting existing habitat which would have otherwise been lost (so called ‘averted loss’ offsets; Sonter et al., 2020b). Offsets are controversial due to questions of permanence (Bull et al., 2013), equivalence (Quétier and Lavorel, 2011), equity (Ives and Bekessy, 2015; Jones et al., 2019c), and for generating gain against a background rate of biodiversity decline (Maron et al., 2015a, 2018). However, where high-quality habitat remains but is threatened by unregulated sectors, averted loss offsets may result in the best possible biodiversity outcomes (Simmonds et al., 2019). Biodiversity is an inherently complex concept, so proxy measures are used to calculate losses and gains (Quétier and Lavorel, 2011). In forested ecosystems where the majority of species are forest-dependent, forest loss can be a useful measure.

Quantifying the biodiversity gains from averted loss offsets requires estimation of the counterfactual scenario – the loss which would have occurred without protection (Maron et al., 2015a). While the counterfactual is inherently unknowable, statistical approaches exist to approximate it and consequently evaluate the impact of interventions on outcomes such as deforestation (Carlson et al., 2018; Börner et al., 2020; West et al., 2020). Statistical matching is commonly used to estimate the counterfactual based on outcomes in matched control units, yet can be contingent on arbitrary modelling choices (Desbureaux, 2021). Recent advances which test the robustness of estimates to a range of valid, alternative matching model specifications (Desbureaux, 2021) and different regression models (Carlson et al., 2018; Ferraro and Simorangkir, 2020) can improve the quality of inference.

The Ambatovy nickel and cobalt mine (Figure 7) is one of the largest lateritic nickel mines in the world. It is located within the biodiversity-rich eastern rainforests of Madagascar which are highly threatened by deforestation, driven principally by shifting agriculture (Tabor et al., 2017; Poudyal et al., 2018a). From the outset, Ambatovy promoted itself as a world-leader in sustainable mining and committed to ensure NNL, and preferably net gain, of biodiversity (Von Hase et al., 2014; Bidaud, Hrabanski and Meral, 2015). Its offset strategy was a pilot for the influential Business and Biodiversity Offset Programme (Von Hase et al., 2014) which shaped guidelines widely used in mitigating biodiversity loss from development (Bidaud, Hrabanski and Meral, 2015; Simmonds et al., 2019). We use statistical matching and regression models to estimate the avoided deforestation achieved by Ambatovy's four biodiversity offsets and check the robustness of our results to 116 alternative matching model specifications (Figure 8). We provide encouraging evidence that this high-profile project, in one of the world's hottest biodiversity hotspots, is on track to achieve No Net Loss of forest and critically reflect on this finding in the broader context of NNL.

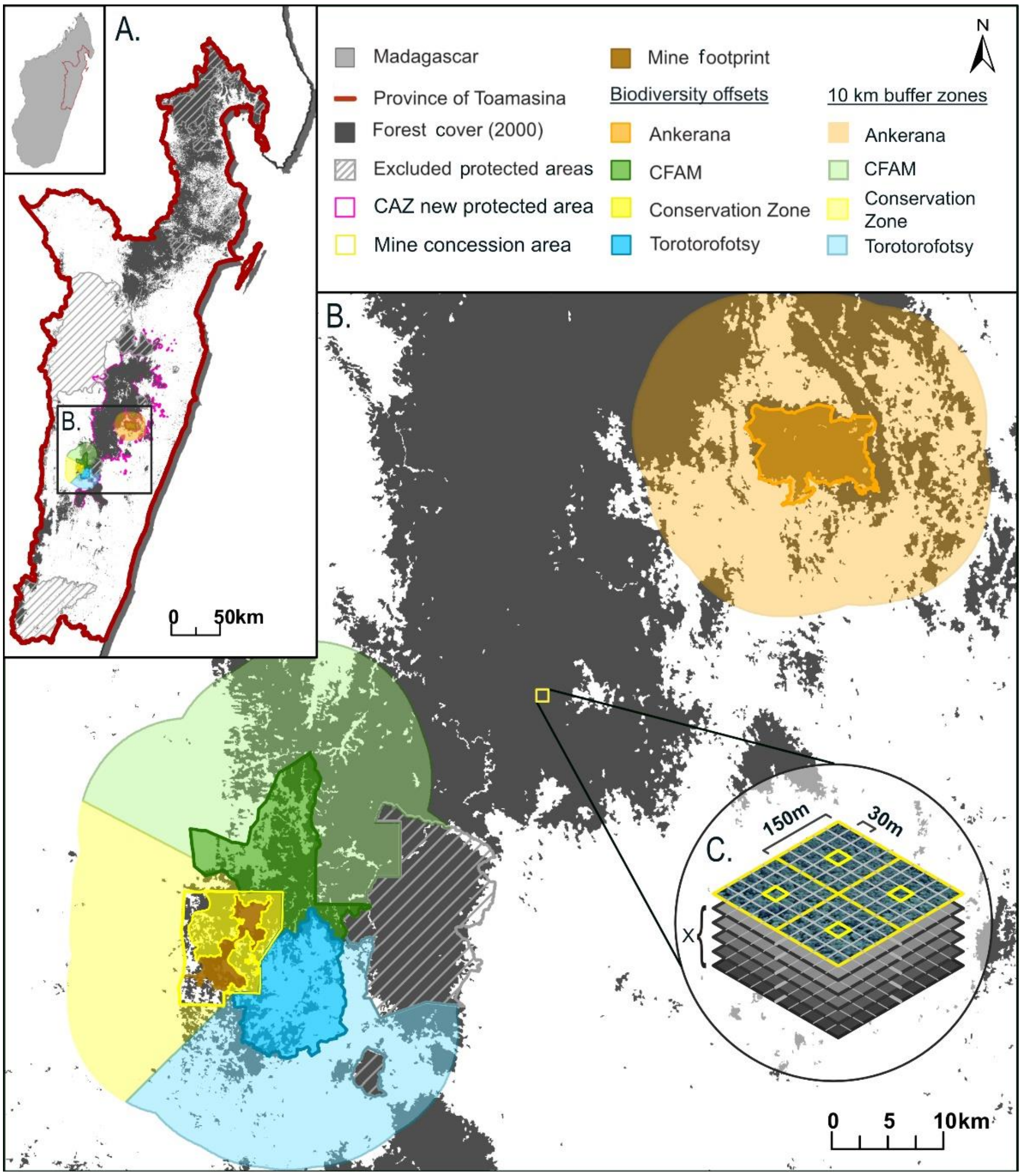

Figure 7: Study area in eastern Madagascar showing the location of Ambatovy's biodiversity offsets and our study design. A) The study area is the former province of Toamasina. Control pixels were sampled from pixels which were forested at baseline in 2000 (grey), excluding those within 10 km of a biodiversity offset, or within established protected areas (grey dashed). The Corridor Ankeniheny-Zahamena (CAZ) new protected area was included in sampling (see Methods). B) Ambatovy's four biodiversity offsets: the Conservation Zone (yellow) which is within the mine concession area, the Corridor Forestier Analamay-Mantadia (CFAM; green), Torotorofotsy (blue), and Ankerana (orange). The 10 km buffer zone (which excludes established protected areas) around each offset is shown in lighter shades and was used to explore deforestation leakage. C) Our grid-based sampling strategy (see Methods). The top layer illustrates the selection of our sub-sample of pixels. Data layers labelled x represent the outcome variable and covariates; all data used in this study are publicly available (Supplementary Table 1.4).

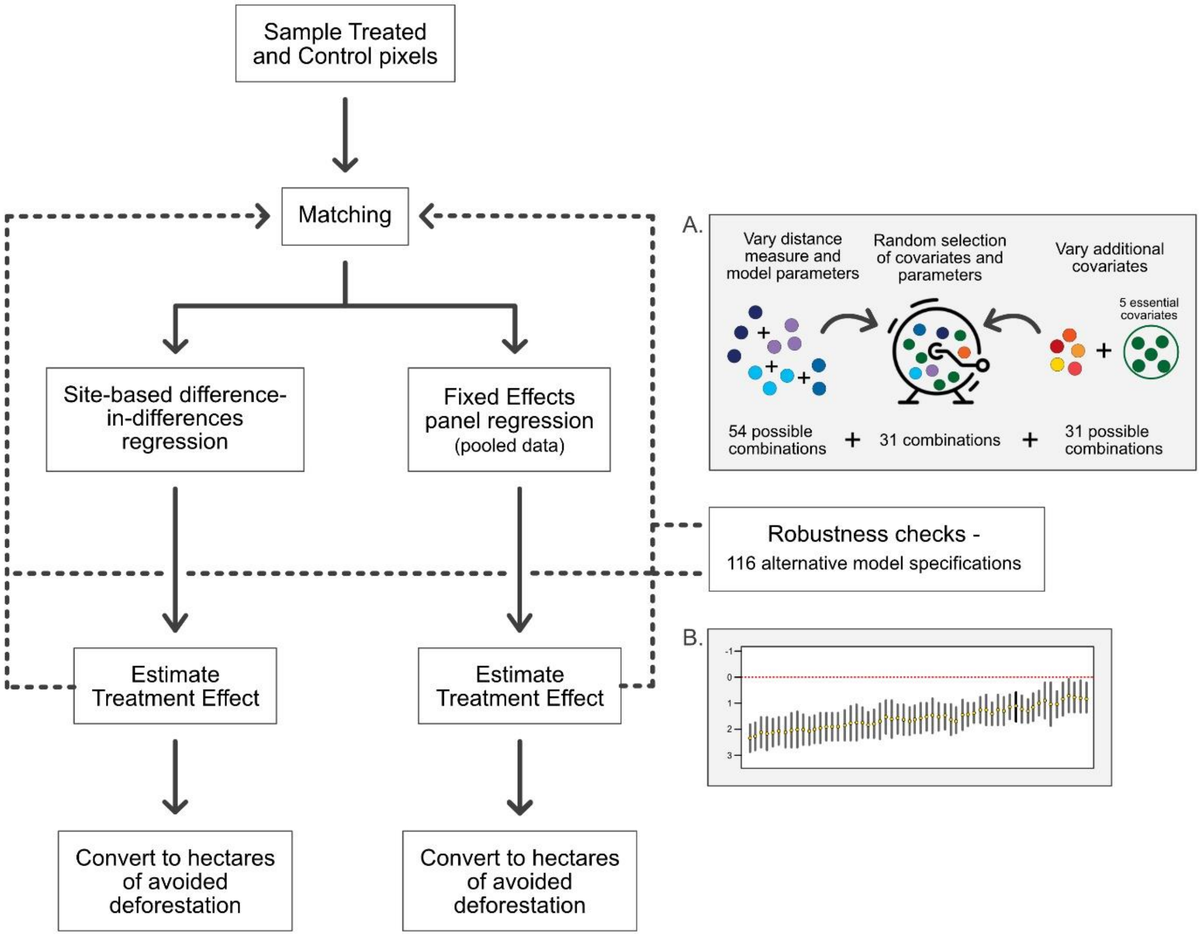

Figure 8: Flowchart of methods. Statistical matching was used to match sampled pixels from each offset to control pixels sampled from the wider forested landscape with similar exposure to drivers of deforestation (Supplementary Table 1.4). Difference-in-differences regressions were run for each matched offset-control sample to estimate the effect of protection within each offset (termed site-based difference-in-differences). Pooled data was used in a fixed effects panel regression to estimate the impact of protection across the whole offset portfolio. Resulting estimates were converted into hectares of avoided deforestation. To test the robustness of results to arbitrary modelling choices, the matching and outcome regressions were repeated using 116 alternative matching model specifications (Box A) to produce a range of estimates (Box B). The statistical distance measure used in matching (e.g., Mahalanobis), caliper size, ratio of matched control to treated units, and matching with or without replacement (shades of blue/purple) were varied in all 54 possible combinations. Holding these choices constant, we constructed 31 models based on all possible combinations of five additional covariates (shown in shades of red/orange) with a core set of five essential covariates (green). Finally, we explore the robustness of the results to 31 randomly selected combinations of distance measure, model parameters and additional covariates.

Results

Ambatovy's offset strategy is based on averted loss. It aims to generate biodiversity gains to offset the losses incurred at the mine site by preventing an equivalent amount of biodiversity loss within four biodiversity offset sites (which face a high rate of deforestation from shifting agriculture; Von Hase et al., 2014). To this end the company, and its NGO partners, implemented conservation activities aimed at slowing forest clearance within the four offsets. These included ecological monitoring, establishing community forest management associations and supporting them with the monitoring and enforcement of resource-use restrictions, environmental education programmes, and promoting alternative income-generating activities in surrounding communities (Ambatovy, 2017; Bidaud et al., 2017). Occasionally the local police are brought in to assist with enforcement (Bidaud et al., 2017).

According to our site-based difference-in-differences regressions (see methods) of the four biodiversity offsets associated with the Ambatovy mine, two significantly reduced deforestation relative to the counterfactual (Ankerana and the Conservation Zone; $p < 0.01$). Protection reduced deforestation by an average of 96% (95% CI: 89 to 98%; $p < 0.001$, N= 38) per year in Ankerana and 66% (27 to 84%; $p < 0.01$, N= 38) per year in the Conservation Zone (Figure 9; Supplementary Table 1.9). One offset showed no significant effect (Torotorofotsy; -41 to +510%; $p = 0.28$, N= 38), while the remaining offset (Corridor Forestier Analamay-Mantadia [CFAM]) could not be assessed due to the lack of parallel trends in outcomes between the offset and matched control sample in the pre-intervention period - a critical assumption in difference-in-difference analyses. In CFAM, there was a significant declining trend in deforestation prior to protection whilst the matched control sample showed a significant increasing trend (Supplementary Figure 1.5). Therefore, CFAM could not be used in the difference-in-differences analysis.

Including all four offsets in a single analysis using a fixed effects panel regression (see methods), we estimate that protection reduced deforestation by an average of 58% per year (95% CI: 37 to 73%, N = 152) across all four biodiversity offsets, relative to the estimated counterfactual (Figure 9). We also tested the effect of excluding CFAM and estimate a greater reduction in deforestation of 72% per year (54 to 83%, N = 114; Supplementary Table 1.12 and Supplementary Figure 1.8). Given the two estimates are not significantly different (Z test, p > 0.2), we present the more conservative estimate, which incorporates the effect of all four offsets, as our main result. Our results are also robust to the alternative specification of site and year as random effects (-53%, -27 to -69%; Supplementary Table 1.12).

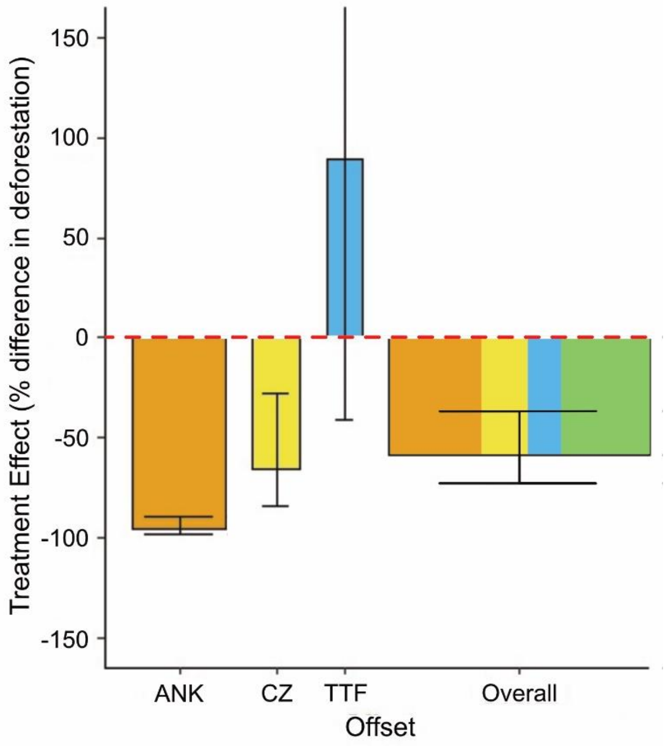

Figure 9: The estimated percentage reduction in annual deforestation within each offset (from the site-based difference-in-differences regressions) and overall, across the entire offset portfolio (from the fixed effects panel regression). The treatment effect is expressed as the average percentage difference in annual deforestation between the offset(s) and the estimated counterfactual following protection. Error bars represent 95% confidence intervals (the upper bound for TTF extends to +510%). The width of the bar is proportional to the area of forest within each offset at the year of protection (Supplementary Table 1.2). ANK: Ankerana (orange), CZ: the Conservation Zone (yellow), TTF: Torotorofotsy (blue). Corridor Forestier Analamay-Mantadia (CFAM; green) could not be included in the site-based difference-in-differences analysis due to lack of parallel trends in the pre-intervention period (Supplementary Figure 1.5). N = 38 for Ankerana, the Conservation Zone and Torotorofotsy and N = 152 for the Overall result.

Results are robust to alternative model specifications

Arbitrary modelling choices, particularly associated with the decisions made in a matching analysis, are inevitable yet can exert a significant influence on estimated impacts (Silberzahn et al., 2018). Following Desbureaux (2021) we show that our results are robust to 116 alternative matching model specifications, all of which are a priori valid (Figure 10). The vast majority of models for both Ankerana and the Conservation Zone confirm the results from the main model specification (see Methods for details of the main model), presented in Figure 9, of significant avoided deforestation. Where some models show an insignificant result (e.g., for the Conservation Zone), in most cases these models are not a posteriori valid. By this we mean that more than 90% of treated units were unmatched (i.e., a match within the caliper of the statistical distance measure could not be found), mean covariate balance exceeded the accepted threshold, or parallel trends were not achieved. Exploring alternative model specifications also did not substantially change our results for Torotorofotsy; 78 of the 79 a posteriori valid models showed no significant impact of protection on deforestation, one suggested protection was associated with an increase in deforestation. For CFAM, the vast majority of alternative specifications, like our main model, were not a posteriori valid as they failed the parallel trends test. Of the seven a posteriori valid models, six showed no significant effect whilst one showed protection was associated with a significant increase in deforestation relative to the counterfactual. Our result of a significant overall reduction in deforestation across all four offsets from the fixed effects panel regression was robust for 106/116 alternative model specifications and none showed a significant increase in deforestation. Therefore, the evidence of avoided deforestation presented in Figure 9 is robust.

We explored which modelling choices had the greatest influence on estimated impacts and found that the choice of statistical distance measure and model parameters had the most consistent, significant effect whilst the effect of including additional covariates is mixed (Supplementary Table 1.13).

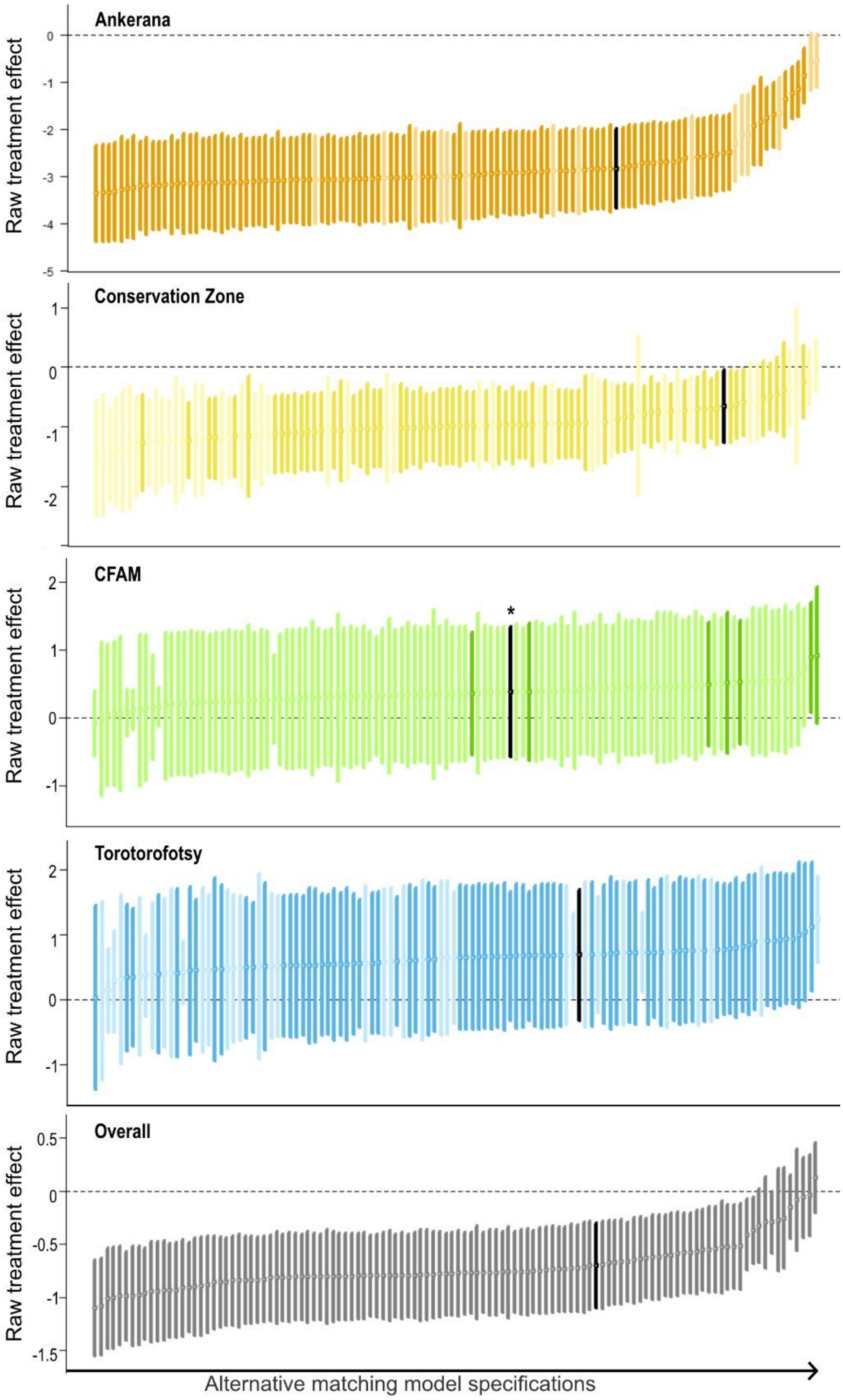

Figure 10: Raw estimates of treatment effect (points) and corresponding 95% confidence intervals (bars) derived from 116 alternative matching model specifications. The alternative specifications included 54 possible combinations of matching distance measure and model parameters, 31 possible combinations of the five additional covariates with the core set of essential covariates, and 31 randomly selected combinations of distance measure, model parameters and additional covariates (see Methods). Results from our main model specification, presented in Figure 9, are shown

in black. An asterisk indicates that the main model was not a posteriori valid. All alternative specifications are a priori valid, but models that are not a posteriori valid (i.e., more than 90% of treated units were unmatched, acceptable covariate balance or parallel trends were not achieved) are shown in lighter shades. See Supplementary Figure 1.11 and 1.12 for full details of parameters and covariates associated with each result. Values are reported un-transformed and represent the effect of treatment on the log(y + 1) transformed count of annual deforestation.

No Net Loss of forest nearly achieved by the offsets

The mine has destroyed or significantly degraded 2,064 ha of natural forest at the footprint and upper reaches of the slurry pipeline (henceforth mine site; Von Hase et al., 2014). The offsets have been in operation for between seven and 12 years (Von Hase et al., 2014). Using site-based difference-in-differences regressions we estimate that between the year of protection and January 2020, 1,922 ha (95% CI: 669 – 5,260 ha) of deforestation has been avoided within Ankerana, and 26 ha (5 – 71 ha) has been avoided within the Conservation Zone (Figure 11; see Supplementary Methods 1). This equates to 1,948 ha of total avoided deforestation (over 94% of the forest loss caused by the mine), with the majority achieved in Ankerana. Using the fixed effects panel regression incorporating all four offsets, we estimate an overall reduction in deforestation of 1,644 ha (674– 3,122 ha) between 2009, when the first offset was protected, and January 2020 (Figure 11). This represents more than 79% (33 – 151%) of the forest loss caused by the mine. From 2014, when all the offsets became protected, an average of 265 ha of deforestation was avoided each year until 2020. If this rate continued, by the end of 2021 2,174 ha of deforestation will have been avoided, fully offsetting forest loss at the mine site. Using the upper and lower bounds of estimated avoided deforestation (674 ha and 3,122 ha) suggests NNL will be achieved between 2018 and 2033. In 2014 the company estimated they would achieve NNL between 2022 and 2035 (Von Hase et al., 2014). Our data therefore suggests Ambatovy is on track to achieve NNL of forest earlier than anticipated.

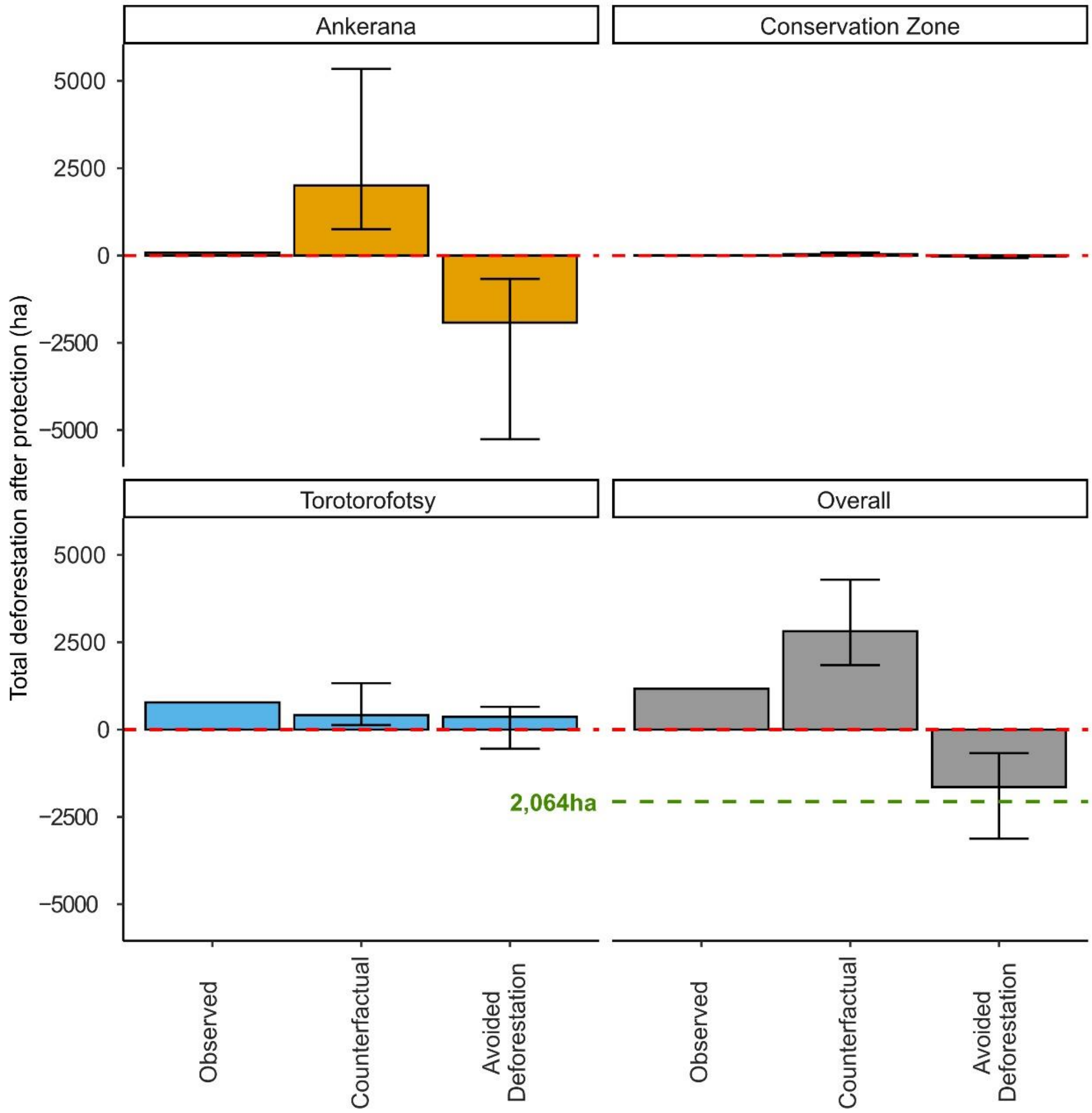

Figure 11: The total observed, counterfactual and the resulting estimate of avoided deforestation within each offset (estimated using site-based difference-in-differences regressions) and overall (using the fixed effects panel regression) between the year of protection and January 2020. The counterfactual is an estimate of the deforestation which would have occurred in the absence of protection and was calculated using the estimated treatment effect (N= 38; Supplementary methods 1). Avoided deforestation is the difference between the observed and counterfactual deforestation; negative values indicate the offset resulted in a reduction in deforestation. The error bars show the 95% confidence interval of the estimates of counterfactual deforestation (derived from the upper and lower confidence intervals of the treatment effect) and the resulting estimates of avoided deforestation. The green dashed line indicates the 2,064 ha of forest loss caused by the mine itself. The number of years following protection is nine for Ankerana, 11 for the Conservation Zone, six for Torotorofotsy and 11 Overall (deforestation within later protected offsets is only counted from the year of protection).

Our estimate of the reduction in deforestation achieved within the Conservation Zone (26 ha, 1.6% of the total reduction in deforestation achieved within the offsets) is likely attributable to a combination of conservation management and the site's location within the mining concession. The company and its predecessor (Phelps Dodge Madagascar) have been present in the concession area since the early 1990s, albeit with a hiatus from 1998 to 2003 (Supplementary Figure 1.1). Therefore, for most of the 19 year study period, access to the concession area, including the Conservation Zone, has been restricted (Bidaud et al., 2017). This de-facto protection reduced deforestation within the Conservation Zone to low levels before it was officially designated as an offset (Figure 12).

A number of studies have documented leakage effects from conservation interventions whereby impacts within the project area are simply displaced outside the boundaries, negating the effect of the intervention at the landscape-scale (Ford et al., 2020). These leakage effects are not observed in our analysis of Ambatovy's offsets (Supplementary Results) as we found that protection of the biodiversity offsets had no significant effect on deforestation within a 10km radius (Supplementary Table 1.16; $p = 0.15$).

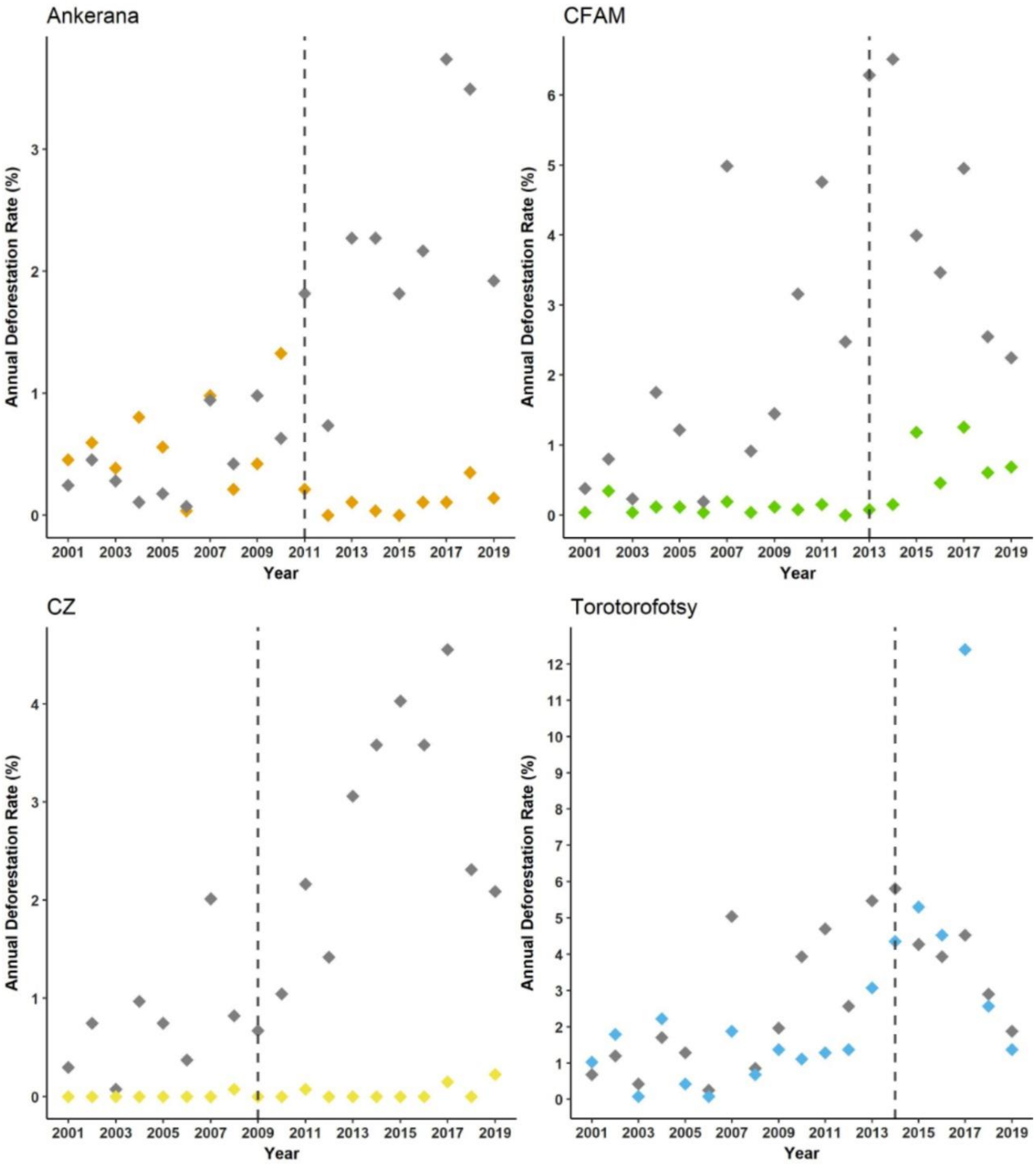

Figure 12: Comparison of the annual deforestation rate within the sample of pixels from each offset and the matched controls over the whole study period. The offset sample is shown in colour whilst the matched control sample is shown in grey. The dashed line indicates the year of protection. The offset and matched control samples contain an equal number of pixels (2862 for Ankerana, 2626 for CFAM, 1340 for the Conservation Zone and 1170 for Torotorofotsy) as the ratio of treated to control units in the matching was set to 1:1. For each offset, N = 38.

Putting these results in a broader context

Despite two thirds of the 12,000+ biodiversity offsets which have been implemented worldwide occurring within forested ecosystems (Bull and Strange, 2018), by 2019 less than 0.05% of these had been evaluated to assess the effectiveness of forest offsets at achieving NNL, and none of these evaluations used robust methods (zu Ermgassen et al., 2019b). Although, there have been several robust evaluations of wider offset policies (Sonter, Barrett and Soares-Filho, 2014; Sonter et al., 2020b). This makes our estimation of the effectiveness of Ambatovy's biodiversity offsets at avoiding deforestation valuable. Our results suggest that by January 2020, the mine had offset 79% (33 – 151%) of the forest loss incurred at the mine site and is on track to achieve NNL by the end of 2021.

In recent years there has been an explosion of studies using robust counterfactual methods to evaluate the effectiveness of other conservation interventions aimed at slowing tropical deforestation. Borner et al (2020) synthesise these findings, using Cohen's d normalised effect sizes to compare the effectiveness of 136 conservation interventions at reducing deforestation. Converting our estimate of the total avoided deforestation achieved by Ambatovy's biodiversity offset policy (1,644 ha according to the fixed effects model) to a Cohen's d effect size yielded an estimate of -0.51 (classed as a 'medium effect'; Cohen, 1988; see Supplementary Results 1). This increases to -1.03 for the individual effect of Ankerana and -0.63 for the Conservation Zone (classed as 'large effects'; Cohen, 1988). Comparison to the normalised effect sizes of the 136 other conservation interventions compiled by Borner et al shows that overall Ambatovy's biodiversity offsets were more effective at reducing deforestation than 97% of the other interventions and all bar one of the protected area interventions (Supplementary Figure 1.10).

Discussion

We lack the empirical evidence to explain why Ambatovy's offsets, as a whole, were so successful at reducing deforestation compared to other forest conservation interventions. We speculate this may stem from the fact that offsetting is inherently centred on achieving measurable impact (No Net Loss). All activities are designed specifically to meet this goal and progress can be regularly evaluated. Furthermore, large companies may possess the sufficient funds to ensure, when they are committed, that they deliver this outcome. In contrast, public protected areas tend to be more focussed on measures such as coverage and investment and less explicitly impact-oriented (Pressey et al., 2021). Another important question is why conservation efforts were so successful in Ankerana but not in Torotorofotsy. It may be that enforcement of conservation restrictions was particularly effective within Ankerana, supported by evidence that local communities lost access to resources after the site was protected (Bidaud et al., 2017; discussed in more detail below).

Methodological caveats

An important caveat to our positive central result relates to the uncertainty inherent in impact evaluation using observational data (Ferraro and Hanauer, 2014). The validity of causal inference rests on our ability to accurately model counterfactual deforestation in the offset sites (what would have happened in the absence of the intervention) using data from matched pixels in the wider landscape which were not protected as offsets. In difference-in-differences analyses this assumes that all important factors influencing selection to treatment and the outcome of interest have been controlled for (or proxied) in the matching, so that the matched offset and control samples have similar trends in deforestation prior to the intervention (Ferraro and Hanauer, 2014). Omitted variables may leave outstanding differences between the two samples which can bias results (Ferraro and Hanauer, 2014). Our choice of matching covariates is based on a good understanding of the local drivers of deforestation and selection to the treatment (Tabor et al., 2017; Poudyal et al., 2018a; see Supplementary Methods 1), and our robustness checks demonstrate our results are robust to alternative specifications (Figure 10).

Our small sample size (N = 38 for the difference-in-differences regressions), limited by the length of the time series of the deforestation data (Hansen et al., 2013), reduces the precision of our estimates. In addition, methods for impact evaluation using observational data are constantly evolving, with recent research highlighting the challenges of evaluating projects with staggered implementation dates (Callaway and Sant'Anna, 2021). Despite these caveats, which are the result of inherent challenges from such a real-world evaluation, our methodology represents a substantial advance in impact evaluation applied to biodiversity offsets. Whilst our results seem relatively robust to alternative modelling specifications, this is only one case study. We hope this work will stimulate further impact evaluations of biodiversity offsetting and emphasize the importance for future researchers to take considerable care over data selection and modelling choices (particularly the matching covariates, distance measure and model parameters) to ensure analyses are context-specific, appropriate, and robust.

Wider concerns with offsetting

Biodiversity offsets in general, and averted loss offsets in particular, are controversial (Gordon et al., 2015; Maron et al., 2015a; Simmonds et al., 2019). General criticisms include whether a concept as complex as biodiversity can be meaningfully reduced to proxies, questions of permanence (Bull et al., 2013; Virah-Sawmy, Ebeling and Taplin, 2014), and the potential social and equity issues of trading biodiversity (including access to ecosystem services) in one place for that in another (Ives and Bekessy, 2015; Jones et al., 2019c). Specific criticisms of averted loss offsets focus on the accuracy of counterfactual scenarios of loss against which gains are measured (Maron et al., 2015a, 2018) and the mismatch between stakeholder expectations and how much averted loss offsets can actually deliver (Gordon et al., 2015; Simmonds et al., 2019). We explore each of these criticisms in turn. In all cases they present clear and important caveats to our positive central result.

The aim of Ambatovy's offset policy is to achieve No Net Loss of biodiversity, whereas our study uses forest cover as an imperfect proxy. Rarely is the appropriate biodiversity data at the required spatial and temporal scale available to facilitate independent evaluation of NNL commitments. In forested ecosystems where most species are forest-dependent (Goodman and Benstead, 2005), forest loss is a transparent, and crucially measurable (Hansen et al., 2013), proxy for biodiversity loss. Furthermore, offsetting development-induced deforestation to achieve NNL of forest is a desirable outcome in itself, given its implications for biodiversity, ecosystem services and carbon storage. However, our measure of deforestation (Hansen et al., 2013) does not capture damage to forest biodiversity occurring at smaller scales, from activities such as selective logging, artisanal mining, and harvesting of forest products for food, fuel, and building materials (Burivalova et al., 2015). More significantly, our method does not capture outcomes for species. In a context of high microendemism with many threatened species there is a real risk large developments such as Ambatovy could lead to species extinction. To mitigate this risk the company surveyed areas scheduled for clearance to identify, catch, and relocate priority species to conservation areas outside the mine footprint (see Supplementary Methods 1 for other mitigation measures), and conducted follow up monitoring of certain species (Von Hase et al., 2014). Whether the impacts of the mine on biodiversity are truly offset will depend on species responses to the changing pressures as well as the presence and efficacy of protection of these species within the offsets, which we were unable to capture in our analysis.

While we present strong evidence that Ambatovy has effectively conserved forest within its biodiversity offsets, questions remain regarding the likely permanence of this achievement. Although Ankerana and Torotorofotsy have been incorporated into the national protected area network and CFAM has been proposed as a new protected area (Ambatovy, 2017), continued effective management after the mine's involvement ceases remains in doubt, given chronic under-investment in Madagascar's protected areas (Jones et al., 2019a). If the offsets become de-facto unprotected after the company pulls out (expected between 2040 and 2050; Von Hase et al., 2014), deforestation is likely to resume and forest within the previously protected offsets may be lost. Offsets are intended to persist for as long as the impacts of the development remain (Bull et al., 2013). Although Ambatovy have committed to restoring the impact site and have taken steps to prepare, tropical forest restoration is notoriously difficult (Crouzeilles et al., 2017). If restoration fails, and the offsets are no longer protected, a future acceleration in biodiversity loss will jeopardise Ambatovy's claims to NNL.

Communities around Madagascar's forests depend on forests for land to practice shifting agriculture and to provide wild products for food, fuel, and building materials (Bidaud et al., 2017; Poudyal., 2018a). The mine and its associated biodiversity offsets have removed or reduced access to these provisioning ecosystem services. To compensate for this loss of access, Ambatovy invested in promoting alternative income-generating activities (including training and the provision of materials) in communities around the mine site and offsets (Ambatovy, 2017; Bidaud et al., 2017). However, research conducted within four affected communities (two near the Conservation Zone and two near Ankerana) found that local people did not consider these benefits to outweigh the significant opportunity costs of the conservation restrictions (Bidaud et al., 2017). The compensatory activities failed to reach those most affected by the restrictions, and there was a temporal mismatch between the immediate loss of access to resources following establishment of the offsets, and the time required for the alternatives to yield benefits (Bidaud et al., 2017). This indicates that poor, rural communities living around the biodiversity offsets are bearing the cost of achieving NNL. For infrastructure developments such as Ambatovy to truly contribute towards sustainable development, SDG 1 (No Poverty) cannot be traded-off for SDG 15 (Life on Land). Instead, project proponents should strive to achieve No Net Loss for both people and planet (Jones et al., 2019c).

An important criticism of averted loss offsets focuses on the accuracy of estimation of the counterfactual scenario; the baseline against which biodiversity losses and gains are measured (Maron et al., 2018). Many offset policies use historical background rates of deforestation to define the counterfactual, but previous studies have shown that this can overestimate the deforestation which would have occurred and consequently overstate the impact of the intervention (Virah-Sawmy, Ebeling and Taplin, 2014; West et al., 2020). We found that the baseline deforestation rates used by Ambatovy in their loss-gain calculations (based on the highest and lowest background deforestation rates at the district level between 1990 and 2010; Von Hase et al., 2014) are actually lower than the counterfactual rates we estimate here using robust methods for impact evaluation, meaning their estimates were conservative (Supplementary Table 1.1). However, there is an important caveat to this: the mine resulted in in-migration to the region (Ambatovy, 2017; Bidaud et al., 2017) which may have indirectly increased pressures on forest resources within the wider landscape, as observed with Rio Tinto's QMM ilmenite mine in Southern Madagascar (Virah-Sawmy, Ebeling and Taplin, 2014). If any mine-related pressures were captured within the period used to define the 'background' rate of deforestation this would no longer represent baseline conditions in the absence of the mine and inflate the counterfactual (and the resulting estimates of biodiversity gains). Ambatovy employs approximately 9000 people (Ambatovy, 2017), many of whom moved to the area from other regions of Madagascar (Ambatovy, 2017; Bidaud et al., 2017). The influx of migrant workers likely increased local demand for food, charcoal, and fuelwood, which may have increased forest clearance and bushmeat hunting (Razafimanahaka et al., 2012; Ambatovy, 2017). Such indirect impacts associated with industrial development are notoriously difficult to quantify, and therefore offset (Lechner et al., 2017). Neither our approach, nor Ambatovy's loss-gain calculations, could account for the indirect impacts of the mine on regional deforestation.

Another criticism of averted loss offsets is that they are premised on a background rate of biodiversity decline which can be slowed to generate the required biodiversity gains (Maron et al., 2018; Simmonds et al., 2019). Therefore, even if 'No Net Loss' as defined by best practice guidelines is achieved (IFC, 2012), loss of biodiversity has still occurred (Gordon et al., 2015; Maron et al., 2015a). This is not what many stakeholders would understand as No Net Loss of biodiversity (Bekessy et al., 2010). However, given Madagascar's high rates of deforestation (Vieilledent et al., 2018), and poor outcomes from tropical forest restoration (Crouzeilles et al., 2017), averted loss is likely to be the better offsetting option (Simmonds et al., 2019). Yet Madagascar has little remaining forest left to lose. Given the importance of the country's biodiversity and the multitude of threats facing it (Jones et al., 2019a), future developments could aim to go beyond

NNL and contribute towards the overall conservation of Madagascar's remaining biodiversity (Simmonds et al., 2019).

Hope for mitigating the environmental impacts of mines

There are over 6,000 industrial mines operating worldwide, covering an estimated 57,000 km^2 (Maus et al., 2020) and impacting around 10% of global forested lands (World Bank Group, 2019). Low-income countries, like Madagascar, desperately need economic development. Mining, if well-regulated, can be part of the solution. From the start Ambatovy promoted itself as a world-leader in sustainable mining and has some of the strongest commitments to conservation among 29 large-scale mines operating within forests (World Bank, 2019). Given this, and the resulting substantial investment the company made in NNL, failure would have been worrying for the concept of mitigating biodiversity loss from development. However, the achievements are notable, especially considering the challenging institutional and political context in which Ambatovy operates (Jones et al., 2019b). Our results provide encouraging evidence that Ambatovy's economic contributions to Madagascar (tens of millions of dollars a year; Ernst Young, 2019), were made whilst minimising trade-offs with the island's precious remaining forest habitat. There are many important caveats to this finding, as to any claim of No Net Loss achieved through offsetting, however the result certainly demonstrates the value of high aspirations combined with substantial investment in mitigating the biodiversity impacts of mining.

Methods

Study Site and context

Ambatovy is a very large nickel, cobalt, and ammonium sulphate mine in central-eastern Madagascar owned by a consortium of international mining companies (Ambatovy, 2018a). It represents the largest ever foreign investment in the country ($8 billion by 2016; Ambatovy, 2018a) and a significant source of fiscal income (Ernst Young, 2019). In 2018, the company contributed approximately $50 million USD in taxes, tariffs, royalties, and other payments (Ernst Young, 2019), and employed over 9,000 people (93% of whom were Malagasy; Ambatovy, 2018b). Commercial production began in January 2014 (Von Hase et al., 2014; Supplementary Figure 1.1). As key components in batteries supply of nickel and cobalt is critical to the green energy transition and demand for these metals is predicted to increase significantly in future (Hund et al., 2020).

The mining concession covers an area of 7,700 ha located in the eastern rainforests of Madagascar (Figure 7) which have very high levels of biodiversity and endemism (Berner, Dickinson and Andrianarimisa, 2009; Phillipson et al., 2010). After avoidance and minimisation measures were applied (Supplementary Methods 1) the mine was predicted to clear or significantly degrade 2,064 ha of high-quality natural forest at the mine footprint and upper pipeline (Von Hase et al., 2014). Any impacts on plantations or secondary habitat are not included in this estimate. Losses at the impact site were not discounted in relation to a background rate of decline meaning the company took responsibility for the full area of forest lost (Bidaud, Hrabanski and Meral, 2015). Independent verification by our team (by measuring the size of the mine footprint on Google Earth) confirms the extent of forest loss at the mine footprint (Supplementary Figure 1.2). Clearance of the footprint accounts for most of the forest loss associated with the mine as losses associated with the pipeline are small (Berner, Dickinson and Andrianarimisa, 2009).

Ambatovy aims to generate biodiversity gains to offset the mine-induced losses by slowing deforestation driven by shifting agriculture elsewhere (Ambatovy, 2017). To this end the company designated four sites, totalling 28,740 ha, to be protected as biodiversity offsets; Ankerana, Corridor Forestier Analamay-Mantadia (CFAM), the Conservation Zone and Torotorofotsy (Berner, Dickinson and Andrianarimisa, 2009; Figure 7). The offsets are considered like-for-like (Sonter, Barrett and Soares-Filho, 2014) and were selected based on similarity to the impact site in terms of forest structure and type, geology, climate, and altitude (Von Hase et al., 2014). The large combined area of the offsets relative to the impacted area was designed to allow flexibility, account for uncertainty, and incorporate as many of the affected biodiversity components as possible (Von Hase et al., 2014). Ankerana is the flagship offset, selected based on its size, connectivity to the CAZ forest corridor and the presence of ultramafic outcrops thought to support the same rare type of azonal forest lost at the mine site (Berner, Dickinson and Andrianarimisa, 2009). Extensive surveys conducted within Ankerana to establish biological similarity concluded the offset to be of higher conservation significance than the forests of the mine site due to the presence of rare lowland tropical forest (Von Hase et al., 2014).

The Conservation Zone is directly managed by the company, given its location within the concession area, whilst the other offsets are managed in partnership with local and international NGOs (Von Hase et al., 2014; Bidaud, Hrabanski and Meral, 2015). Ambatovy funds the management of Ankerana by Conservation International and local NGO partners (although prior to 2015 Ankerana was directly managed by Ambatovy via a Memorandum of Understanding with Conservation International; Von Hase et al., 2014), supports BirdLife partner Asity with the management of Torotorofotsy, and several local NGOs including Voary Voakajy (Bidaud, Hrabanski and Meral, 2015) are involved in CFAM (Ambatovy, 2017). The company is also working to secure formal, legal protection for CFAM (Ambatovy, 2017) as part of a proposed Torotorofotsy-CFAM Complex New Protected Area (although progress on this has stalled).

Overview of methods

To estimate the impact of the offsets on deforestation and determine whether this has prevented enough deforestation to offset forest loss at the mine site, we combined several complementary methods for robust impact evaluation (Figure 8). First, we used statistical matching to match a sample of pixels from each biodiversity offset to pixels from the wider forested landscape with similar exposure to drivers of deforestation. Then we used a site-based difference-in-differences regression for each matched offset-control sample, and a fixed effects panel regression on the pooled data, to estimate the effect of protection. We systematically explored how arbitrary modelling choices (including the statistical distance measure used in matching, caliper size, ratio of control to treated units, matching with or without replacement and which, if any, additional covariates were included) affected our inference, exploring the robustness of our results to 116 alternative model specifications.

Matching

The former province of Toamasina was selected as the geographic area from which control pixels were sampled as it encompasses forests of the same type as the concession area with varying degrees of intactness and accessibility. The four biodiversity offsets are located within this province (Figure 7).

The unit of analysis is a 30 x 30 m pixel that was forested in the baseline year 2000 (Harper et al., 2007; Vieilledent et al., 2018). It is important that the scale of analysis aligns with the scale at which the drivers of deforestation (in this case, small-scale shifting agriculture) operate (Avelino, Baylis and Honey-Rosés, 2016). The median agricultural plot size (from 564 measured plots) in the study region is approximately 36 m x 36 m (Poudyal et al., 2018b). We took a sub-sample of pixels to reduce computational effort whilst maintaining the capacity for robust statistical inference (Blackman, 2013; Rasolofoson et al., 2015). We used a grid-based sampling strategy ensuring a minimum distance between sample units to reduce spatial autocorrelation (Robalino and Pfaff, 2012), and equal coverage of the study area (Blackman, 2013). A 150m x 150m resolution grid, aligned to the other 30m resolution data layers (Figure 7C), was overlaid on the province and the 30x30m pixel at the centre of each grid square was extracted to produce a sub-sample of pixels that are 120m away from their nearest neighbour. 120m is larger than the minimum distance between units used in another matching study in Madagascar (68m; Rasolofoson et al., 2015) but smaller than that used in other studies (200m; Bruggeman, Meyfroidt and Lambin, 2015), and so strikes an appropriate balance between the avoidance of spatial autocorrelation and maximising the possible sample cells.

Protected areas in the study area managed by Madagascar National Parks were excluded from our control sample as they are actively managed and therefore do not represent counterfactual outcomes for the biodiversity offsets in the absence of protection (Figure 7). However, control pixels were sampled from within the Corridor Ankeniheny-Zahamena (CAZ) new protected area as legal protection was only granted in 2015 and resources for management are limited and thinly spread (Hewson et al., 2019). Additionally, Ankerana and parts of CFAM overlap with the CAZ and would have experienced the same management, and likely trajectory, as the rest of the CAZ, had they not been designated biodiversity offsets. Areas within 10km of an offset boundary were excluded from the control sample to reduce the chance of leakage (where pressures are displaced rather than avoided) biasing results (Ford et al., 2020; West et al., 2020). 10km was selected as it is a commonly used buffer zone within the literature (Blackman, 2013; West et al., 2020).

To test for leakage effects, we used Voronoi polygons to partition the buffer area for CFAM, the Conservation Zone and Torotorofotsy (which overlap) into three individual buffer areas according to the nearest offset centroid and took a sub-sample of pixels from each (Figure 7). Areas that overlapped with the established protected areas of Mantadia National Park and Analamazotra Special Reserve were excluded from the buffer zones.

The outcome variable is the annual deforestation rate sourced from the Global Forest Change dataset (Hansen et al., 2013). Following Vieilledent et al (2018) these data were restricted to only include pixels classed as forest in a forest cover map of Madagascar for the year 2000 (Harper et al., 2007; Vieilledent et al., 2018), reducing the probability of false positives (whereby tree loss is identified in pixels that were not forested). The resulting tree loss raster was snapped to the forest cover 2000 layer to align cells, resulting in a maximum spatial error of 15m. The Global Forest Change (GFC) product (Hansen et al., 2013) has been shown to perform reasonably well at detecting deforestation in humid tropical forests (Galiatsatos et al., 2020). In the north-eastern rainforests of Madagascar, Burivalova et al (2015) found GFC data performed comparably to a local classification of very high-resolution satellite imagery at detecting forest clearance for shifting agriculture (although it was not effective at detecting forest degradation from selective logging). As clearance for shifting agriculture is considered the principal agent of deforestation in the study area (Poudyal et al., 2018a) and the forests of the study area are tropical humid (> 75% canopy cover), the GFC data is an appropriate tool for quantifying forest loss. However, the GFC forest loss data only represents the first incidence of forest loss over the study period. It does not measure forest gain and therefore cannot capture secondary forest regeneration, such as which occurs on fallow land within shifting agricultural systems. This means that preventing forest loss from shifting agriculture within the offsets, which in *sustainable* systems is transient, may not compensate for the long-term loss of forest at the mine site. However, evidence from Madagascar shows that in many places population pressure and declining land availability have reduced the length of the fallow periods in shifting agricultural systems, preventing secondary forest regeneration, and ultimately leading to long-term forest loss (Styger et al, 2007). Recent evidence suggests GFC data has temporal inconsistencies, with loss detection improving markedly after 2015 (Palahí et al., 2021). While this may influence the comparison of deforestation rates before and after offset protection (as more deforestation is captured in the later period), this likely affects our control and treated samples equally and so is unlikely to impact our results.

The choice of covariates is extremely important in matching analyses. They must include, or proxy, all important factors influencing selection to treatment and the outcome of interest so that the matched control sample is sufficiently similar to the treated sample in these characteristics to constitute a plausible counterfactual, otherwise the resulting estimates may not be valid (Ferraro and Hanauer, 2014). Based on the literature and a local theory of change we selected five covariates which we believe capture, or proxy for the aspects of accessibility, demand, and agricultural suitability which drive deforestation in the study area (McConnell, Sweeney and Mulley, 2004; Rasolofoson et al., 2015; Eklund et al., 2016; Poudyal et al., 2018a). These are slope, elevation, distance to main road, distance to forest edge and distance to deforestation (see Supplementary Methods 1 for further details). These five essential covariates comprise the main matching specification and form the core set used in all alternative specifications that we tested in the robustness checks. We also defined five additional variables (annual precipitation, distance to river, distance to cart track, distance to settlement, and population density) and tested the effect of including these in the robustness checks. The additional covariates were so defined because they were of poorer data quality (population density, distance to settlement), correlated with an essential variable (annual precipitation, population density) or simply considered less influential (distance to river, distance to cart track; see Supplementary Methods 1).

Statistical matching was conducted in R Statistics using the MatchIt package version 4.1 (Ho et al., 2011). To improve efficiency and produce closer matches we pre-cleaned the data prior to matching to remove control units with values outside the calipers of the treated sample in any of the essential covariates (see Supplementary Methods 1 for details on caliper definition). Following the recommendations of Schleicher et al (2019) we tested several matching specifications and selected the one which maximised the trade-off between the number of treated units matched and the closeness of matches as the main specification (Supplementary Table 1.7). This was 1:1 nearest-neighbour matching without replacement, using Mahalanobis distance and a caliper of one standard deviation. This specification produced acceptable matches (within one standard deviation of the Mahalanobis distance) for all treated units within all offsets. The maximum post-matching standardised difference in mean covariate values between treated and control samples was 0.05, well below the threshold of 0.25 considered to constitute an acceptable match (Stuart, 2010). This indicates that, on average, treated and control units were very well matched across all covariates.

Matching was run separately for each offset. The resulting matched datasets were aggregated by treated status (offset or control) and year to produce a matrix of the count of pixels that were deforested each year (2001-2019) in the offset and the matched control sample. Converting the outcome variable to a continuous measure of deforestation avoids the problem of attrition associated with binary measures of deforestation and is better suited to the framework of the subsequent regressions (Desbureaux and Damania, 2018).

Robustness checks

Statistical matching requires various choices to be made (Schleicher et al., 2019), many of which are essentially arbitrary. There therefore exist a range of possible alternative specifications which are all a priori valid (although some may be better suited to the data and study objectives; Stuart, 2010) but which could influence the results (Silberzahn et al., 2018; Desbureaux, 2021). We tested the robustness of our results to 116 different matching model specifications (Figure 10). First, we tested the robustness of the estimates to the use of three alternative matching distance measures (standard propensity score matching using generalized linear model regressions with a logit distribution, propensity score matching using RandomForest, and Mahalanobis distance), three different calipers (0.25, 0.5 and 1SD), different ratios of control to treated units (one, five and 10 nearest neighbours), and matching with/without replacement. Holding the choice of covariates constant (using only the essential covariates), the combination of these led to the estimation of 54 different models. Second, we tested the robustness of results to the inclusion of the five additional covariates. Holding the choice of distance measure and model parameters constant, we constructed 31 models based on all possible combinations of additional covariates with the core set of essential covariates. Finally, we explore the robustness of results for 31 randomly selected combinations of distance measure, model parameters and additional covariates. All 116 specifications are a priori valid, assuming the covariates capture or proxy for all important factors influencing outcomes but may fail to satisfy the parallel trends condition or produce matches for insufficient number of treated observations (<10%), rendering them a posteriori invalid. It remains important to test the assumptions of the alternative models as failure to do so may lead to erroneous conclusions about effect size and direction being drawn from invalid models. Results are presented through specification graphs based on codes developed in Ortiz-Bobea et al (2021).

Additionally, we tested the robustness of our results from the site-based difference-in-differences regressions to an alternative temporal specification using an equal number of years before and after the intervention (eight for Ankerana and the Conservation

Zone, six for CFAM and five for Torotorofotsy) and dropping individual years from the analysis. This did not change the significance or magnitude of our results (Supplementary Table 1.10, Supplementary Figures 1.6 and 1.7).

Outcome Regressions

Deriving estimates of causal effect from statistical comparisons of outcomes between treated and control samples relies on the assumption that the latter is a robust counterfactual for the former. In a difference-in-differences analysis this assumes that in the absence of the intervention the treated sample would have experienced the same average change in outcomes over the before-after period as the control sample (Cunningham, 2021). Parallel trends in outcomes between treated and control prior to the intervention is an essential pre-requisite for this assumption. We tested this for each matched offset- control dataset using the following formula:

Eqn 1: $\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 Year_t + \beta_2 CI_i + \beta_3 Year * CI_{it} + \epsilon_{i,t}$

where the outcome is the log(y+1) transformed count of deforestation within sample *i* at year *t* and CI is a binary variable indicating whether the observation is from the offset (1) or control (0) sample.

Parallel trends in deforestation between offset and matched control samples in the years before the intervention were present for all offsets except for CFAM (Supplementary Figure 1.5). Consequently, CFAM could not be used in the site-based difference-in-differences analysis. However, its effect is still captured in the results from the fixed effects panel regression as this is not based on an identifying assumption of parallel trends between groups in the pre-treatment period (Cunningham, 2021).

To estimate the impact of protection within each individual offset we ran an ordinary least squares difference-in-differences regression for each matched offset-control dataset using the following formula:

Eqn 2: $\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 BA_t + \beta_2 CI_i + \beta_3 (BA \times CI)_{i,t} + \epsilon_{i,t}$

where BA and CI are binary variables indicating whether the observation occurred before (0) or after (1) the intervention, in the offset (1) or control sample (0). Given the non-normal properties of count data and the presence of zero values a log(y+1) transformation was applied to the outcome variable (Ives, 2015; Desbureaux and Damania, 2018). The coefficient β_3 and the corresponding confidence intervals were back-transformed (see Supplementary Table 1.9) to obtain an estimate of the percentage difference in average annual deforestation between the offset and the matched control sample after protection, controlling for prior differences between samples (i.e., the estimated counterfactual).

To estimate the overall impact of Ambatovy's biodiversity offset policy at reducing deforestation we pooled the data for all four offsets and their corresponding matched control samples and ran a fixed effects panel regression. The pooled data (N = 152) comprise an observation for each site (*i*=8, four offset and four control) for each year (*t* =19). The fixed effects panel regression quantifies the effect of protection on the log-transformed count of deforestation controlling for site and year fixed effects, according to the following formula:

Eqn: 3 $$\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 Tr_{i,t} + \propto_i + \gamma_t + \epsilon_{it}$$

where Tr is a binary measure indicating the treated status of sample *i* in year *t* (*Tr* = 1 for observations from offset sites in the years following protection and 0 for all other observations), $\propto_i$ and γ_t represent site and year fixed effects respectively and ϵ_{it} represents the composite error. The coefficient of interest (β_1) and the associated confidence intervals were backtransformed to obtain the percentage difference in average annual deforestation across all four biodiversity offsets following protection (the treatment effect).

Evaluating deforestation leakage

To determine whether protection of the four biodiversity offsets simply displaced deforestation into the surrounding forested landscape we repeated the matching and outcome regressions with the sub-sample of units from each buffer zone assigned as the treated group (Blackman, 2013; West et al., 2020; Supplementary Results 1).

Data and code availability statement

All input data and computer code used in this study are available in the GitHub repository accessible here: https://github.com/katie-devs/Biodiversity_offset_effectiveness.

Chapter 3: Mapping to explore the challenges and opportunities for reconciling artisanal gem mining and biodiversity conservation

Photo: A rough sapphire mined in the Ilakaka area and bought by Mr Daou, a gem trader. Photo credit: Author.

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Highlights

- Mining in areas important for biodiversity conservation can cause conflict
- In Madagascar we map areas where gem potential and high biodiversity overlap
- 11-14% of land important for biodiversity in Madagascar may host gem deposits
- But 80% of land with gem potential (7 million hectares) is outside these areas
- There, mining could be promoted and supported to minimise environmental trade-offs

Abstract

Artisanal and small-scale mining (ASM) provides a vitally important livelihood for millions of people in many low- and middle-income countries. ASM can result in habitat clearance, increased hunting pressure, pollution, and sedimentation of waterways. Consequently, where mineral and biological wealth coincide, there are trade-offs. Here, we combine geological data with four datasets capturing conservation priorities, to evaluate where, and to what extent, mining may impact biodiversity, and to explore opportunities for both to co-exist. We use Madagascar as a case study: a biodiversity hotspot rich in economically important minerals where artisanal gem mining has conflicted with biodiversity conservation. We identify areas of Madagascar most likely to host primary deposits of gems and find that 11% - 14% of the most important area for biodiversity on the island could host primary gem deposits. However, we also identify 7 million hectares (80%) of potentially prospective land which is outside of these areas. Establishing decentralised, community-managed zones for licensed ASM in such areas could help to incentivise formalisation and minimise social and environmental trade-offs. Our mapping approach could be applied in other countries to encourage the establishment of designated zones for ASM in places where mining does not conflict with conservation.

Introduction

Artisanal and small-scale mining (ASM) has expanded rapidly in recent decades to become a major livelihood in many low- and middle-income countries, involving an estimated 45 million people in 2020 (World Bank, 2020). Much ASM occurs in countries which are resource-rich but economically poor (IGF, 2017), where ASM can contribute towards poverty alleviation by providing alternative or additional means of income generation, particularly in rural areas with few other options (Hirons, 2020). Engaging in ASM can help to buffer shocks, sustain agricultural livelihoods, and raise funds for investments which are otherwise unattainable (Hilson and Garforth, 2012; Hilson and Maconachie, 2020). However, many of these places are also hotspots for biodiversity (e.g. the Amazon, East Africa, Indonesia and Madagascar), where ASM's contributions to development may involve significant environmental trade-offs (Villegas et al., 2012; Hirons, 2020).

ASM is a labour-intensive and sometimes risky form of mineral extraction and processing characterised by limited use of machinery (Hilson and McQuilken, 2014; Lahiri-Dutt, 2018). It requires little capital investment and, as such, is highly accessible (Yakovleva, 2007). ASM operates mostly outside of the legal economy and formal regulatory structures, and this informality can lead to environmental degradation, poor health and safety, crime and corruption (Duffy, 2007; Verbrugge, 2015; Smith et al., 2016; Gerety, 2017). Historically, much of the narrative around ASM has focussed on these negative social and environmental impacts (Hilson and McQuilken, 2014). However, in recent decades there has been growing recognition of the key role that ASM plays in poverty alleviation and its potential to contribute towards development (Hilson and McQuilken, 2014). This has led to growing calls to formalise the sector to improve conditions, increase efficiency and to mitigate the environmental impacts (Hilson et al., 2017).

The environmental impacts of ASM

Direct environmental impacts of ASM include: deforestation and habitat loss (Espejo et al., 2018; Macháček, 2019; Álvarez-Berríos, L'Roe and Naughton-Treves, 2021; Barenblitt et al., 2021; Laing and Moonsammy, 2021); soil disturbance leading to the sedimentation of waterways, impacting freshwater biodiversity, water quality and flow (Hollestelle, 2012; Lobo et al., 2016); and chemical pollution (Nkuba, Muhanzi and Zahinda, 2022). Mercury contamination from artisanal gold mining is a major problem in many countries (although not currently Madagascar, Klein 2022b), with serious implications for both human (Gibb and O'Leary, 2014) and ecosystem (Boening, 2000) health. ASM can also generate substantial indirect impacts, particularly when it occurs at scale in remote areas (Villegas et al., 2012; Hirons, 2020). Miners need fuel and wood for constructing shelters and mineshaft supports, resulting in tree felling (Schure et al., 2011; Macháček, 2019; Nkuba, Muhanzi and Zahinda, 2022). A growth in local demand for food can spur land conversion for agriculture (Maconachie and Binns, 2007) and increase hunting of threatened species (Hollestelle, 2012; Spira et al., 2019). Artisanal mining can open up remote frontiers to other forms of resource extraction and miners may turn to other, more environmentally damaging forms of income generation, such as charcoal production, as the value of finds decreases (Villegas et al., 2012; Kinyondo and Huggins, 2021; Zhu and Klein, 2022). When hundreds, or even thousands of people converge upon a remote, biodiverse area (such as a Protected Area) to mine, the collective impact on biodiversity can be severe (Villegas et al., 2012; Asner and Tupayachi, 2017). Consequently, where the world's mineral and biological wealth coincide, there can be substantial trade-offs.

Madagascar: a biological and mineral hotspot

Madagascar is internationally renowned for its biodiversity (Myers et al., 2000), but the island is also incredibly rich in economic minerals (Yager, 2019). Madagascar is a poor country and is unsurprisingly using its mineral wealth to support development (EDBM, 2021). While the government has been promoting expansion of the formal mining sector (Canavesio, 2014), ASM has grown rapidly over the past 30 years to become the second most important rural livelihood after agriculture, involving hundreds of thousands of people and indirectly supporting an estimated 2.5 million more in downstream industries (World Bank, 2010; Hilson, 2016). Most ASM targets gold and high-value gemstones, such as ruby and sapphire (Cartier, 2009; Cook and Healy, 2012).

Both Madagascar's mineral and biological wealth stem from a dynamic geological history involving the formation and break-up of supercontinents (Pezzotta, 2001; Richard, 2022). Most of Madagascar's gem deposits, as well as those of neighbouring Mozambique, Tanzania and Kenya, were formed 650 – 500 Ma during the East African and Kuungan orogenies (Rakotondrazafy et al., 2008; Giuliani et al., 2020) when much of Madagascar, and subsequently India, collided with East Africa during the assembly of Gondwana (Fritz et al., 2013). The eastern two-thirds of Madagascar comprises a mosaic of Precambrian crustal blocks that were finally assembled during this period (Supplementary Figure 2.2; Tucker et al., 2014). Continental convergence led to regional metamorphism and intrusive magmatism which produced the high temperatures, pressures, and fluids necessary for the formation of gems. Understanding the geological conditions (i.e. the temperatures, pressures and chemical compositions of rocks) required for gem formation allows us to identify which areas of Madagascar are most likely to be prospective for gems.

Madagascar's gem deposits remained mostly untapped until the discovery of sapphires in the far south of the island in 1992 (Cook and Healy, 2012). This initiated a cascade of discoveries across the island, each attracting a rush of migrant miners, sometimes numbering in the tens of thousands (Canavesio and Pardieu, 2019). Since then ruby and sapphire have been found in numerous locations across the island (Figure 14; Rakotondrazafy et al., 2008), making Madagascar a leading global producer of high-quality gems (Shor and Weldon, 2009; Giuliani et al., 2020).

Environmental and social trade-offs of ASM in Madagascar

People engage in artisanal mining in Madagascar for a variety of reasons: to meet basic needs; diversify livelihoods and reduce risk; raise income to invest in business, housing or education; as a last line of defence against destitution (Cartier, 2009; Lawson, 2018); or to spend on luxury goods (Walsh, 2003). Artisanal mining can also facilitate female empowerment (Lawson, 2018). As such, ASM plays a vitally important role supporting the lives and livelihoods of millions of people across Madagascar, but it can also generate negative social and environmental impacts (Walsh, 2003; Duffy, 2007;

Canavesio, 2009; Cook and Healy, 2012; Cabeza et al., 2019). ASM for gems has impacted important areas for biodiversity as the following examples illustrate.

In 1996, sapphires were discovered near the village of Ambondromifehy in the north-west and within two years an estimated 14,000 people were mining in the area, including within the adjacent Ankarana Special Reserve (Walsh, 2003; Tilghman, Baker and Deleon, 2007). Miners felled trees to clear the land for mining and to obtain wood for fuel and mine supports (Cook and Healy, 2012). Repeated disturbance displaced wildlife and impeded forest regeneration. The number of miners operating within the reserve and the inability of the authorities to evict them, exacerbated by long-standing conflicts over resources, created de-facto conditions of open access in the northern part of the reserve (Baker-Médard, 2012). This enabled an increase in other, more destructive forms of resource use, namely charcoal production and harvesting of precious woods (Tilghman, Baker and Deleon, 2007; Cook and Healy, 2012).

The giant Ilakaka sapphire rush which started in 1998 has affected an extensive area of south-west Madagascar (Figure 13; Canavesio, 2009). Whilst much of this region comprises species-poor savannah, ASM has impacted highly biodiverse dry forests within Zombitse-Vohibasia National Park (Tilghman, Baker and Deleon, 2007; Cook and Healy, 2012). In the early 2000s, forest within and around the protected area were cleared for agriculture to meet the growing demand for food from the burgeoning mining population (Cook and Healy, 2012). Then, in 2003, sapphires were discovered in the buffer zone around the protected area and mining gradually spread into the interior (Tilghman, Baker and Deleon, 2007). ASM has, directly and indirectly, caused substantial forest loss within Zombitse-Vohibasia National Park, as well as increased soil erosion and sedimentation of waterways (Cook and Healy, 2012).

This study

We evaluate where, and to what extent, gem mining could occur within other important areas for biodiversity across Madagascar, and explore ways to minimise trade-offs between ASM, rural livelihoods and biodiversity conservation. We quantify the spatial overlap between the potential distribution of primary gem deposits and four datasets capturing biodiversity conservation priorities. We focus on ruby, sapphire and emerald as these constitute Madagascar's largest gem exports by quantity and value (Cartier, 2009). Using a simplified mineral systems approach we identify areas most likely to host primary ruby, sapphire and emerald deposits based on the underlying geology, and validate the resulting map against a database we compiled of known gem deposits. Next, we explore the spatial overlap with areas of importance for biodiversity; Key Biodiversity Areas (Birdlife International, 2021); Conservation Priority Areas, which capture the distribution of many endemic species (Kremen et al., 2008); protected areas (Rebioma, 2017); and natural forests (Hansen et al., 2013).

Figure 13: Ilakaka before (left) and ten years after (right) the discovery of sapphires which triggered Madagascar's largest gem rush and transformed the area into a gem mining and trading hub. © Pierrot Men.

Methods

Identifying areas potentially prospective for gemstones

Potentially prospective refers to areas with the right geological conditions for the formation of gemstones at the broad-scale. We use the qualifier ‘potentially’ because: a) small-scale variation means the right conditions will not be present across the entire area, and b) ground truthing and geological exploration is necessary to determine whether an area is truly prospective (i.e. likely to contain economic deposits of gemstones).

We use a top-down, mineral systems approach (Wyborn, Heinrich and Jacques, 1994) to identify broad areas potentially prospective for primary ruby, sapphire and emerald deposits based on the critical geological processes and lithologies required for formation. This technique was designed to aid targeting of mineral exploration by identifying new prospective areas at larger scales (Hagemann, Lisitsin and Huston, 2016). The focus on large-scale processes of mineralisation, which are often generic, can enable the identification of areas prospective for multiple minerals, and avoids limitations in the availability of high-resolution data needed for traditional targeting methods (e.g. deposit models; Hagemann, Lisitsin and Huston, 2016)

A mineral systems approach requires an understanding of the geological processes and conditions in which the specific minerals are formed. Ruby and sapphire are gem-quality variants of the mineral corundum ($Al2O3$) and typically occur in rocks which are aluminium-rich and silica-poor, and have been metamorphosed at moderate pressures and relatively high temperatures (Simonet, Fritsch and Lasnier, 2008; Giuliani et al., 2020). Corundum formation often requires the circulation of a fluid to supply aluminium or other trace elements and remove silica from the host rock, via diffusion along geochemical gradients (Simonet, Fritsch and Lasnier, 2008; Giuliani et al., 2020). Emerald is green gem-quality beryl ($Be2Al2Si6O18$) and requires beryllium and trace amounts of chromium and/or vanadium to form. Beryllium is rare in the upper crust and is typically supplied through the intrusion of magma, or by fluids circulating from depth (Giuliani et al., 2019). As such, emeralds are usually associated with intrusive granites, pegmatites or shear zones (zones of rock with enhanced permeability which act as fluid conduits) intersecting chromium-rich rocks (Giuliani et al., 2019). See Supplementary Information for more details.

Our analysis is restricted to primary deposits; those where the gems have not been significantly affected by processes (i.e. erosion and deposition) at the Earth's surface and remain in-situ in the host rock. Secondary deposits are those where gems have been removed from the host rock by erosion and weathering and deposited downslope or within contemporary or paleo river systems. We have topographic data that would enable us to map contemporary river systems, but it is more challenging to map paleo river systems (e.g. within the sedimentary rocks of western Madagascar) and data for these do not exist at a consistent scale across Madagascar. Therefore, as we could not comprehensively assess the potential distribution of secondary deposits, we chose not to include these in our identification of potentially prospective areas.

In Madagascar, the critical large-scale geological processes required for gem formation include: 1) regional metamorphism and magmatism associated with the East African and Kuungan orogenies (Rakotondrazafy et al., 2008; Giuliani et al., 2020); 2) presence of key lithologies in which gems are likely to have formed; notably metamorphosed mafic-ultramafic rocks, low-silica sedimentary rocks such as carbonates, and alkaline volcanic rocks that may contain gems transported from depth (Giuliani et al., 2019, 2020); and 3) major km-scale areas of significant fluid flow, which are typically mapped as shear zones (see Supplementary Information).

The first critical process, regional metamorphism and magmatism, has occurred throughout much of the island's Precambrian basement, excluding the Antongil domain (BGS-USGS-GLW, 2008; Schofield et al., 2010; Fritz et al., 2013). In order to map the other two critical factors, we used the Geological Map of Madagascar at the 1: 1,000,000 scale (Roig et al., 2012) to identify: a) major shear zones, and b) geological units with prospective lithologies (marble, mafic-ultramafic rocks, aluminous metasedimentary rocks, skarns, alkaline volcanic rocks) based on the classifications of Giuliani et al (2020; Supplementary Table 2.1). Shear zones can introduce fluids bearing elements such as beryllium and aluminium which can lead to metasomatism of the rocks within and around the shear zones (Giuliani et al., 2020). However, these rocks must be of a suitable lithology for ruby, sapphire, or emerald to form. Therefore, we only selected shear zones which at some point intersect our selected geological units, which are all silica-poor. Since many of Madagascar's major shear zones are associated with metavolcanics and metasedimentary rocks, most are considered prospective.

Geological data

The 1:1M Geological Map of Madagascar (Roig et al., 2012) was produced by the World Bank funded Projet de Gouvernance de Ressources Minerales (PGRM) which aimed to facilitate development of the mining sector in Madagascar by improving geological knowledge and data availability, governance and management (Cook and Healy, 2012). The map represents the finest resolution, most up-to-date and complete visualisation of Madagascar's geology available.

The geological units in this map represent a simplification of more detailed mapping, and some of these units encompass a range of different lithologies, intimately associated, which cannot be differentiated on a map of this scale (e.g. the basic paragneiss of the Tsaratanana thrust sheet incorporates smaller-scale areas of prospective mafic gneiss and schist which are not shown (Tucker et al., 2014). In these cases, we took a conservative approach. Where the unit description does not clearly indicate a prospective lithology, and where no corundum or emerald deposits are known from that area, we did not include it in our selection. The units identified thus represent those that are considered most likely to be prospective, but it is still possible that primary gem deposits could be found outside these areas.

We first assessed all the lithological units on the map legend and decided which had the potential to be prospective for gems (Supplementary Table 2.1). Then we produced a polyline shapefile of the map which we overlaid on a georeferenced image of the original map and used this to identify and merge polyline segments outlining potentially prospective units. Finally, we digitised the shear zones shown in the raster image and merged with the shapefile of potentially prospective units to form our map of gem potential.

Validating our map of gem potential against known gem deposits

To provide a first-order validation of our map of gem potential, we compiled a spatial database of known gem deposits (categorised according to whether they are primary or secondary; Supplementary Table 2.3) and calculated the distance from each point to the nearest area we identified as potentially prospective (Supplementary Table 2.4). Whilst known secondary deposits are not needed to validate our map of gem potential, which is targeted towards primary deposits, they were included in this analysis to explore the distance between secondary deposits and potential source rocks.

Known gem deposits in Madagascar were identified from the peer-reviewed and grey literature, and the Mindat website (Mindat, 2022). Rakotondrazafy et al (2008), Canavesio and Pardieu (2019) and Cook and Healy (2012) provided many key references. We searched the Journal of Gemmology, and Gems and Gemmology using the search term Madagascar for case study analyses of gems from specific locations. We also searched the grey literature to find expedition reports published on the websites of field gemmologists (e.g. Perkins, 2016) and gemmology institutes (e.g. Pardieu and Rakotosaona, 2012). Vincent Pardieu shared the locations of numerous sites he had visited in east and south-west Madagascar.

Mindat (an open spatial database of global mineral occurrences and mine sites compiled by 4500 contributors and verified by a team of 50 experts) was principally used to locate deposits that had been named, but not georeferenced, in other sources. Where available co-ordinates were coarse resolution, or where distance to the nearest settlement was given, we scanned the area on Google Earth to try to visually identify any mine sites. Mindat entries with a margin of error greater than 5km were not included if no other sources of information could be found.

Our review was not systematic and there are undoubtedly many known gem occurrences in Madagascar which are not reported in the international literature. Therefore, our database should not be considered comprehensive but rather an indicative and informative sample of the distribution of known gem deposits across Madagascar.

Biodiversity data

Biodiversity is inherently complex and difficult to summarise in a single measure (Purvis and Hector, 2000). To mitigate this, we use four different measures, or proxies, of biodiversity, and calculate the proportion of each which is potentially prospective for gems (Supplementary Table 2.2). These datasets are: 1) protected areas (Rebioma, 2017), 2) Key Biodiversity Areas (Birdlife International, 2021), 3) Conservation Priority Areas (Kremen et al., 2008), 4) natural forests (Harper et al., 2007; Hansen et al., 2013; Vieilledent et al., 2018). The overlap with areas of gem potential is not intended to be compared between measures as each measure uses different methodology, biological data, and is subject to different constraints. While there is some spatial overlap between the four layers, there are still considerable differences (Table 1).

Protected areas are established and, in theory, managed to conserve biodiversity. Madagascar's latest cohort of protected areas (granted temporary status in 2005 and formally protected in 2015) was designed to capture important biodiversity features, informed by conservation planning and gap analyses ([including Kremen et al, 2008]; Gardner et al., 2018). However, protected areas do not, and cannot, capture all areas important for biodiversity. Therefore, we use three additional datasets to ensure we capture the wider distribution of biodiversity outside the protected area network. Key Biodiversity Areas and Conservation Priority Areas both represent areas of high conservation priority based on species richness and level of threat, incorporating factors such as species range size, endemism, habitat loss and extinction risk (Kremen et al., 2008; IUCN, 2016), but they use different underlying species data. The Key Biodiversity Areas for Madagascar mostly comprise Important Bird Areas and sites identified by the Critical Ecosystem Partnership Fund (CEPF, 2014; pers comm. A Plumptre) using data from a wide range of taxa and expert elicitation. The Conservation Priority areas were defined to maximise the proportional representation of >2000 endemic species from six taxonomic groups (ants, butterflies, lemurs, frogs, geckos and plants) on 10% of the land surface (Kremen et al., 2008). Forest is a useful indicator of biodiversity as most terrestrial Malagasy species are forest-dependent (Goodman, 2022). Furthermore, forests also provide essential ecosystem services such as carbon storage, clean water provision, and erosion mitigation, which could be compromised by the environmental impacts of ASM (Laing and Moonsammy, 2021).

To produce a recent map of forest cover we masked the Global Forest Change dataset (Hansen et al., 2013) to a national-scale map of natural forests (excluding plantations) for the year 2000 (Harper et al., 2007; Vieilledent et al., 2018). Following Vieilledent et al (2018), we then removed all pixels classed as deforested between 2001 and 2020. The resulting map represents forest cover in Madagascar in January 2020.

Protected areas officially classified as marine protected areas and those within a marine portion greater than 80% were removed from the dataset (Supplementary Table 2.2). The remaining protected areas were clipped to the boundary of Madagascar. The same procedure was applied to remove marine portions of Key Biodiversity Areas.

Table 1: The extent of spatial overlap between the four biodiversity datasets. Values refer to the percentage of biodiversity layer 1 which is within biodiversity layer 2. E.g. 44% of forests are within protected areas.

	Biodiversity layer 1			
Biodiversity layer 2	KBA	Priority Areas	Protected areas	Forests
KBA	N/A	46%	74%	55%
Priority Conservation Areas	30%	N/A	31%	28%
Protected areas	55%	36%	N/A	44%
Forests	49%	38%	53%	N/A

Spatial overlay analysis

Raster overlay was used to calculate the proportion of each biodiversity layer which is potentially prospective for primary ruby, sapphire, or emerald deposits (see Supplementary Information). Following Eklund et al (2022) we disaggregated the results for forest by forest type (using the biome classification from the Resolve Ecoregions project; Dinerstein et al., 2017), to evaluate whether certain types of forest (humid, dry or spiny) are more likely to overlap with areas of high gemstone potential (these results are presented in Supplementary Table 2.5 and Figure 2.2).

We then calculated the percentage of each individual locality (Key Biodiversity Area/Conservation Priority Area/protected area or forest block) which is potentially prospective for gems using Tabulate Intersection on the polygon data (forest and Priority Area layers were first converted from raster, see Supplementary Methods 2).

Ethical considerations regarding the presentation of results

Our analysis is a large-scale identification of areas most likely to host primary gem deposits based on the underlying geology. It does not provide detailed locations of where gems will be found (both because of uncertainties associated with the method, and the scale of analysis). However, to avoid signposting potentially prospective areas and generating perverse outcomes, such as encouraging mining within protected areas (Lindenmayer and Scheele, 2017), we have chosen to present our results in a way that obscures identification of these areas (even at the coarse resolution of the image). As such, we only present maps showing the percentage of each locality that is potentially prospective for gems, not the area within these localities that is potentially prospective (i.e. we do not overlay the map of gem potential on each of the biodiversity layers). This is to avoid highlighting that, for example, the south-west corner of a protected area may contain gems. For this reason, we have also chosen not to make publicly available the detailed spatial data showing the area of gem potential (shown in Figure 14). However, we do publish our spatial database of known gem deposits as these are already known and information is accessible online. We hope that the maps presented below will provide valuable information for policy-makers working in Madagascar on the potential for gem mining to occur in certain areas.

Results

The known gem deposits map well onto the areas we identified as potentially prospective for primary gem deposits. Of the 13 primary deposits of ruby, sapphire and emerald in our database, 10 were located within a potentially prospective unit (including all sapphire and emerald deposits) and the other three were located within 2 km (Figure 14; Supplementary Table 2.4). This is considered within the margin of error for the geological map due to the limited amount of rock exposure on the ground.

Our results show that approximately 8.8 million hectares of land in Madagascar is potentially prospective for primary deposits of ruby, sapphire or emerald, representing ~15% of the land surface (Figure 14). 7 million hectares of this (~80%) occurs outside of the most important areas for biodiversity (combining all four biodiversity layers). Potentially prospective areas occur across much of the Precambrian basement in the eastern two-thirds of the island (Figure 14 and Supplementary Figure 2.1).

We find that 11% of the total terrestrial extent of Key Biodiversity Areas (1,017,857 ha), 14% of Priority Areas (839,447 ha), 11% of the terrestrial protected area estate (741,994 ha) and 12% of forested land (991,704 ha) is potentially prospective for primary deposits of ruby, sapphire and emerald (Supplementary Table 2.5). A substantial proportion of highly biodiverse, potentially prospective land lies outside of the protected area network: 41% (414,086 ha) of KBA land with gem potential is unprotected, 67% (559,928 ha) of Priority Areas, and 47% (466,479 ha) of forests (Supplementary Table 2.5).

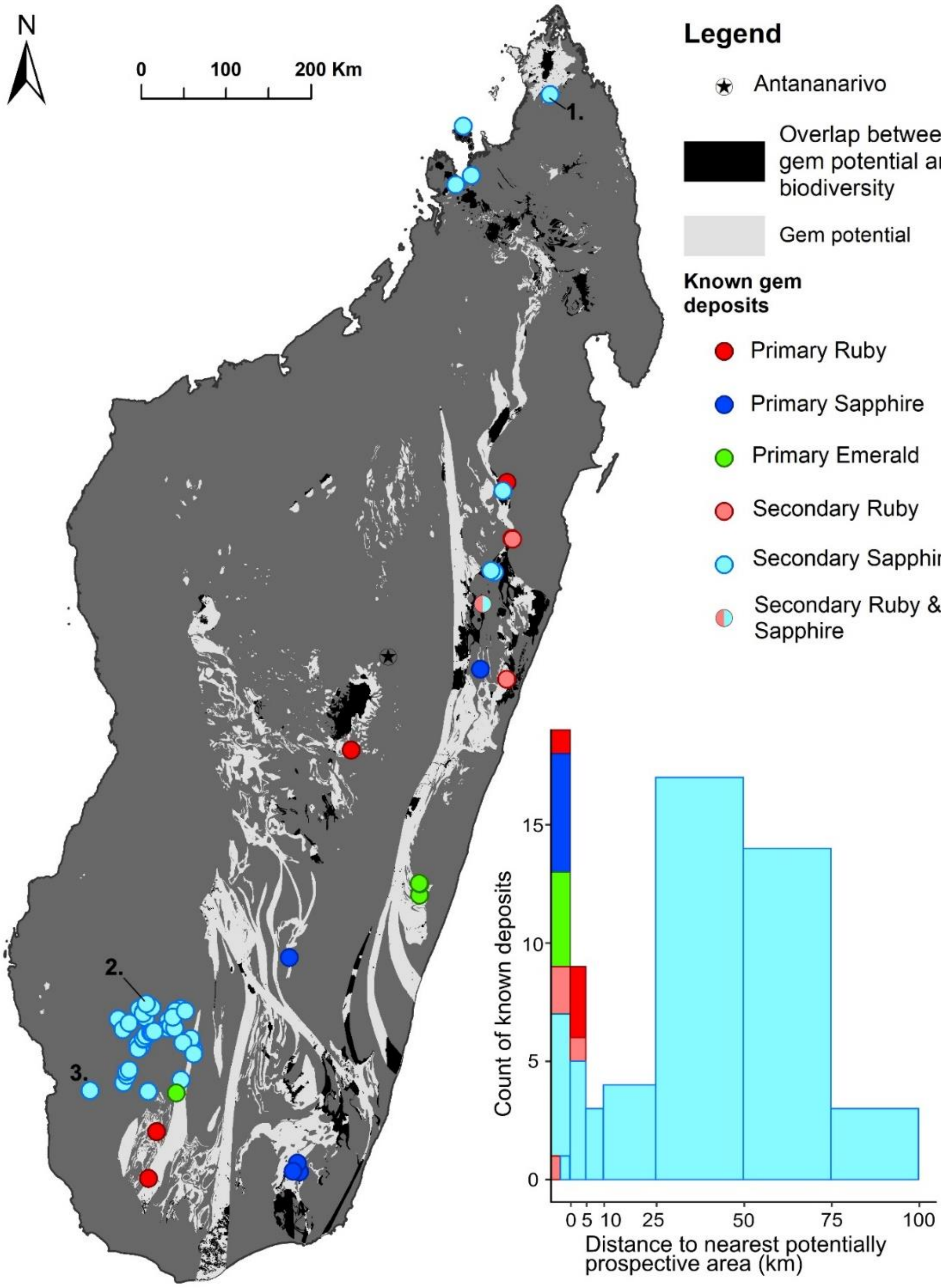


Figure 14: Our map of gem potential and the location of known gem deposits. Light grey represents the area of gem potential outside of protected areas, Key Biodiversity Areas, Priority Areas, and forests (80%). Potentially prospective land within any of these important areas for biodiversity is shown in black (20%). The histogram shows the frequency distribution of distances between known gem deposits and the nearest polygon we identified as potentially prospective for primary ruby, sapphire or emerald. Points and bars are symbolised according to the type of deposit (i.e. the type of gem and whether the deposit is primary or secondary). The large cluster of secondary sapphire deposits in the south-west are part of the giant Ilakaka deposit. Places named in the text are indicated by numbers: 1 = Ambondromifehy, 2 = mine sites near Zombitse-Vohibasia National Park, 3 = Soabiby.

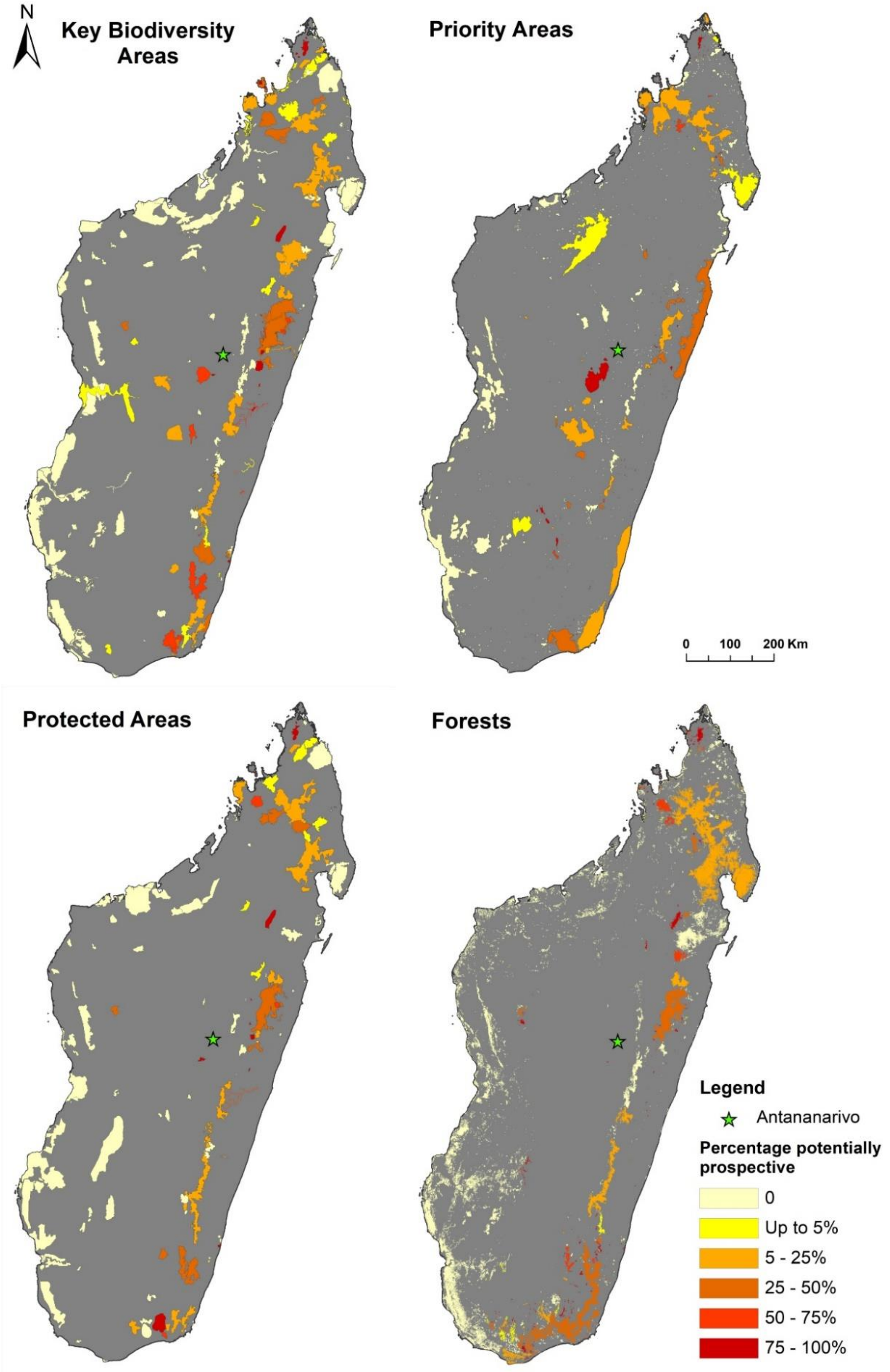


Figure 15: The percentage of each locality (individual Key Biodiversity Area, Priority Area, protected area and forest block) which is potentially prospective for gems. Darker colours indicate a greater proportion of the area is potentially prospective.

Figure 15 shows the percentage of each individual locality (Key Biodiversity Area, Priority Area, protected area, or forest block) which is potentially prospective for primary gem deposits. Most localities in the north and east of the island have potential for gems to occur in at least 5% of their area. 14 Key Biodiversity Areas (6%), 158 Priority Areas (12%), 11 protected areas (10%) and 304 forest blocks (7%) have potential for gems to be found in more than 75% of their area (Figures 15 and 16). These localities are mostly small (median size = 135ha). However, overall, most localities (over 50%) within each biodiversity layer, are not mapped as containing any potentially prospective geology (Figure 16). For example, localities in the south-west and west which overlie Mesozoic sedimentary sequences have not been subject to the metamorphic conditions necessary for the formation of gems (Figure 15 and Supplementary Figure 2.1) and are therefore not considered prospective for primary deposits (although some contain secondary deposits exploited by artisanal miners, eg. Zombitse-Vohibasia National Park and Amoron'I Onilahy Protected Landscape).

Our results are supported by the data on the 69 known gem deposits (both primary and secondary). Including a 500m buffer zone, there are 11 (16%) known deposits within Key Biodiversity Areas, 11 (16%) within Priority Areas, 8 (12%) within protected areas (the Coridor Ankeniheny-Zahamena, Zahamena National Park, Ankarana Special Reserve, Zombitse-Vohibasia National Park, and Amoron'I Onilahy Protected Landscape), and 11 (16%) within a forest (although many of these deposits occur within multiple overlapping biodiversity features; Figure 17).

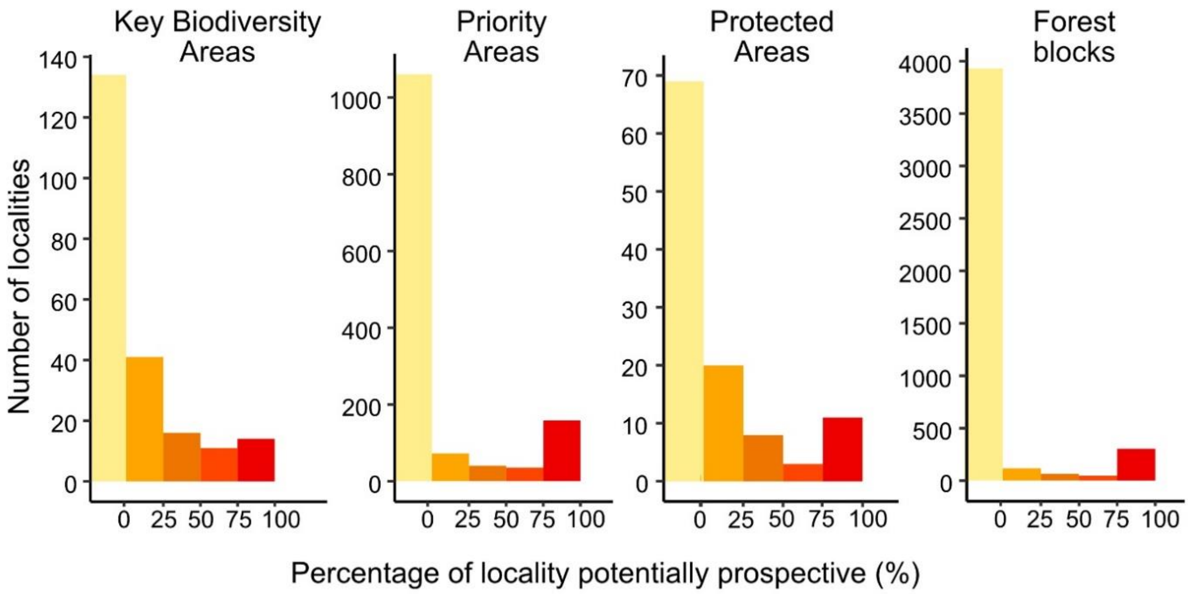


Figure 16: Histogram shows the number of localities within each biodiversity layer grouped according to the percentage of the locality which is potentially prospective for primary gem deposits. Pale yellow bars represent the number of localities which do not contain any potentially prospective land. Forest blocks are only those larger than 84ha (Supplementary Methods 2).

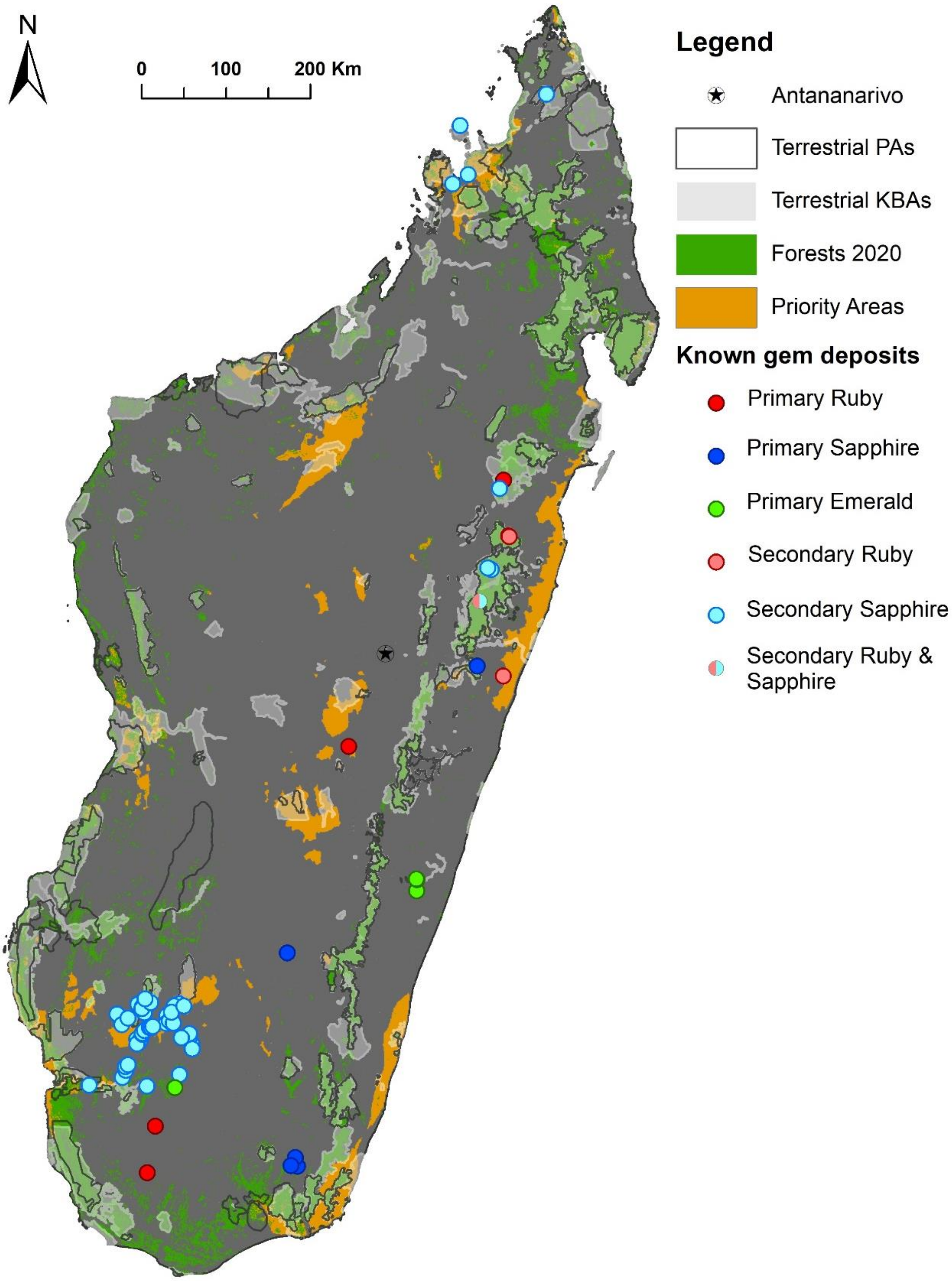


Figure 17: Location of known gem deposits in relation to important areas for biodiversity (Terrestrial Protected Areas [PAs], Key Biodiversity Areas [KBAs], Forest cover in 2020, and Priority Areas).

Discussion

This study has revealed areas of potential future conflict between artisanal and small-scale gem mining and biodiversity conservation in Madagascar, but also opportunities for co-existence. Our results show that 11-14% of the most important area for biodiversity on the island could potentially host primary gem deposits and therefore be impacted by gem mining in future. This has global significance as high rates of endemism in Madagascar combined with the very restricted ranges of some species (Goodman, 2022) means habitat loss or degradation from mining could potentially lead to species extinction. However, we also show that 80% of the potentially prospective land (7 million hectares) lies outside these important areas for biodiversity, where the environmental trade-offs of gem mining could be minimised.

First, we explore how our approach could inform efforts to formalise ASM in countries with a nascent or growing sector through the establishment of designated zones for ASM. We then explore how this could apply within the legal and political context of Madagascar. Next, we consider the conditions which would be needed for legalised ASM within protected areas to be managed effectively. We finish by discussing the limitations of this study and potential avenues for future research.

Informing the establishment of designated zones for ASM

Our methods can be used to identify areas with the potential to host primary gem deposits outside of important areas for biodiversity. The top-down identification of potentially prospective areas, which contain the right geological conditions for the mineralisation of gems, can be used to target more detailed geological analysis and on-the-ground geological exploration to identify zones within these areas which are truly prospective (i.e. likely to contain primary gem deposits). This could inform efforts to formalise ASM through the establishment of designated zones where licensed ASM can be promoted and supported (Corbett, O'Faircheallaigh and Regan, 2017), while minimising impacts on biodiversity.

Formalisation, bringing informal ASM into the legal economy, has emerged as a core policy response to the challenges of ASM (Hilson and McQuilken, 2014). Legalising ASM can enable better regulation, taxation, and improved environmental performance as license holders can be required to conduct environmental impact assessments or site remediation (Hilson et al., 2017; but see Álvarez-Berríos, L'Roe and Naughton-Treves, 2021). It can also facilitate access to credit and technical support for miners, enabling investment in labour or technology to increase production and improve health and safety practices (Siegel and Veiga, 2009; Nopeia et al., 2022). In some countries (e.g. DRC, Mozambique) ASM is only legal within certain designated zones for miners in possession of a license (Hilson, 2020). However, these zones are often not defined on any geological basis and therefore may not contain any workable economic mineral deposits (Dondeyne et al., 2009; Geenen, 2012). It is essential that any designation of ASM zones is grounded in the geology, to ensure that zones are truly prospective for the relevant minerals (Corbett, O'Faircheallaigh and Regan, 2017; Hilson, 2020).

There are considerable political and practical barriers which need to be overcome for ASM to be formalised generally, and within designated zones. There is often a lack of political will to formalize ASM (Corbett, O'Faircheallaigh and Regan, 2017; Hilson et al., 2017) rooted in a bias towards large-scale mining, elite vested interests, outdated discourses about the characteristics of artisanal miners, and a lack of understanding of the importance of ASM for rural livelihoods (Duffy, 2007; Geenen, 2012; Hilson et al., 2017; Vuola, 2022). A lack of political capacity to enforce the regulations is exacerbated by the remote location of much ASM and centralised governance structures (Geenen, 2012; Corbett, O'Faircheallaigh and Regan, 2017; Hilson, 2020), and by inappropriate regulations (Hilson et al., 2017). Many formalisation efforts have failed because the duration and size of license squares do not reflect the nature of the deposits or the often transient, part-time nature of ASM (Dondeyne et al., 2009; Siegel and Veiga, 2009; Hirons, 2020). Additionally, there are practical challenges in demarcating designated zones for ASM amid existing land claims, both formal and customary (Corbett, O'Faircheallaigh and Regan, 2017; Álvarez-Berríos, L'Roe and Naughton-Treves, 2021). In many countries where ASM is an important contributor to livelihoods, little land is truly unowned and unoccupied, and state attempts to acquire land for designated ASM zones could amount to further enclosure of the commons (Alden Wily, 2014; Mitchell, 2016). Finally, miners are typically risk-adverse and therefore must believe that the benefits of formalisation will outweigh the costs (Siegel and Veiga, 2009). Miners may be more willing to obtain a license and operate within designated zones if they know the area is likely to contain gemstones (Nopeia et al., 2022).

Establishing designated zones for ASM in Madagascar

Mining in Madagascar is regulated by the Mining Code of 2005, although a revised Code has recently been approved by the National Assembly and is proceeding through the courts, but has not yet been promulgated (L'Express de Madagascar, 2023). The revised Code includes a new provision for the creation of artisanal mining zones (in addition to individual permits for artisanal miners, Permis Réservé aux Exploitants Artisanaux, which can cover up 50km2; Code Minier, 2023). These zones are to be proposed by decentralised authorities and approved by the Minister of Mines. Artisanal miners wishing to work within these zones must form a collective and obtain an authorisation permit (Autorisation minière d'exploitation artisanale) which is valid for six months and renewable once (Code Minier, 2023). Similar provisions permitting the creation of gold panning corridors have been in force since 2005 (Code Minier, 2005). However, a recent a court audit found that no panning corridors have been established in Madagascar's main gold mining region (Cour des Comptes, 2022). Unfortunately, poor governance and capacity shortfalls severely limit the application and enforcement of the Mining Code in practice.

In the absence of the state, communities have established a variety of novel governance regimes, often drawing on customary arrangements, to regulate and govern ASM (Klein, 2022a, 2022b). In some cases, this has improved health and safety, community cohesion, benefit-sharing and mitigated environmental impacts (Klein, 2022a, Cook and Healy, 2012; Baker-Médard, 2012, cf. Canavesio, 2009). For example, in Soabiby in south-west Madagascar the local community was able to impose respect for local rules and customs on thousands of migrant sapphire miners, preventing mining within sacred forest areas and enabling land-owners to extract rents from miners (Baker-Médard, 2012). Given the current inability of the state to regulate ASM and broad distrust of state institutions (Walsh, 2003; Klein, 2022b), a decentralised, community-based approach towards establishing and managing designated zones for ASM could prove more effective, better at reconciling with existing land claims, and consequently more socially acceptable (Corbett, O'Faircheallaigh and Regan, 2017; Hilson, 2020; Klein, 2022a, 2022b).

Designated zones for ASM may be best suited to establishing new, or formalising existing, long-term mining sites in Madagascar. They may struggle to provide strong enough incentives to discourage the 'rush type' mining common in Madagascar (Cartier, 2009), or mining in Protected Areas. Especially as Protected Areas are sometimes targeted for ASM in active resistance against the perceived appropriation of resources (minerals) by state/conservation interests, and the history of exclusion (Baker-Médard, 2012; Klein, 2022b).

The conditions needed for ASM within protected areas to be managed effectively

ASM within protected areas is illegal in many countries, including Madagascar (Code Minier, 2005; IGF, 2017). Yet, efforts to keep ASM out of protected areas, often involving the police or military, have often failed (Dondeyne et al., 2009; Villegas et al., 2012). In the worse cases, the resulting conflict has threatened lives (Baker-Médard, 2012; Gerety, 2017). Allowing a small amount of tightly-regulated ASM by license holders within sustainable use zones of a protected area has been attempted as an approach to address the impact caused by unregulated ASM within protected areas (e.g. in Gabon, Villegas et al., 2012; Hollestelle et al., 2012, and Daraina, Madagascar, Cook and Healy 2012). This approach could also help mitigate the impact of conservation restrictions and land enclosures on local livelihoods (Vuola, 2022).

However, effective management and regulation of ASM within protected areas requires strong rule of law, good governance, and effective, non-corrupt policing to monitor and enforce rules (Álvarez-Berríos, L'Roe and Naughton-Treves, 2021). Without these foundations, which are lacking in many ASM hotspots (including Madagascar; IGF, 2017), permitting ASM within protected areas risks creating an open-access situation, leading to uncontrolled mining and environmental damage, jeopardising conservation goals (Villegas et al., 2012). Outcomes of efforts so far to regulate ASM within protected areas have been mixed. An influx of migrant miners caused the failure of the agreement in Gabon (Hollestelle, 2012). In Daraina, Madagascar, efforts of the conservation NGO Fanamby to regulate artisanal gold mining within the Loky-Manambato protected area have met with varying success and faced considerable challenges (Fanamby, 2021), including from rising insecurity during the political crisis of 2009 (Cook and Healy, 2012). In places without the capacity to prevent, or strictly manage, mining within protected areas, formalizing ASM outside of protected areas is the best solution (although this still requires considerable governance capacity).

Limitations of the study

The strength of our results rests on the quality of the data. The Geological Map of Madagascar (Roig et al., 2012) is a relatively broad scale (1:1,000,000) generalisation of more detailed mapping, which was itself constrained by the limited amount and accessibility of bedrock exposure across much of Madagascar. Consequently, there is uncertainty in the location of boundaries between geological units and the map cannot capture small-scale variation, meaning we were unable to capture small areas of gem potential (<1km) within larger non-prospective units. We were unable to map the potential distribution of secondary deposits as maps of alluvial sediments are not available at a consistent scale across Madagascar. This is an important limitation, given that some of the largest gem rushes exploited secondary deposits. Finally, it was not possible to map the potential spread of gold deposits with the existing data available. Yet artisanal gold mining is widespread in Madagascar, including within Protected Areas, and is a source of conflict between mining and conservation (Cook and Healy, 2012; Cabeza et al., 2019). These limitations highlight the need for accessible, detailed geological data to underpin policy decisions.

None of the biodiversity datasets used in this study perfectly captures the distribution of Madagascar's biodiversity, and there will still be valuable biodiversity outside of these areas. However, using four datasets allows us to capture a variety of species and habitats and, by combining them, identify the areas of highest biodiversity value where the trade-offs from mining would be greatest.

Future research priorities

To date, there have been no robust, quantitative evaluations of the impacts of ASM on biodiversity in Madagascar. This needs to be addressed to ensure policy responses to ASM, particularly within protected areas, are appropriate and proportionate. A better understanding of local ASM governance is also needed to ensure formalisation policies are tailored to fit the context (Siegel and Veiga, 2009; Klein, 2022a).

Conclusion

ASM supports an estimated 45 million people within 80 low- and middle-income countries (World Bank, 2020). It is also a significant source of minerals, supplying 20% of global gold, up to 30% of cobalt, and 80% of the world's sapphires (World Bank, 2020). Yet ASM's positive contributions to development and mineral supply can involve substantial environmental trade-offs, impacting some of the most biodiverse regions on earth. Our approach could be applied in other biodiversity hotspots with a nascent or growing ASM sector to identify potentially prospective areas outside important areas for biodiversity where ASM could be promoted and supported. Policies to encourage ASM within designated zones of known mineral potential, but low biodiversity, could help to mitigate conflicts between mining and conservation, facilitate distribution of financial and technical support to improve practices, and contribute towards formalisation of the sector.

Data availability

The database of known gem deposits compiled in this study is available here: https://github.com/katie-devs

Chapter 4: No evidence of increased forest loss from a mining rush in a biodiversity hotspot

Photo: The 2016 sapphire rush at Bemainty within the Coridor Ankeniheny-Zahamena Protected Area. Photo credit: Rosey Perkins

This chapter is being prepared for submission to Communications Earth and Environment.

Abstract

Artisanal and small-scale mining (ASM) is an important livelihood activity in many of the world's biodiversity hotspots. However, there is substantial international concern about the negative impacts of ASM on biodiversity. Risks to biodiversity from ASM can become particularly pronounced during a mining rush – a rapid, uncontrolled expansion of ASM. Here, we evaluate what can happen when a mining rush occurs within a highly biodiverse, protected forest, focusing on the 2016 sapphire rush at Bemainty in eastern Madagascar. This rush generated significant media attention which claimed the rush caused hundreds of hectares of deforestation and threatened endangered lemur populations. We interrogate these claims, using the synthetic control method to evaluate the impact of the mining rush on forest cover, combined with field data from interviews and a lemur survey to better understand the wider impacts and trade-offs of mining. We find that the mining rush did not cause a significant increase in forest loss relative to estimated counterfactual loss from other causes in the absence of mining. Evidence from lemur surveys shows lemur populations appear to remain healthy three years after the rush. This evidence, supported by insights from interview data, suggests that mining at Bemainty had limited impacts on the surrounding forest, especially relative to other drivers of change. Our results highlight the heterogeneity of environmental impacts from ASM and emphasize the need for more robust, case-study evaluations to inform policies which are evidence-based, proportionate and fair.

Introduction

Artisanal and small-scale mining (ASM), a labour-intensive form of mining with limited use of machinery (Hilson and McQuilken, 2014), is a globally important livelihood activity, supporting an estimated 45 million people in 80 low- and middle-income countries (World Bank, 2020). Much ASM occurs in places which are also hotspots of biodiversity (Villegas *et al.*, 2012), such as the Amazon (Asner and Tupayachi, 2017), West and Southern Africa (Obodai *et al.*, 2019), Madagascar (Cook and Healey, 2012), and Indonesia (Meutia, Lumowa and Sakakibara, 2022). Where ASM occurs in areas of high biodiversity, there can be substantial trade-offs (Espejo *et al.*, 2018; Hirons, 2020; Barenblitt *et al.*, 2021a; Laing and Moonsammy, 2021). Yet, in most places the impacts of ASM on biodiversity have not been robustly quantified (World Bank, 2019; Hirons, 2020).

ASM can impact biodiversity in a variety of ways (see Chapter 1 for more detail). It can lead to habitat loss and deforestation as miners clear land for mining and harvest wood for fuel or construction materials (Villegas *et al.*, 2012; Macháček, 2019; Barenblitt *et al.*, 2021b). ASM can release toxic chemicals used in mineral processing, such as mercury and cyanide, into the air and water (Donato *et al.*, 2007; Diringer *et al.*, 2015). Artisanal gold mining is the largest global source of mercury pollution (UN Environment, 2019), which can have devastating effects on biodiversity, reducing fitness and increasing mortality of organisms up the food chain (Boening, 2000; Sandheinrich and Wiener, 2011; Sierra-Marquez *et al.*, 2018). Mining, panning, and releasing tailings along waterways can increase erosion and river siltation, impacting water quality, downstream water availability, and freshwater biodiversity (Mol and Ouboter, 2004; Lobo *et al.*, 2016). ASM can also generate indirect impacts by driving in-migration and opening up remote areas to other forms of natural resource exploitation (eg. logging, hunting or farming; Villegas *et al.*, 2012; Kinyondo and Huggins, 2021).

Here, we focus primarily on the impacts of ASM on forest cover. Much of the evidence of ASM-related deforestation comes from descriptive accounts from case studies (eg. Gandiwa and Gandiwa, 2012; Villegas *et al.*, 2012; Macháček, 2020). Quantitative evidence is limited and mostly concentrated within certain regions or countries, such as the Amazon and Ghana, where ASM is extensive (World Bank, 2019). These studies use satellite imagery or secondary forest change data to quantify deforestation in known ASM areas (eg. Asner and Tupayachi, 2017; Dezécache *et al.*, 2017; Espejo *et al.*, 2018; Obodai *et al.*, 2019). For example, Espejo *et al* (2018) identified nearly 100,000 ha of deforestation associated with artisanal gold mining in the Madre de Dios region of Peru between 1984 and 2017. Comparative studies have shown that the forest impacts of ASM can be highly variable (Villegas *et al.*, 2012; World Bank, 2019). The most extensive analysis, quantifying deforestation around 21 ASM sites in 12 countries, found that the rate of forest loss within a 5km buffer zone varied between 0.1% and 46% (World Bank, 2019). However, none of these quantitative studies use robust counterfactual methods to isolate the impact of mining by controlling for other drivers of forest change (Chapter 1).

The environmental risks associated with ASM become particularly concerning when there is a rapid, uncontrolled expansion of mining within biodiverse ecosystems (World Bank, 2019). Mining rushes occur when the discovery of a potentially rich deposit sparks a large, rapid migration of people to the site of the deposit to mine (ICMM, 2010). Awareness spreads (e.g. by word of mouth, social and news media) and the mining population can increase rapidly (Villegas *et al.*, 2012; Canavesio and Pardieu, 2019). People may travel from different regions, or even countries, to take part (Bosee Jønsson and Fahy Bryceson, 2009). At some point the population peaks, and then often declines rapidly as the deposit becomes more depleted and difficult to access (Canavesio and Pardieu, 2019; Fahy Bryceson, Bosse Jønsson and Clarke Shand, 2020). A limited number of miners may remain long-term and continue mining. The discovery of a new deposit elsewhere or the intervention of external actors (eg. the police or army) will often cut short the evolution of a mining rush (Bosee Jønsson and Fahy Bryceson, 2009). Mining rushes are large, and when they occur in remote, biodiverse areas the collective impact can be potentially serious.

Madagascar is a hotspot for both minerals and biodiversity (Myers *et al.*, 2000; Pezzotta, 2001). The ASM sector has grown rapidly over the past 30 years to become the second most important rural livelihood after agriculture, supporting an estimated half a million people (World Bank, 2010, 2019; Cook and Healy, 2012). The rapid expansion of ASM across the island was sparked by a series of discoveries of high-value ruby and sapphire deposits (Canavesio and Pardieu, 2019). These discoveries triggered rushes, where thousands of people from across the island moved to the area to mine (Cook and Healy, 2012; Canavesio and Pardieu, 2019). Although mining within protected areas is illegal in Madagascar (*Code Minier*, 2005), some of these mining rushes occurred within protected areas, for example Zombitse-Vohibasia National Park and Ankarana Special Reserve (Figure 14; Tilghman, Baker and Deleon, 2007; Cook and Healy, 2012; Devenish *et al.*, 2023).

We focus on the particularly high-profile sapphire rush at Bemainty in eastern Madagascar, which began in earnest in September 2016, following limited sapphire mining from 2012 (Figure 18). This rush generated significant national and international media attention (Carver, 2017; Tullis, 2019) as it occurred deep within the rainforests of the Coridor Ankeniheny-Zahamena (CAZ), a category VI protected area home to globally important biodiversity, including many endemic and threatened species (Goodman, Raherilalao and Wohlauser, 2018; Gamba *et al.*, 2022; Safford *et al.*, 2022). At its peak, over 10,000 people were illegally mining in several valleys stretching approximately 4km (Canavesio and Pardieu, 2019), but estimates were as high as 30,000 (Pardieu *et al.*, 2017). A National Geographic article blamed miners for causing hundreds of hectares of deforestation and threatening populations of endangered lemurs (Tullis, 2019). Others criticised this narrative suggesting that land clearance in the valley long pre-dated the start of mining and was instead driven by land conversion to agriculture (Pardieu, 2019). The World Bank (2019a) study introduced above estimated a deforestation rate of 43% within the mining area at Bemainty and 4.5% within a 5 km buffer zone. However, there are issues with this analysis which means this deforestation cannot be robustly attributed to the mining activity. Firstly, the deforestation rate is calculated for the period 2000 – 2016 but mining at Bemainty only started in 2012 (and the rush didn't begin until 2016), meaning this forest loss could have occurred before mining began. Secondly, the raw global dataset used in this analysis (Hansen *et al.,* 2013), detected much of this deforestation in valleys which were in fact cleared long ago

(Supplementary Figures 3.5). The emphasizes the importance of using robust methods to evaluate the environmental impacts of mining, particularly as perceptions of these impacts strongly influence policy.

Here we evaluate whether the mining rush led to an increase in deforestation and forest degradation (defined here as temporary tree cover loss) in the Bemainty drainage basin relative to a counterfactual of no mining. Counterfactual outcomes are estimated using a synthetic control; a weighted combination of control drainage basins designed to be as similar as possible to Bemainty in factors influencing forest loss. We also draw on interviews and lemur surveys conducted at the mine site to further explore the impacts of the mining rush at Bemainty. To our knowledge this is the first study to use robust counterfactual methods to evaluate the environmental impact of ASM.

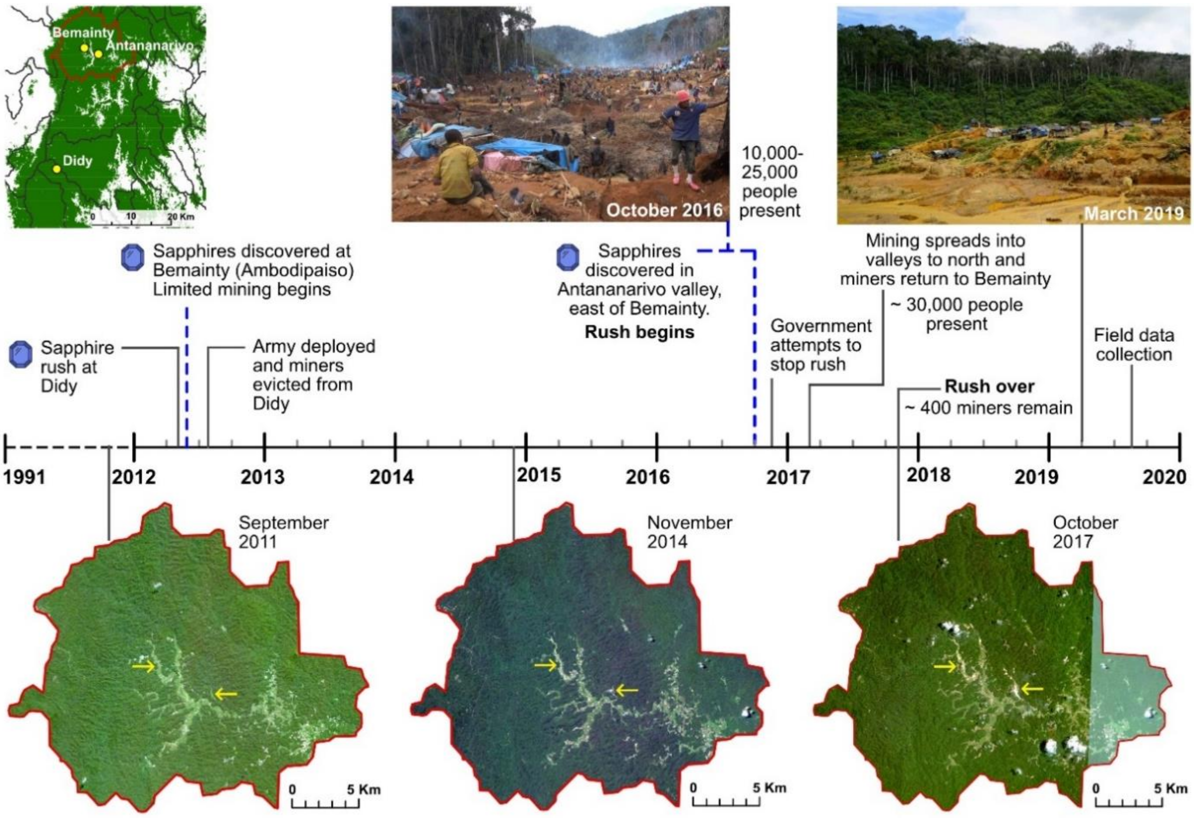


Figure 18: Timeline of the development of mining in the Bemainty drainage basin over the study period. Yellow arrows point to the Ambodipaiso (left) and Antananarivo (right) mining valleys. Dashed blue lines indicate the start of mining in the Bemainty valley in 2012 and the onset of the rush in September 2016. Satellite images were captured by the RapidEye sensor and obtained from Planet (Planet Team, 2017). Photo credit: Rosey Perkins.

Results

Forest loss

We find no evidence that artisanal gem mining at Bemainty, which began in 2012 and surged during the rush of 2016-2017 (Figures 19 and 20), caused a significant increase in deforestation or forest degradation (collectively termed forest loss), relative to a counterfactual of no mining estimated using a synthetic control. For both outcomes this finding is consistent across three different measures (e.g., raw hectares of deforestation, deforestation rate and cumulative deforestation) and two scales of analysis (first sampling control basins from the CAZ, and second from the wider province of Toamasina, Supplementary Figure 3.12).

While deforestation at Bemainty did increase between 2016 and 2017 and was higher than the synthetic control in 2017 (particularly for cumulative deforestation), this difference is marginal and well within the range of statistical noise established using placebo tests. It is therefore considered a non-significant effect. Furthermore, seven of the eight similarly forested drainage basins in the CAZ also experienced an increase in deforestation between 2016 and 2017, indicating that this increase was likely driven by external factors affecting a wider area (Supplementary Table 3.3).

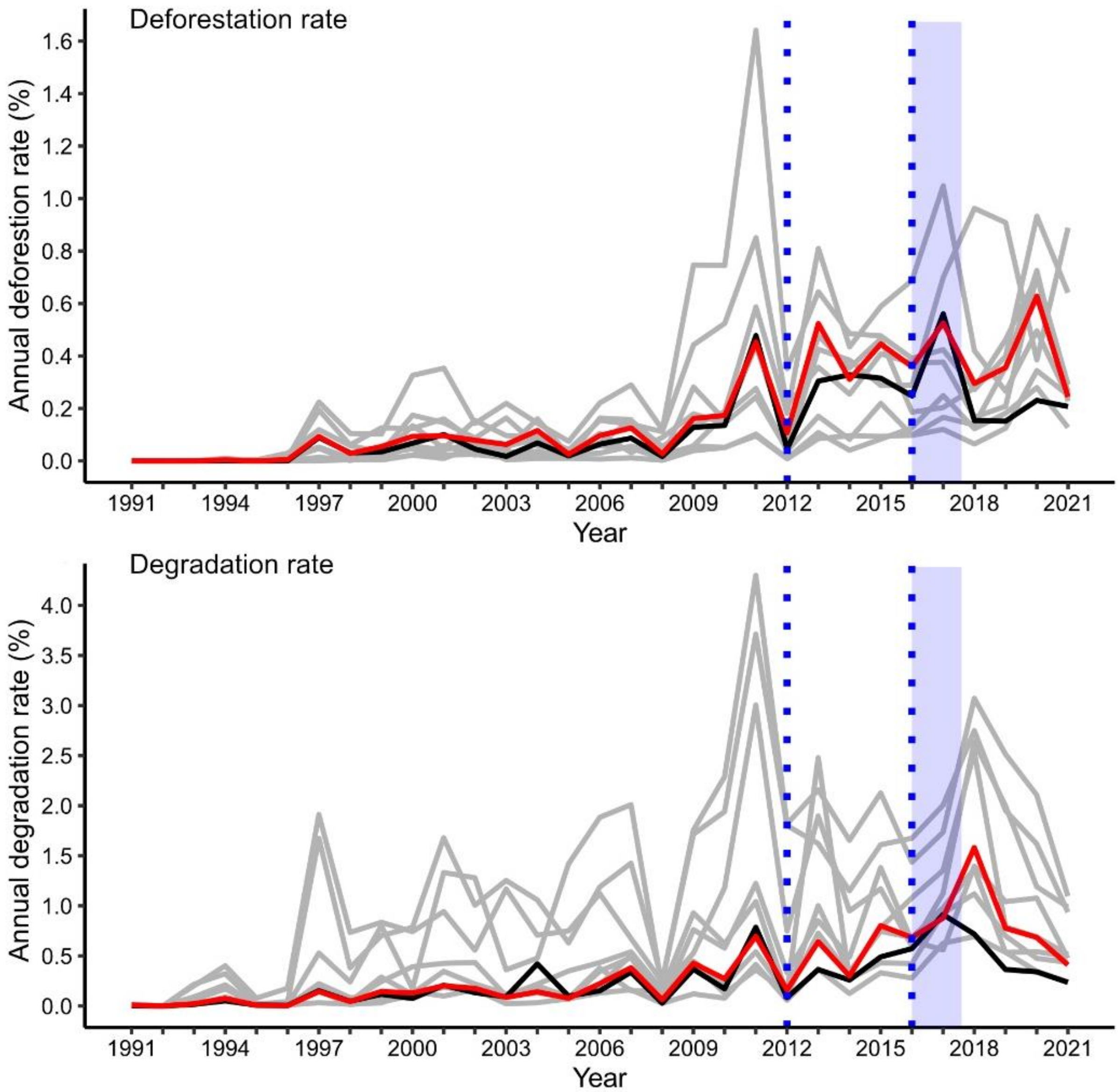


Figure 19: The annual deforestation and degradation rate within the Bemainty basin (black) compared to the synthetic control (red) over the study period. Light grey lines show outcomes in the eight control drainage basins in the CAZ (i.e. the donor pool from which basins were selected to comprise the synthetic control). The dotted blue lines indicate the preliminary onset of mining in 2012 (left) and the start of the mining rush in 2016 (right). The light blue shaded area indicates the duration of the peak mining rush. These results are from our primary analysis focussed on the CAZ.

There are some signs that mining may have in fact been associated with reduced, rather than increased forest loss (i.e., deforestation or degradation) at Bemainty. After the onset of mining, deforestation and degradation were mostly lower in Bemainty than the synthetic control (Figures 19 and 20). However, in almost all cases this difference is within the range of statistical noise (although it is very close to the lower boundary in many cases), and therefore cannot be differentiated from uncertainty in the estimation method (Abadie, 2021; Figure 20). Isolated observations of significantly lower deforestation in Bemainty in certain years (eg. 2013) are not consistent across all outcome measures and scales of analysis (Figure 20, Supplementary Figure 3.12).

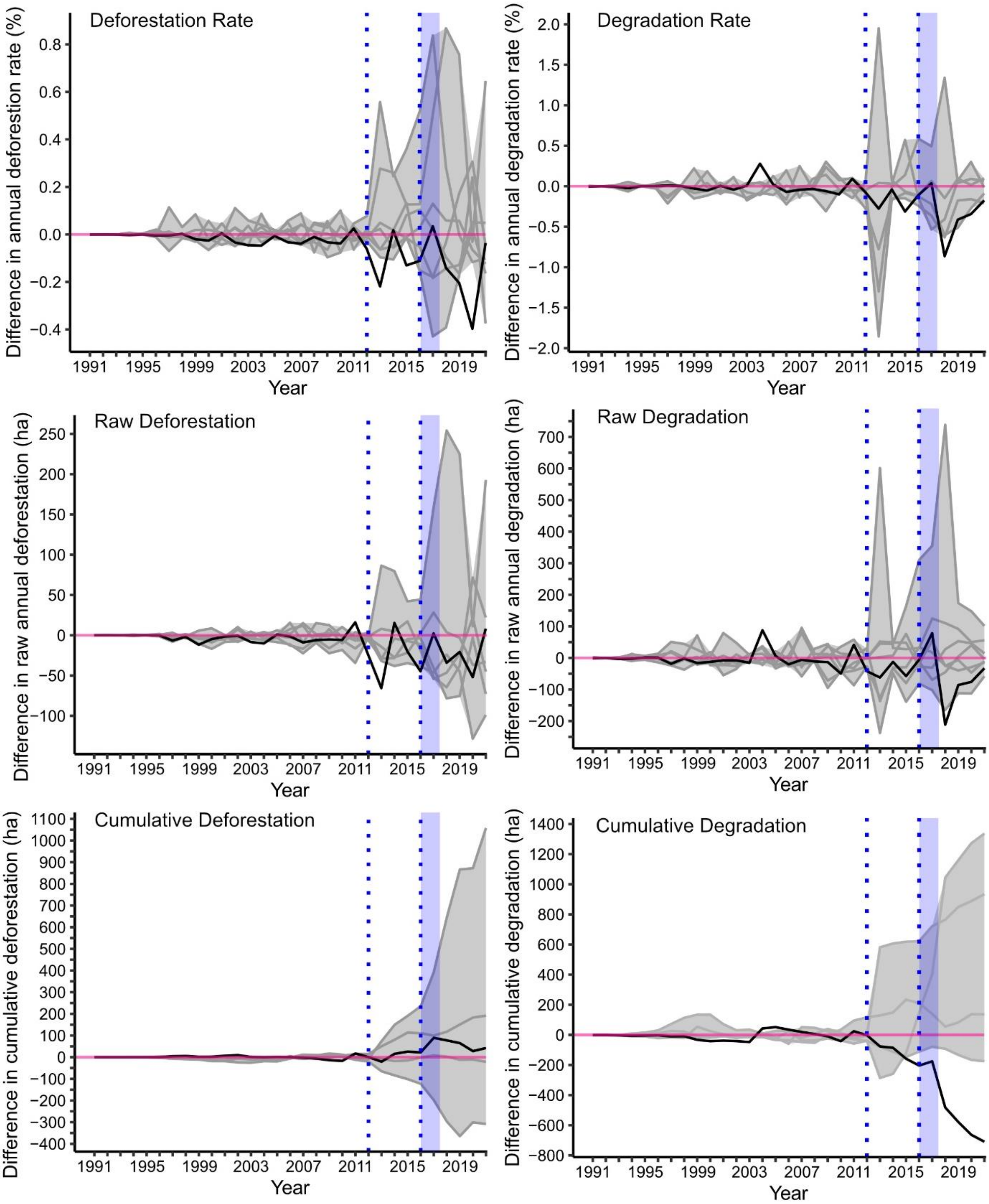


Figure 20: Assessing the significance of our results using placebo tests to quantify the range of noise in the estimation method. In the placebo tests each control basin in the donor pool was falsely assigned treated status and a synthetic control constructed for each. Grey lines represent the difference in outcomes between each false-treated basin and its synthetic control (only pairs where the synthetic control is an acceptable match to the false-treated unit are included, see Methods). The range of values from the placebo tests thus represents the statistical noise in estimation post-intervention (shaded grey area). The difference in forest loss outcomes between Bemainty and its

synthetic control is shown in black. A strong significant effect is indicated where the black line falls outside the shaded grey area. The dotted blue lines indicate the onset of mining in 2012 (left) and the start of the mining rush in 2016 (right). The light blue shaded area indicates the duration of the peak mining rush. Results are from the primary analysis using drainage basins from the CAZ as the donor pool (see Supplementary Figure 3.12 for results from the wider analysis).

Lemur populations

Over six weeks between October and November 2019, we repeatedly surveyed five transects, stretching from villages in the Bemainty valley into the surrounding forest (see Methods). In total, we recorded 735 observations of 10 different lemur species. We made 252 visual observations of ten species, and 483 auditory observations of three species. The most common species recorded were the critically endangered Indri (472 observations), followed by the critically endangered Black and White Ruffed lemur (V*arecia variegate;* 184 observations). All of the lemur species recorded during the survey are known to occur in the area (Goodman, Raherilalao and Wohlauser, 2018)

Neither R.H nor independent researchers visiting the site at the peak of the rush witnessed bushmeat openly on sale (Rosey Perkins, *pers comm;* Pardieu, 2019) but five lemur traps, for small-bodied lemurs were discovered during the survey.

Table 2: Number of lemurs recorded during 2019 surveys in the Bemainty valley.

Species	Number of auditory encounters	Number of visual encounters
Avahi laniger Eastern wooly lemur	0	2
Cheirogaleus crossleyi Furry-eared dwarf lemur	0	1
Eulemur fulvus Common brown lemur	2	26
Eulemur rubriventer Red-bellied lemur	0	13

Hapalemur griseus Eastern lesser bamboo lemur	0	5
Indri indri Indri	332	140
Lepilemur mustelinus Weasel sportive lemur	0	6
Microcebus lehilahytsara Goodman's mouse lemur	0	1
Propithecus diadema Diademed sifaka	0	23
V*arecia variegata* Black and white ruffed lemur	149	35
Total	483	252

Interview data

Of the 73 respondents interviewed, 29 identified themselves as miners and 44 as farmers. All farmers except one were interviewed in the four established villages of Bemainty, Sahananto, Ambanany Sahambato, and Sahamatra. All miners interviewed were found in the temporary settlements of Antananarivo and Milliard, which were constructed during the mining rush. As such, the classification of farmers and miners broadly aligns with the distinction between local residents and migrants, although the boundaries are blurred and there are some exceptions. Local residents can, and did, engage in mining, which means some self-confessed miners may also be local residents. However, most of the local residents (i.e., from the established villages) who reported directly engaging in mining (eight of 43) still identified themselves as farmers. Perhaps because mining was performed on a part-time or temporary basis alongside agriculture. Self-confessed farmers living in the established villages may also be migrant miners, as evidenced by the respondent in Sahananto who arrived in the first wave of mining and decided to remain. Nonetheless, the broad alignment between farmers and local residents, and miners and migrants, becomes apparent in the interview responses.

Most farmers interviewed (66%; N = 44) stated that the environment had changed or degraded since 2016. Of these 65% cited mining as a cause of environmental degradation.

"The forest here has been destroyed by people who mine sapphires. They are migrants." (Farmer, Bemainty).

Conversely, miners claimed that locals were responsible for deforestation in the area and that their impacts are comparatively much smaller:

"We are accused that we cut the forest but it is not the case, we use very little trees compared to the local community and we do not burn the forest" (Miner, Antananarivo).

Both miners and farmers reported that trees were harvested for firewood or construction materials. However, members of both groups emphasized that they do not cut mature trees, or they only use dry wood as firewood. No-one reported cutting trees for charcoal production. Only five respondents mentioned deforestation for shifting agriculture (two miners and three farmers).

While some respondents (34%) said they or others hunt bushpig or birds, no respondents (miners or farmers) reported hunting lemurs themselves. However, four farmers claimed that lemurs were hunted in the area. Both miners and farmers stated that it was *fady* (taboo) to hunt and eat Indri. Many miners emphasized that they do not hunt at all and some explained that they must respect the fady in order to find sapphires.

"Miners do not hunt. We are here in Antananarivo for sapphire mining, not for hunting. And the presence of Indri indri brings us a good luck for finding sapphires so we do not kill them." (Miner, Antananarivo)

When asked about their perceptions of sapphire mining, many farmers (55%) described the negative socio-economic impacts of the rush on the local community and/or the lack of benefits. Farmers described increased crime and insecurity, rising costs of staple foods, and declining water quality and availability, which affected rice production.

"It attracted bandits to the area. As an example, the chief of Bemainty village was shot by bandits and died. The rice production is worse because of the sapphire rush. The miners use water for extraction so we do not have water for our ricefields." (Farmer, Bemainty)

"Sapphire activity destroys the environment in this area. More people means more dirt. People defecate everywhere. Many people died during sapphire mining. People said that it brings positive benefits but where is that now? You can see how poor we are here." (Farmer, Bemainty)

Most miners talked about the potential to make money from sapphire mining. Yet, many stated that sapphires were becoming harder to find and several miners (including farmers who participated in mining) reported not finding any sapphires at all. See Supplementary Results 3 for more responses.

Discussion

We found no evidence to support claims that the high-profile mining rush at Bemainty had a substantial negative impact on the surrounding forests. We show that the presence of 10,000-25,000 miners did not cause more deforestation or forest degradation that we estimate would have happened from other causes in the absence of mining. Additionally, field data collected three years after the rush shows that apparently healthy populations of critically endangered lemur species (Indri and Black and white ruffed lemur) were present in the area. Here, we explore possible explanations for the limited impacts of the rush on the surrounding forests and lemur populations. We then evaluate the main trade-offs of mining at Bemainty and reflect on the wider implications of these findings for understandings of ASM. We finish with a call for more robust, interdisciplinary evaluations of the impacts of ASM.

Limited impacts of the mining rush on the surrounding forests

We suggest that four main factors contributed to the negligible impact of the mining rush on deforestation and degradation. These are: the geological characteristics of the deposit, the legacy of past forest clearance, miners natural resource use, and the relatively larger footprint of deforestation for agriculture.

First, miners at Bemainty were exploiting a secondary sapphire deposit, where gems eroded from a host rock had been deposited in the alluvial gravels of the streambed (Giuliani *et al.*, 2020). These geological characteristics confined mining activity to a narrow ribbon along the valley bottom and restricted the lateral spread, limiting the amount of land clearance. This echoes findings from other case studies which found that the spatial distribution of deposits was a key determinant of the severity of deforestation in mining areas (World Bank, 2019).

Second, the miners did not need to clear much forest for mining as much of the fertile valley floor had already been cleared for agriculture by local communities long before the rush (Supplementary Figure 3.5). The Ambodipaiso valley, where mining began in

2012 and returned in 2017, had been cleared since the 1970's (Supplementary Figure 3.5; Harper *et al.*, 2007). The Antananarivo valley, where the mining rush began in 2016, had been partly cleared for shifting agriculture by November 2013 (Supplementary Figure 3.1; Pardieu *et al.*, 2017). However, while mining was restricted from spreading laterally, it did spread the length of several valleys and there appears to have been some mining-induced deforestation as activity spread north out of the Antananarivo valley in 2017 (Figures 18 and 22).

Third, the impact of miners harvesting timber for firewood and construction materials (Villegas *et al.*, 2012; Macháček, 2019) was likely small-scale and limited. In interviews miners stressed that they preferred to collect dry wood for firewood, only harvested small trees for construction materials, and did not engage in charcoal production. The former aligns with independent reports from the field (Pardieu, 2019), and previous studies showing that rural Malagasy prefer to collect deadwood or harvest single branches for firewood, either by choice (for ease), or because of customary rules (Kull, 2002; Casse *et al.*, 2004). These small-scale impacts from selective harvesting are unlikely to have caused substantial deforestation but may nonetheless have affected forest structure and therefore biodiversity (Allnutt et al, 2013). However, we were unable to detect small-scale impacts in our degradation analysis which is based on 30m resolution satellite imagery.

Fourth, clearance for agriculture is a major driver of forest loss in the CAZ (Tabor *et al.*, 2017; Goodman, Raherilalao and Wohlauser, 2018; Hewson *et al.*, 2019) and its large relative footprint likely contributes to the non-significant impacts of the mining rush. Our results show that mining-related forest loss in the Bemainty basin did not exceed estimated counterfactual loss from other causes (predominantly agriculture) in the absence of mining. This means that 10,000-25,000 people mining in the valley did not cause more deforestation than several hundred local people (Supplementary Table 3.4) clearing land for agriculture. This highlights the considerably lower per capita deforestation footprint of artisanal mining compared to agriculture in this study area, raising interesting questions for wider rural development policy, which has typically overlooked the former and prioritised the latter (Hilson and McQuilken, 2014; Hilson, 2016). In fact, our results suggest that the mining rush may potentially have reduced land clearance for agriculture, contrary to findings from elsewhere (eg. Sierra Leone, Maconachie and Binns, 2007; Sakaraha, Cook and Healy, 2012). Previous research, from Madagascar and elsewhere, has shown that farming and mining are often complementary activities, with farmers engaging in mining during quieter agricultural periods (Maconachie and Binns, 2007; Kamlongera, 2011; Hilson and Garforth, 2012; Stoudmann *et al.*, 2021). At the height of the mining rush many farmers may have abandoned farming to mine, resulting in fewer new fields being cleared (this was reported in a mining area north of the CAZ; Stoudmann *et al.*, 2021, and in other countries; Boadi *et al.*, 2016; Poignant, 2023). Other farmers may have been less willing to invest in clearing new land due to the increased insecurity, fear the land would be occupied by miners, or because there was less water available for irrigating rice fields (Supplementary Results 3; Baffour-Kyei *et al.*, 2021).

Evidence that lemur populations at Bemainty remain healthy

Tullis (2019) claims that the mining rush threatened populations of endangered lemurs through hunting and destruction of forest habitat. Elsewhere in Madagascar ASM has impacted wildlife populations by eroding customary practices and taboos (*fady)* governing natural resource use (Walsh, 2003; Canavesio, 2009; Cook and Healy, 2012). In the south of the CAZ, Jenkins *et al* (2011) linked the expansion of artisanal gold mining and influx of migrant miners to a weakening of *fady* protecting the endangered Indri, resulting in increased hunting.

However, our results suggest that this was not the case at Bemainty. No respondents (miners or farmers) reported hunting lemurs, and most stated that it is *fady* to hunt or eat Indri. This taboo is fortified by the belief amongst miners that the presence of Indri brings good luck for finding sapphires. While such interview questions are highly vulnerable to social desirability bias, this evidence is supported by the apparently healthy populations of Indri recorded during the lemur survey (Table 2). Together this suggests that at Bemainty, traditional taboos protecting the species appear to have withstood the pressures of human mobility and sudden population growth. The relatively high numbers of Black and white ruffed lemurs observed during the study indicates hunting pressure is generally low, as this species tends not to be taboo in eastern Madagascar (Jenkins *et al.*, 2011) and has been extirpated due to hunting in other parts of the CAZ (Schmid and Alonso, 2005).

However, there are several important caveats to these results. Firstly, lemur hunting is a sensitive topic in Madagascar as it is widely known to be illegal (Jenkins *et al.*, 2011; Borgerson *et al.*, 2016). Therefore, respondents may not have answered questions about lemur hunting truthfully, meaning it could be more prevalent than reported in direct questioning (Razafimanahaka *et al.*, 2012). Secondly, the mining rush may have attracted external hunters to the area by improving access and highlighting the lack of law enforcement (as reported in Cameroon, Denison Mundi 2022; and DRC, Spira *et al,* 2019). Hunting by other actors besides miners and farmers is not well captured in the interviews (respondents who reported hunting by others may not have been willing, or able, to identify the perpetrators). Thirdly, we are not able to evaluate the impact of the rush itself on lemur populations as we do not have a robust counterfactual for populations in the absence of mining. Nor do we have before-after data. Furthermore, our lemur survey was conducted three years after the rush when mining activity was much reduced. Lemur populations could have been initially impacted by mining but recovered by 2019 as the mining activity declined. Nevertheless, the combined evidence from the lemur survey, interviews, and independent reports from site visits suggests that hunting pressure from the mining rush at Bemainty was low.

Trade-offs of mining at Bemainty

ASM provides a vital source of income and employment in Madagascar, but in some places this has brought serious environmental costs (Tilghman, Baker and Deleon, 2007; Cook and Healy, 2012; World Bank, 2019). However, our results suggest that at Bemainty, the economic contributions made by ASM did not involve substantial trade-offs to forest cover or lemur populations. For a time, ASM at Bemainty supported the livelihoods up to 25,000 people (mostly migrants from outside the area). While we do not have data on average mining incomes from Bemainty, artisanal miners in

Madagascar can generally find enough small stones to cover basic needs while larger finds can improve livelihoods (although unlucky miners can become further impoverished; Lawson, 2018; Cartier, 2009; Walsh, 2012). Income from mining or related services can be used to invest in land, livestock, business, or children's education, helping to alleviate poverty and strengthen livelihoods (Supplementary Results 3; Lawson, 2018; Stoudmann et al., 2021; although c.f. Walsh, 2003).

However, our qualitative data reveal that the uncontrolled nature of the mining rush brought other concerning trade-offs which undermined the economic benefits (see Supplementary Results 3). The mining rush increased crime and insecurity, and poor sanitation increased the spread of disease. Local food security was negatively impacted as the mining rush affected rice production and inflated the price of basic goods (Supplementary Results 3). This is evidence is supported by independent reports from site visits (Perkins, 2016, 2017, Pardieu *et al.,* 2017). These negative impacts affected both migrant miners and the local community. However, for local farmers these costs were perceived to outweigh the benefits of mining, which were considered to be unequally shared with the local community (accruing mostly to the migrant miners). These poor conditions may also have pushed migrant miners to leave, with or without a valuable find (insecurity was reported a reason for miners leaving a site in north-west Madagascar; Walsh, 2012).

The mining rush also compromised the integrity of the protected area. Mining within protected areas is illegal in Madagascar but authorities often lack the capacity and/or political will to keep artisanal miners out (Tilghman, Baker and Deleon, 2007; Cook and Healy, 2012). The persistence of such a high-profile, illegal activity within a protected area can have serious implications. It can fuel corruption, weaken local governance (Duffy, 2007) and undermine the authority of conservation restrictions; potentially encouraging other illicit activity and reducing conservation effectiveness (Cabeza *et al.*, 2019; Jones *et al.*, 2019b; Tullis, 2019). Therefore, while we show that the mining rush did not directly increase forest loss, it could have had indirect impacts which were potentially more impactful.

ASM can result in other environmental trade-offs which we were unable to assess in this study. ASM can increase erosion and siltation of waterways (Rajaee *et al.*, 2015; Lobo *et al.*, 2016). Indeed photos from the site show that the mining caused significant soil disturbance, increased turbidity, and disrupted water flow (Figures 3 and 18). However, we were unable to assess the impacts of this on freshwater biodiversity and ecosystem services. We were also unable to assess the impacts of the mining rush on wider biodiversity.

Contextualising our results and implications for future research

Our study is the first to apply counterfactual methods to evaluate the environmental impact of ASM. We combine this approach with lemur surveys and interviews to gain a more comprehensive understanding of the impacts of ASM at Bemainty. Important insights from interviews into the concerning social impacts of the mining rush highlight the importance of combining satellite-derived analyses of environmental change with information collected on the ground.

We hope our results emphasize the need for more robust evaluations of the impacts of ASM under different conditions. The results shown here are just one case study, and our findings will not necessarily apply to other mining rushes, within Madagascar or globally. The limited forest impacts of the mining rush at Bemainty likely resulted from context-specific factors, including geological characteristics, land-use history, the large footprint of alternative land uses; and the low-tech mining methods which did not require chemical inputs. Under different conditions, ASM can have serious environmental impacts (Asner and Tupayachi, 2017; Espejo *et al.*, 2018; Barenblitt *et al.*, 2021a).

Future studies can improve on our approach by using high-resolution data to capture smaller-scale forest impacts, and incorporating a wider range of social and ecological data collected in the field (i.e., species data, water and soil sampling, household surveys or interviews), to gain a broader understanding of the benefits and trade-offs of mining.

Conclusion

Contrary to media claims, we found that an artisanal mining rush involving tens of thousands of people within a protected tropical forest in Madagascar did not increase forest loss. Instead, we found that ASM at Bemainty had a much smaller per-capita deforestation footprint than shifting agriculture, which remained the dominant driver of forest loss in the study area. Additional evidence from interviews and a lemur survey conducted in the field support the findings of limited trade-offs to forests and lemur populations from the mining rush.

While this is just one case study, these findings emphasize that the environmental impacts of ASM are highly heterogeneous and should be considered relative to other land uses. There is a need for more case-study evaluations using robust methods to build an evidence-base of the impacts of ASM under different conditions. This would help to inform policy responses to ASM which are evidence-based, proportionate, and which focus on maximising the benefits and minimising the trade-offs.

Methods

Overview

To evaluate the impact of the mining rush at Bemainty on the surrounding forest, we need to estimate how much forest loss would have occurred in the absence of mining, i.e. the counterfactual. We estimate counterfactual outcomes using a synthetic control; a weighted combination of several existing control units weighted to be as similar as possible to Bemainty in characteristics which influence deforestation, and pre-mining forest loss trends. Then, we compare observed forest loss at Bemainty to counterfactual outcomes in the synthetic control, using placebo tests to assess significance. We run the analysis for three different measures of deforestation and degradation, at two scales of analysis. We draw upon additional field data (interviews and lemur surveys) to contextualise our findings and further explore the impacts of mining on forest biodiversity.

Study area

Sapphires were first discovered near the village of Bemainty within the rainforests of the Coridor Ankeniheny-Zahamena (CAZ) in eastern Madagascar in April 2012. Soon over 1,000 people were mining in the Ambodipaiso valley, north of Bemainty village (Figure 18; Desbureaux, 2012; Perkins, 2017). Visual analysis of RapidEye satellite imagery shows that by November 2013, mining had spread the ~3.5km length of the valley (Supplementary Figure 3.1). Some riverbank disturbance is still visible in June 2015, but the duration of the active mining phase is unknown.

This initial phase received little media attention at the time as it was mostly eclipsed by the much larger sapphire rush in the forest 40km to the south, in the commune of Didy (Desbureaux, 2012). The rush near Didy, involving 10,000- 40,000 people, began in April 2012 but was relatively short-lived. In July 2012, the government deployed the army, and miners and foreign buyers were evicted from the area (Desbureaux, 2012; Pardieu and Rakotosaona, 2012). Despite these efforts mining persisted, albeit at a smaller-scale, at both Didy and Bemainty (Madagascar Tribune, 2013).

In September 2016, sapphires were discovered in a second valley east of Bemainty village by gold miners prospecting on land cleared for *tavy* (shifting agriculture, visible in Supplementary Figure 3.1; Perkins, 2016; Pardieu *et al.*, 2017). Word of this discovery quickly spread and tens of thousands of miners from across the country flocked to the area to mine. By mid-October 2016, an estimated 10,000- 25,000 people were working in the valley known as Antananarivo, after Madagascar's busy capital city (Canavesio and Pardieu, 2019). Efforts of the authorities to disrupt the rush (by establishing road blocks and evicting foreign buyers from the nearest trading town; Madagascar Tribune, 2016) had little effect. By February 2017 an estimated 30,000 people were still working in the area (Pardieu *et al.*, 2017). As the rush developed, mining spread north into several tributary valleys. In May 2017 the epicentre shifted back to the original site in the Ambodipaiso valley (Perkins, 2017). Over the following months many miners left the area and by October 2017 only approximately 400 miners remained, marking the end of the rush (Pardieu, 2019). Although mining continued at a much smaller scale until at least October 2019.

Mining at Bemainty was labour-intensive, informal and illegal, as it occurred within a protected area (*Code Minier*, 2005; Perkins, 2016). Miners dug pits 2-3 m deep near the river and sieved excavated gravels in the stream (Perkins, 2016; Pardieu *et al.*, 2017). While most miners started out independent, by February 2017, most were working for sponsors and using more efficient water pumps and hoses to sieve the gravels (Perkins, 2017).

Here, we take 2012, the year when artisanal gem mining first began at Bemainty, as the year of the 'intervention', as this is when mining-related forest loss may have begun. Although the large mining rush did not start until 2016, using the period from 2012 allows us to explore the impacts of mining at different scales. Therefore, in our results we mark both the onset of mining in 2012, and the start of the rush in 2016.

Unit of analysis

We use the Level 9 drainage basins from HydroBASINS as our unit of analysis (Lehner and Grill, 2013). HydroBASINS is a global map of watershed boundaries and drainage basins at hierarchically nested scales, from the continental to the local, derived from digital elevation models. At each smaller scale, drainage basins are sub-divided into their four largest tributary basins with an individual area of at least 100km^2, and five smaller inter-basins (Lehner and Grill, 2013).

Drainage basins are an appropriate unit at which to measure the impacts of the mining rush, as basin geography influences the distribution of gemstones (i.e. selection to treatment) and forest loss outcomes. The gems mined at Bemainty are secondary deposits which have been removed from a host rock within the catchment via erosion or weathering, been transported and deposited within river sediments in the valley bottom (Giuliani *et al.*, 2020). Drainage basin geography may also restrict the potential spread of impacts, as miners may be less inclined to travel over watershed ridges to harvest materials.

We chose to use the Level 9 basins (the second smallest in this area), as we considered this best captured the hypothesised scale of impacts (Supplementary Figure 3.2). Survey data from 418 villages in Masoala National Park in north-eastern Madagascar shows that on average, villagers would travel up to a maximum 1.9 hours to collect forest products (Allnutt *et al.*, 2013). We applied this threshold to map the potential impact zone around the two mining valleys at Bemainty and found it best matched the scale of the Level 9 basin (Supplementary Figure 3.2). While this potential impact zone is likely an overestimate, as short-term migrant miners may be especially unlikely to travel far from the mine site to access resources, we wanted to ensure we captured all potential impacts within our treated unit and avoided spillovers into neighbouring control units.

Selection of covariates

We chose a selection of biophysical, demographic, and geographic covariates which have been shown to influence deforestation and degradation in Madagascar, and globally (Supplementary Table 3.1). Covariates must represent baseline conditions unimpacted by the intervention of interest (Abadie, 2021). Therefore, for our time-variant covariates we use values prior to onset of mining in 2012. Our covariates are: population density in 2011, population growth rate 2001-2011, mean distance to settlement, mean elevation, mean slope, mean annual precipitation, mean distance to cart track, mean distance to road, mean distance to river, mean distance to forest edge in 2011, percentage forest cover in 2011 and area of the drainage basin (Supplementary Table 3.1). These covariates capture aspects of accessibility, demand for land and forest products, suitability for agriculture and vulnerability to natural disturbances, which influence forest loss in Madagascar (McConnell, Sweeney and Mulley, 2004; Brinkmann *et al.*, 2014; Burivalova *et al.*, 2015; Andriatsitohaina *et al.*, 2020). Constructing a synthetic control which is as similar as possible to Bemainty in these factors controls for the influence of these confounding factors on forest loss and consequently helps to isolate the impact of the intervention.

Selection of the donor pool

The synthetic control is constructed from several control units selected from a pool of potential control units known as the donor pool (Abadie, Diamond and Hainmueller, 2010; Abadie, 2021). If control units in the donor pool are already similar to the treated unit in a few key factors, this can help improve the accuracy of the synthetic control (West *et al.*, 2020; Fick *et al.*, 2021).

Units in the donor pool should not have experienced any intervention or event over the study period which the treated unit would not also have experienced in the absence of the intervention, as this could cause outcomes to diverge from the counterfactual (Ferraro and Hanauer, 2014; Abadie, 2021). This complicates the selection of control units in our study area as there are multiple Protected Areas with different implementation dates and degrees of management. The Bemainty gem rush occurred within the forests of the Coridor Ankeniheny-Zahema. The CAZ was granted temporary protected status in 2005 and formally gazetted in 2015. As the transition to formal protection occurred after mining began at Bemainty, this change in status (and theoretically management) could potentially confound the impact of the mining rush (Ferraro and Hanauer, 2014). For example, conservation actions reducing forest loss from other causes at the same time as the mining rush could falsely indicate the mining rush had reduced forest loss. In this context, the most appropriate control units are those which experienced the same change in circumstance, but which did not have a mining rush (Abadie, 2021). Therefore, our primary analysis only includes drainage basins which intersect the CAZ in the donor pool (N = 47).

We then filtered this selection to only include basins with similar forest cover to the Bemainty basin (96%) before the intervention (i.e. in 2011). We chose 70% forest cover as the threshold for inclusion as this allowed us to include all the mostly forested drainage basins in the CAZ, striking an appropriate balance between the number of basins included and the degree of similarity. Next, we removed basins known to contain other gem mining sites (i.e., Didy, Figure 18), using the database compiled in Chapter 3. We also removed basins with more than 10% overlap with another Protected Area, where outcomes may be influenced by different conservation management, implemented at different times. These other Protected Areas include Andasibe-Mantadia National Park, Analamazoatra Special Reserve and the biodiversity offsets associated with Ambatovy mine, which have been effective at reducing deforestation (Chapter 2). Where less than 10% of a basin intersected another Protected Area, we edited the boundary of the basin to exclude the overlapping section. This, for example, allowed us to retain a large, and potentially well-matched basin in the centre of the CAZ where 6.5% overlapped with the Ankerana biodiversity offset. In Chapter 2 I showed that the deforestation reductions achieved within the Ankerana offset did not spillover into the surrounding forests, so we did not need to establish a wider zone of exclusion. This pre-matching filtering left eight basins in the donor pool from the CAZ in the primary analysis (Figure 21).

Eight basins is a small donor pool, particularly for the placebo tests used to assess significance. Therefore, to increase the size of the donor pool and test the robustness of our results, we ran a second analysis sampling control basins from a wider area (West *et al.*, 2020) - the former province of Toamasina (Supplementary Figure 3.3). Using the same filtering criteria as above we identified 13 forested basins to comprise the donor pool. This donor pool comprises eight basins from the CAZ as before plus five additional, unprotected forested basins from the wider province. Whilst the CAZ is officially protected, resources and conservation activities are thinly spread (Hewson *et al.*, 2019). Therefore, unprotected forests are likely to represent a more appropriate counterfactual for the CAZ than long-established and well-managed protected areas. Unfortunately, this still limits the size of the donor pool as there are very few drainage basins in the study area with over 70% forest cover which are unprotected. However, widening the selection criteria would risk including basins with substantial differences which could confound our analysis.

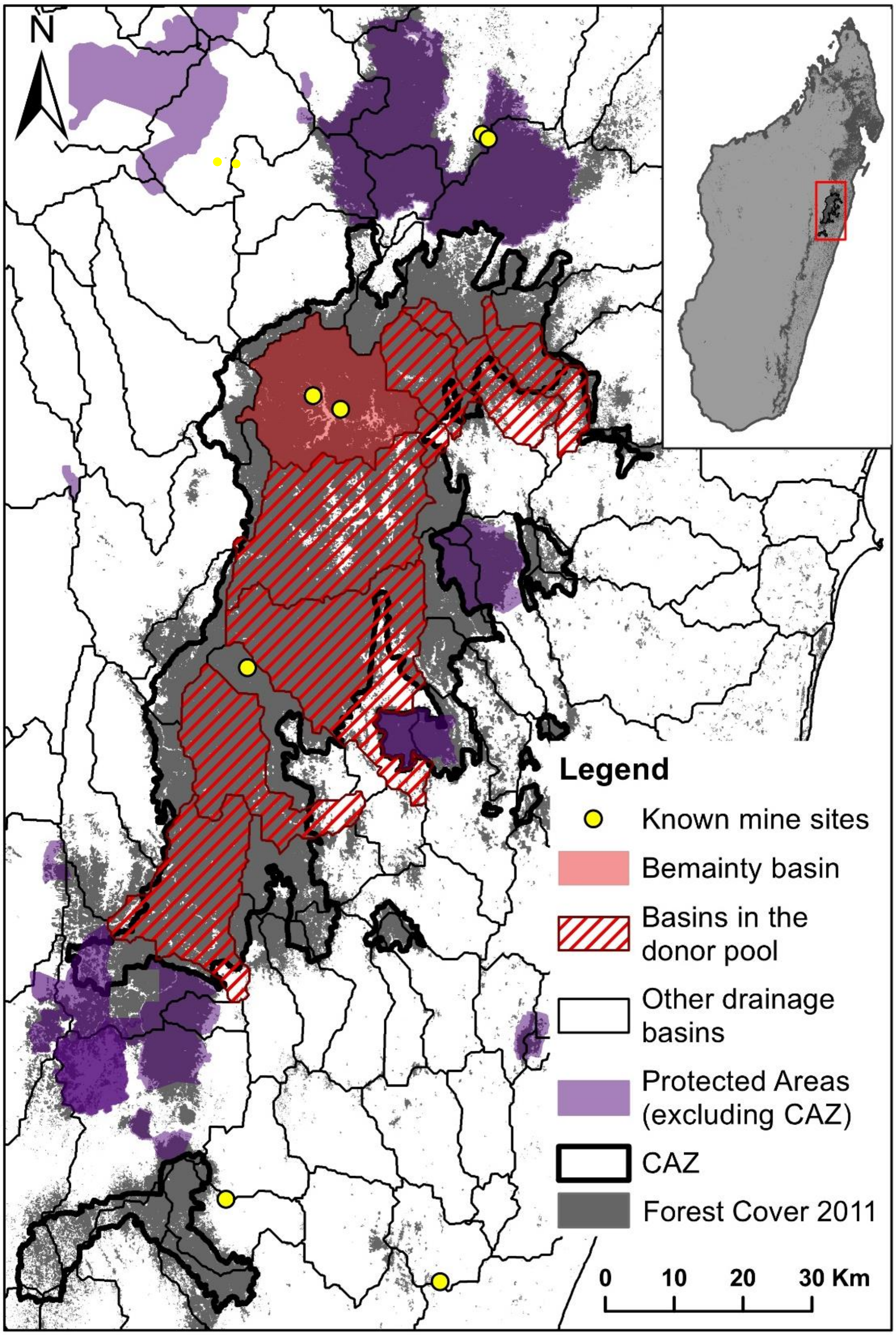


Figure 21: Map shows the treated Bemainty basin (red) and the eight drainage basins included in the donor pool (red hashed) for the primary analysis focussed on the CAZ (outlined in black). Drainage basins which overlap with Protected Areas or biodiversity offsets (shown in purple), or which contain other known gem mining sites (yellow points) were excluded. Yellow points in the Bemainty basin show the Ambodipaiso (left) and Antananarivo (right) mining valleys. See Supplementary Figure 3.3 for a similar map showing the donor pool drawn from the ex-province of Toamasina in the wider analysis.

Outcome variable

We ran our synthetic control approach for two different outcomes – deforestation and forest degradation (together termed forest loss) – at each scale of analysis.

Data were derived from the Tropical Moist Forests product (TMF) of Vancutsem *et al* (2021)*.* The TMF dataset maps the annual extent and land cover changes within tropical moist forests globally from 1990-2021 at 30 m resolution. Loss of canopy cover in a given year is defined as either deforestation or degradation based on the duration of clearance. Deforestation is defined as the long-term conversion of forest to non-forested land, lasting over 2.5 years. Degradation is considered temporary loss of canopy cover, lasting less than 2.5 years, after which there is some forest recovery (Vancutsem *et al.*, 2021).

We use the Deforestation Year, Annual Change, Transition Map and Annual Disruptions TMF data products. Following Vieilledent *et al* (2018) all layers were masked to a map of forest cover in Madagascar in 1990 (Harper *et al.*, 2007; Vieilledent *et al.*, 2018). This map is based on a national-scale remote sensing study and is therefore considered a more accurate representation of the forest present in Madagascar at the start of the study period than a global study (the difference is shown in Supplementary Figures 3.4 and 3.6).

Our deforestation outcome variable is the amount of deforestation per basin, per year obtained from the Deforestation Year data. We do not use the equivalent Degradation Year data as this only represents the *first year* degradation was observed in a pixel. However, pixels can be degraded multiple times during the study period. The gem rush at Bemainty occurred in the valley bottom, close to the village, where the adjacent forest is more likely to have been degraded earlier (721ha, 2% of forest in the Bemainty basin was degraded 2-3 times over the study period). To avoid missing degradation which occurred on previously degraded, then recovered, land we adapt the raw Annual Disruptions dataset to obtain annual data on degradation events. The Annual Disruptions dataset contains the number of times a disruption (defined as an absence of canopy cover) was observed per pixel (for pixels forested in 1990) in all satellite images from that year. Using Google Earth Engine we reclassify the data to a binary measure of whether a disruption was observed (1) or not (0) each year. Consecutive years of disruption observations represent the duration of the loss of canopy cover. However, we are primarily interested in the year of clearance (i.e. when each degradation event began). Therefore, where there are a series of disruptions spanning consecutive years, we retain the first but remove all subsequent observations in that episode (by reclassifying to zero). Then, we masked this layer to pixels classed as degraded in the final Transition Map classification. Finally, we calculated the area of degradation events per basin, per year as our outcome variable. By capturing pixels which are cleared for a few years and then show regrowth, a pattern which can be repeated multiple times during the study period, our measure of degradation captures the dynamics of shifting agriculture. This allows us to compare the impacts of the mining rush to the impacts of the most common alternative land use in the study area.

We measured each outcome in three different ways and repeated the analysis for each: 1) the annual deforestation/degradation rate as a percentage of forest cover present at the start of each year; 2) raw hectares of deforestation/degradation; 3) cumulative hectares of deforestation/degradation. The TMF data does not provide a specific set of annual forest cover maps so to obtain these we reclassified the TMF Annual Change datasets to only include the forest classes, including forests at any successional stage (i.e. undisturbed tropical moist forest, degraded tropical moist forest, and forest regrowth classes; see Supplementary Methods 3).

Synthetic control

We use the synthetic control approach to estimate counterfactual forest loss at Bemainty in the absence of mining, and consequently infer the impact of mining. The synthetic control is a weighted average of several existing control units in the donor pool, weighted to maximise similarity to the treated unit in covariates and pre-intervention forest loss outcomes (Abadie and Gardeazabal, 2003; Abadie, Diamond and Hainmueller, 2010; Supplementary Methods 3). It is based on the rationale that in cases such as this, where the intervention is applied to a single area and where there are few appropriate control units available, a weighted combination of controls may represent a better counterfactual than any individual control (Abadie and Gardeazabal, 2003; Roopsind, Sohngen and Brandt, 2019). Weighting the control units to maximise similarity in covariates known to predict anthropogenic forest loss helps to control for the influence of these confounding factors, while a similar pattern of pre-intervention outcomes helps to control for the influence of unobserved factors (Abadie, Diamond and Hainmueller, 2010; Sills *et al.*, 2015; see Chapter 1). Consequently, outcomes in the synthetic control in the post-intervention period can represent a credible counterfactual for outcomes in the treated unit in the absence of the intervention.

We construct our synthetic controls using the Synth package in R (Abadie, Diamond and Hainmueller, 2011). The study period is 1991-2021 (1991-2011 pre-intervention and 2012-2021 post-intervention). The quality of the synthetic control was assessed through the similarity in pre-intervention outcomes between Bemainty and the synthetic control. We used the Mean Square Prediction Error (MSPE) in the pre-treatment period as a measure of similarity and visually compared plotted outcomes to check for bias (Figure 19; Abadie, Diamond and Hainmueller, 2010; West *et al.*, 2020).

Following West et al (2020) and Abadie, Diamond, and Hainmueller (2015), we conducted in-time placebo tests as a validation exercise and robustness check. We falsely-assigned treatment (i.e., the start of mining) to 2009 (three years before the actual start of mining in 2012) and constructed a synthetic control using 1991-2008 forest loss outcomes. If the resulting synthetic control closely reproduces outcomes in Bemainty between 2009 and 2012 (i.e., a period without mining), this indicates that the method can produce a credible estimates of forest loss in Bemainty without mining in the real post-intervention period (i.e., the counterfactual; Abadie, 2021). Although this synthetic control will likely differ from that constructed in the main analysis (as using the full 1991-2021 pre-intervention data will likely change the weightings), it still presents a useful validation of the method (West *et al.*, 2020). In-time placebos also act as a robustness check. If a similar magnitude effect is demonstrated after false-treatment (i.e., 2009) compared to real treatment (i.e., post 2012), the latter is likely not attributable to the intervention (Abadie, Diamond and Hainmueller, 2015). Results from these tests are presented in Supplementary Figures 3.10 and 3.11.

Assessing the significance of our results

To determine whether any difference in post-intervention outcomes between Bemainty and the synthetic control is a significant effect of mining we use 'in-space' placebo tests (Abadie, Diamond and Hainmueller, 2010). We iteratively assign false treatment in 2012 to every control basin in the donor pool, construct a synthetic control for each, and compare outcomes between each false-treated unit and its synthetic control (plotted in grey in Figure 20). As these false-treated (control) basins did not experience a mining rush, any difference in outcomes over the post-intervention period results from unobserved heterogeneity and can be considered noise in the synthetic control estimation (West *et al.*, 2020). If the difference in outcomes between Bemainty and its synthetic control exceeds this range of noise, the effect of mining on deforestation and degradation can be considered significant.

For the synthetic controls constructed in the placebo tests to be an appropriate comparison, they must closely reproduce pre-intervention outcomes in their matched false-treated unit. Following Abadie *et al* (2010) and West *et al* (2020), we remove pairs where the MSPE is over 5x the MSPE of the synthetic control for the Bemainty basin.

Field data collection

One of our team (R.H) visited Bemainty in 2019 to conduct a lemur census and semi-structured interviews with people in the area. Data were collected over a six-week period between October and November 2019. We obtained ethical approval (IRB-2019-513, University of San Diego) and permissions from the Ministry of Environment and Sustainable Development (Number 295/19/MEED/SG/DGEF/DGRNE), Conservation International (the management authority for the CAZ), and the local authority. Four community members were recruited to assist with the lemur censuses.

Lemur surveys

Lemur surveys were conducted along five transects (roughly 7km long) from villages in the Bemainty valley into the adjacent forests, along existing paths. Each transect was repeated 5-6 times. In total 27 transect surveys were conducted, covering approximately 189km. Surveys were timed to coincide with peak activity of diurnal lemurs (06:00 – 11:00, and 13:30 – 17:30). The field team also conducted a single night survey on an additional transect around Bemainty village to explore which nocturnal species were present.

We used similar methods and survey effort (eg. area covered) to other surveys of diurnal lemurs conducted in Madagascar (Banks, Ellis and Wright, 2007; Gilles and Reuter, 2014). Transects were walked at a speed of 1 - 2km/hr. We recorded all visual and auditory encounters and noted the time of the encounter, species, and number of individuals.

Semi-structured interviews

We conducted 73 semi-structured interviews in five villages in the Bemainty valley (Figure 22). The purpose of the interviews was to gain a contextual understanding from members of the local community and sapphire miners about the impacts of mining. Interviews were opportunistic and participants were found by knocking on doors. As such, interviews were conducted mostly in the mornings and evenings when people were more likely to be home. Interviews were conducted in the local Sihanaka dialect and only adults over 18 were interviewed. The purpose of the study was explained, and respondents were asked if they were happy to be interviewed.

Respondents were asked a series of questions about their natural resource use and perceptions of local environmental change, mining, tree cutting, and hunting. Respondents were also asked to name their job. There were only three responses: farmer, miner, or farmer and vice-president of an association.

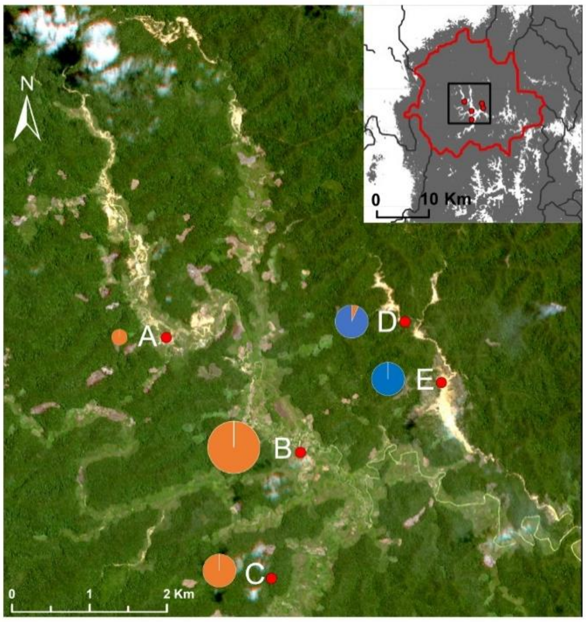


Figure 22: Satellite image of the Bemainty valley from November 2017 (the clearest image available after the mining rush had peaked; Planet Team, 2017) showing the location of villages where interviews were conducted. Pie charts represent the proportion of respondents at each village who identified themselves as farmers (orange) or miners (blue). The size of the pie charts corresponds to the number of respondents. A = Sahananto (2 respondents), B = Bemainty (25), C = Sahamatra, D= Milliard (15), E = Antananarivo (15). Inset map shows the location of the villages within the Bemainty drainage basin (outlined in red).

Chapter 5: Discussion

There is a popular perception that mining, particularly artisanal and small-scale mining (ASM), is inherently environmentally destructive and '*worse*' than alternative land uses (e.g. agriculture; Lahiri-dutt, 2014; Klein, 2022). This perception has influenced attitudes and policies towards mining, feeding discourses which define the range of 'acceptable' policy responses and preclude consideration of alternative, more flexible approaches (Tschakert and Singha, 2007; Hirons, 2011; Klein, 2022). Here, I explore how my research challenges these perceptions, and critically evaluate the policies they prescribe. Drawing on evidence from the literature and this thesis, I argue that these perceptions lack evidence, are disproportionately influenced by the worst cases, and homogenise the impacts of mining which are, in fact, highly heterogenous, depending on the form, scale and technologies used. Like all natural resource uses mining inevitably generates some degree of environmental damage, which needs to be mitigated for the sector to maximise its sustainable development potential. Yet, the main policy responses to the environmental challenges of mining (both industrial and ASM) prescribed by these dominant discourses, have often failed, and improvements are needed. In some places where ASM and conservation interests coincide, alternative, more pragmatic approaches are required to negotiate the complex trade-offs between ASM, conservation, and rural livelihoods.

The following discussion is organised as follows. First, I interrogate these popular perceptions and, drawing on evidence from the Introduction argue that, for ASM, there is a serious lack of evidence to support perceptions that mining is inherently environmentally destructive and worse than alternatives. I then present evidence to contest this narrative from a case study of an artisanal gem rush in Madagascar (Chapter 4). In the next section, I explore how these popular perceptions have influenced discourses defining policy responses to the challenges of mining. I critically evaluate several of the main policies (biodiversity offsetting for industrial mining, formalisation, and certification for ASM) and discuss the conditions and improvements needed for these measures to be effective. I then consider the suitability and feasibility of formalisation and certification schemes to improve the environmental outcomes of ASM in Madagascar – a country which epitomises many of the challenges facing the sector. Finally, I examine some alternative approaches to managing conflicts and trade-offs between mining and conservation more broadly.

It is important to stress that I do not seek to minimise or disregard the environmental damage that mining can bring, but rather challenge narratives that mining is inherently '*worse'* than alternative land uses, which have prevented more constructive engagement with the sector. I also emphasize the need for more evidence on the impacts (both positive and negative) of mining under different conditions, to inform policy which is evidence-based, appropriate, and balanced.

Popular perceptions of mining

Mining is often perceived as intrinsically environmentally destructive and worse than alternative land uses (Lahiri-Dutt, 2014). To illustrate this point, consider a farmer choosing to mine their fields for gold, rather than planting crops. Mining is automatically seen as 'destructive' and 'bad', while farming is considered comparably 'natural' and 'good'.

Mining *can* be more destructive than an equivalent area of an alternative land use but this is not guaranteed. The environmental impacts of mining depend on the target mineral and the geological characteristics of the deposit (which determine the scale, mining methods, and technologies used), the level of capitalisation, local ecological sensitivity, and the impacts of alternative land uses (Espejo *et al.*, 2018; Sonter, Ali and Watson, 2018; Hirons, 2020; Timsina *et al.*, 2022). Open pit mining entails the complete removal of surface vegetation and topsoil, which, for the largest mines, devastates vast areas of land (Dudka and Adriano, 1997; Sonter *et al.*, 2014). This is different to agriculture (even intensive monocultures or plantations), for example, where land retains some (albeit much reduced) carbon storage and biodiversity (Foster *et al.*, 2011; Virah-Sawmy, Ebeling and Taplin, 2014). Abandoned mine sites are also slower and less likely to recover natural vegetation than abandoned farmland (Kalamandeen et al, 2020;

Peterson and Heemskirk, 2001). However, for underground mining, where the ore is mined from tunnels and shafts drilled deep into the rock, much of the surface vegetation and soil remains intact (Altun, Yilmaz and Yildirim, 2010; Farjana *et al.*, 2019). Furthermore, for ASM, while mining may be more destructive than agriculture per unit area, the per capita footprint can be much lower, meaning impacts are less extensive (Chapter 4).

The consequences of these negative perceptions tend to be more pronounced for ASM than industrial mining (here referring to mechanised medium or large-scale operations which possess the required licenses to operate lawfully). For industrial mining the potential economic contributions to government revenue often trump concerns over environmental damage and mining is approved, despite frequent public opposition (e.g., Huff, 2016). However, for ASM (which does not have the same government support) perceptions of the relative impacts of agriculture and ASM, while clearly a simplification (as small-holder farmers are also often vilified as environmentally destructive; Kull, 2000), have had significant implications. In Sub-Saharan Africa these biases helped keep small-holder agriculture at the centre of rural development strategies for decades despite mounting evidence of the declining viability of agriculture and the growing importance of ASM and other non-farm livelihoods (Hilson and Garforth, 2012). This contributed to the marginalisation of ASM within the development discourse and wasted opportunities for early engagement to support and regulate the growing sector (Banchirigah and Hilson, 2010; Hilson and McQuilken, 2014; Hilson, 2016).

Narratives of the environmental destructiveness of ASM, combined with wider narratives of lawlessness, criminality, and chaos (Tschakert and Singha, 2007; Lahiri-Dutt, 2014; Klein, 2022), have strongly influenced policy responses. This has resulted in the stigmatization and marginalisation of the sector, and precluded more constructive engagement (Hirons, 2011; Lahiri-Dutt, 2014). Yet, as I show in the Introduction, there is limited evidence to support these claims. Existing evidence on the environmental impacts of ASM is mostly qualitative or descriptive (e.g., Cook and Healy, 2012; Gandiwa and Gandiwa, 2012; Villegas *et al.*, 2012; Macháček, 2019). Quantitative evidence of the impacts of ASM on land cover or biodiversity is extremely limited and mostly concentrated in certain regions where ASM is particularly extensive (e.g., Amazonia, Ghana; Asner and Tupayachi, 2017; Espejo *et al.*, 2018; Barenblitt *et al.*, 2021).

In the following section, I present evidence from the first robust counterfactual evaluation of the environmental impacts of ASM (Chapter 4) which contests these dominant narratives.

The environmental trade-offs of ASM in Madagascar

ASM is widespread across Madagascar (Canavesio and Pardieu, 2019; Chapter 3), and is considered an important threat to biodiversity, including within Protected Areas (Cook and Healy, 2012; Goodman, Raherilalao and Wohlauser, 2018; IUCN, 2022). Over the past 30 years, ASM has expanded across the island to become the second most important rural livelihood after agriculture, directly supporting an estimated 500,000 people, mirroring trends across Sub-Saharan Africa (World Bank, 2010; Hilson and Garforth, 2012; IGF, 2017). The development of artisanal gem mining (particularly for high-value ruby and sapphire) in Madagascar has been characterised by a series of gem rushes, whereby the discovery of valuable deposits draws thousands of people from across island into the area to mine (Cartier, 2009; Canavesio and Pardieu, 2019). Some of these rushes attracted substantial negative media coverage focussed on poor social and environmental outcomes (e.g., Carver, 2017; Tullis, 2019), which has fed negative perceptions of ASM more broadly (Walsh, 2012; Klein, 2022).

In Chapter 4, I conduct the first evaluation using robust counterfactual methods of the environmental impacts of ASM, using the sapphire rush at Bemainty in eastern Madagascar as a case study. This rush, involving 10-25,000 people (mostly migrants from across the island), occurred deep within the protected rainforests of the Corridor Ankeniheny-Zahamena. Given the number of the people involved, the lack of regulation, and the location within an area of global conservation significance, the risks to biodiversity were significant. I evaluated the impact of this mining rush on deforestation and forest degradation (considered temporary loss of forest), using a synthetic control to estimate counterfactual forest loss in the absence of mining. I found that the mining rush did not significantly increase deforestation or forest degradation above estimated counterfactual forest loss from other causes (namely agriculture). Lemur survey data collected in the field revealed apparently healthy populations of critically endangered lemur species which have been heavily impacted by hunting and/or ASM elsewhere (Schmid and Alonso, 2005; Jenkins *et al.*, 2011). Combined with evidence from interviews, this suggests hunting pressure in the area was relatively low, particularly from miners. These findings contest media claims that the mining rush caused hundreds of hectares of deforestation and threatened lemur populations (Tullis, 2019), highlighting the importance of using robust methods to evaluate the impacts of mining.

This evidence directly challenges popular perceptions of mining and has important implications for homogeneous understandings of ASM and policy responses, both in Madagascar, and globally. We showed that 10,000- 25,000 people mining in the heart of a protected rainforest did not cause more forest loss than several hundred people clearing land for shifting agriculture. This indicates that at Bemainty ASM had a much lower per capita deforestation footprint than agriculture. While we do not have data on average incomes from artisanal gem mining in Madagascar, in general miners can make enough money to meet basic needs while larger finds can improve living standards and fund investments in the future (Cartier, 2009; Walsh, 2012; Lawson, 2018; Klein, 2020). Elsewhere in Sub-Saharan Africa it has been reported that miners can earn more (and often substantially more) from ASM than agriculture, which has become increasingly unviable (Banchirigah and Hilson, 2010; Schure *et al.*, 2011; Hilson and Garforth, 2012; Villegas *et al.*, 2012; Macháček, 2019; Baffour-Kyei *et al.*, 2021). Therefore, in places with rich deposits where the environmental trade-offs are minimal, ASM can potentially represent a more efficient land use, providing greater returns per unit area than agriculture.

However, there are several important caveats to this. First, the study only evaluates the impact of mining on forest cover and the status of lemur populations three years after the rush. We were unable to capture smaller-scale forest impacts, wider impacts on other species, or the aquatic ecosystem which, given the amount of sediment pollution, may have been substantial (Lobo *et al.*, 2016; Bansah *et al.*, 2018). Second, the economic contributions of ASM are inconsistent and can involve concerning social trade-offs, affecting both miners and the local community (Lawson, 2018; Stoudmann *et al.*, 2021; Supplementary Results 3). Negative social impacts from ASM include exploitation, child labour, truancy, poor health and safety, increased disease, crime, and corruption (see Introduction; Duffy, 2007; Canavésio, 2010; Smith *et al.*, 2016; Bansah *et al.*, 2018; De Haan, Dales and McQuilken, 2020; Ofosu *et al.*, 2020). For unlucky miners ASM can become a poverty trap (Cartier, 2009). Finally, this is just one case study, and these findings will not necessarily apply to ASM elsewhere.

However, some of the factors which limited the environmental impacts of the mining rush at Bemainty may also limit the impacts of ASM elsewhere (particularly for alluvial gem mining). These factors include the mineral (i.e., gemstones), the location of the deposit within alluvial sediments, the geographical spread of the deposit, the basic mining methods used, and the large footprint of alternative land uses. Mining at Bemainty was reliant on manual labour and no chemicals or heavy machinery were used. Small pits were dug by hand and the extracted sediment sieved in the river (Figure 23; Perkins, 2016; Pardieu *et al.*, 2017). The only machinery used were low-power water pumps to remove water from pits or wash the extracted sediment (Perkins, 2017). Artisanal gem mining does not require the use of chemicals (like artisanal gold mining can use mercury or cyanide), and often exploits alluvial deposits, which may be unsuitable for more mechanised mining. Therefore, the simple methods used at Bemainty are relatively common to artisanal gem mining sites (and some gold mining sites) across Madagascar and Sub-Saharan Africa (Tilghman, Baker and Deleon, 2007; Canavésio, 2010; Schure *et al.*, 2011; Villegas *et al.*, 2012; Cabeza *et al.*, 2019; World Bank, 2019; Denison Mundi, 2022). This means that artisanal gem mining may have similarly limited impacts elsewhere (eg. Schure *et al.*, 2011; Funoh *et al.*, 2017), particularly in places where the environmental impacts of alternative livelihood activities (i.e. the other activities those miners would be undertaking in the absence of mining), is substantial. For example, in forested hills of eastern Madagascar shifting rainfed rice cultivation is a dominant livelihood activity (Styger *et al.*, 2007; Poudyal *et al.*, 2018), and a major driver of deforestation, accounting for 99% of forest loss in one case study in the north-east (Llopis *et al.*, 2019).

Figure 23: Miners manually sieving extracted sediment at Bemainty. Photo: Rosey Perkins.

However, this sort of artisanal gem mining can be considered a 'light touch' form of mining, and under different conditions ASM can have substantial environmental impacts (see Chapter 1). Where ASM involves the use of heavy machinery, toxic chemicals, or is particularly extensive, it can cause serious environmental damage (Figure 24; Asner and Tupayachi, 2017; Espejo *et al.*, 2018; Obodai *et al.*, 2019; Barenblitt *et al.*, 2021). Mercury pollution from artisanal and small-scale gold mining can have potentially serious and wide-ranging effects on human health and biodiversity (Boening, 2000; Sandheinrich and Wiener, 2011; Gibb and O'Leary, 2014; UN Environment, 2019). In some places small-scale gold mining has become increasingly mechanised. Mechanisation can deplete deposits quicker, increasing the rate of expansion into new areas (World Bank, 2019). In the Peruvian Amazon and Brazil, for example, excavators and loaders are used in open pit mining, and high power water cannons and suction pumps are used to dislodge alluvial sediments (Espejo *et al.*, 2018; Siqueira-Gay and Sánchez, 2021; Figure 24B). Mechanisation has increased the scale and impacts of illegal gold mining, which has become one of the leading drivers of deforestation in the region, including within Protected Areas (Alvarez-Berríos and Mitchell Aide, 2015; Asner and Tupayachi, 2017; Espejo *et al.*, 2018; Kalamandeen *et al.*, 2018; Siqueira-Gay and Sánchez, 2021). In Southern Ghana, illegal artisanal and small-scale gold mining, termed *galamsey,* is extensive, involves substantial mercury use, and is becoming increasingly mechanised (Owusu, Bansah and Mensah, 2019; Achina-Obeng and Aram, 2022). Between 2005 and 2019 ASM quadrupled in extent and caused an estimated 47,000 ha of deforestation, including 700 ha within protected forest reserves (Barenblitt *et al.*, 2021).

These high-profile, and relatively well-documented cases where ASM has caused extensive environmental damage have strongly influenced perceptions of ASM *in general,* feeding narratives that ASM is inherently environmentally destructive and worse than alternatives. However, evidence from the case-study in Madagascar shows that this is not always the case. Instead, these findings emphasize that the impacts of ASM are highly heterogenous (Hirons, 2011), depending on the location, extent, level of mechanisation, and chemical use (Villegas *et al.*, 2012; World Bank, 2019). The tendency to tar all ASM with the same brush simplifies complex realities and can contribute to inappropriate and ineffective policy responses.

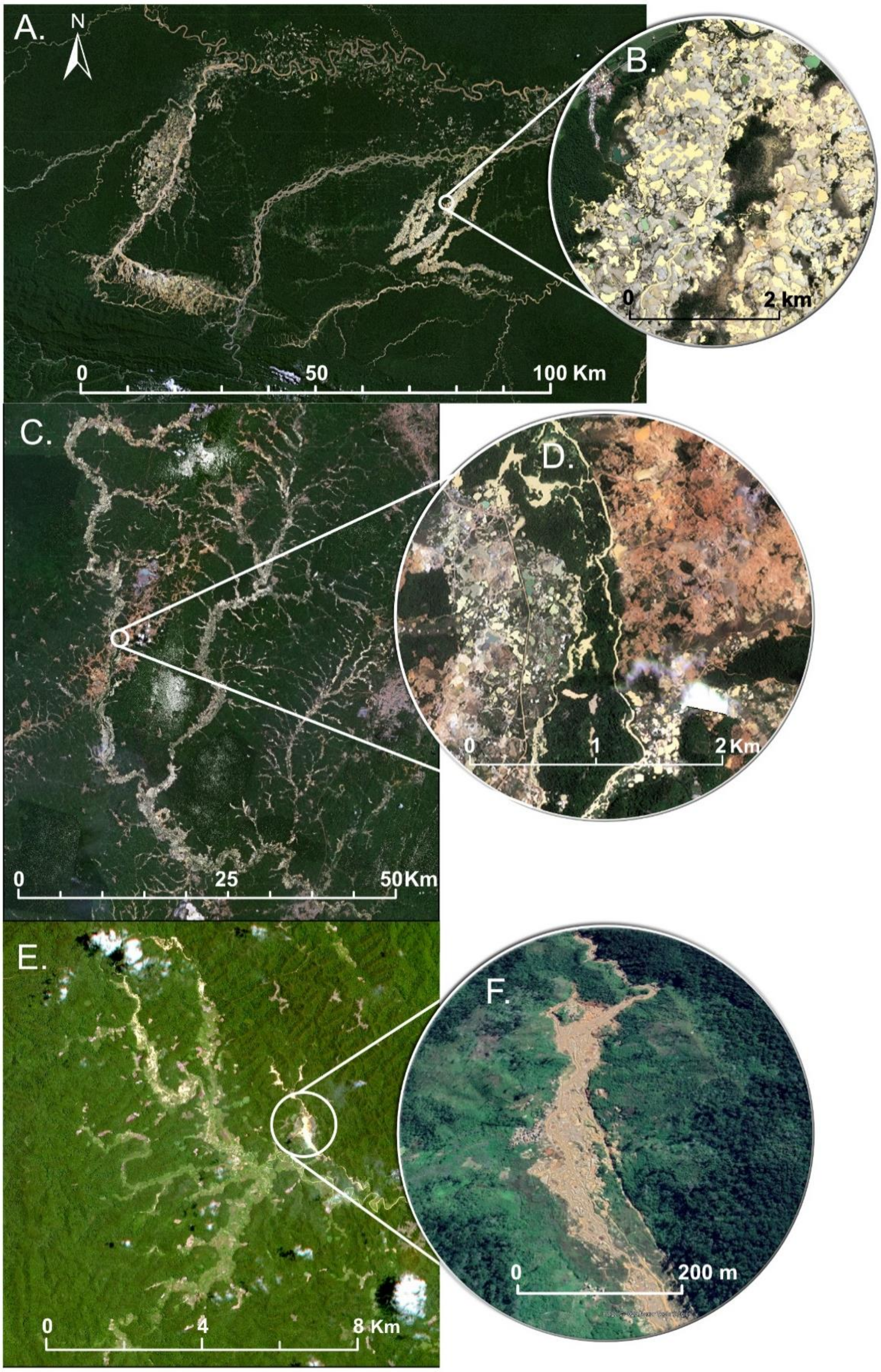


Figure 24: Panel of satellite images from Madre de Dios, Peru (A and B); Southern Ghana (C and D); and Bemainty, Madagascar (E and F). The comparison emphasizes the difference in extent between ASGM in Peru and Ghana, which extends for hundreds of kilometres, and artisanal gem mining in Madagascar, which tends to be more localised (with the exception of Ilakaka). Image A, B, C, D and E were sourced from Planet (Planet Team, 2017). Image F was taken from Google Earth.

Mitigating the environmental trade-offs of mining

Perceptions of mining as inherently environmentally destructive and worse than alternatives have influenced policy responses to the environmental challenges of mining (particularly for ASM). These perceptions have fed discourses that the negative impacts of mining should be addressed through suppression (for ASM), regulation, and control (Hirons, 2011; Lahiri-Dutt, 2014; Klein, 2022). For industrial mining this manifests as necessary laws and international standards mandating Environmental Impact Assessments, pollution regulation, mitigation, and biodiversity offsetting. However for ASM, this has resulted in policies which have marginalised or criminalised the sector (e.g. by creating a framework for legal ASM which is unattainable to most miners; Tschakert and Singha, 2007; Hirons, 2011; Hilson *et al.*, 2017).

In the following sections I critically evaluate three of the main policy responses, informed by the dominant discourses and narratives, to mitigating the environmental impacts of mining: biodiversity offsetting for industrial mining, formalisation, and certification schemes for ASM. I then consider the applicability of formalisation and certification to improving the environmental outcomes of ASM, using the case study of Madagascar.

Policies for mitigating the environmental impacts of industrial mining: the mitigation hierarchy and biodiversity offsetting

The mitigation hierarchy and biodiversity offsetting are key policy mechanisms for mitigating the impact of industrial mining on biodiversity and have been implemented globally (Bull and Strange, 2018). The mitigation hierarchy requires that damage to biodiversity from mining should first, and preferentially, be avoided, minimised, and eventually restored, and unavoidable biodiversity loss offset through equivalent gains elsewhere (McKenney and Kiesecker, 2010; Quétier and Lavorel, 2011; Bull *et al.*, 2013). One way for mining companies to offset biodiversity loss is by preventing an equivalent amount of biodiversity loss from occurring elsewhere, through conservation actions within designated biodiversity offset sites (known as avoided loss offsetting; Maron *et al.*, 2015). By 2018, an estimated 12,000 biodiversity offsets had been implemented worldwide and considerable time and resources invested in the concept (Bull and Strange, 2018). Yet, there was very little evidence that they actually work (zu Ermgassen *et al.*, 2019b). To help fill this knowledge gap, I evaluated the effectiveness of the high-profile biodiversity offset strategy of the Ambatovy mine in Madagascar at achieving No Net Loss of forest (Chapter 2). Using statistical matching and regression models to estimate counterfactual deforestation within each of Ambatovy's four biodiversity offsets sites, I found that by January 2020 the mine had managed to prevent 1,644 ha (674 -3,122 ha, 95% CI) of deforestation, offsetting nearly 80% of forest loss at the mine site. If reductions in deforestation continued at the same average rate, Ambatovy would have achieved No Net Loss of forest by the end of 2021.

While these results are encouraging for Ambatovy, Madagascar, and the concept of offsetting in general, there are serious caveats, and improvements are needed for offsetting to become a sustainable means for mitigating the environmental impacts of mining. Firstly, in the effort to minimise trade-offs to biodiversity, other important trade-offs emerged. Bidaud *et al* (2017) show that effective conservation of the offsets resulted in substantial opportunity costs borne by poor, rural communities who lost access to the land and forest resources upon which they depend. Alternative livelihood projects implemented by the mine to compensate for this loss of resources were well-received, but generally considered insufficient to cover the costs. As such, Ambatovy's efforts to achieve No Net Loss of biodiversity compromised local livelihoods and wellbeing (Bidaud *et al.*, 2017) and contravened the mine's own commitments to international standards on social protection (i.e. IFC Performance Standard 5).

Social costs from biodiversity offsetting are not unique to Ambatovy. Concerning social impacts from biodiversity offsetting have also been reported at the QMM ilmenite mine in Southern Madagascar, where conservation restrictions increased food insecurity and poverty (Seagle, 2012; Huff and Orengo, 2020), and elsewhere (Koh, Hahn and Ituarte-Lima, 2017; Tupala, Huttunen and Halme, 2022). By substituting biodiversity in one place for that in another, offsetting removes or redistributes access to the benefits that nature provides (ecosystem services; Ives and Bekessy, 2015; Jacob *et al.*, 2016; Jones *et al.*, 2019c; Sonter *et al.*, 2020c). Local people living near the impact site lose out, as do people living near offset sites who may lose access to land and resources (Sonter *et al.*, 2018, 2020c). To address these impacts, some have advocated for integration of ecosystem services into the No Net Loss framework (Jacob *et al.*, 2016; Jones *et al.*, 2019c; Sonter *et al.*, 2020c). Proponents argue this that would require the impacts of development (including mitigation measures) on local people's access to ecosystem services to be properly accounted for, and appropriately compensated (Jones *et al.*, 2019c; e.g., Barnes *et al.*, 2023). Ambatovy have taken promising, if rather late, action on this with the creation of an Ecosystem Services team. This team is tasked with assessing the benefits that local people gain from the forest and designing appropriate compensation measures.

Secondly, evidence that biodiversity offsetting can successfully compensate for loss of natural habitat should not be taken as a green light for unrestricted mining development. Biodiversity offsetting is highly uncertain, costly, and difficult to get right (zu Ermgassen *et al.*, 2019b). It is also gameable, as developers can manipulate the calculation of biodiversity losses and gains through their choice of counterfactual scenario (Virah-Sawmy, Ebeling and Taplin, 2014; Gordon *et al.*, 2015; Maron *et al.*, 2015). Voluntary biodiversity offset projects such as Ambatovy, in countries where offsetting is not a legal requirement, often have no external monitoring and evaluation, or institutional oversight (Bull *et al.*, 2013; Bidaud, Hrabanski and Meral, 2015; Koh, Hahn and Boonstra, 2019). Governments may lack the resources, technical capacity, political will, or responsibility (if NNL is not a policy or legal requirement) to monitor progress (Maron *et al.*, 2016). As such, monitoring and evaluation often relies on self-reporting and self-regulation. This is a serious issue, as it means there is often no real accountability on whether companies deliver on their commitments, or consequences if they fail (Maron *et al.*, 2016). Ambatovy invested substantial resources into its mitigation and offsetting programme and put considerable effort into prior avoidance and minimisation (Supplementary Methods 1). Not all companies will do the same. A lack of monitoring and enforcement allows ineffective offset projects, which fail to compensate for biodiversity loss, to quietly continue, meaning the considerable impacts to biodiversity from industrial mining remain un-mitigated.

A final major caveat is that avoided loss offsetting is not a sustainable option for mitigating the environmental impacts of mining in the long-term. Avoided loss offsetting relies on a continued rate of biodiversity decline, which can be slowed to generate the required biodiversity 'gains' (Maron *et al.*, 2015). NNL commitments aim to offset the project-specific losses of biodiversity (i.e. the forest cleared by a mine), bringing biodiversity loss back to counterfactual levels as if the project had never happened. As such, the baseline NNL commitments strive for is one of continued biodiversity decline (Maron *et al.*, 2015, 2016). Yet, if global biodiversity loss continues there may come a time when there is no equivalent habitat left with which to offset the impacts of a development. This was the case for the QMM ilmenite mine in Southern Madagascar. This controversial project destroyed 1,560 hectares of ecologically distinct littoral forest, but there was not enough unimpacted littoral forest remaining with which to offset this loss, so the mine had to conserve an alternative forest type instead (Virah-Sawmy, Ebeling and Taplin, 2014). This meant that the loss of unique littoral forest biodiversity from the mine site went uncompensated. Offsetting the impacts of development relative to a baseline of biodiversity loss is unsustainable and conflicts with global agreements to halt biodiversity loss (i.e., the Kunming-Montreal Global Biodiversity Framework). Therefore, a stronger approach is needed in which developers are required to go beyond offsetting project-specific impacts and contribute towards overall conservation or ecosystem restoration, for example, by aligning with jurisdictional conservation targets (Simmonds *et al.*, 2019), or aiming for Net Positive (Maron *et al.*, 2023). However, such approaches must build on the robust, verifiable foundations of the mitigation hierarchy (Maron *et al.*, 2023).

Policies for mitigating the environmental impacts of ASM: formalisation

In contrast to industrial mining, artisanal and small-scale mining operates mostly outside of the legal economy (IGF, 2017; World Bank, 2020), meaning legal frameworks for regulating the sector have very little effect. This informality and the lack of formal regulation also contributes to the negative socio-economic and environmental impacts of ASM (Villegas *et al.*, 2012; Hilson and McQuilken, 2014; IGF, 2017). As such, policy responses to the challenges of ASM have focussed on bringing the sector into the legal economy through a process of formalisation (Hilson and Maconachie, 2017; Hilson *et al.*, 2017).

The underlying theory of formalisation reflects the ideas of economist Hernando de Soto that informal economic activities (in this case ASM) should be integrated into the legal economy by providing formal land rights (Siegel and Veiga, 2009; Hilson *et al.*, 2017). For ASM, the idea is that issuing formal mining rights through licenses gives miners tenure security (although this means little if not enforced, Schure *et al.,* 2011), enables better regulation (as the authorities know who is mining where) and forces trade through formal channels, enabling the collection of taxes which can be reinvested in the sector and community development (IGF, 2017; De Haan, Dales and McQuilken, 2020; Hirons, 2020)

However, formal mining rights alone are not enough improve the social and environmental outcomes of ASM, formalisation must be accompanied by support and effective regulation (Siegel and Veiga, 2009; Hilson *et al.*, 2017; Álvarez-Berríos, L'Roe and Naughton-Treves, 2021). In Guyana ASM is mostly formalised, due to concerted political effort to strengthen the sector (and capture taxes and royalties), and a relatively cheap and easy licensing process (Clifford, 2011; Hilson and Maconachie, 2017; Hirons, 2020). Yet, ASM remains the largest driver of deforestation in the country – responsible for 81% of forest loss in 2022 (Guyana Forestry Commission, 2023) – and causes an estimated US$72 million worth of damage from mercury pollution (Laing and Moonsammy, 2021). These environmental impacts result from a lack of financial support to help miners comply with costly and contested environmental requirements, the difficulties of monitoring compliance across a vast forested landscape, and corruption undermining enforcement (Clifford, 2011; Hook, 2019). For formalisation to be an effective means of improving the socio-environmental outcomes of ASM it must be accompanied with support to help miners comply with regulations, and capacity to monitor and enforce the rules. Support could involve training, technical assistance, and improved access to credit to enable miners to adopt cleaner technologies, improve working practices, increase efficiency, and mitigate environmental damage (Hinton, Veiga and Veiga, 2003; Tschakert and Singha, 2007; Siegel and Veiga, 2009). It could also involve establishing formal purchasing schemes to ensure miners receive a fair price for their product, or processing centres where miners can bring gold to be processed using cleaner technologies (eg. Shamwa Mining Centre, Hinton, Veiga and Veiga, 2003; Schure *et al.*, 2011). Such measures could increase mining incomes, helping to alleviate poverty and incentivise formalisation (Schure *et al.*, 2011).

Yet despite these promises, formalisation has mostly failed – up to 80% of artisanal and small-scale mining worldwide remains informal and illegal (IGF, 2017). Formalisation is extremely costly (Schure *et al.*, 2011), and governments often lack the capacity or political will to enforce the regulations and provide the necessary support and incentives (Clifford, 2011; Hilson *et al.*, 2017). The lack of political will to engage with ASM stems from a long-standing bias towards industrial mining, or because corrupt elites benefit from continued informality (Duffy, 2007; Corbett, O'Faircheallaigh and Regan, 2017; Hilson *et al.*, 2017). These issues are often compounded by a lack of trust towards the authorities within mining communities (Schure *et al.*, 2011; Hilson, 2020). Furthermore, regulations are often ill-suited to realities on the ground and do not account for the diversity of ASM, reflecting the lack of participation of miners in decision-making (Hilson et al, 2007, 2017; Verbrugge, 2015; Geenen, 2012). Licensing processes are often overly bureaucratic, costly, and may involve unfeasible requirements (such as conducting an environmental impact assessment; Verbrugge, 2015; Hilson *et al.*, 2017; Hirons, 2020; Nopeia *et al.*, 2022). This restricts participation in the formal economy to those with knowledge and capital, in most places excluding the majority of poor artisanal miners and entrenching existing socio-economic inequalities and vulnerabilities (Geenen, 2012; Verbrugge, 2015; Hirons, 2020).

Yet despite the failures to date, formalisation can still contribute to improving the conditions and mitigating the environmental impacts of ASM if designed appropriately, sufficiently resourced, and combined with long-term support. Formalisation programmes which are designed from the bottom-up, with active participation from mining communities, to fit local realities and existing governance structures (but not entrench existing inequalities) are more likely to succeed (Klein, 2022a; Siegel and Veiga, 2009; Schure *et al* 2011). Increased positive engagement with ASM (whether it achieves widespread legalisation or not) could help to normalise the sector and challenge negative perceptions.

Certification schemes

In the absence of state capacity, and in line with neoliberal discourses of environmental management (Hirons, 2011), market-based approaches have been promoted as another means of regulating ASM and improving socio-environmental outcomes (Hilson, Hilson and McQuilken, 2016; IGF, 2017).

Consumer awareness of the social and environmental ills of gold and gemstone mining has grown (Fisher, 2018; Van Bockstael, 2018). This has translated into greater demand for transparency in supply chains and a willingness to pay for ethically-produced products (De Angelis, Adıgüzel and Amatulli, 2017; Moraes *et al.*, 2017). Certification schemes such as Fairmined, Fairtrade Gold, and the Responsible Jewellery Council, leverage consumers' willingness-to-pay for ethical products to command a price premium, enabling businesses, from mines to retailers, to gain a higher price for products produced to certain social and environmental standards (e.g. without child labour, chemical free, and using environmentally responsible practices; ARM, 2014; Van Bockstael, 2018; Cartier, 2019). Theoretically, this incentivises businesses to implement more ethical and sustainable practices so they can benefit from certification.

Different schemes target different actors at different scales. The Responsible Jewellery Council is the leading sustainability standard for the jewellery sector, but mostly targets gem cutters, traders, wholesalers, manufacturers, and retailers (Responsible Jewellery Council, 2019). Only 12 mining companies are certified (Responsible Jewellery Council, 2023). As such, the Responsible Jewellery Council Code of Practice focusses on promoting responsible sourcing and requires members to follow global standards on supply chain due diligence (OECD, 2016; Responsible Jewellery Council, 2019). It states that when sourcing from ASM, businesses should *"Regularly assess [...] the risks of unsafe working conditions, uncontrolled mercury use and significant environmental impacts (including impacts to biodiversity), and seek opportunities for ASM community development"* (Responsible Jewellery Council, 2019; p. 16). Other certification schemes, such as

FairMined and Oro Verde in Latin America, employ a bottom-up approach to improving the sustainability of mineral supply and specifically target ASM (Villegas *et al.*, 2012; ARM, 2014; Echavarria, 2014). These schemes help to connect organisations of artisanal and small-scale miners with international markets for ethical gold, enabling them to sell their products for a higher price (Echavarria, 2014). Miners' organisations receive an additional 10% of the gold price to invest in community development and an extra 5% if they do not use mercury (ARM, 2014; IGF, 2017). In addition, they receive training and technical assistance to improve efficiency, health and safety, and mitigate environmental impacts (IGF, 2017).

However, both approaches have their limitations. The Responsible Jewellery Council assumes that demand for greater sustainability and traceability from certified downstream industries (mineral processing, trade, manufacturing, and retail), while much needed, will trickle-down the supply chain to improve ASM practices on the ground. However, supply chains for gemstones in particular are complex, opaque, and very difficult to trace (Cartier, Ali and Krzemnicki, 2018). Gems may have passed through several countries and been traded informally many times before reaching the retailer (Cook and Healy, 2012; Cartier, Ali and Krzemnicki, 2018), making this theory of change questionable. Bottom-up approaches have struggled with scalability (currently only seven ASM organisations have Fairmined certification and only three are Fairtrade Gold certified; Fisher, 2018; Fairmined, 2023), and adapting to the diversity of ASM (Hilson and McQuilken, 2016). Most importantly, certification schemes are not accessible to all. Participation is often restricted to larger, industrial operations (for the Responsible Jewellery Council), or established co-operatives of small-scale miners with legal mining titles who are more likely to be able to meet the demanding standards and bureaucratic requirements (Echavarria, 2014; Hilson, Hilson and McQuilken, 2016; IGF, 2017). These stringent standards and license requirements exclude the majority of poor, unlicensed artisanal and small-scale miners who are most in need of support (Hilson, Hilson and McQuilken, 2016; Fisher, 2018). This makes certification ill-suited to the realities of most ASM which, particularly in Sub-Saharan Africa, remains predominantly informal. Consequently certification has limited potential to improve social and environmental outcomes of ASM without prior investment in formalisation (Hilson, Hilson and McQuilken, 2016).

The potential for policies to improve the environmental outcomes of ASM in Madagascar

Madagascar exemplifies many of the challenges in regulating and improving the social and environmental performance of ASM. While the country has a legal framework for formalising ASM, the vast majority of ASM remains informal; a situation which is unlikely to change anytime soon. The area coverage (at least $625m^2$) and duration (eight years) of mining licenses (*Code Minier,* 2023*)* does not reflect the reality of most artisanal mining on the island, which is very small-scale, and often transient, part-time, or seasonal. Official licenses may lack local legitimacy (in contrast to existing socially-accepted informal governance arrangements) and thus fail to confer tenure security (Klein, 2022a; Cook and Healy, 2012; Hilson *et al.,* 2017). There are limited repercussions for mining illegally (especially outside of Protected Areas) and few benefits from obtaining a license (Klein, 2022a). Furthermore, there is strong distrust of state institutions in many mining communities, due to corruption and a long-history of state-corporate attempts at suppression and eviction (Klein 2020; 2022a). Therefore, even if regulations were improved to make licenses more appropriate, much cheaper, and easier to obtain, it is possible that many people would choose to continue mining informally to avoid involvement with state institutions and taxation (Klein, 2022a; similar sentiment was noted in Cameroon; Schure *et al.,* 2011). Therefore, given current governance shortfalls, low trust, and a lack of capacity to support the sector and enforce the regulations, formalisation currently has very limited potential to improve the social and environmental outcomes of ASM in Madagascar.

Certification also has a very limited potential to influence ASM in Madagascar as the stringent requirements are out of reach for the vast majority of the island's artisanal miners. As of 2017, there were no certification schemes active for ASM on the island (IGF, 2017). However, certification and mine-to-market approaches (where producers connect directly with sellers of ethical products, increasing traceability and transparency) could play a role in incentivising and funding improved environmental practices for formal small, medium, and large-scale mines in Madagascar (e.g., the Prosperity Earth garnet mine in the north-west).

Yet, in the absence of co-ordinated formal efforts to regulate and improve the sector, a range of independent, bottom-up approaches have emerged, with varying degrees of success and sustainability. These approaches have been led by NGOs, local communities, and local authorities, and have been implemented regardless of the legal status of mining.

Individuals and mining-affected communities have initiated grassroots efforts to mitigate the environmental impacts of mining or restore degraded landscapes. In Bekily, in south-west Madagascar, village associations refilled abandoned mine shafts, which posed a real danger to livestock and children, and reforested land degraded by artisanal sapphire mining (Flores Zavala, 2017). Later, the German Development Agency (GIZ) became involved and provided saplings to the association in Bekily and several other impacted villages (Flores Zavala, 2017). At Soabiby in south-west Madagascar, the local community mobilised the authority and existing institutions for natural resource governance to impose respect for local customs on thousands of migrant sapphire miners, preventing mining in sacred forest areas, and even extracting rents and commission from miners (Baker-Médard, 2012, but c.f Canavesio, 2009, 2010). In the goldfields of Northern Madagascar Klein (2022a) describes the variety of informal governance structures which have emerged to effectively regulate and govern the goldfields as a mineral commons (Klein, 2022b). These governance institutions, which often draw on customary arrangements, create and enforce rules and norms concerning land access, claim staking, taxation, labour and revenue-sharing, acceptable behaviour, dispute arbitration and enforcement (Klein, 2022a). They aim to maintain social order, community cohesion and mitigate social harm from ASM. They may also act to mitigate negative environmental impacts; for example, by keeping mining out of sacred forests and rice paddies, or prohibiting hunting of certain species (Baker-Médard, 2012; Klein, 2022a; Chapter 4). Far from being the unregulated and chaotic spaces conceived by the dominant narratives, (Walsh, 2012; Lahiri-Dutt, 2014), ASM sites are in fact governed by a complex, flexible array of forms and institutions, created and generally accepted by local communities (Klein, 2022a).

Local authorities can play an important role in such informal ASM governance. At a gold rush site near Arivonimamo in Central Madagascar, commune authorities intervened to improve security, sanitation, and health and safety (Cook and Healy, 2012). In Antamimbary in Northern Madagascar, Klein (2022a) describes how local authorities have syncretised (or locally adapted) state institutions for mining governance by issuing their own licenses, collecting their own taxes, and managing land claims. These forms of governance can be considered informal-formalisation – local authorities are implementing structures to regulate ASM independent of the formal legal framework, which, while locally recognised as legitimate, remains officially illegal.

These grassroots initiatives and institutions have helped to mitigate the harmful social and environmental aspects of ASM in Madagascar, improve practices, and share the benefits (Baker-Médard, 2012; Klein, 2022a).

Alternative visions for reconciling mining with biodiversity conservation

So far, I have examined several of the main policy approaches (biodiversity offsetting, formalisation, and certification schemes) for mitigating the environmental impacts of mining. These policies are informed by, and largely in keeping with, dominant narratives. Next, I will explore some alternative, more controversial possibilities, which challenge these dominant narratives and the restrictive policies they prescribe.

Narratives of the inherent environmental destructiveness of ASM mean it is often portrayed as a threat to biodiversity and in conflict with conservation. But in some cases, could ASM actively help conservation? At Bemainty, I show that ASM had a lower per-capita deforestation footprint than agriculture, the principal livelihood activity in the area. Theoretically, in places with rich deposits on the forest edge could ASM not be promoted as an alternative livelihood strategy to reduce dependence on agricultural practices which drive deforestation? In Madagascar, such a strategy could be envisioned on the edge of a Protected Area, implemented by PA managers with support from NGOs or agencies involved in ASM (e.g. the GIZ), and local authorities. Proponents would provide training, technical and financial support, and access to markets to create, or strengthen, a local ASM sector with higher potential earnings than agriculture, thus incentivising mining (Villegas, Turay and Sarmu, 2013). Mining would be subject to environmental rules prohibiting the use of chemicals and requiring restoration of abandoned mining areas. To ensure increased income from mining is not invested in clearing more land for agriculture (Llopis *et al.*, 2019), other incentives would be needed. Could Payments for Ecosystem Services schemes such as REDD+, which pay Protected Area managers for reductions in deforestation, be used to fund ASM-based alternative livelihood projects and channel payments to communities as incentives to reduce overall deforestation? Although, there would be huge practical challenges to such an approach. Participation would need to be restricted to local communities to prevent a mining rush that could end up conversely increasing deforestation (although it can be difficult to differentiate locals and migrants in ongoing migration frontiers; Jones *et al.*, 2018). This requires strong local governance and external support for enforcement, which may not be available (discussed further below). Furthermore, such an approach may not be sustainable in the long-term as deposits become depleted.

ASM is considered most concerning when it occurs within Protected Areas. Mining within Protected Areas is illegal in Madagascar and most other countries, and policy responses, prescribed by narratives of environmental destructiveness, are centred on suppression (Dondeyne *et al.*, 2009; Villegas *et al.*, 2012). Yet attempts to exclude or evict ASM from Protected Areas often fail (Dondeyne *et al.*, 2009; Cook and Healy, 2012; Villegas *et al.*, 2012). Inadequate enforcement and ineffective attempts at eviction enable mining to persist within PAs, and can increase conflict between miners, local communities (who may also be miners), and Protected Area managers (Dondeyne *et al.*, 2009; Villegas *et al.*, 2012). This is currently the case at Itremo Protected Area in Madagascar where local communities are in conflict with PA managers who are attempting to prevent them from mining within the large reserve (Sarobidy Rakotonarivo, *pers comm)*. The illegality of mining within Protected Areas can exacerbate negative social and environmental impacts and encourage other illicit activities (eg. harvesting precious timber or bushmeat hunting; Funoh *et al.*, 2017; Spira *et al.*, 2019). Miners operating within Protected Areas may be fearful of arrest and keen to make money and get out quick, potentially incentivising more destructive practices. Given the practical difficulties of monitoring large, forested Protected Areas (Clifford, 2011) and serious governance and capacity shortfalls in many ASM hotspots, it is likely that in many places this situation will persist (Villegas *et al.*, 2012). Therefore, alternatives approaches should be considered.

In Chapter 3 I discuss several cases where Protected Area managers have attempted to permit and regulate a limited amount of ASM within designated zones of their Protected Areas. In Northern Madagascar, the Malagasy conservation NGO Fanamby recognised the importance of artisanal gold mining to local livelihoods and the futility of trying to prevent mining within the Loky Manambato Protected Area (a Category V multiple use PA). As such, Fanamby decided to permit mining within designated corridors in Sustainable Use Zones of the PA. Mining is subject to certain rules prohibiting the use of mercury, acid, or felling trees without approval and activity must be confined to the corridors (Cook and Healy, 2012; World Bank, 2019). Fanamby supported miners to self-organise into a mining association and register with the municipality; enabling the collection of taxes (although this is a case of informal-formalisation as miners did not possess the offical permits required by federal law; World Bank, 2019; Cook and Healy 2012). Compliance with the environmental rules is monitored by the association. Permitting mining within these designated corridors helped to prevent mining from spreading into core zones of the Protected Area and mitigated environmental damage (World Bank, 2019). Unfortunately, the agreement struggled during the period of political instability and economic hardship following the coup of 2009, when mining spilled into other zones of the Protected Area (Cook and Healy, 2012). However, Fanamby continues to work with artisanal gold miners operating within the PA to encourage compliance, formalisation, and rehabilitation of abandoned mine sites (Fanamby, 2021). Fanamby's efforts represent an alternative, potentially more pragmatic, vision for the relationship between mining and conservation, focussed less on conflict and more on coexistence. This approach recognises the enormous contributions of ASM to local livelihoods and attempts to balance both the environmental trade-offs of mining and the social trade-offs of conservation. Such an approach, if well implemented and tightly-regulated, has the potential to mitigate land-use conflicts between ASM and conservation. In Itremo, for example, local communities could be permitted to mine in less sensitive designated zones outside the rare ancient grasslands for which the PA was designated (Sarobidy Rakotonarivo, *pers. comm).*

Of course, such an approach is risky and faces considerable practical and ideological barriers and challenges. Practically, it is likely only suitable in certain circumstances where the number of miners can be limited (a similar agreement in a protected forest in Gabon failed due to an influx of migrant miners; Hollestelle, 2012; Vuola, 2022), there is a strong relationship (and trust) between PA managers and local communities, strong local institutions, and buy-in from local authorities. It may be vulnerable to external shocks and requires effective, non-corrupt policing to fall back on. Furthermore, while mining could be permitted within Protected Area categories V and VI (a global agreement on mining and Protected Areas signed at the World Conservation Congress in 2000 states that mining is acceptable in these Protected Areas if the nature and extent of activities do not compromise conservation objectives; Dudley, 2013), many countries (including Madagascar) have legally prohibited mining in all Protected Areas (IGF, 2017). This illegal status may discourage NGOs and Protected Area managers from engaging with ASM for fear of condoning an illicit activity (Villegas *et al.*, 2012).

Alternative approaches permitting ASM in, or near, Protected Areas may also face insurmountable ideological barriers. Deeply engrained perspectives of mining as inherently environmentally destructive position ASM at odds with conservation in the popular imagination. This means that, regardless of legal status, funding or engaging with mining may not be permissible to the donors who fund much conservation and may not meet the criteria for Payments for Ecosystem Services schemes. Additionally, international conservation NGOs may fear constructive engagement with mining could undermine their social legitimacy.

However, there is precedent for a controversial strategy which conflicts with popular perceptions of conservation to be adopted, and to demonstrate some success. Trophy hunting involves controlled hunting of typically large, iconic species by high-paying clients on private land or within Protected Areas (Category V or VI). Well-regulated trophy hunting with low levels of off-take can be a sustainable and economically efficient land use (Lindsey, Roulet and Romañach, 2007; Nelson, Lindsey and Balme, 2013; 't Sas-Rolfes, 2017). The high revenues from trophy hunting can incentivise sustainable management of wildlife populations, enable conservation of large areas, and prevent other, more intensive land uses (Di Minin, Leader-Williams and Bradshaw, 2016). Revenue can also be levied to fund wider conservation initiatives (Nelson, Lindsey and Balme, 2013; Di Minin, Leader-Williams and Bradshaw, 2016).

Similarities can be drawn between trophy hunting and the alternative approaches to ASM and conservation outlined above. Like trophy hunting, ASM exploits a high-value resource and permitting regulated exploitation within Protected Areas could replace other more impactful or extensive land uses. Both ASM and trophy hunting are often considered inherently 'bad' in the popular imagination, and perceptions are heavily shaped by high-profile cases (eg. Madre de Dios and the killing of Cecil the Lion in 2015; Macdonald *et al.*, 2016; 't Sas-Rolfes, 2017; Evans *et al.*, 2023). Both also clash with popular perceptions of what conservation is, how it should be achieved, and what trade-offs are acceptable (Batavia *et al.*, 2019; Evans *et al.*, 2023). Yet, despite the controversy and substantial opposition, trophy hunting has become a huge industry in Sub-Saharan Africa (Saayman, van der Merwe and Saayman, 2018) and is now widely accepted by conservationists as a potentially effective tool for conservation (van Houdt *et al.*, 2021).

Although, there are considerable differences between ASM and trophy hunting, particularly in the volume and distribution of financial benefits, which means alternative approaches to ASM are less likely to receive the same government backing. There may only be a few cases which meet the conditions outlined above, where ASM could be permitted and regulated in Protected Areas (also given the fact that economic mineral deposits are relatively rare). Income from ASM, while higher than alternatives, is still relatively low (complex supply chains mean miners receive a tiny proportion of the final product value), and only a small percentage of this (if any), will be paid to the government in taxes. This means that there may be little economic incentive for governments to support controversial, alternative approaches for ASM, unlike trophy hunting for which the potential economic benefits were clear.

The example of trophy hunting shows that pragmatic approaches permitting a limited amount of tightly-regulated extractive resource use within Protected Areas can contribute towards conservation. It also shows that such approaches can be adopted, despite negative popular perceptions and substantial opposition (although opposition to trophy hunting remains strong and has recently had some success with a new UK law banning the import of hunting trophies; Challender *et al.*, 2023). Unfortunately, there is currently very little consideration in the policy or research space of such alternatives for managing ASM and conservation (except Villegas *et al.,* 2012), meaning the status quo is likely to persist and cases such as Fanamby will remain isolated examples.

Conclusions and wider implications

The findings of this research have wider implications and applicability beyond the Madagascar case study.

In Chapter 2 I showed that biodiversity offsetting can be an effective mechanism for mitigating the environmental impacts of industrial mining, if well-designed and sufficiently resourced. This research has implications for mines with existing biodiversity offset strategies, as it shows that progress towards meeting their commitments can be independently evaluated, increasing accountability. It also proves that it is possible to offset the impacts of mining on forest cover (and so mitigate impacts on biodiversity), setting standards to which other mining projects should aim, and be held accountable. Hopefully these findings, and the spectre of independent evaluation, will encourage other mines to devote sufficient resources to their biodiversity compensation schemes and greater attention to avoiding and minimising impacts as much as possible, given the difficulties and resources needed to make offsetting work. I hope that my study will encourage similar evaluations of other mitigation and offset strategies to help build an evidence-base of what works and what doesn't to inform and improve future efforts. While the concept of No Net Loss is imperfect and unsustainable in its current form (for the reasons outlined above), it is at least a defined and verifiable target. This contrasts with more vague Net Positive Impact commitments which are becoming increasing popular but are potentially less direct and de-emphasize the need for prior mitigation and like-for-like-compensation (Maron *et al.*, 2023). By showing that NNL of forest is achievable and auditable, perhaps NNL/Net Gain can remain on the table, buying time for necessary policy improvements to bring commitments in line with broader jurisdictional and global biodiversity targets (Simmonds *et al.*, 2019).

For artisanal and small-scale mining, the methods developed in Chapter 3 can be applied in other countries with a nascent or growing ASM sector to identify zones of known mineral potential where ASM could be promoted and supported with limited trade-offs to biodiversity. Incentivising mining within such designated zones could help to minimise the environmental trade-offs of mining and aid formalisation efforts (Corbett, O'Faircheallaigh and Regan, 2017; Nopeia *et al.*, 2022).

In Chapter 4, I show that an artisanal mining rush involving up to 25,000 people in the heart of protected forest in Madagascar had limited impacts on the surrounding forest and did not cause more forest loss than several hundred people clearing land for agriculture. This evidence challenges popular perceptions that mining (particularly ASM) is inherently environmentally destructive and worse than alternative land uses. Despite the lack of underlying evidence, these negative perceptions continue to feed discourses and influence policy responses which have very real impact, affecting the lives and livelihoods of millions of people around the world.

Mining is here to stay and will likely need to expand to meet the mineral requirements for the low-carbon energy transition (IEA, 2021). While there is an undeniable need to reduce demand for mined products (particularly for luxury, non-essential goods) by addressing global consumption levels and increasing recycling, these changes are unlikely to be sufficient to reduce the need for mining, at least in the short term. Therefore, we need effective and equitable solutions to mitigate the environmental impacts of mining, to ensure mineral supply does not trade-off biodiversity and ecosystem services. To reach such solutions we need evidence. Evidence on the impacts of mining (both industrial and ASM) at different scales and under different conditions, preferably using robust methods to quantify impacts relative to alternatives. Such evidence could help to challenge perceptions and dominant discourses of mining that restrict the range of policy responses, paving the way for more flexible, pragmatic, and evidence-based engagement and policy. To maximise mining's contributions to sustainable development, policies should also aim to improve the conditions and enhance the social and economic benefits of mining, both for participants and local communities. We also need more evidence on the effectiveness of mitigation measures, to increase accountability and ensure that the environmental impacts of mining are truly being mitigated.

Appendix 1

Chapter 2: On track to achieve No Net Loss of forest at Madagascar's biggest mine

Supplementary Methods 1

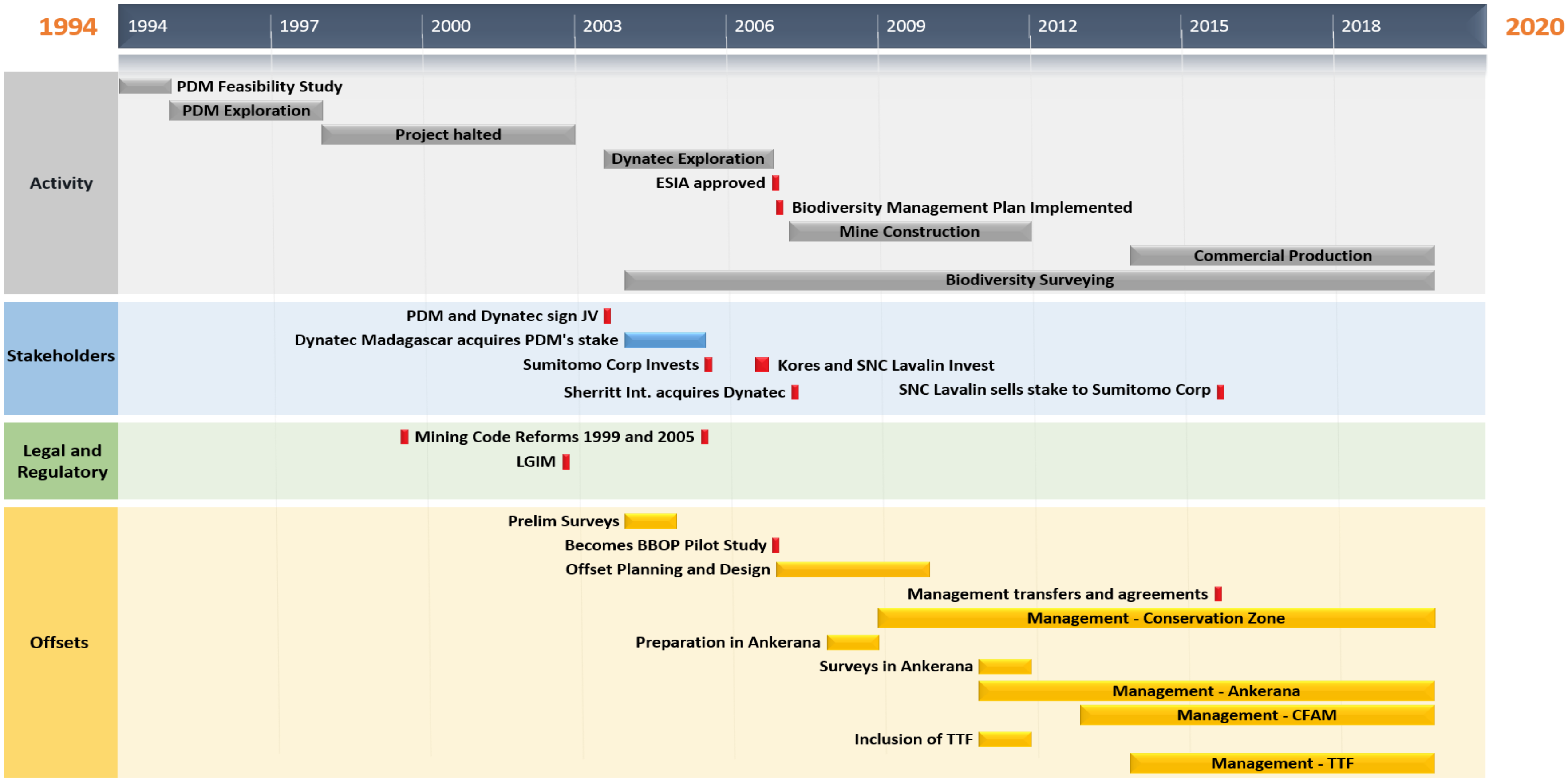
Study site and context

The mitigation hierarchy states that damage to biodiversity resulting from development must first, and preferentially, be avoided, minimised and restored (McKenney and Kiesecker, 2010), with offsetting reserved for any remaining unavoidable impacts (Bull *et al.*, 2013). As part of its avoidance measures Ambatovy set aside several patches of rare azonal forest (totalling 306 ha) overlying the ore deposit for conservation, foregoing mineral extraction (Von Hase *et al.*, 2014). Additionally, the slurry pipeline was routed to avoid fragments of primary forests and sensitive habitats (e.g. breeding habitat for the critically endangered Golden Mantella frog *Mantella aurantiaca*; Von Hase *et al.*, 2014).

To minimise impacts on biodiversity, prior to the mine construction Ambatovy surveyed the area scheduled for clearance and the adjacent forests to ensure locally endemic species found within the footprint were also found elsewhere, ensuring the mine did not lead to species extinction (Berner, Dickinson and Andrianarimisa, 2009). During the construction phase, priority species with restricted mobility were salvaged from sites before clearance and relocated to conservation areas outside the mine footprint (Berner, Dickinson and Andrianarimisa, 2009). Floral species of concern were transplanted in an on-site nursery (Berner, Dickinson and Andrianarimisa, 2009). Forest clearance was paced, radiating from a central point to give mobile species time and space to disperse (Von Hase *et al.*, 2014). To aid future restoration efforts the removed top-soil was preserved (Von Hase *et al.*, 2014).

Ambatovy has committed to restoring the mine site to a multi-functional forest (Ambatovy, 2017). Plant nurseries (including five community nurseries) and the forests within the Conservation Zone biodiversity offset will provide a source of seeds and propagules to aid forest restoration (Ambatovy, 2017). Whilst the mine is still in the early phase of operations the company has conducted trials of forest restoration to test and develop methods and in 2017, 6 ha of land was rehabilitated (Ambatovy, 2017).

The BBOP case study where Ambatovy's NNL strategy is documented, and the Environmental Impact Assessment are available for download here: https://ambatovy.com/ang/media/reports/



Supplementary Figure 1.1: Timeline of key events in the development of Ambatovy. This includes exploration and mining activity, stakeholders, relevant legal and regulatory changes in Madagascar, and progress in the biodiversity offset programme. PDM = Phelps Dodge Madagascar (original concession-holder and predecessor of Ambatovy), ESIA = Environmental and Social Impact Assessment, JV = Joint Venture, LGIM = Law on Large Scale Investments in Mining - Law n°2001-031, BBOP = Business and Biodiversity Offset Partnership, CFAM = Corridor Forestier Analamay- Mantadia, TTF = Tototorofotsy.

To quantify biodiversity losses and gains Ambatovy used a modified version of the habitat hectares approach which combines the area of habitat impacted with a composite measure of habitat quality (habitat hectares = area x quality; Parkes, Newell and Cheal, 2003). Prior to mine construction the forest was mapped and surveyed to calculate the impact area and measure the structural and compositional attributes selected as indicators of habitat quality (Berner, Dickinson and Andrianarimisa, 2009). The density of three species of critically endangered lemurs was also integrated into the habitat quality metric (Berner, Dickinson and Andrianarimisa, 2009). The company estimated that, in total, 2,064 ha of natural forest would be cleared or significantly degraded at the mine footprint and upper pipeline, translating to a loss of 1,467 habitat hectares (Von Hase *et al.*, 2014).

Ambatovy aimed to compensate for these losses by reducing deforestation from shifting agriculture within four sites designated as biodiversity offsets (Von Hase *et al.*, 2014). To calculate the expected biodiversity gains from protecting these sites Ambatovy had to establish the baseline (how much biodiversity would be lost in the absence of protection) and estimate conservation effectiveness (how much of this loss could be prevented through protection).

The baseline was defined using historical background rates of deforestation in the district (Moramanga for the Conservation Zone, CFAM and Torotorofotsy, and Brickaville for Ankerana; Von Hase *et al.*, 2014). Conservation effectiveness was based on the deforestation rate within the nearest protected areas (Mantadia National Park and Analamazoatra Special Reserve for the Conservation Zone, CFAM and Torotorofosty; and Mangerivola National Park for Ankerana), assuming that equivalent rates are achievable through protection of the offsets (Von Hase *et al.*, 2014). This was more realistic than other offset policies which assume 100% conservation effectiveness, or zero deforestation, within the offsets following protection (Virah-Sawmy, Ebeling and Taplin, 2014).

To account for uncertainty and increase the likelihood of achieving NNL Ambatovy developed four possible scenarios of baseline deforestation and conservation effectiveness and ensured the required biodiversity gains could be achieved for all scenarios by 2040, before the company ceases operations (Von Hase *et al.*, 2014). These scenarios were based on the highest and lowest rates of deforestation in the associated district between 1990 and 2010 and the highest and lowest rates of deforestation within the nearest protected areas over the same period.

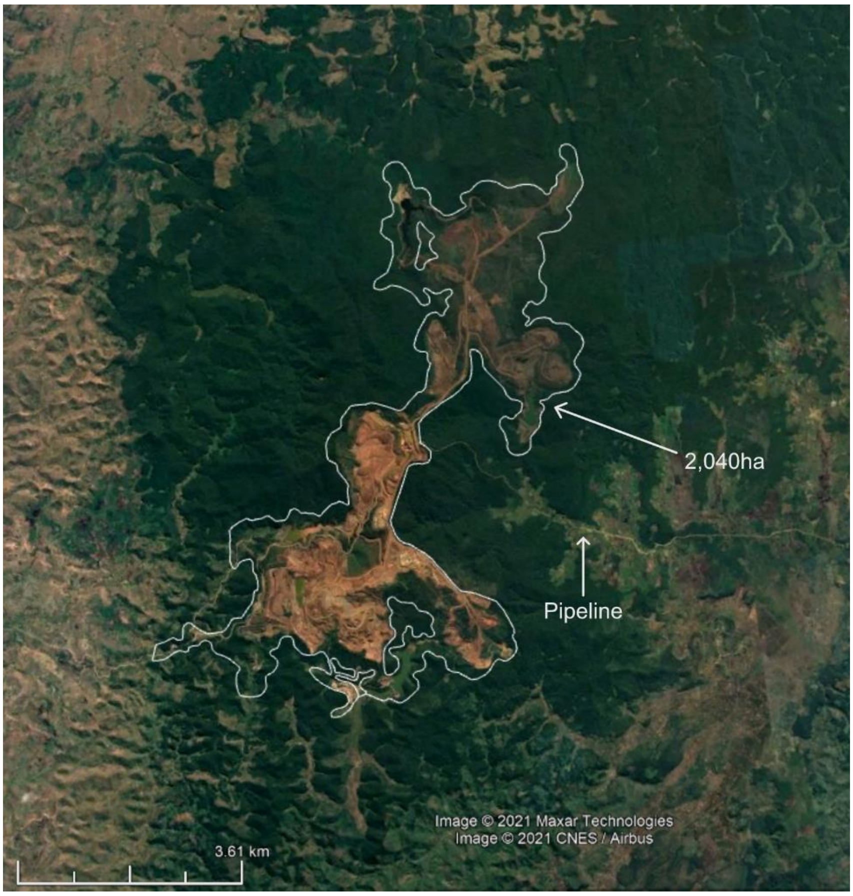
For each scenario the habitat hectares that could be gained through protection were estimated. First, the offset sites were surveyed to assess habitat quality. Then, gains in habitat area were estimated by subtracting the expected rate of deforestation within the offset following protection (i.e. the measure of conservation effectiveness) from the baseline deforestation rate. The most optimistic scenario, based on a high background rate of deforestation and high conservation effectiveness, predicted NNL of forest would be achieved during 2022 (Von Hase *et al.*, 2014).

Supplementary Table 1.1; Comparison of the baseline deforestation rates used by Ambatovy in their loss-gain calculations and the counterfactual deforestation rates estimated here using statistical matching and differences-in-differences regressions**.** We could not estimate the counterfactual deforestation rate for CFAM as it did not meet the requirements of parallel trends necessary for the matched controls to represent counterfactual outcomes.

Offset	**Baseline annual deforestation rate used by Ambatovy (%)**		**Counterfactual annual deforestation rate (%) after protection estimated using matching and difference-in-differences regressions**
	Highest	Lowest	
ANK	0.5	0.3	3.67
CZ	1.4	0.5	0.12
CFAM	1.4	0.5	N/A
TTF	1.4	0.5	3.1

Deforestation rates are highly spatially and temporally specific, influenced by a range of physical, social, economic, and political factors operating at that point in time (Ingram and Dawson, 2006). Therefore, attempts to extrapolate across space and time are highly uncertain. Our counterfactual deforestation rates were derived from observed outcomes in matched controls *over the same post-intervention period* (controlling for pre-intervention differences between treated and control samples). Therefore, we avoid extrapolating over time and mitigate the uncertainty of extrapolating over space through matching, as the matched samples have a similar probability of deforestation under baseline conditions (based on the measures of accessibility and agricultural suitability). Therefore, we consider our estimates of counterfactual deforestation a more reliable estimation of the deforestation which would have occurred within the offsets in the absence of protection than the historical rates employed by Ambatovy.

These results indicate that Ambatovy underestimated the amount of deforestation which would have occurred, absent protection, in Ankerana and Torotorofotsy and consequently the potential biodiversity gains which could be accrued through protection. However, Ambatovy's estimate was higher in the Conservation Zone.



Supplementary Figure 1.2; Independent estimate of the area of forest loss at the mine footprint through manual digitisation of a Google Earth image (Map data: Google, Maxar Technologies, CNES/Airbus, 2021). The image is dated 19/6/2021.

Ambatovy estimated that 2,064 ha of forest would be lost or significantly degraded at the mine footprint and upper pipeline and used this figure to calculate the residual biodiversity loss in habitat hectares. The company expected these losses to accrue between 2007 and 2022. Biodiversity loss associated with the pipeline was only calculated for the upper 2km which crosses the primary forests of the concession area as, for most of the route, the pipeline traverses a modified landscape of secondary vegetation of low biodiversity value (Dickinson and Berner, 2010). Consequently, losses associated with the pipeline are small, amounting to 21.5 ha of forest or four habitat hectares (0.3% of the total estimated biodiversity loss resulting from the mine; Berner, Dickinson and Andrianarimisa, 2009). Likewise, the processing plant and tailings facility at Toamasina were constructed on degraded secondary land and were therefore not included in the loss calculations (Von Hase *et al.*, 2014). Our independent estimate of the area of forest loss at the mine footprint (2,040 ha) supports Ambatovy's total estimate of forest loss at the footprint and upper pipeline (2,064 ha).

Supplementary Table 1.2: Forest cover and loss statistics for each offset and the entire offset portfolio over the 19 year study period. These figures correspond to the total area of the offset.

Offset	**Ankerana**	**CFAM**	**Conservation Zone**	**Torotorofotsy**	**Total**
Total Offset Area (ha)	6,904	9,423	3,787	8,626	28,740
Forest Area 2000 (ha)	6,459	5,916	3,035	2,653	18,062
% of total area forested in 2000	94	63	80	31	63
Forest area at year of protection (ha)	6,068	5,824	3,031	2,216	17,139
% of total area forested at year of protection	88	62	80	26	60
Forest area 2020 (ha)	5,985	5,529	3,017	1,437	15,968
% of total area forested by 2020	87	59	80	17	56
% Reduction in Forest Cover 2000-2019	7.3	6.5	0.6	46	12
Average annual deforestation rate before protection (%)	0.61	0.13	0.02	1.3	N/A
Average annual deforestation rate after protection (%)	0.15	0.72	0.04	5.9	N/A

To calculate forest area in 2000 we clipped the tree loss layer (the Global Forest Change dataset; Hansen *et al.*, 2013) masked to the map of forest cover 2000 (Vieilledent *et al.*,

2018) layer) to the boundary of each offset polygon. The total number of pixels within this layer represents the area of forest within each offset in 2000. From this we subtracted the pixels that were classed as deforested between 2001 and the year of protection to give forest area in hectares at the year of protection.

These results show that Ankerana is the most forested offset and since protection has experienced very little deforestation. The Conservation Zone is also highly forested and only lost 0.6% of its forest cover over the whole study period. The near-total lack of deforestation *before* protection in this offset underlines the impact the presence of the mine itself had on reducing forest loss.

Contrary to expectations, the average annual deforestation rate increased following protection in CFAM, the Conservation Zone (although it remained negligible) and Torotorofotsy. However, this does not necessarily mean the offsets had no effect as the increase in deforestation could have been higher without protection (as in the Conservation Zone, Supplementary Figure 1.4).

The situation in Torotorofotsy is particularly worrying. Between 2014, when the site became designated as a biodiversity offset, and January 2020 35% of its forests were cleared. Nearly 40% of this loss occurred in 2017 alone. An average deforestation rate of 5.9% per year between 2014 and 2020 is well above the national rate of 1.63% per year for the same period (calculated using the same data and methods as above). However, our results suggest there was no significant difference in deforestation between Torotorofotsy and the estimated counterfactual over this period, based on our representative sub-sample of pixels. In other words, this high rate of deforestation can be explained by the accessibility and suitability of the site for alternative uses (in this case rice production) as the matched control units which have similar characteristics also experienced high deforestation over this period.

Matching

Supplementary Table 1.3: Deforestation rates before and after protection in our sub-sample of pixels compared to the total rates for the whole offset. This indicates that our sample pixels, which comprise approximately 4% of pixels forested in 2000, were representative of deforestation outcomes within each offset. Sample sizes refer to the total number of pixels.

	Ankerana		**CFAM**		**Conservation Zone**		**Torotorofotsy**	
	Total	Sample (n=2,862)	Total	Sample (n=2,626)	Total	Sample (n=1,340)	Total	Sample (n=1,170)
Average annual deforestation rate % (2000-Year of Protection)	0.61	0.58	0.13	0.10	0.02	0.01	1.3	1.3
Average annual deforestation rate % (Year of Protection-2020)	0.15	0.12	0.72	0.64	0.04	0.04	5.9	6.1

Pre-cleaning the data

To obtain our pool of control units we used a grid-based sampling strategy to extract pixels from the province of Toamasina that were forested in the Year 2000 and outside the formal protected area network (excluding the CAZ) and the buffer zones of the biodiversity offsets. This produced 634,465 potential control pixels. To improve efficiency (which was particularly necessary when conducting the robustness checks) we pre-cleaned the data prior to matching. To do so, we defined a set of calipers based on the distribution of covariate values within each treated (offset) sample and removed control units with values outside this caliper which would never have been matched. This reduced the spread of values within the remaining control sample bringing the added benefit of producing closer matches by making the calipers in the matching algorithm (which is based on the standard deviation of the data) more restrictive.

First, we combined our sample of pixels from each offset with the full set of 634,465 potential control pixels. Then we filtered each dataset removing all observations with values greater than $\max(x) + \sigma(x)$ and smaller than $\min(x) - \sigma(x)$; where x refers to the covariate values in the offset sample and σ, the standard deviation. This was repeated for all five essential covariates in all four offset-control datasets. This resulted in the removal of up to 92% of the potential control pixels with covariate values way outside the range of the offset sample which would never have been matched.

Selection of covariates

In deforestation analyses selected covariates are primarily associated with accessibility and suitability of the site for alternative land uses, typically agriculture or the extraction of forest products (Andam *et al.*, 2008; Gaveau *et al.*, 2009; Blackman, 2013).

The covariates selected for use in this study are presented in Supplementary Table 1.4. These are known drivers of deforestation, both in Madagascar and globally, and have been used in other impact-evaluation studies of deforestation (Supplementary Table 1.5). Following Eklund *et al* (2016) annual precipitation, combined with slope and elevation is a proxy for agricultural suitability. Distance to forest edge and distance to recent deforestation reflect the frontier effect and the increased probability of deforestation occurring near previously cleared sites (McConnell, Sweeney and Mulley, 2004; Robalino and Pfaff, 2012). Distance to road, cart track and the nearest settlement, plus land surface characteristics such as elevation and slope are proxies for accessibility, demand, the ability to clear forest undetected, and the ease of transporting harvested products to market (McConnell, Sweeney and Mulley, 2004; Brinkmann *et al.*, 2014).

The five additional covariates were so defined because of poorer data quality (population density and distance to settlement), correlation with essential variables (annual rainfall is highly correlated with elevation [0.7]) or because they are simply considered less influential drivers of deforestation in this context (distance to cart track, distance to river). Distance to settlement does not differentiate between the size of settlement. However, demand and utilisation of forest resources varies significantly between villages, towns, and cities. While evidence for a significant relationship between population density and deforestation in Madagascar is mixed (Green and Sussman, 1990; Jarosz, 1993; Gorenflo *et al.*, 2011; Brinkmann *et al.*, 2014), possibly due to data limitations (McConnell, Sweeney and Mulley, 2004; Gorenflo *et al.*, 2011), population density is generally considered a key factor influencing deforestation globally and is a commonly used covariate in matching analyses (Supplementary Table 1.5). We chose not to include population density as an *essential* covariate as the available data at the appropriate spatial resolution is poorly measured in the study area. The data is based partly on night-light which, in a country where 73% of the population lacks access to electricity (World Bank Group, 2021), is not the most reliable indicator. However, population density is indirectly controlled for in our matching analysis as it is collinear with our five essential covariates (Supplementary Table 1.6).

Supplementary Table 1.4; List of covariates used in the statistical matching with their description, resolution, and source. When Euclidean Distance was calculated the output cell size was set to 30m to match the resolution of the outcome variable (tree loss) layer. To align the data layers, covariates with 30m resolution were snapped to the tree loss layer, resulting in a maximum spatial error of 15m. All data was projected to WGS 1984 UTM Zone 38S. All data is publicly available online

Covariate	**Description**	**Resolution**	**Essential or Additional**	**Source**
Distance to Road (m)	Euclidean Distance in metres to the nearest main road. Calculated in ArcMap 10.5 from the roads layer using the Euclidean Distance tool	30m	Essential	Roads - FTM (Foiben Taosarin-tanin 'I Madagasikara)
Distance to Forest Edge (m)	Distance in metres to the nearest forest edge in 2000.	30m	Essential	Vieilledent et al (2018)
Distance to former deforestation 1990-2000 (m)	Euclidean Distance in metres to the nearest pixel deforested between 1990 and 2000. These were identified by extracting pixels classed as forest in 1990 but non-forest in 2000.	30m	Essential	Vieilledent et al (2018)

Elevation (m)	Digital Elevation Model	30m	Essential	Shuttle Radar Topography Mission SRTM, 1 arc-second Digital Elevation Model. Downloaded from USGS Earth Explorer.
Slope (°)	Calculated from the DEM using the slope function in ArcMap 10.5	30m	Essential	
Annual Precipitation (mm)	Average annual precipitation 1970-2000.	490m	Additional	WorldClim v.2 Bioclimatic Variables. Fick and Hijmans (2017)
Distance to River (m)	Euclidean distance in metres to the nearest river.	30m	Additional	Digital Chart of the World
Distance to Cart Track (m)	Euclidean Distance in metres to the nearest cart track.	30m	Additional	Cart Tracks – FTM
Distance to Settlement	Euclidean distance in metres to the nearest settlement	30m	Additional	NGA OCHA ROSA Madagascar

				Populated Places (2007)
Population Density	Estimated population density in Year 2000. Values represent people per pixel.	90m	Additional	WorldPop Version 2

Supplementary Table 1.5: Covariates used in other matching and regression studies as predictors of deforestation. Studies marked with an asterix * are based in Madagascar. Covariates shown are red were selected for use in this study.

Previous Studies	**Used Statistical Matching**	Distance to road	Distance to settlement	Population Density	Population Pressure	Elevation	Slope	Aspect	Distance to forest edge	Distance to River	Distance to recent deforestation	Vegetation type/Ecoregion	Annual Rainfall	Population Growth	Distance to nearest agricultural cell	Distance to border of NP	Infrastructure	Conservation Activity	Awareness of deforestation	Amount of Hatsaky, charcoal production and cattle ranching.	Cropland	Agricultural Suitability	Irrigated rice suitability
Brinkmann et al, 2014 *		✓	✓	✓		✓										✓	✓	✓	✓	✓	✓		
Agrawal et al, 2005*		✓		✓		✓	✓																
Vagen, 2006 *		✓	✓			✓	✓							✓	✓								
Rasolofoson et al, 2015 *	✓	✓	✓	✓		✓	✓		✓		✓											✓	✓
Eklund et al, 2016 *	✓	✓	✓			✓	✓			✓	✓	✓	✓										
McConnell et al, 2004 *			✓	✓	✓	✓	✓		✓	✓					✓								
Green and Sussman, 1990 *				✓			✓																
Nagendra et al, 2003		✓	✓			✓																	
Honey-Roses et al, 2011	✓	✓				✓	✓	✓															
Cuenca et al, 2016	✓	✓	✓			✓	✓																
Jones and Lewis, 2015	✓	✓	✓			✓	✓		✓	✓													
Andam et al, 2008	✓	✓	✓	✓		✓	✓		✓	✓													
Bruggeman et al, 2015	✓	✓	✓			✓	✓					✓											
Buntaine et al, 2014	✓	✓		✓		✓	✓				✓												
Costedoat et al, 2015	✓	✓	✓	✓		✓	✓																
Sills et al, 2015	✓			✓		✓	✓																
Arriagada et al, 2012	✓						✓																
Simmons et al, 2018	✓						✓																
Assuncao et al, 2015																							

Supplementary Table 1.6: Correlation between population density and the five essential variables in our full pre-matching sample of pixels. Overall refers to the full sample of control and treated pixels (i.e. from all four offsets). Columns labelled 2,3,4 and 5 refer to the full control sample plus the sample of pixels from the named offset. ANK = Ankerana, CZ = Conservation Zone, CFAM = Corridor Forestier Analamay-Mantadia, TTF = Torotorofotsy. Results obtained from a linear regression with population density as the dependent variable and distance to road, distance to edge, distance to recent deforestation, slope, and elevation as predictors.

	Dependent variable = Population Density				
	(1) Overall	(2) ANK + ctrl	(3) CFAM + ctrl	(4) CZ + ctrl	(5) TTF + ctrl
Dist_road	-0.003***	-0.003***	-0.003***	-0.003***	-0.003***
	(0.00002)	(0.00002)	(0.00002)	(0.00002)	(0.00002)
Dist_edge	-0.008***	-0.009***	-0.008***	-0.008***	-0.008***
	(0.0002)	(0.0002)	(0.0002)	(0.0002)	(0.0002)
Dist_defor	0.002***	0.002***	0.002***	0.002***	0.002***
	(0.0001)	(0.0001)	(0.0001)	(0.0001)	(0.0001)
Slope	0.197***	0.144***	0.170***	0.158***	0.163***
	(0.026)	(0.026)	(0.027)	(0.027)	(0.027)
Elevation	-0.110***	-0.106***	-0.108***	-0.107***	-0.107***
	(0.001)	(0.001)	(0.001)	(0.001)	(0.001)
Constant	286.550***	288.334***	288.376***	289.104***	288.836***
	(0.675)	(0.677)	(0.678)	(0.679)	(0.679)
Observations	641,437	636,301	636,065	634,779	634,609
R^2	0.107	0.109	0.108	0.109	0.109
Adjusted R^2	0.107	0.109	0.108	0.109	0.109

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

This shows that population density is significantly correlated with the five essential covariates used in our main matching specification.

Implementation of matching

Mahalanobis matching has been shown to produce better balance across individual covariates than propensity score matching and is appropriate and effective when there are small number of covariates upon which close matches are desired (Stuart, 2010; King and Nielsen, 2019).

Following the recommendations of Schleicher et al (2019) we tested several matching specifications and compared the resulting match quality before the deciding upon the main matching specification. All specifications used nearest-neighbour matching with Mahalanobis distance on the five essential covariates but the size of caliper (0.25, 0.5 and 1 standard deviation), matching with/without replacement and the ratio of treated to control units (1:1 and 1:5) were varied. Match quality was assessed through the post-matching standardised difference in mean covariate values between treated and control samples (Supplementary Table 1.7). Values less than 0.25 are generally considered to represent an acceptable match but the closer to zero the better (Stuart, 2010).

In selecting the most appropriate matching specification there is a trade-off between the quality of matches and the number of treated units for which a match can be found (Schleicher *et al.*, 2019). Setting a caliper of 0.25 standard deviations resulted in a very close matches but left hundreds of treated units un-matched. Rejecting treated units could bias the results if the un-matched units are non- random, i.e. if they share a common characteristic, such as location. Therefore, we chose the specification which matched all treated units in all offsets yet still produced a very good covariate balance (maximum standardised difference in means < 0.05). This was 1:1 nearest neighbour matching without replacement, using Mahalanobis distance and a caliper of 1sd. Neither matching with replacement nor with ratio a of 1:5 yielded a consistent improvement in balance in comparison.

Supplementary Table 1.7; Post-matching standardised difference in mean covariate values between each offset and matched control samples for each of the matching specifications tested**.** All specifications used nearest-neighbour matching with Mahalanobis distance on the five essential covariates. Unless otherwise specified in the column heading matching was conducted without replacement and with a ratio of 1:1. Only the parameter specified in the column heading was varied.

Covariates	**1 sd caliper**	**0.5 sd caliper**	**0.25 sd caliper**	**0.5 sd caliper + matching with replacement**	**1 sd caliper + ratio of control to treated units of 1:5**
Ankerana (N=2862 pixels)					
Slope	-0.0003	-0.0007	0.0011	0.0008	0.0091
Elevation	0.0035	0.0079	0.0073	-0.0091	-0.022
Distance to road	-0.017	-0.012	0.0003	-0.04	-0.033
Distance to edge	0.04	0.04	0.022	0.019	0.06
Distance to deforestation	0.033	0.029	0.013	0.019	0.031
Number of un-matched treated units	0	57	426	5	0
CFAM (N = 2626)					
Slope	-0.012	-0.0091	-0.0035	-0.0058	-0.0061
Elevation	-0.055	-0.047	-0.016	-0.014	-0.2
Distance to road	-0.0041	-0.0043	-0.0014	0.013	0.027
Distance to edge	0.013	0.0062	0.0099	0.016	-0.044
Distance to deforestation	0.039	0.031	0.013	0.019	0.09
Number of un-matched treated units	0	95	649	15	0
Conservation Zone (N= 1340)					
Slope	0.002	0.0036	-0.0011	0.0043	0.03
Elevation	0.039	0.04	0.028	0.026	0.077

Distance to road	-0.0083	-0.0065	0.0036	-0.012	-0.053
Distance to edge	0.013	0.011	0.011	0.013	-0.016
Distance to deforestation	0.034	0.037	0.015	0.021	0.074
Number of un-matched treated units	0	10	343	2	0
Torotorofotsy (N= 1170)					
Slope	-0.0072	-0.0074	-0.009	-0.011	-0.0009
Elevation	0.0042	0.0062	0.0022	-0.0032	-0.021
Distance to road	0.019	0.021	0.015	0.0011	0.04
Distance to edge	0.0065	0.0067	0.0028	0.0099	-0.045
Distance to deforestation	0.017	0.019	0.016	0.016	0.032
Number of un-matched treated units	0	2	158	2	0

Calculating counterfactual and avoided deforestation

To estimate the amount of deforestation which would have occurred each year in the offsets in the absence of protection we use the estimated treatment effect to convert observed deforestation to counterfactual levels. For example, results from our site-based difference-in-differences regression showed that protection reduced average annual deforestation by an estimated 96% in Ankerana. In other words, observed deforestation was 4% of the estimated counterfactual. To convert the area of observed deforestation each year to the counterfactual levels we used the following formula:

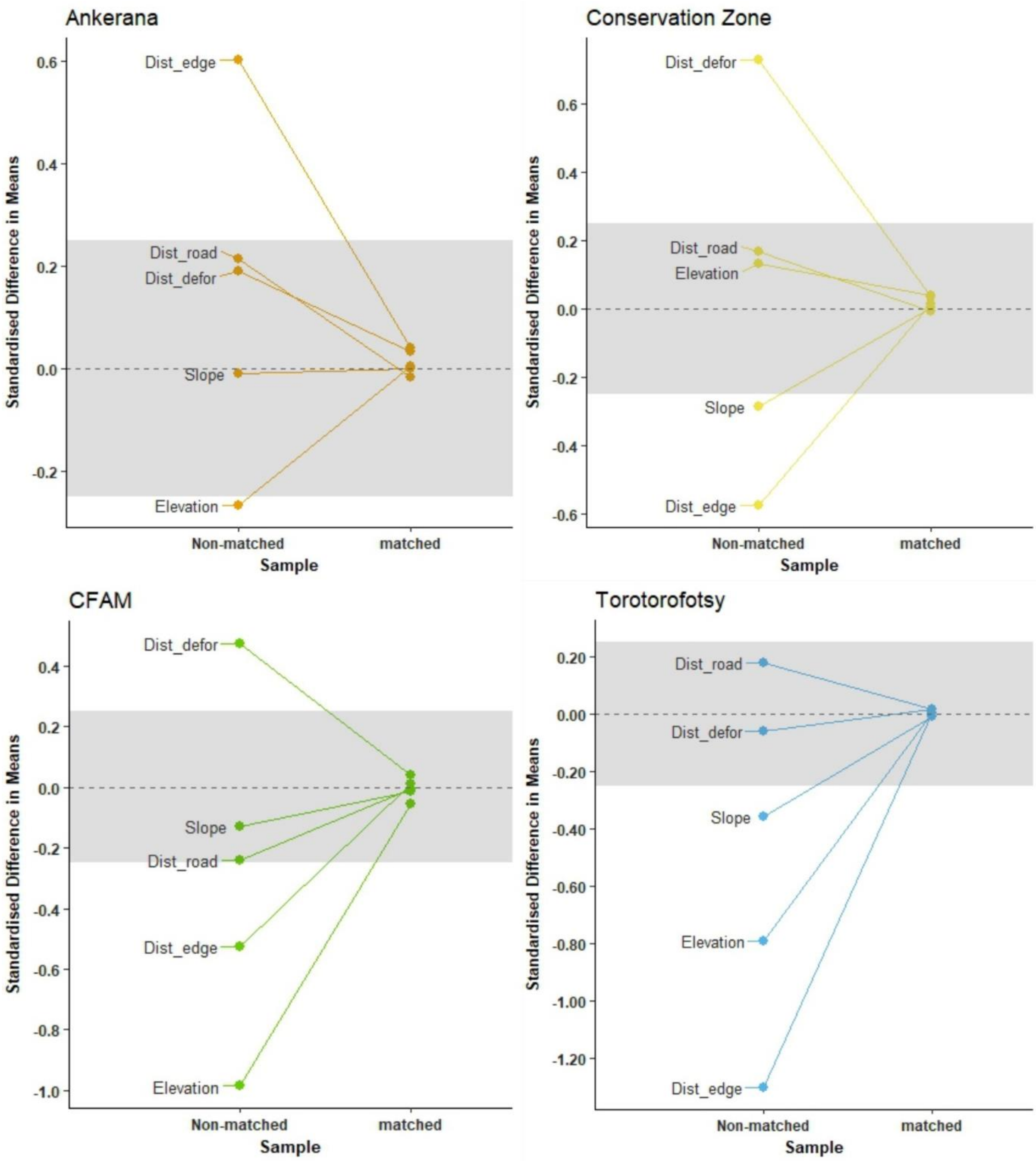
$$Counterfactual\ deforestation = observed\ deforestation\ \times \frac{100}{(100 - Treatment\ Effect)}$$

Avoided deforestation is subsequently defined as the difference between the observed and estimated counterfactual deforestation.

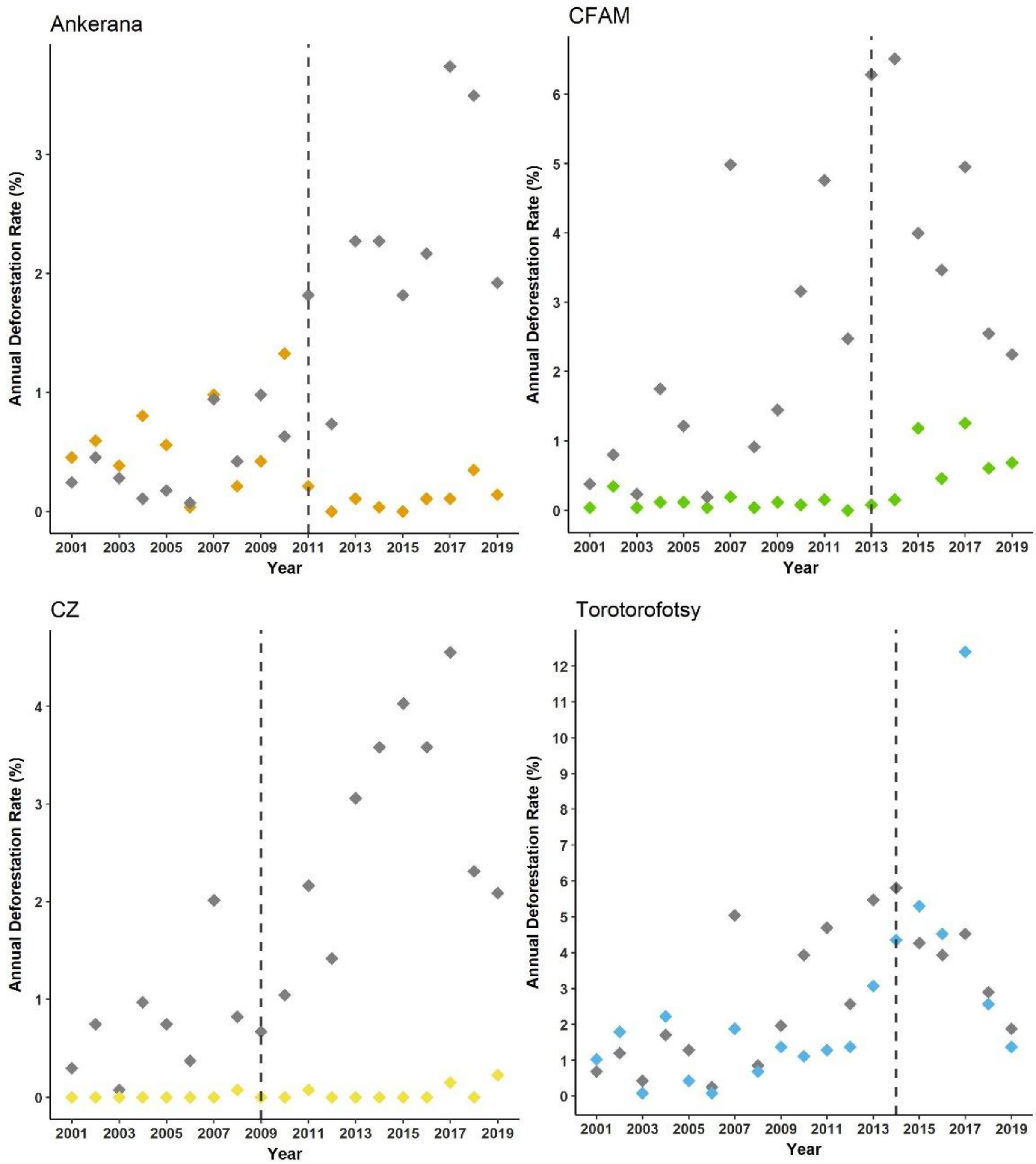
We calculated the upper and lower confidence intervals around our estimates of counterfactual, and consequently avoided deforestation, using the upper and lower confidence intervals of the estimated treatment effect.

Supplementary Results 1

Matching



Supplementary Figure 1.3; Evaluation of the quality of matches produced with the main matching specification. This is assessed through the standardised difference in mean covariate values between offset and matched control samples. The shaded grey area indicates the ± 0.25 interval widely considered an acceptable match. The maximum standardised difference in mean covariate values was 0.05, well below the acceptable threshold.

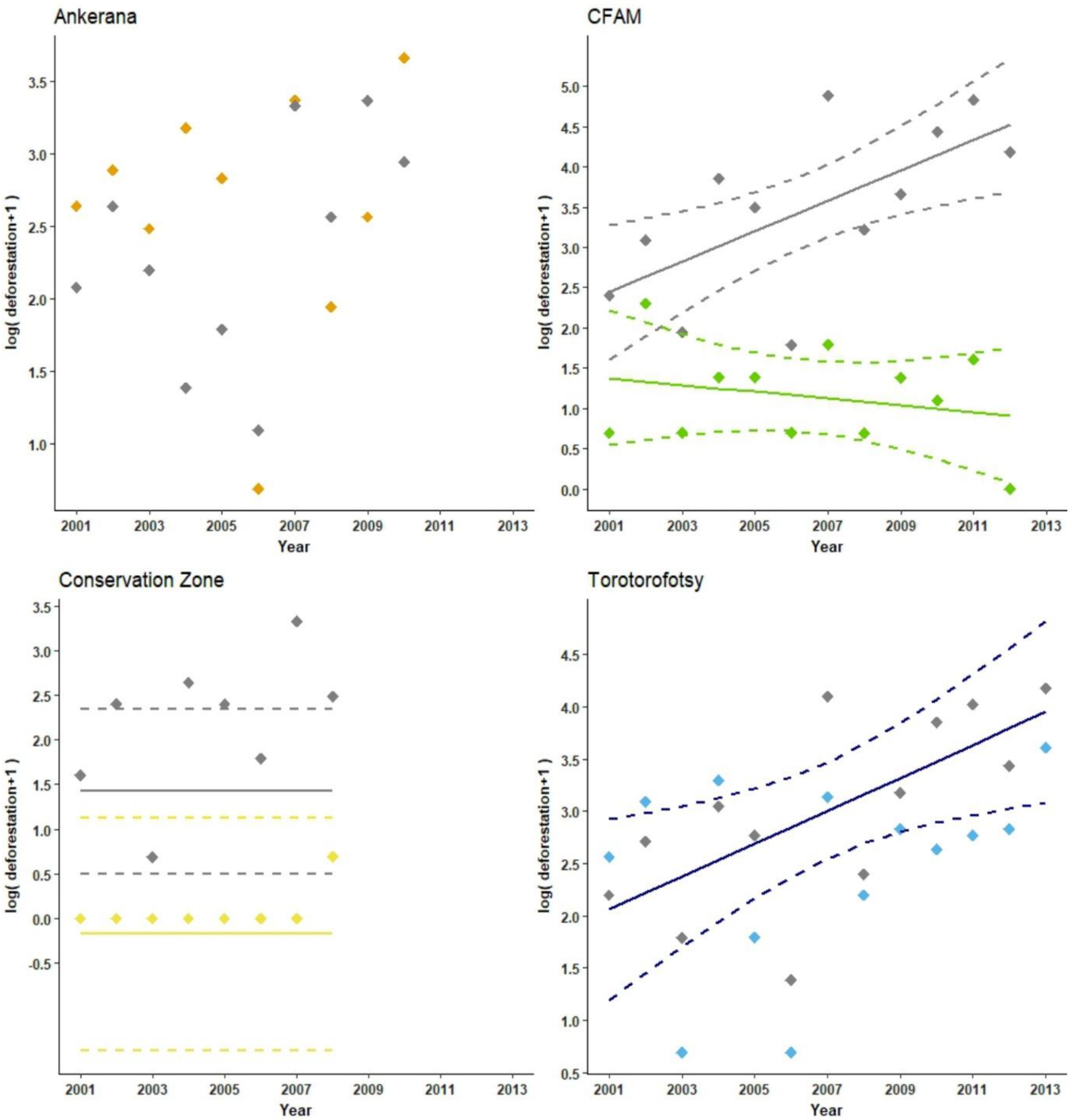


Supplementary Figure 1.4; Comparison of the annual deforestation rate within the sample of pixels from each offset and the matched controls over the whole study period. The offset sample is shown in colour whilst the matched control sample is shown in grey. The dashed line indicates the year of protection. The offset and matched control samples contain an equal number of pixels (2862 for Ankerana, 2626 for CFAM, 1340 for the Conservation Zone and 1170 for Torotorofotsy) as the ratio of treated to control units in the matching was set to 1:1. For each offset, N = 38.

Before Ankerana was protected it experienced similar rates of deforestation to the matched control sample, but this diverged considerably following protection. For CFAM, deforestation became increasingly higher within the matched control sample relative to the offset sample in the years before protection of the offset. However, contrary to expectations, after protection deforestation increased within the offset but declined within the matched control sample. Torotorofotsy experienced a similar magnitude and pattern of deforestation to the matched control both before and after protection. Excluding 2017, which was a record year for deforestation within Torotorofotsy (12.4% of all sample pixels were deforested in that year), forest loss within both the offset and matched control sample has declined since 2014 when the offset was protected. Forest loss within the Conservation Zone was extremely low throughout the study period with zero deforestation occurring in the sample for 15 out of the 19 years. By contrast, deforestation within the matched control was much higher and increased rapidly after 2009. An escalation in deforestation rates for several years from 2009 was also found in the matched control samples for the other three offsets. This coincides with a period of political and economic instability triggered by the political coup in 2009. This increased poverty, insecurity, and corruption, and severely weakened the capacity of the authorities to enforce forest laws, leading to increased deforestation across Madagascar (Desbureaux et al, 2016; Tabor et al, 2017).

Outcome Regressions

Testing the assumptions



Supplementary Figure 1.5; Test for parallel trends in deforestation between each offset and its matched control sample in the pre-intervention period. Points show the log(y+1) transformed count of deforested pixels within each offset (shown in colour) and its matched control sample (grey). Lines plot the significant coefficients from the linear regression model:

$$\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 Year_t + \beta_2 CI_i + \beta_3 Year * CI_{it} + \epsilon$$

where *i* denotes the sample, *t* denotes the year and CI is a binary variable indicating whether the observation is from the offset (1) or control (0) sample (see Supplementary Table 1.8). Diagonal lines indicate significant temporal trends in the data (Year is a significant predictor) whilst paired horizontal lines indicate a significant difference in deforestation, on average, between the two samples. Dashed lines represent the 95% confidence intervals around the significant coefficients. Lines are coloured according to whether the coefficient corresponds to the offset or the matched control sample except for Tototorofotsy where the line applies to both. This is because the slope of the relationship between year and the log-transformed count of deforestation does not differ significantly between the two samples. N= 20 for Ankerana, 24 for CFAM, 16 for the CZ and 26 for Torotorofotsy.

In Ankerana neither year nor treated status were significant predictors of the annual deforestation rate (shown by the lack of lines in the Figure above). This indicates that there were no temporal trends in deforestation in either the offset or the matched control sample in the pre-intervention period. In CFAM, there was a significant declining trend in deforestation prior to protection whilst the matched control sample showed a significant increasing trend (Supplementary Figure 1.4 suggests these trends may have reversed post-intervention). This violates the assumption of parallel trends meaning CFAM cannot be used in the difference-in-differences analysis. There were no significant trends in deforestation over time in the Conservation Zone nor its matched control sample, however, on average, deforestation was significantly lower within the offset than the matched control sample (shown by the two horizonal lines in Supplementary Figure 1.5). Torotorofotsy did experience a trend in deforestation over time which was not significantly different to the trend in the matched control sample, hence the single trend line. Furthermore, there was no significant difference in deforestation, on average, between the two samples. Therefore, Ankerana, the Conservation Zone, and Torotorofotsy show parallel trends in deforestation to their matched control sample in the pre-intervention period and can therefore be used in the difference-in-differences analysis. However, there is an important caveat to this in that the small sample size in these regressions (N = 16 for the Conservation Zone) produces large uncertainty, reducing the likelihood of finding a significant difference in trend even if the true trends are not parallel.

Raw results from parallel trends test

Supplementary Table 1.8; Raw results from the test for parallel trends in deforestation between each offset and its matched control sample in the pre-intervention period. CI is a binary variable indicating treatment status (i.e., whether the observation is from an offset (1) or matched control sample (0). The interaction between Year and CI is the coefficient of interest, which indicates whether the relationship between the Year and the log(y+1) transformed count of deforestation differs significantly between treated and control samples. This coefficient is only significant (p = 0.0153) for CFAM.

Offset		Intercept	Year	CI	Year:CI
Ankerana	estimate	1.6779	0.1204	0.8589	-0.1042
	std.error	0.5516	0.0889	0.7801	0.1257
	statistic	3.0418	1.3543	1.1010	-0.8287
	p.value	0.0078	0.1945	0.2872	0.4195
CFAM	estimate	2.2526	0.1893	-0.8352	-0.2313
	std.error	0.4540	0.0617	0.6421	0.0872
	statistic	4.9616	3.0685	-1.3009	-2.6510
	p.value	0.0001	0.0061	0.2081	0.0153
CZ	estimate	1.4261	0.1649	-1.5994	-0.1072
	std.error	0.4238	0.0839	0.5993	0.1187
	statistic	3.3650	1.9653	-2.6685	-0.9030
	p.value	0.0056	0.0730	0.0205	0.3843
TTF	estimate	1.9016	0.1575	-0.0023	-0.0755
	std.error	0.4709	0.0593	0.6660	0.0839
	statistic	4.0379	2.6546	-0.0035	-0.9001
	p.value	0.0005	0.0145	0.9972	0.3778

Note: N= 20 for Ankerana, 24 for CFAM, 16 for the CZ, and 26 for Torotorofotsy

Site-based difference-in-differences regression

Supplementary Table 1.9; Results from the site-based difference-in-differences regressions for each offset-control sample that met the condition of parallel trends. Results are from the regression:

$\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 BA_t + \beta_2 CI_i + \beta_3 BA * CI_{i,t} + \in$,

where BA and CI are a binary variables indicating whether the observation is from the period before (0) or after (1) protection (BA), from the offset (1) or matched control (0) sample (CI). BA:CI is the coefficient of interest which represents the effect of being in an offset after protection on the log-transformed count of deforestation. Back-transforming this estimate $((\exp(estimate) - 1) \times 100)$, gives the treatment effect, expressed as the percentage difference in average annual deforestation between the offset and the estimated counterfactual following protection. Counterfactual deforestation is estimated by adjusting the average annual deforestation within the matched control sample after the intervention, to account for the pre-intervention difference in deforestation between the two samples.

Offset		Intercept	BA	CI	BA:CI
Ankerana	estimate	2.3401	0.2859	1.7573	-3.1827
	std.error	0.2347	0.3319	0.3410	0.4823
	statistic	9.9704	0.8614	5.1530	-6.5994
	p.value	0.0000	0.3951	0.0000	0.0000
CZ	estimate	2.1683	-2.0816	1.2768	-1.0746
	std.error	0.1997	0.2824	0.2625	0.3712
	statistic	10.8572	-7.3704	4.8647	-2.8949
	p.value	0.0000	0.0000	0.0000	0.0066
TTF	estimate	3.0042	-0.5310	0.7785	0.6374
	std.error	0.2290	0.3238	0.4075	0.5763
	statistic	13.1195	-1.6397	1.9104	1.1060
	p.value	0.0000	0.1103	0.0645	0.2765

Note: N= 38

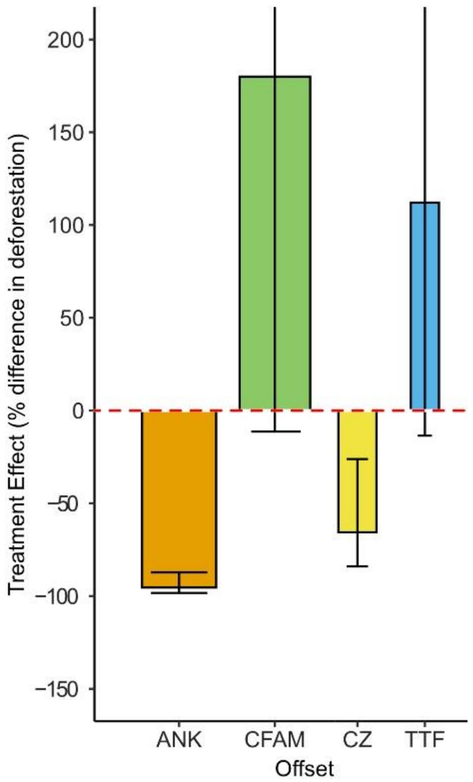
These results show highly significant reductions in deforestation of 96% (95% CI: 89 to 98%) in Ankerana and 66% (27 to 84%) in the Conservation Zone. In Torotorofotsy, average annual deforestation was higher in the offset than the estimated counterfactual after protection, but this difference was not significant.

Site-based difference-in-differences regression with alternative temporal specification

As an additional robustness check we repeated our site-based difference-in-differences regressions using an equal number of years before and after treatment and corresponding alternative baseline year for each offset.

Supplementary Table 1.10: Results from the site-based difference-in-differences regression with alternative temporal specification. The estimate of BA:CI represents the effect of being in an offset after protection on the log-transformed count of deforestation. These values were back-transformed as detailed above to give the treatment effect, expressed as the percentage difference in average annual deforestation between the offset and the estimated counterfactual**.**

Sample	BA:CI estimate	Standard error	Parallel trends	N years before and after	Treatment effect	Upper CI	Lower CI	df
ANK	-3.0789	0.5206	TRUE	8	-95.3992	-86.0437	-98.4833	15
CFAM	1.0297	0.5211	TRUE	6	180.0124	781.5570	-11.0586	11
CZ	-1.0693	0.3910	TRUE	8	-65.6742	-21.0103	-85.0834	15
TTF	0.7520	0.4157	TRUE	5	112.1211	443.2284	-17.1705	9

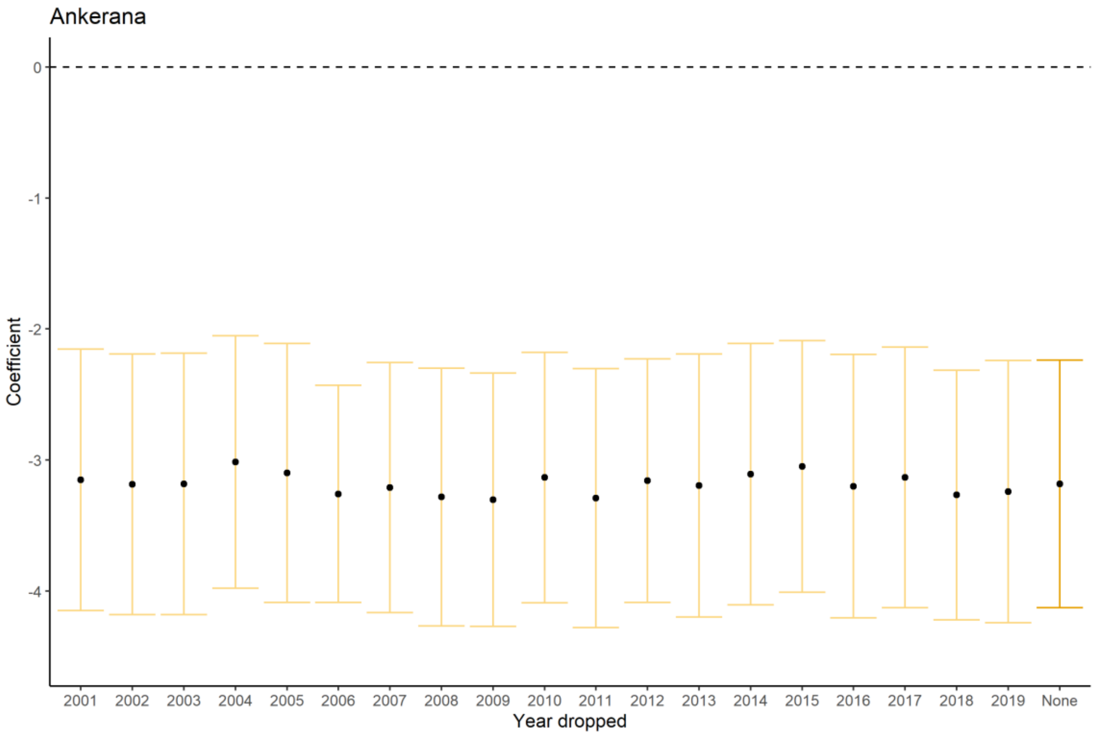


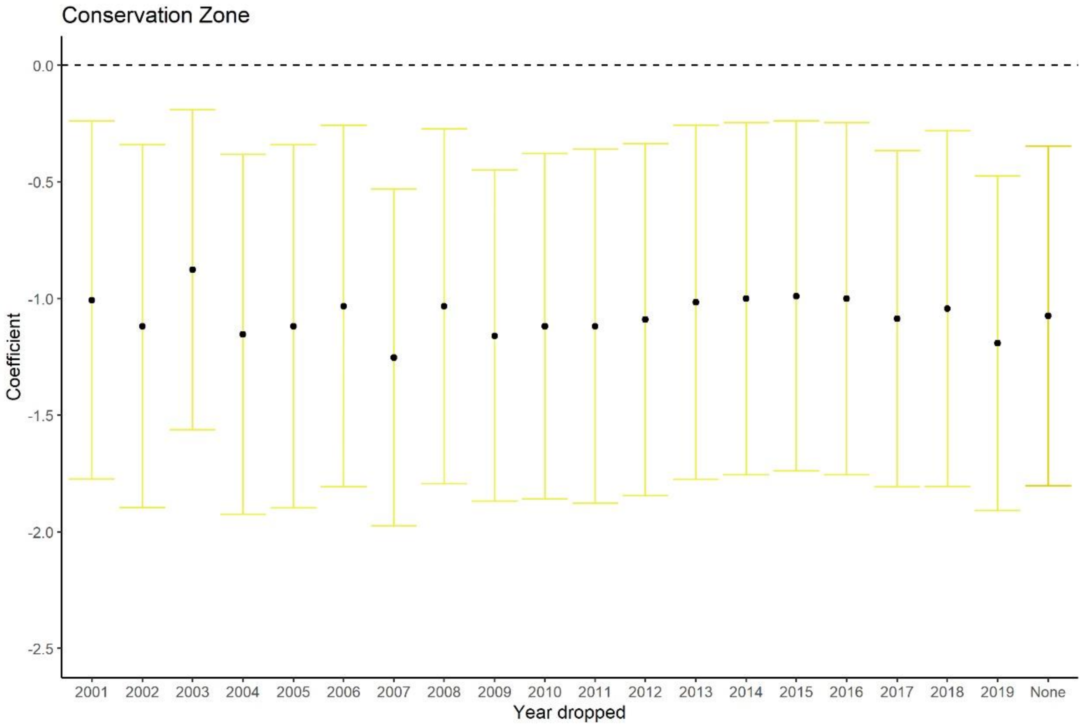
Supplementary Figure 1.6: The estimated percentage reduction in deforestation within each offset from the site-based difference-in-differences regression with alternative temporal specification. Error bars represent 95% confidence intervals (the upper bound extends to +443% for TTF and +782% for CFAM). The width of the bar is proportional to the area of forest within each offset at the year of protection (Supplementary Table 1.2). ANK = Ankerana, CFAM = Corridor Forestier Analamay-Mantadia, TTF = Torotorofotsy. N = 16 for Ankerana, 12 for CFAM, 16 for the Conservation Zone and 10 for Tototorofotsy.

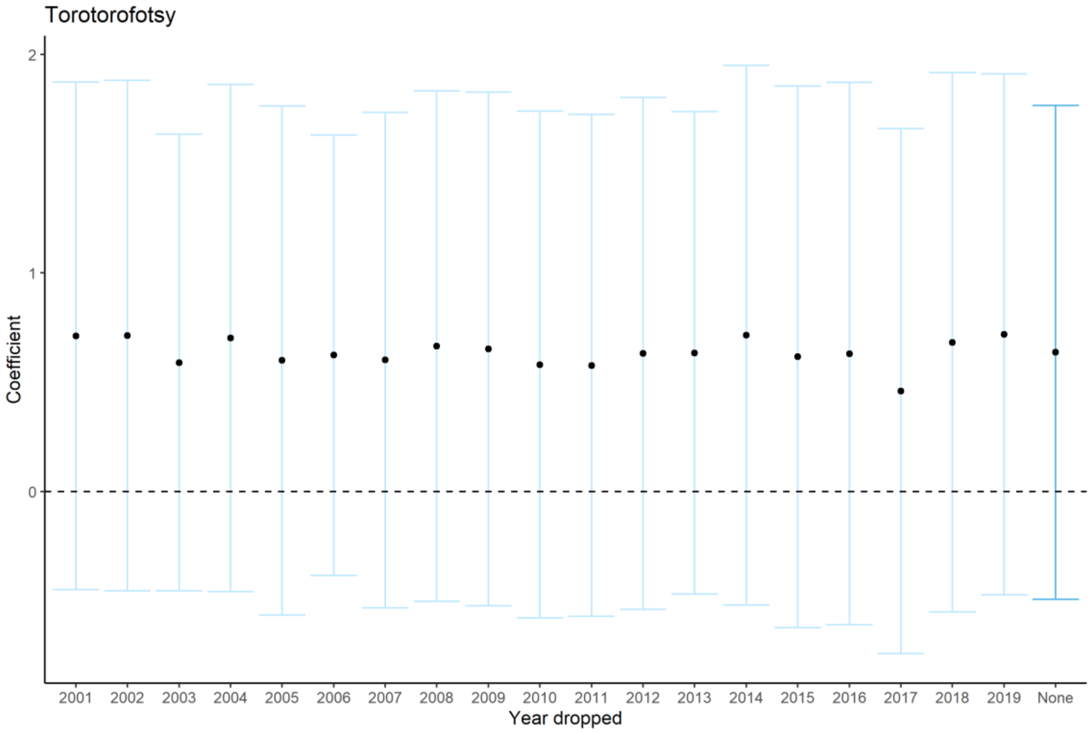
These results are consistent with those obtained from our main modelling specification. They show a significant reduction in deforestation of 95.4% in Ankerana and 65.7% in the Conservation Zone. This is extremely close to the estimates from the main specification of 96% and 66% respectively. Whilst the estimated increase in deforestation within Torotorofotsy relative to the counterfactual was higher than the main specification (+112% compared to +89%), the effect remained insignificant. In contrast to the main specification, CFAM met the condition of parallel trends meaning it could be assessed in the difference-in-differences analysis. This showed higher deforestation within CFAM than the estimated counterfactual but this difference was not statistically significant. However, the small sample size (eg. N = 10 for Torotorofotsy) produces very large uncertainty, decreasing the likelihood of showing a significant effect, either positive or negative. Given this caveat, our finding of a significant negative effect in Ankerana and the Conservation Zone indicates the strength of the signal of reduced deforestation within these offsets. Overall, this analysis shows that our results are robust to an alternative temporal specification.

Site-based difference-in-differences regression dropping individual years

Given the relatively small sample size (N = 38) of our difference-in-differences regressions we tested whether our results were influenced by a single data value by repeating the each regression 19 times, each time dropping an observation for one year.







Supplementary Figure 1.7: Results from the site-based difference-in-differences regression dropping an individual year from the analysis. Points represent the estimated raw treatment effect; the coefficient of BA*CI from the difference-in-differences regression. Year dropped refers to the year of the observation which was removed from the analysis. Bars show the 95% confidence intervals around the estimated treatment effect. Results from the main specification including all years (Supplementary Table 1.9) are shown in the darker shade. N = 36 for all estimates except where Year dropped is None where N = 38. Results are only included for the three offsets which passed the parallel trends test.

This shows that our results are robust to removal of individual years from the analysis and are therefore not likely to be influenced by a single data point.

Site-based difference-in-differences regression including a time trend

Supplementary Table 1.11: Comparison of results from our site-based difference-in-differences regression with and without a time trend. The difference-in-differences regression with a time trend takes the form:
$\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 BA_t + \beta_2 CI_i + \beta_3 BA * CI_{i,t} + Year_t + \epsilon_{i,t}$.
BA*CI is the coefficient of interest. The column 'With Year' shows the results of the difference-in-differences regression including Year as a predictor. The standard error is shown in brackets below the estimate.

	Ankerana		Conservation Zone		Torotorofotsy	
	Main	With Year	Main	With Year	Main	With Year
CI (Treated Status)	0.286	0.286	-2.082***	-2.082***	-0.531	-0.531*
	(0.332)	(0.319)	(0.282)	(0.244)	(0.324)	(0.303)
BA (Before-After)	1.757***	0.973*	1.277***	0.336	0.778*	-0.102
	(0.341)	(0.518)	(0.262)	(0.350)	(0.407)	(0.530)
BA*CI	-3.183***	-3.183***	-1.075***	-1.075***	0.637	0.637
	(0.482)	(0.463)	(0.371)	(0.321)	(0.576)	(0.540)
Year		0.083*		0.099***		0.093**
		(0.042)		(0.028)		(0.039)
Constant	2.340***	1.886***	2.168***	1.723***	3.004***	2.355***
	(0.235)	(0.324)	(0.200)	(0.214)	(0.229)	(0.346)
Observations	38	38	38	38	38	38

Adjusted R^2	0.643	0.671	0.866	0.900	0.278	0.366

$^{*}p< 0.1$, $^{**}p< 0.05$, $^{***}p<0.01$

This shows that the addition of time trends to our canonical difference-in-differences model does not change our estimated treatment effect. In fact, it decreases the standard error of the coefficient of interest (BA*CI), increasing the significance of our results.

Fixed Effects Panel Regression

We conducted a secondary analysis using a fixed effects panel regression to obtain an overall estimate of treatment effect across all four biodiversity offsets, controlling for time-invariant unobserved heterogeneity. Results show that protection reduced deforestation across all four biodiversity offsets by an average of 58% (37 -73%) per year (Column 1, Supplementary Table 1.12).

To test whether inclusion of the fixed effects was necessary we ran an F-test for individual and time effects, comparing the fixed effects model to a pooled OLS regression of the following form:

$$\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 Tr_{i,t} + \epsilon_{i,t}$$

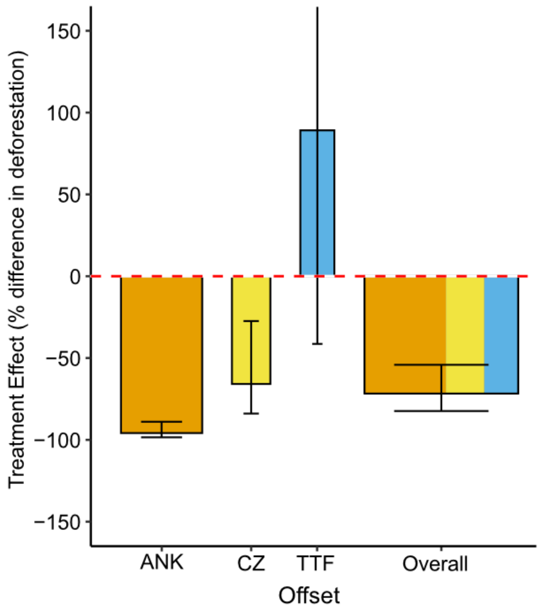
This revealed significant ($p < 0.05$) heterogeneity between sites and over time, supporting the inclusion of these variables as fixed effects in the modelling process.

Although fixed effects panel regressions are not based on an identifying assumption of parallel trends between groups in the pre-treatment period (Cunningham, 2021), we tested the effect of excluding CFAM and its matched control sample (which show diverging pre-treatment trends in deforestation) from the regression (Column 2, Supplementary Table 1.12). We found that this increased the precision and magnitude of the estimated treatment effect. Excluding CFAM increased the estimated average reduction in deforestation from 58% to 72% per year (95% CI: 54 to 83%) within the remaining three offsets. This translates to 2,221 ha (1039 to 4132 ha) of avoided deforestation between the year of protection of each offset and January 2020, exceeding the 2,064 ha of forest loss at the mine site which was required to be offset. Therefore, according to this estimate, Ambatovy has already achieved No Net Loss of forest. However, in the main text, we prefer to highlight the more conservative estimate, which incorporates the effect of all four offsets.

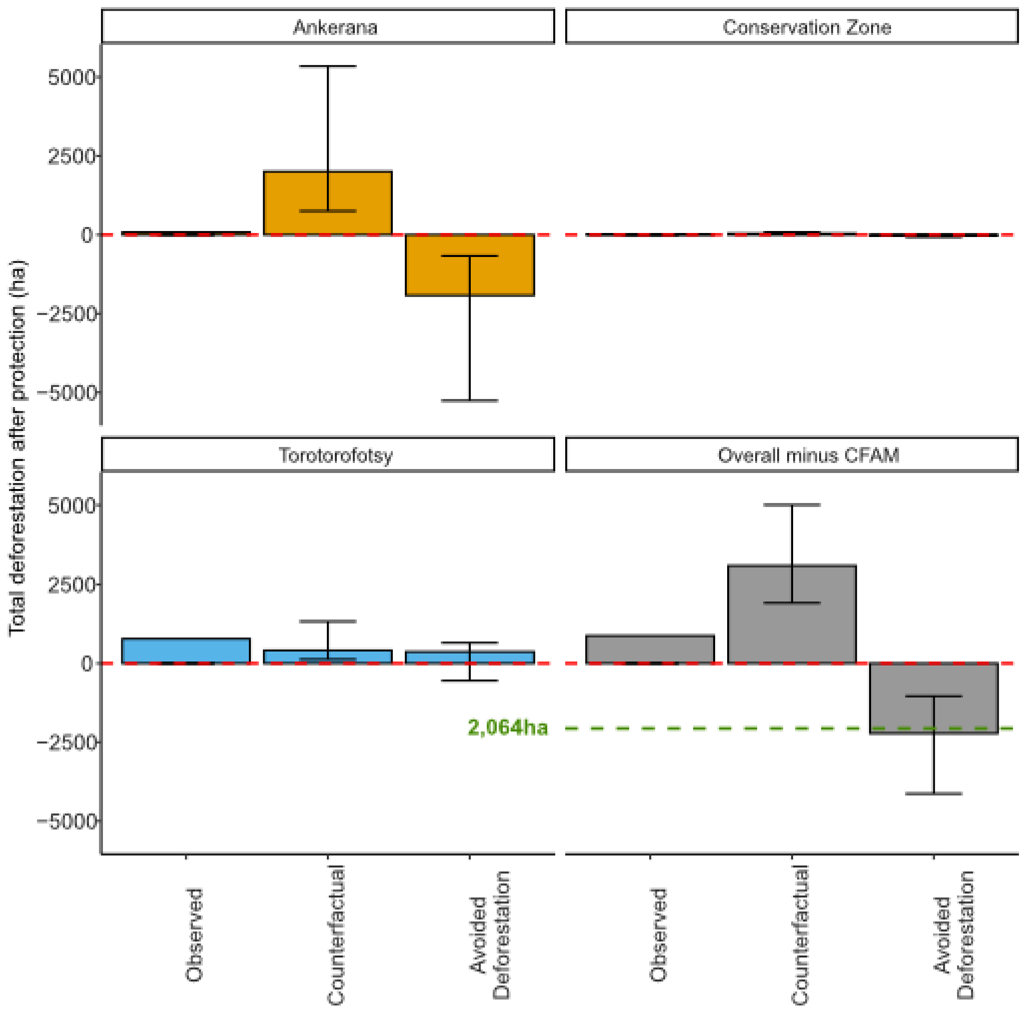
Finally, we tested the robustness of our results from the fixed effects panel regression to the alternative specification of site and year as random effects (Column 3, Supplementary Table 1.12). This gives a significant overall reduction in deforestation of 53% (27% to 69%) per year, following protection of the four biodiversity offsets. This estimate is within the confidence intervals, and extremely close, to the estimate derived from the fixed effects panel regression.

Supplementary Table 1.12; Results from the fixed effects panel regression on the pooled data. The fixed effects panel regression takes the form $\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 Tr_{i,t} + \propto_i + \gamma_t + \epsilon_{it}$,
where Tr is a binary measure indicating the treated status of observation *i* in year *t* (Tr = 1 for observations from offset sites in the years following protection and 0 for all other observations), $\propto_i$ and γ_t represent site and year fixed effects respectively (modelled as random effects in model 3), and ϵ_{it} represents the composite error. Tr is the coefficient of interest which represents the effect of being in an offset after protection on the log(y+1) transformed count of deforestation. The estimate is back-transformed as described above to express the treatment effect as the percentage difference in average annual deforestation following protection. The table shows the results from the main fixed effects model specification (Column 1) and two alternative specifications to test the robustness of results to the exclusion of CFAM (Column 2) and the designation of site and year as random effects (Column 3). Models 1 and 3 were run on the full pooled data comprising an observation for each site (i=8, four offset and four control) for each year (t =19). Model 2 only included observations for Ankerana, the Conservation Zone and Torotorofotsy (i = 6, t= 19).

Term	**Treatment effect (Tr)**		
Model	All four offsets	Excluding CFAM and its matched control sample	With Site and Year as random effects
	(1)	(2)	(3)
estimate	-0.8774	-1.2631	-0.7476
std.error	0.2154	0.2453	0.207
statistic	-4.0732	-5.1489	-3.612
p.value	0.0001	0.000002	0.0004
N	152	114	152
df	125	89	125



Supplementary Figure 1.8: The estimated percentage reduction in annual deforestation within each offset (from the site-based difference-in-differences regressions) and overall, across the three offsets which met the condition of parallel trends (from the fixed effects panel regression excluding CFAM). The treatment effect is expressed as the average percentage difference in annual deforestation between the three offset(s) and the estimated counterfactual following protection. Error bars represent 95% confidence intervals (the upper bound for TTF extends to +510%). The width of the bar is proportional to the area of forest within each offset at the year of protection (Supplementary Table 1.2). ANK: Ankerana (orange), CZ: the Conservation Zone (yellow), TTF: Torotorofotsy (blue). N = 38 for Ankerana, the Conservation Zone and Torotorofotsy and N = 114 for the Overall result.



Supplementary Figure 1.9: The total observed, counterfactual, and the resulting estimate of avoided deforestation within each offset (estimated using site-based difference-in-differences regressions) and overall, for the three offsets which met the condition of parallel trends (using the fixed effects panel regression excluding CFAM), between the year of protection and January 2020. The counterfactual is an estimate of the deforestation which would have occurred in the absence of protection and was calculated using the estimated treatment effect (N = 38; Supplementary methods 1). Avoided deforestation is the difference between the observed and counterfactual deforestation; negative values indicate the offset resulted in a reduction in deforestation. The error bars show the 95% confidence interval of the estimates of counterfactual deforestation (derived from the upper and lower confidence intervals of the treatment effect) and the resulting estimates of avoided deforestation. The green dashed line indicates the 2,064 ha of forest loss caused by the mine itself. The number of years following protection is nine for Ankerana, 11 for the Conservation Zone, six for Torotorofotsy and 11 Overall (deforestation within later protected offsets is only counted from the year of protection).

Comparison of our estimated effect size to those of other interventions aimed at slowing deforestation

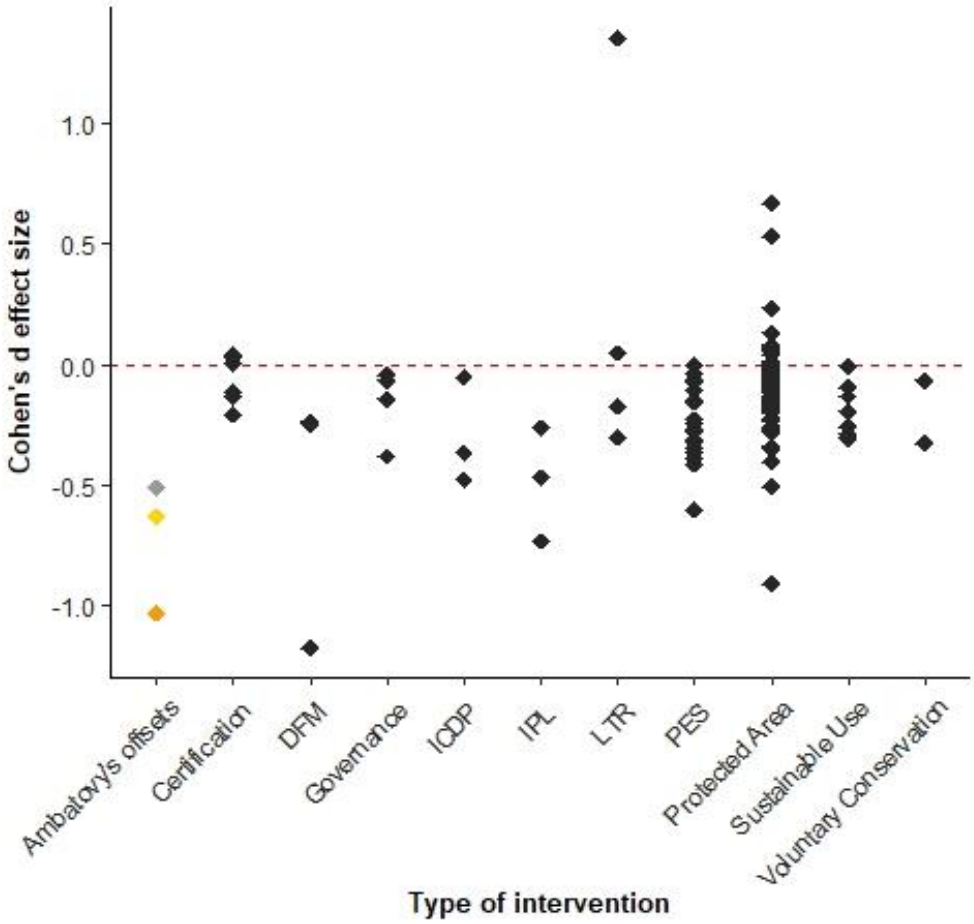
In a recent review Borner et al (2020) compiled and summarised the results of 99 studies using counterfactual methods to evaluate the effectiveness of various forest conservation interventions at reducing deforestation. From these studies the authors obtained estimates of effect size for 136 conservation interventions, which they converted to a normalised Cohen's d effect size for comparison. The interventions were grouped by type (eg. protected areas, Payments for Ecosystem Services, land titling reform) to assess whether certain forms of intervention were more successful than others.

To enable comparison of our results to those compiled by Borner et al we converted our estimate of avoided deforestation to a Cohen's d effect size using the following formula (Cohen, 1988):

$$Cohen's\ d\ effect\ size = \frac{(mean\ treated - mean\ counterfactual)}{standard\ deviation(counterfactual)}$$

where the numerator refers to the difference in average annual deforestation between the offset(s) and the estimated counterfactual (calculated as described above) following protection, and the denominator is the standard deviation of the counterfactual annual deforestation in the post-intervention period. While Cohen's d is usually calculated using standard deviation of the pooled samples, we follow Borner et al (2020) in using the standard deviation of the control sample.

The Cohen's d statistic is -1.03 for Ankerana classed as a 'large effect' (Cohen, 1988), -0.63 for the Conservation Zone and – 0.51 overall (across the entire offset portfolio), both classed as 'medium effects'.



Supplementary Figure 1.10: Comparison of normalised Cohen's d effect sizes for Ambatovy's biodiversity offsets and 136 other conservation interventions compiled by Borner et al. Coloured points show the statistically significant results from this study converted to a normalised Cohen's d effect size. Orange = Ankerana, yellow = the Conservation Zone and grey = the Overall effect of Ambatovy's four biodiversity offsets (from the fixed effects panel regression). Black points represent the normalised effect sizes of 136 conservation interventions grouped by type from Borner et al. DFM = Decentralised Forest Management, ICDP = Integrated Conservation and Development Programmes, LTR = Land Titling and Reform, PES = Payments for Ecosystem Services. Negative values indicate the intervention led to a reduction in forest loss.

This comparison shows that overall Ambatovy's biodiversity offsets were more effective at reducing deforestation than 97% of the other interventions and all bar one of the protected area interventions. Ankerana was the second most effective intervention overall and the most effective protected area intervention. These results are particularly striking and reinforce the need for future work to evaluate the reasons behind

Ambatovy's apparent success at conserving its biodiversity offsets as this could help to inform and improve offsetting and conservation practices more broadly.

Robustness checks

We evaluated the extent to which our primary results, derived from the main matching specification, are affected by arbitrary modelling choices following the procedure proposed in Desbureaux (2021). In this study, arbitrary modelling choices concerned the selection of covariates (5 essential covariates included), matching distance measure (Mahalanobis), value of the calipers (1 SD), matching without replacement, and the number of nearest neighbours to match on (1 nearest neighbour).

We tested the robustness of our results to the inclusion of five additional covariates (Supplementary Table 1.4), alternative matching distance measures (standard PSM and Random Forest PSM), caliper values (0.25 and 0.5 SD), matching with replacement, and different numbers of nearest neighbours (5 and 10 nearest neighbours), in three stages.

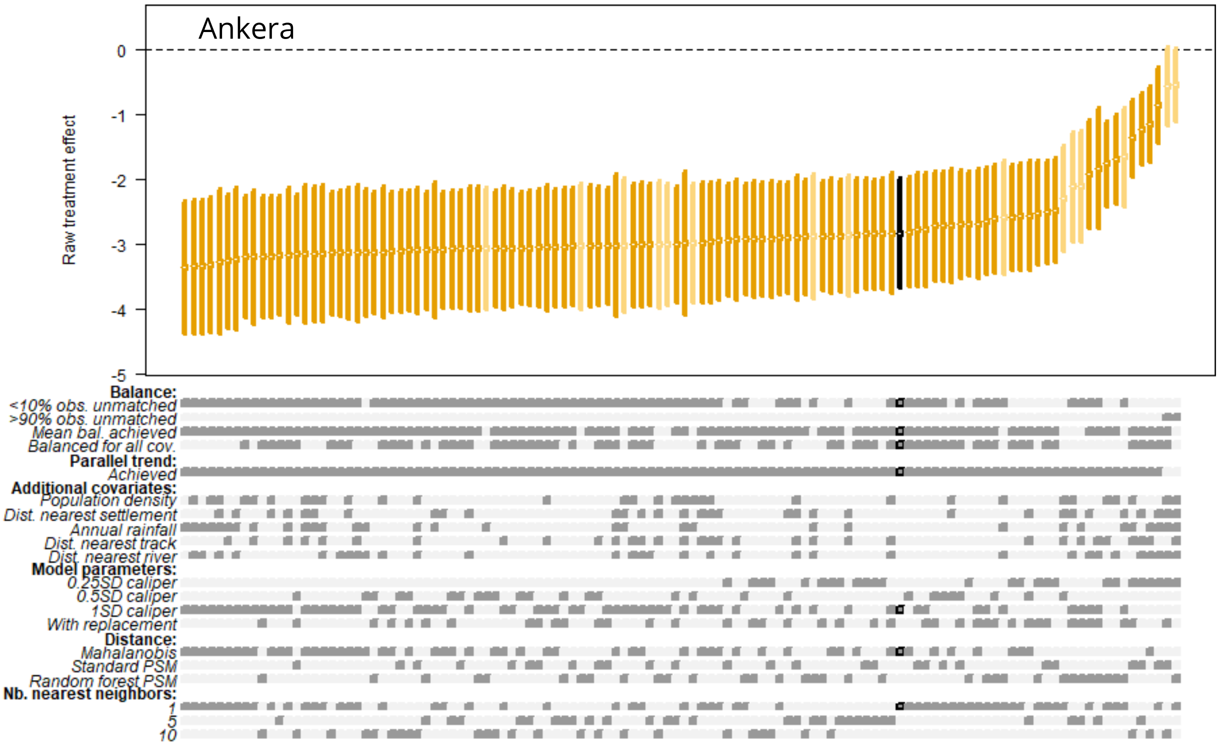
First, holding the choice of covariates constant (using only the essential covariates) we tested the robustness of results to alternative matching distance measures and model parameters (calipers, number of nearest neighbours, matching with/without replacement). This led to the estimation of 54 different models.

Second, we tested the robustness of results to the inclusion of the five additional covariates. Holding the choice of distance measure and model parameters constant, we constructed 31 models based on all possible combinations of additional covariates with the core set of essential covariates.

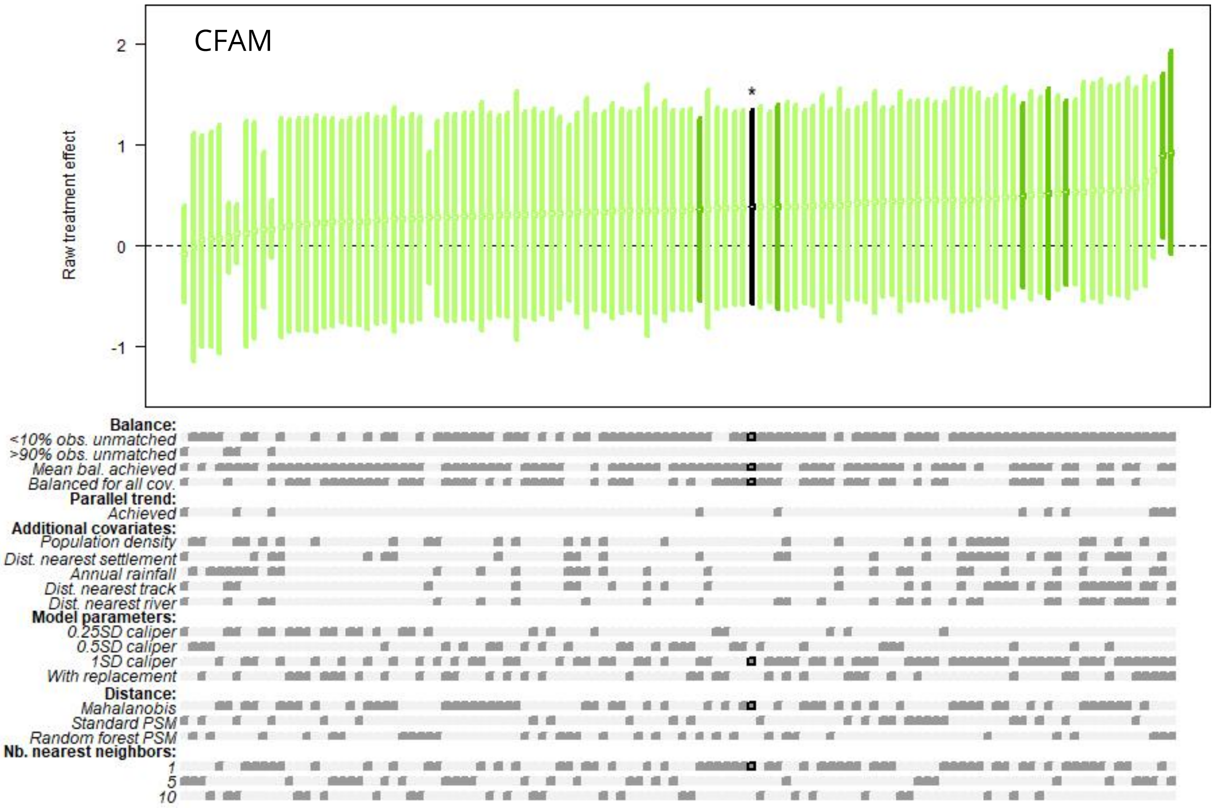
Finally, we explore the robustness of results for 31 randomly selected combinations of distance measure, model parameters and additional covariates.

All models are *a priori* valid, as they follow the best practice guidelines defined by Schleicher et al (2019). However, they are considered *a posteriori* invalid if they meet any of the following three conditions: 1) no adequate matches are found for over 90% of treated observations; 2) the post-matching average covariate balance (defined as the standardised difference in means) is above the accepted threshold of 0.25; and 3) the resulting matched data violate the assumption of parallel trends in outcomes between treated and control samples in the pre-intervention period. Failure to match a large number of treated observations leads to their rejection from the sample which could bias results if the remaining observations are no longer representative of the original sample. Failure to achieve an acceptable post-matching covariate balance means the matched control sample cannot be considered an appropriate counterfactual for the treated sample.

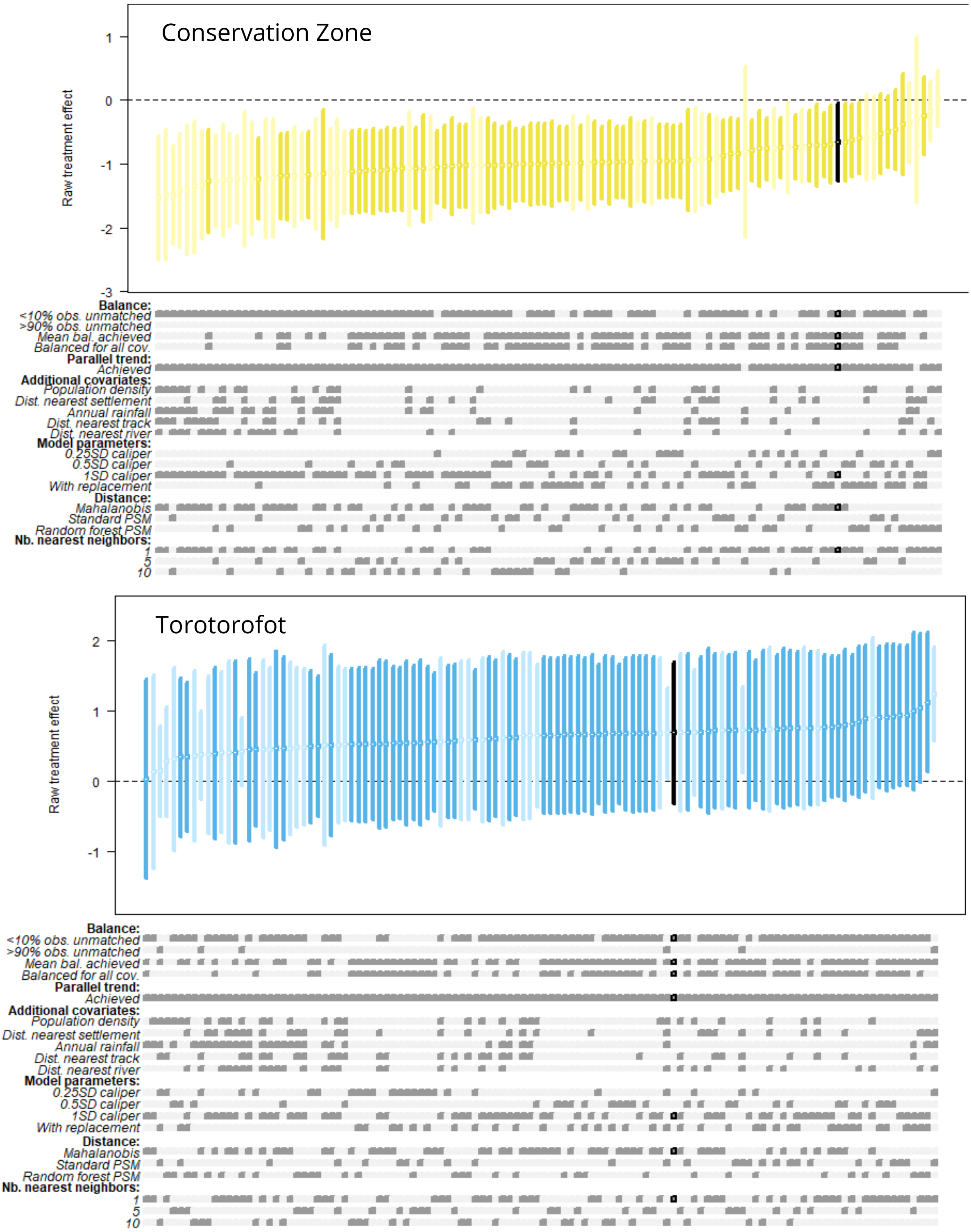
To aid interpretation of how alternative modelling choices affect the direction, significance, and magnitude of our results we expand upon Figure 4 presented in the main text to show which model specifications are associated with each result (Supplementary Figure 1.11). We then test which modelling choices exert the greatest influence on our estimated impacts.



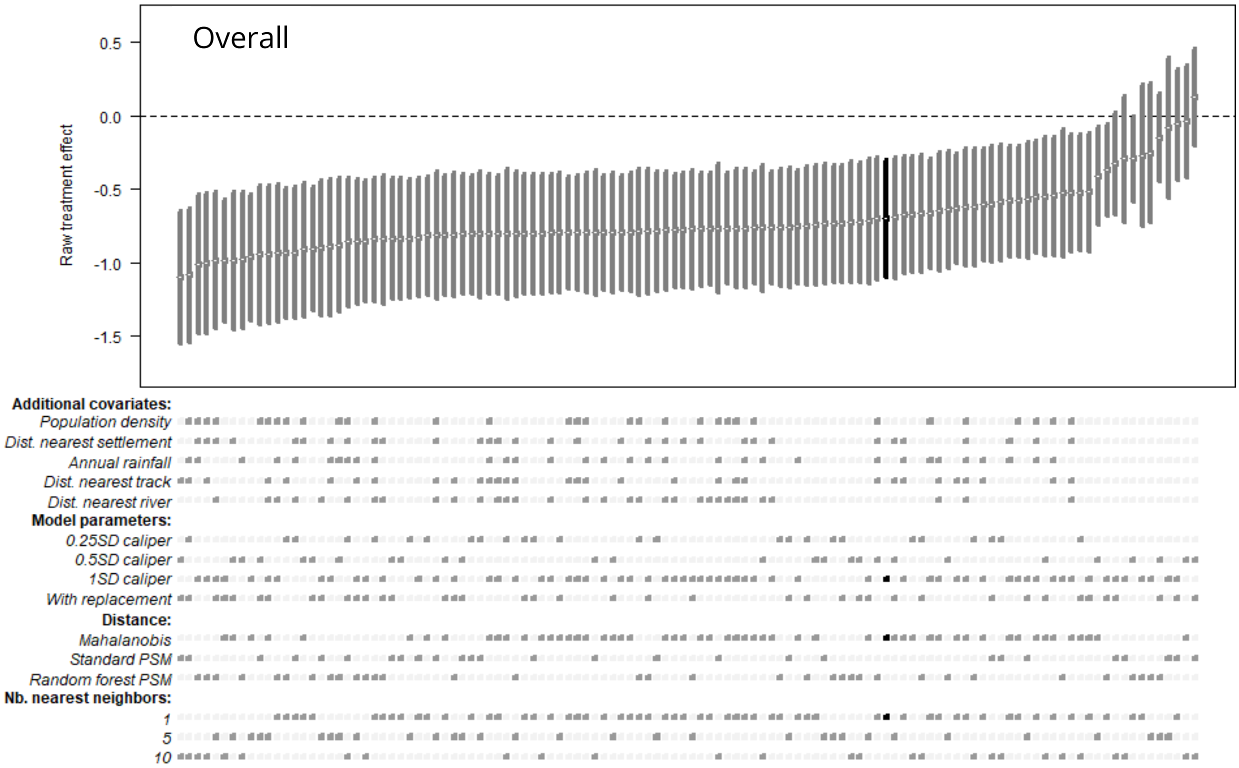
Ankera
Raw treatment effect
0
-1
-2
-3
-4
-5
Balance:
<10% obs. unmatched
>90% obs. unmatched
Mean bal. achieved
Balanced for all cov.
Parallel trend:
Achieved
Additional covariates:
Population density
Dist. nearest settlement
Annual rainfall
Dist. nearest track
Dist. nearest river
Model parameters:
0.25SD caliper
0.5SD caliper
1SD caliper
With replacement
Distance:
Mahalanobis
Standard PSM
Random forest PSM
Nb. nearest neighbors:
1
5
10



CFAM
Raw treatment effect
2
1
0
-1
*
Balance:
<10% obs. unmatched
>90% obs. unmatched
Mean bal. achieved
Balanced for all cov.
Parallel trend:
Achieved
Additional covariates:
Population density
Dist. nearest settlement
Annual rainfall
Dist. nearest track
Dist. nearest river
Model parameters:
0.25SD caliper
0.5SD caliper
1SD caliper
With replacement
Distance:
Mahalanobis
Standard PSM
Random forest PSM
Nb. nearest neighbors:
1
5
10



Supplementary Figure 1.11: Raw estimates of treatment effect (points) and corresponding 95% confidence intervals (bars) derived from 116 alternative matching model specifications for each of the four biodiversity offsets. The dark grey squares in the panel below each plot indicate the model specifications (additional covariates, model parameters and matching distance measure) associated with each estimate and the outcome of the post-matching validity checks (the percentage of treated observations unmatched, whether an acceptable mean covariate balance and parallel trends have been achieved). In each plot models which do not pass these validity checks, and are consequently considered *a posteriori* invalid, are shown in lighter shades. Our primary result, derived from the main matching specification, is shown in black. An asterix indicates that the main model was not *a posteriori* valid. Values are reported un-transformed and represent the effect of treatment on the log(y + 1) transformed count of annual deforestation.



Supplementary Figure 1.12: Raw estimates of treatment effect (points) and corresponding 95% confidence intervals (bars) derived from 116 alternative matching model specifications for the pooled data. The dark grey squares in the panel below the plot indicate the model specifications (additional covariates, model parameters and matching distance measure) associated with each estimate. Our primary result, derived from the main matching specification, is shown in black. Values are reported un-transformed and represent the effect of treatment on the log(y + 1) transformed count of annual deforestation.

The effect of arbitrary modelling choices on our results

Arbitrary modelling choices can exert a significant influence on the estimated impact of conservation interventions (Desbureaux, 2021). We explore which modelling choices have the greatest influence on our estimated treatment effect by regressing the 456 coefficients estimated in our robustness checks on a series of dummy variables representing the associated modelling choices. Results are summarised in Supplementary Table 1.13.

Overall, the results suggest that the choice of matching algorithm (Mahalanobis matching, standard propensity score matching, or Random Forest propensity score matching) and model parameters (caliper value, matching with/without replacement and the number of nearest neighbours) have the most consistent effect on the estimated impact of the offsets. The effect of including additional covariates in the matching process is less clear with some covariates having a significant effect in some offsets but not others, except for annual rainfall which has a mostly significant, yet ambiguous effect. These results are pretty much aligned with the conclusions of Desbureaux (2021).

Supplementary Table 1.13: The effect of arbitrary modelling choices on the estimated impact of Ambatovy's biodiversity offsets. The coefficients of treatment effect obtained in the robustness checks (shown in Supplementary Figures 1.11 and 1.12) were regressed on a series of dummy variables representing the modelling choices associated with each result (1 if the choice was made and 0 if not). Columns 1-5 refer to the pooled data and the regressions include dummy variables for each offset to allow the effect of the offset itself to be distinguished from the effect of the modelling choices. Column 1 includes all 456 estimated coefficients, regardless of whether the models are *a posteriori* valid or not. Columns 2, 3, 4 and 5, only include estimates from matching models where less than 10% of the treated pixels are unmatched (column 2), where acceptable covariate balance was achieved on average (column 3) and for all covariates (column 4), and where parallel trends were achieved (column 5). In columns 6 to 9, we estimate the impact of modelling choices on the estimated treatment effects for each individual offset. Standard error is shown in brackets beneath the estimated coefficient.

	All (1)	<10% unmatched (2)	Mean diff <.25 (3)	Max diff <.25 (4)	Parallel Trend (5)	ANK (6)	CFAM (7)	CZ (8)	TTF (9)
Pop Density	-0.037	-0.059^{**}	-0.068	0.124	-0.059	0.009	-0.081^{***}	0.023	-0.114^{***}
	(0.036)	(0.025)	(0.055)	(0.093)	(0.040)	(0.089)	(0.030)	(0.050)	(0.036)
Dist Sett.	0.124^{***}	0.078^{***}	0.099^{**}	0.047	0.091^{**}	0.258^{***}	0.009	0.120^{**}	0.065^{*}
	(0.036)	(0.025)	(0.046)	(0.058)	(0.040)	(0.089)	(0.030)	(0.050)	(0.036)
Annual Rain	-0.059	-0.135^{***}	-0.045	0.472^{***}	-0.096^{**}	0.119	-0.081^{***}	-0.184^{***}	-0.148^{***}
	(0.036)	(0.026)	(0.051)	(0.097)	(0.041)	(0.090)	(0.030)	(0.051)	(0.036)

Dist Track	0.078**	0.068***	0.084*	0.024	0.082**	0.203**	0.120***	-0.019	-0.004
	(0.036)	(0.025)	(0.045)	(0.057)	(0.040)	(0.089)	(0.030)	(0.050)	(0.036)
Dist_River	0.103***	0.004	0.141***	0.111*	0.067*	0.304***	0.057*	-0.031	0.026
	(0.036)	(0.026)	(0.046)	(0.062)	(0.040)	(0.090)	(0.031)	(0.051)	(0.036)
Caliper 0.25	0.191***		0.184***	0.139***	0.203***	0.756***	-0.169***	0.173***	-0.112***
	(0.039)		(0.042)	(0.044)	(0.044)	(0.095)	(0.033)	(0.058)	(0.038)
Caliper 0.5	0.089**	-0.002	0.096**	0.116**	0.103**	0.214**	-0.065**	0.157***	0.018
	(0.038)	(0.027)	(0.042)	(0.045)	(0.043)	(0.096)	(0.032)	(0.052)	(0.039)
With repl.	0.144***	0.166***	0.103***	0.098**	0.144***	0.259***	0.062**	0.154***	0.076**
	(0.033)	(0.025)	(0.035)	(0.038)	(0.037)	(0.082)	(0.028)	(0.045)	(0.033)
Mahalanobis	-0.182***	-0.193***	-0.163***	-0.079	-0.211***	-0.366***	-0.056*	-0.257***	-0.050
	(0.035)	(0.027)	(0.043)	(0.053)	(0.039)	(0.087)	(0.030)	(0.047)	(0.035)
PSM GLM	-0.026	-0.068**	-0.001	0.067	-0.062	-0.060	-0.001	-0.148***	0.070*
	(0.040)	(0.031)	(0.044)	(0.053)	(0.044)	(0.099)	(0.034)	(0.056)	(0.040)

1 nearest neigh.	0.167***	0.203***	0.149***	0.123***	0.222***	0.202**	0.059*	0.306***	0.137***
	(0.039)	(0.031)	(0.044)	(0.046)	(0.044)	(0.097)	(0.033)	(0.054)	(0.039)
5 nearest neigh.	0.012	0.057*	0.004	0.004	0.054	0.010	-0.017	0.080	0.009
	(0.041)	(0.031)	(0.045)	(0.047)	(0.046)	(0.102)	(0.034)	(0.056)	(0.041)
CFAM	3.150***	3.338***	3.175***	3.170***	3.127***				
	(0.040)	(0.029)	(0.043)	(0.051)	(0.096)				
CZ	1.846***	1.951***	1.906***	1.939***	1.872***				
	(0.040)	(0.029)	(0.048)	(0.052)	(0.039)				
TTF	3.418***	3.607***	3.472***	3.510***	3.453***				
	(0.039)	(0.029)	(0.044)	(0.050)	(0.038)				
Constant	-2.951***	-3.001***	-2.980***	-3.031***	-2.975***	-3.257***	0.382***	-1.076***	0.611***
	(0.054)	(0.038)	(0.063)	(0.076)	(0.058)	(0.119)	(0.040)	(0.065)	(0.048)
Observations	456	349	342	251	349	116	114	110	116
Adjusted R^2	0.954	0.983	0.961	0.963	0.960	0.530	0.344	0.483	0.347

Note: $^{*}p < 0.1$, $^{**}p < 0.05$, $^{***}p < 0.01$

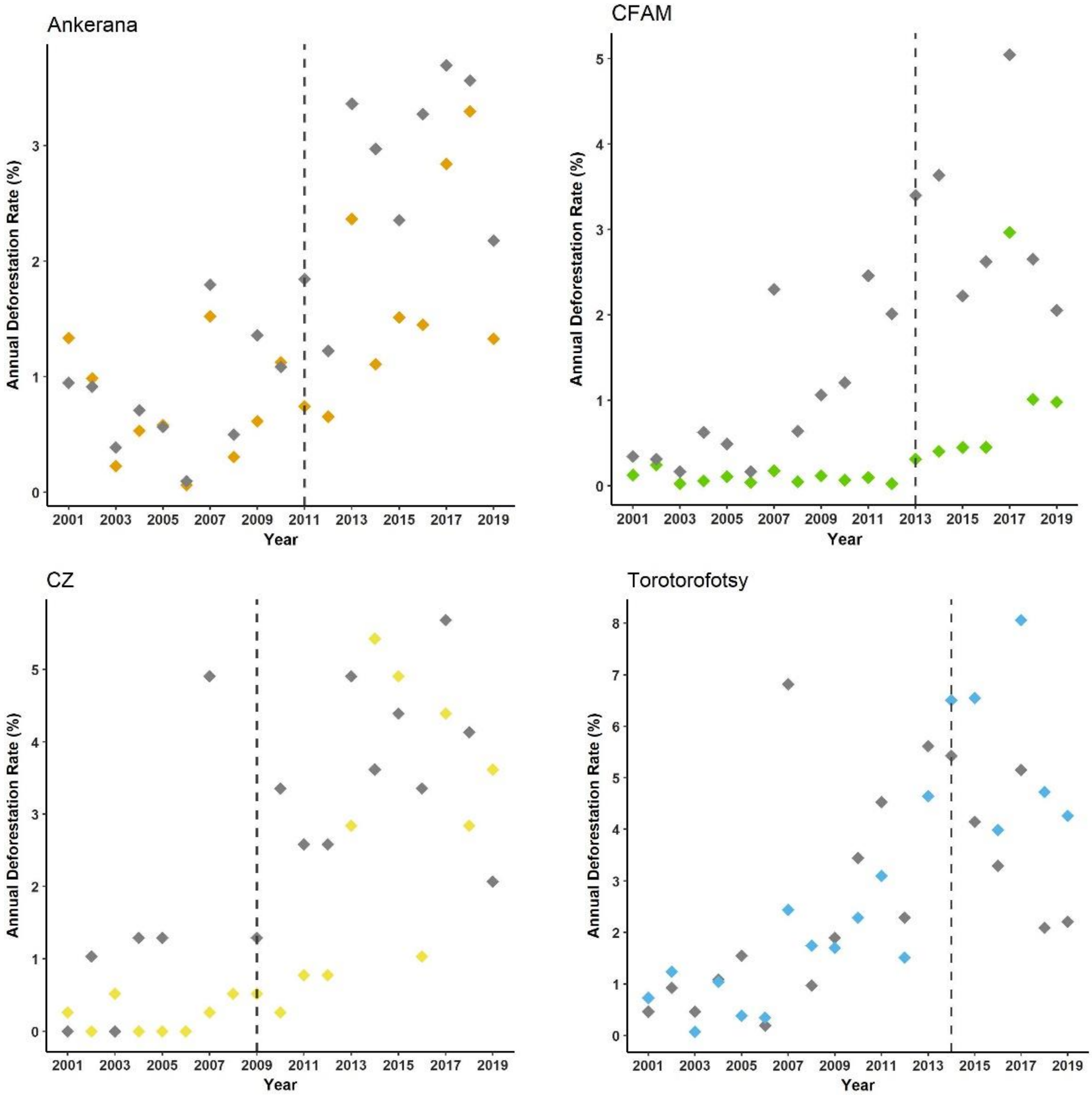
Evaluation of deforestation leakage

To determine whether protection of the biodiversity offsets displaced the anthropogenic drivers of deforestation into the surrounding landscape, we repeated the analysis using pixels sampled from the 10km buffer zone around each offset as the treated sample. The date of the intervention remains the year the adjacent offset was protected. If deforestation within these buffer zones was significantly higher than the estimated counterfactual, it would suggest deforestation has been displaced from the offsets into the surrounding area, undermining the true biodiversity 'gains' achieved through offset protection.

Matching

Acceptable matches (within one standard deviation of the Mahalanobis distance) were found for all buffer units associated with Ankerana, the Conservation Zone and Torotorofotsy. Only 28 out of 10,203 units from the buffer zone of CFAM could not be matched.

The standardised difference in mean covariate values between buffer and matched control samples was within the acceptable threshold of 0.25 for all covariates and all four buffer zones. The maximum post-matching standardised difference in mean covariates values was -0.15, indicating that, on average, buffer and control samples were well-matched.



Supplementary Figure 1.13; Comparison of deforestation outcomes between the sample of pixels from the buffer zone of each offset (shown in colour) and the matched controls (grey) over the whole study period. The dashed line indicates the year the adjacent offset was protected. The buffer zone and the matched control samples have an equal sample size (12,344 for Ankerana, 10,203 for CFAM, 387 for the Conservation Zone and 2581 for Torotorofotsy) as the ratio of treated to control units in the matching was set to 1:1.

Testing the assumptions

Supplementary Table 1.14; Test for parallel trends in deforestation between each buffer zone and its matched control sample in the pre-intervention period. Parallel trends in deforestation between the buffer zone and matched control samples were present for the buffer zones of Ankerana, the Conservation Zone and Torotorofotsy ($p > 0.05$). However, within the buffer zone of CFAM deforestation was declining prior to the intervention ($p = 0.0028$), whilst it was increasing in the matched control sample ($p = 0.001$). Consequently, the buffer zone of CFAM did not meet the assumption of parallel trends and could not be used in the subsequent difference-in-differences analysis. Interestingly, this replicates the findings of the main analysis.

Buffer Zone		**Intercept**	**Year**	**Treated**	**Year:Treated**
Ankerana	Estimate	4.2207	0.0381	0.0739	-0.0521
	std.error	0.6272	0.1011	0.8870	0.1429
	Statistic	6.7296	0.3766	0.0833	-0.3644
	p.value	0.0000	0.7114	0.9346	0.7204
CFAM	Estimate	2.9556	0.1989	-0.4228	-0.2490
	std.error	0.3797	0.0516	0.5370	0.0730
	Statistic	7.7832	3.8549	-0.7872	-3.4120
	p.value	0.0000	0.0010	0.4404	0.0028
Conservation Zone	Estimate	0.3776	0.1741	-0.0908	-0.1383
	std.error	0.6693	0.1325	0.9465	0.1874
	Statistic	0.5642	1.3133	-0.0959	-0.7376
	p.value	0.5830	0.2136	0.9251	0.4749
Torotorofotsy	Estimate	2.4177	0.1812	-0.2174	-0.0077
	std.error	0.4561	0.0575	0.6450	0.0813
	Statistic	5.3007	3.1528	-0.3370	-0.0945
	p.value	0.0000	0.0046	0.7393	0.9256

Note: N= 20 for Ankerana, 24 for CFAM, 16 for the CZ and 26 for Torotorofotsy

Site-based difference-in-differences regressions

Supplementary Table 1.15; Results from the site-based differences-in-differences regression for each buffer zone. The buffer zone of CFAM could not be included due to the lack of parallel trends. Treated and Time are a binary variables indicating whether the observation is from the buffer zone (1) or matched control (0) sample, from before (0) or after (1) the intervention. The coefficient of interest is Treated:Time which represents the effect of being within 10km of a protected biodiversity offset on the log(y+1) transformed count of deforestation.

Buffer Zone		Intercept	Treated	Time	Treated:Time
Ankerana	estimate	4.4300	-0.2126	1.3354	-0.3354
	std.error	0.2248	0.3179	0.3267	0.4620
	statistic	19.7049	-0.6686	4.0882	-0.7261
	p.value	0.0000	0.5082	0.0003	0.4727
Conservation Zone	estimate	1.1609	-0.7130	1.4414	0.1860
	std.error	0.2642	0.3736	0.3472	0.4910
	statistic	4.3943	-1.9083	4.1515	0.3787
	p.value	0.0001	0.0648	0.0002	0.7072
Torotorofotsy	estimate	3.6859	-0.2711	0.8226	0.7250
	std.error	0.2420	0.3423	0.4307	0.6091
	statistic	15.2282	-0.7921	1.9099	1.1903
	p.value	0.0000	0.4338	0.0646	0.2422

Note: N = 38.

Results show no significant difference in average annual deforestation between the buffer zone and the estimated counterfactual for Ankerana, the Conservation Zone and Torotorofotsy following protection of the offsets. Therefore, there is no evidence of deforestation leakage from the protected offsets into the surrounding forested landscape.

Fixed effects panel regression

As in the main analysis the data for all four buffer zones and their matched control samples were pooled to form one dataset with 152 observations comprising an observation for each site (i=8, four buffer zone and four control) for each year (t =19).

Supplementary Table 1.16; Results from the fixed effects panel regression on the pooled buffer zone data. The fixed effects regression takes the form $\log(count\ of\ deforestation + 1)_{i,t} = \beta_0 + \beta_1 Tr_{i,t} + \propto_i + \gamma_t + \epsilon_{it}$, where Tr is a binary measure indicating the treated status of observation *i* in year *t* (Tr = 1 for observations from a buffer zone in the years following protection of the adjacent offset, and 0 for all other observations), $\propto_i$ and γ_t represent site and year fixed effects respectively and ϵ_{it} represents the composite error. The sample size is 152. The coefficient of Tr indicates the treatment effect – the effect of being within 10km of a protected biodiversity offset on the log-transformed count of deforestation.

term	**Tr**
estimate	0.2271
std.error	0.1564
statistic	1.4520
p.value	0.1490

Note: N = 152, df = 125

Results show that overall, protection of the biodiversity offsets had no significant effect on deforestation within a 10km radius. This verifies the findings of the site-based difference-in-differences regressions (but captures the effect of CFAM), that there is no evidence of deforestation leakage from Ambatovy’s four biodiversity offsets into the surrounding forested landscape.

Appendix 2

Chapter 3: Mapping to explore challenges and opportunities for reconciling artisanal gem mining and biodiversity conservation

Supplementary Methods 2

Geological conditions for the formation of ruby and sapphire

Ruby and sapphire are gem-quality variants of the mineral corundum (Al_2O_3). Corundum typically occurs in rocks that are aluminium-rich and silica-poor, and have been metamorphosed at moderate pressures and relatively high temperatures (Simonet, Fritsch and Lasnier, 2008; Giuliani *et al.*, 2020). Metamorphism at these pressure and temperature (*P-T)* conditions falls within the amphibolite to granulite facies and is most commonly indicative of regional metamorphism in zones of continental collision or contact metamorphism (whereby intruding magma heats the surrounding rocks). Corundum formation under these P-T conditions commonly requires the circulation of a fluid to supply Al or other trace elements and remove silica from the host rock (Simonet, Fritsch and Lasnier, 2008). A desilicated environment is critical for corundum formation, as where silica is available aluminium will preferentially react with it to form other minerals such as feldspar or mica.

Giuliani *et al* (2020) reviewed corundum deposits from around the world to produce a classification of the geology and formation of deposits. Primary deposits are classed as magmatic (Type I) or metamorphic (Type II) based on the geological environment in which they are found.

Type I magmatic deposits include two sub-types:

A) gems as xenocrysts or in xenoliths within erupted volcanic rocks, such as alkali basalts.
B) gems in intrusive igneous rocks, such as kimberlite, lamprophyre or syenite

Gems within magmatic deposits were formed at depth within the Earth's crust and then transported to shallower levels by rising magma. As such, other authors classify these types of magmatic deposits as secondary as they have been moved from where they were originally formed (Simonet, Fritsch and Lasnier, 2008).

Type II metamorphic deposits can be divided into:

A) strictly metamorphic, where the chemistry of the rocks is such that gem corundum could form during metamorphism with no introduction or removal of elements by fluids.
B) Metamorphic-metasomatic, where the introduction of fluids has led to the formation of gem corundum.

Host rocks for metamorphic gem corundum deposits typically have relatively low silica content but moderately high Al contents. These may include mafic-ultramafic igneous rocks, which typically also contain chromium, and metasedimentary rocks. In some cases, deposits considered to be metamorphic are found in marble; however marble is depleted in both silica and aluminium so requires aluminium and trace elements to be supplied from impurities, or layers of other sediments, within the limestone protolith (Dzikowski *et al.*, 2014).

Metamorphic-metasomatic deposits are formed through fluid-rock interactions which result in chemical alteration of the host rock and the formation of new minerals (metasomatism) (Putnis and Austrheim, 2010). During periods of regional metamorphism, fluids circulate through thrusts, veins and shear zones, reacting with wall rocks and facilitating the diffusion of elements between rocks of contrasting lithologies along geochemical gradients (Putnis and Austrheim, 2010). Fluid can also be supplied by the intrusion of magma, with metasomatized zones typically focused along the contact of the intrusion (Simonet, Fritsch and Lasnier, 2008). Typically, metasomatic corundum deposits are formed through the interaction of a silica and alumina-rich rock or fluid, with a silica-poor rock. Silica diffuses from the former into the latter, leaving an environment relatively enriched in aluminium where corundum can crystallise, given the required trace elements are present (Simonet, Fritsch and Lasnier, 2008). Metasomatic corundum deposits are associated with desilicated pegmatites intruding mafic-ultramafic rocks or marbles (to form skarns), and desilicated gneiss (Giuliani *et al.*, 2020)

Both types of magmatic and metamorphic deposits have been documented in Madagascar (Rakotondrazafy *et al.*, 2008; Rakotosamizanany *et al.*, 2014; Giuliani *et al.*, 2020).

Formation of emerald

Emerald is green gem-quality beryl ($Be_2Al_2Si_6O^{18}$). Trace amounts of chromium and/or vanadium produce the green colour. Beryllium is rare in the upper continental crust and is typically supplied through the intrusion of magma, or fluid migration along thrusts, veins and shear zones. Deposits formed through the intrusion of magma are classed as tectonic-magmatic and typically occur where granites and pegmatites have intruded mafic-ultramafic or sedimentary rocks (Groat *et al.*, 2008). An example of this occurs at the Ianapera deposit, Madagascar; (Andrianjakavah *et al.*, 2009). As the intruding magma cools beryllium and other incompatible elements become concentrated in late-stage melts and fluids, which can react with mafic-ultramafic or sedimentary wall rocks to form emeralds at the contact zone. Emeralds formed through fluid-rock interactions in the absence of magmatism are termed tectonic-metamorphic and are associated with fluid pathways such as shear zones cross-cutting mafic-ultramafic rocks, black shale and metamorphic rocks (Giuliani *et al.*, 2019). The known emerald deposits of Madagascar are associated with desilicified pegmatites and metasomatism within shear zones cutting mafic-ultramafic host rocks (BGS-USGS-GLW, 2008; Giuliani *et al.,* 2019).

The geology of Madagascar and relevance to the formation of gemstones

Madagascar has a long and dynamic geological history, featuring the formation and break-up of supercontinents. The East African and Kuungan orogenies, 650 – 500 Ma, in which much of Madagascar collided with East Africa and subsequently with India during the assembly of Gondwana, provided the conditions for the formation of many of Madagascar's mineral deposits, including gemstones (Giuliani *et al.*, 2007, 2020; Rakotondrazafy *et al.*, 2008). In fact, gemstones are found across the Neoproterozoic Mozambique belt which extends through Kenya, Tanzania, Mozambique and into Madagascar, and defines a major part of the East African Orogen (Giuliani *et al.*, 2020).

The island of Madagascar can be divided into the Precambrian basement rocks that occupy the eastern two-thirds of the island, and the late-Palaeozoic to recent sedimentary sequences of the western third. The younger sedimentary sequences have not experienced the metamorphic conditions required for the formation of gems, nor alkaline volcanism and are therefore not prospective for primary deposits. The Precambrian basement comprises nine tectonic domains of different ages, origins, and dominant lithologies. From north to south these are; the Bemarivo, Anaboriana-Manampotsy, Antongil-Masora, Tsaratanana, Antananarivo, Ikalamavony, Anosyen, Androyen and Vohibory domains (Supplementary Figure 2.1; Boger et al, 2019; Key et al, 2011).The Antananarivo, Tsaratanana and Antongil-Masora domains are fragments of older continental crust, termed cratons, dating from the Archaean period (Schofield *et al.*, 2010; Key *et al.*, 2011). These domains constitute the oldest core of Madagascar to which the other domains were accreted between 1.6 Ga and 530 Ma (Thomas *et al.*, 2009; Tucker *et al.*, 2014).

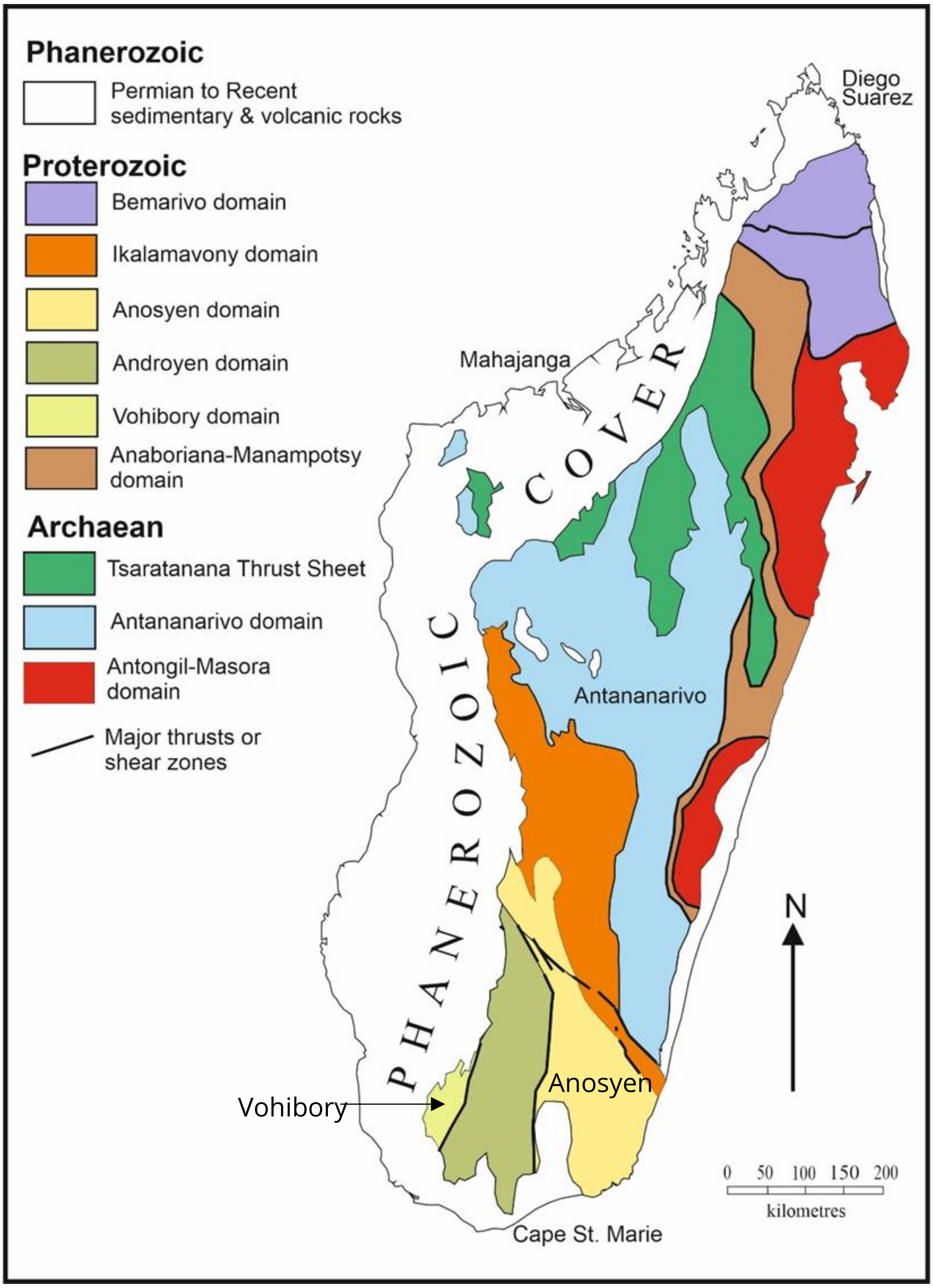
The amalgamation of these building blocks of Madagascar is tied to the formation, and subsequent break-up of the supercontinent of Gondwana (Collins and Pisarevsky, 2005; Fritz *et al.*, 2013; Reeves, 2014; Tucker *et al.*, 2014; Boger *et al.*, 2019). Gondwana existed from ~550 Ma – 170 Ma and consisted of modern-day Africa, Arabia, India, Sri Lanka, Madagascar, Australia, Antarctica, and South America (Reeves, 2014). It formed through a series of orogens (collisions of older fragments of continental crust) known collectively as the Pan-African orogeny (Fritz et al., 2013).

The first of which were the East African and Kuungan orogenies, 650 Ma – 500 Ma, where Arabia, India and Madagascar collided with the Congo Craton of Africa to close the Mozambique Ocean (Fritz et al., 2013). Around the same time Southern Africa, Antarctica and Australia collided to the south. Debate is ongoing regarding the exact timing and location of the collision between Africa, Madagascar, and India. Tucker et al (2014) propose that the Antananarivo and Antongil-Masora domains formed part of the Greater Dharwar Craton of India, to which the Southern Malagasy domains (Androyen, Anosyen and Vohibory) and the northern Bemarivo domain were accreted in the Neoproterozoic to Cambrian. According to this interpretation, the Malagasy Precambrian basement amalgamated at the Western margin of India (the Greater Dharwar Craton) before a combined India and Madagascar collided with East Africa in the Neoproterozoic. In contrast, Collins and Pisarevsky (2005) argue that only the Antongil-Masora domain was joined to the Greater Dharwar Craton during the Archaean, and that the Antananarivo domain, which comprises most of Central Madagascar, was part of the island micro-continent of Azania. In this model, Azania collided with East Africa c. 630 Ma, whilst collision with India only occurred during the late stages of assembly after 550 Ma (Armistead et al., 2020; Collins and Pisarevsky, 2005). However, this argument is complicated by the presence of the extensive c. 850-750 Ma Imorona-Itsindro igneous suite intruding the Antananarivo, Ikalamavony, and (possibly) Antongil-Masora domains, which suggests the likelihood of an earlier Proterozoic collision (Key et al., 2011; Archibald et al., 2016; Zhou et al., 2018).

Regardless of the exact sequence of events, evidence of high-grade metamorphism recorded across south-west, central and northern Madagascar suggests that much of the Malagasy basement was substantially reworked during the East African and Kuungan orogenies of the Neoproterozoic (Thomas *et al.*, 2009; Fritz *et al.*, 2013; Tucker *et al.*, 2014; Boger *et al.*, 2019). Continental convergence led to regional metamorphism and magmatism which produced the high temperatures, pressures, and fluids necessary for the formation of gems. Shear zones were formed or re-activated in areas of high strain and acted as conduits for fluid circulation and metasomatism (Wit *et al.*, 2001). Continental arc-magmatism related to oceanic subduction led to the emplacement of the 560 – 520 Ma Ambalavao – Kiangara – Maevarano granites (Goodenough *et al.*, 2010; Tucker *et al.*, 2014).

Gondwana began to fragment ~168 Ma (Reeves, 2014). By 120 Ma a combined Madagascar and India had separated from Africa, and by 88 Ma India had separated, leaving Madagascar in it's current position by ~60 Ma. It was during this period of rifting, separation, and uplift in the east that the sedimentary sequences which formed the Western 1/3 of the island were deposited (Wescott and Diggens, 1997).

More recent volcanism during the Cenozoic provided the second, and final, period of gem emplacement in Madagascar. The eruption of alkali basalts across parts of northern and central Madagascar 23 – 2.6Ma transported gems formed at greater crustal depths to the surface, forming the magmatic-type ruby deposits discovered at Soamiakatra and Ambondromifey (Rakotondrazafy *et al.*, 2008; Rakotosamizanany *et al.*, 2014).



Supplementary Figure 2.1: Map of the tectonic domains of Madagascar derived from Key et al (2011)

Identifying potentially prospective units

Supplementary Table 2.1: Lithologies identified as potentially prospective for ruby, sapphire or emerald from the Geological Map of Madagascar (Roig *et al.*, 2012)

Geological unit code on map	Domain/Group or Suite	Description	Notes	Type of potential deposit
Ad8	Androyen	Impure marble with calc-silicate minerals		Metamorphic
Ad9	Androyen	Graphitic basic gneiss with pyroxene		Metamorphic
An1	Anosyen – Tranomaro Group	Calc-silicate paragneiss	Tranomaro group prospective for skarns where marble intruded by granite	Metamorphic
An2	Anosyen – Tranomaro Group	Banded metapelitic paragneiss with Mag, Crd and Opx	Tranomaro group prospective for skarns where marble intruded by granite	Metamorphic
An6	Anosyen	Basic orthogneiss (metagabbro)		Metamorphic
At11	Antananarivo	Marble and calc-silicate paragneiss		Metamorphic

At17	Antananarivo	Orthogneiss basique	Metamorphic
At3	Antananarivo – Manampotsy group	Quartzite and Ampasary paragneiss with relics of ultramafic rock	Metamorphic
At4	Antananarivo – Manampotsy group	Andasibe paragneiss and schist	Metamorphic
At5	Antananarivo – Manampotsy group	Ranomafana paragneiss	Metamorphic
At7	Antananarivo	Calc-silicate paragneiss and marble	Metamorphic
At9	Antananarivo	Ultramafic rock	Metamorphic
Atn3	Bemarivo – Antsirabe North suite	Undifferentiated ultramafic rock	Metamorphic
Bm11	Bemarivo	Marble and calc-silicate paragneiss	Metamorphic
Bm12		Amphibolite	Metamorphic
Ik2	Ikalamavony	Calc-silicate paragneiss, Mag + Cpx	Metamorphic
Ik4		Calc-silicate marble with intercalcated amphibolite	Metamorphic
Ik5		Basic paragneiss with amphibolite	Metamorphic

Il2	Imorona - Itsindro	Granite and basic orthogneiss of Itsindro type	Includes metagabbro	Metamorphic
Il4	Imorona - Itsindro	Hazburgite, pyroxenite and periodotite		Metamorphic
It1	Itremo	Dolomitic marble		Metamorphic
Ma1	Masora	Pelitic schist and amphibolite		Metamorphic
Pea	Cenozoic volcanism	Syenite and granite		Magmatic
Pem		Gabbro		Magmatic
Vmpa		Rhyolite, trachyte, phonolite, ignimbrite		Magmatic
Vmpa + Pea		Basalt, Ankaratrite basanite with syenite and granite		Magmatic
Vmpm		Basalt, Ankaratrite basanite		Magmatic
Vo3	Vohibory	Amphibolitised metabasalt		Metamorphic
Vo5		Calc-silicate marble		Metamorphic
Akm1	Ambalavao – Kiangara – Maevarano suite	Charnockitic granite		Metamorphic
Akm2		Granite, monzonite and undifferentiated syenite		Metamorphic
Akm3		Granite and syenite stratoids		Metamorphic
Akm4		Gabbro and diorite		Metamorphic
Akm5		Pyroxenite		Metamorphic

Akm6		Granitic and monzonitic orthogneiss	Metamorphic

N.B – Several units were identified as prospective but not found on the map (Ad3, Ma3)

Biodiversity data

Supplementary Table 2.2: Summary of the datasets capturing conservation priorities used in this analysis, including changes to the original data layers (excluding raster to polygon conversion and vice-versa). All data layers were reprojected to WGS 1984 UTM Zone 38S projected co-ordinate system.

Biodiversity indicator layer	Original format	Raster resolution	Edits	Source
Key Biodiversity Areas (KBA)	Polygon	100m (when converted)	Removed KBA polygons with a marine portion > 80% of the polygon area and clipped the remaining data to the boundary of Madagascar	Birdlife International (2021)
Conservation Priority Areas	Raster	918m	We use the version unconstrained to the existing protected area network. No edits except reprojection.	Kremen et al (2008)

Protected Areas	Polygon	100m (when converted)	Amended by Jorge Llopis to remove PAs classified as Marine Protected Areas and PAs where no supporting evidence of existence could be found (i.e no evidence of legal definition). The resulting dataset was verified by an expert (Dimby Razafimpahanana). We further amended the dataset to remove PAs with a marine portion > 80% and clipped the remaining data to the boundary of Madagascar	Rebioma (2017) edited by Jorge Llopis
Forests 2020	Raster	30m	We created this layer by merging a forest cover map of Madagascar for the Year 2000 (Harper *et al.*, 2007; updated by Vieilledent *et al.*, 2018) with the Global Forest Change data.	Harper *et al,* (2007); Vieilledent *et al,* (2018)*;* Hansen *et al,* (2013)

Our database of known gem deposits

Supplementary Table 2.3: Database of 69 known ruby, sapphire and emerald deposits in Madagascar compiled from a literature search.

Name	Stone	Primary	Source	Notes
Ambalavihy Village	Sapphire	No	Site visited by Vincent Pardieu (VP), location provided.	
Ambalavihy mines	Sapphire	No	Site visited by VP, location provided.	
Ambalmasi	Sapphire	No	Site visited by VP, location provided.	
Ambarinakoho	Sapphire	No	Site visited by VP, location provided.	
Ambarinakoho	Sapphire	No	Site visited by VP, location provided.	
Ambatomianty	Sapphire	No	Site visited by VP, location provided.	
Ambodibakoly	Emerald	Yes	https://www.mindat.org/loc-27840.html	
Ambodipaiso	Sapphire	No	Perkins (2016, 2017)	
Ampandamisivaly	Sapphire	No	Site visited by VP, location provided.	
Ampasimamitaka	Sapphire	No	Site visited by VP, location provided.	
Anakondro	Sapphire	No	Site visited by VP, location provided.	
Analalava Village	Sapphire	No	Site visited by VP, location provided.	
Analasoa Village	Sapphire	No	Site visited by VP, location provided.	

Anavoha	Ruby	Yes	Rakotondrazafy et al (2008); Mercier et al (1999); https://www.mindat.org/loc-264252.html;	Compared location between three sources. Likely correct within 5km
Andilamena	Ruby	Yes	Site visited by VP, location provided. Leuenberger (2001); Hughes, Pardieu and Schorr (2005); Pardieu and Wise (2008)	
Andohasilaka	Sapphire	No	Site visited by VP, location provided.	
Andranondambo	Sapphire	Yes	Schwarz (1996)	
Andrebabe	Sapphire	No	Hughes, Pardieu and Schorr (2005); Pardieu and Rakotosaona (2012); https://www.mindat.org/loc-304128.html.	
Anduharano	Sapphire	No	Site visited by VP, location provided.	
Anena	Sapphire	No	Site visited by VP, location provided.	This is highly likely to be Soabiby mentioned in Baker-Médard (2012).
Ankadilalana	Emerald	Yes	Schwarz (1994); https://www.mindat.org/loc-26409.html	
Ankaranduha (Antsoa)	Sapphire	No	Site visited by VP, location provided.	

Ankazoabo	Sapphire	Yes	https://www.mindat.org/loc-226853.html. Also shown in Figure 2 Pardieu et al (2016)	
Ankilimasy	Sapphire	No	Site visited by VP, location provided.	
Ankilitelo	Sapphire	No	Site visited by VP, location provided.	
Ankotika (Andampy)	Sapphire	No	Cook and Healy (2012); https://www.mindat.org/loc-232739.html	
Antaralava	Sapphire	No	Site visited by VP, location provided.	
Antsimobohitra	Sapphire	No	Site visited by VP, location provided.	
Antsirabe	Sapphire	No	Cook and Healy (2012); https://www.mindat.org/loc-232904.html.	
Antsoa village	Sapphire	No	Site visited by VP, location provided.	
Banque Suisse	Sapphire	No	Site visited by VP, location provided.	
Beforona	Sapphire	Yes	Rakotondrazafy et al (2008). https://www.mindat.org/loc-191486.html.	
Befotaka, Nosy Be	Sapphire	No	Ramdohr and Millisenda (2006)	
Bekily	Sapphire	No	Cook and Healey (2012)	Next to Zombitse-Vohibasia National Park. Located using Figure 13 in Cook and Healy (2012)

Belamoty (Rush 2018)	Sapphire	No	Site visited by VP, location provided.	
Bepeha	Sapphire	No	Site visited by VP, location provided.	
Betsingaly	Sapphire	No	Site visited by VP, location provided.	
Bevilany	Sapphire	No	Site visited by VP, location provided.	
Ambondromifey	Sapphire	No	Schwartz et al (2000)	
Antsiermene	Sapphire	Yes	Schwartz et al (1996)	Around 11km north of Andranondambo. Several mining spots within 4km radius.
Didy	Ruby & Sapphire	No	Pardieu and Rakotosaona (2012)	Placer deposits (Giuliani *et al,* 2020)
Esoki	Sapphire	No	Site visited by VP, location provided.	
Ianapera	Emerald	Yes	Mercier et al (1999), Andrianjakavah et al (2009); https://www.mindat.org/loc-27838.html;	Ruby also found at this locality
Ilakaka	Sapphire	No	Pardieu (2013), Rakotondrazafy *et al,* (2008); https://www.mindat.org/loc-27802.html	There are many mine sites in the immediate vicinity of Ilakaka.
Limit	Sapphire	No	Site visited by VP, location provided.	
Lovakadabo	Sapphire	No	Site visited by VP, location provided.	

Madama Pauline (Vohimena)	Sapphire	No	Site visited by VP, location provided.	
Mahasoa village	Sapphire	No	Site visited by VP, location provided.	
Manamboay	Sapphire	No	Cook and Healy (2012)	Cross-referenced Figure 13 in Cook and Healy (2012) with Google Earth to visually identify mine site.
Mangatuka (Antsoa)	Sapphire	No	Site visited by VP, location provided.	
Maniry	Ruby	Yes	Mercier *et al* (1999)	Map in Mercier *et al* (1999) shows deposits are next to large anorthosite block. Cross-referenced with the location of this block in the Geological Map of Madagascar to locate deposits.
Manombo – Misereno	Sapphire	No	Site visited by VP, location provided.	
Manombo Kel	Sapphire	No	Site visited by VP, location provided.	
Manombo Voavoa	Sapphire	No	Pardieu (2013); https://www.mindat.org/loc-45926.html	

Maromiandry	Sapphire	No	Cook and Healey (2012); https://www.mindat.org/loc-157486.html	Near border of Zombitse-Vohibasia National Park
Morafeno	Emerald	Yes	Schwarz (1994); https://www.mindat.org/loc-27842.html.	
Old Thai sapphire Mine Ankaboka Ambinany	Sapphire	No	Site visited by VP, location provided.	
Sakabe	Sapphire	No	Site visited by VP, location provided.	
Sakalama	Sapphire	No	Site visited by VP, location provided.	
Sakameloka village	Sapphire	No	Site visited by VP, location provided.	
Sakaraha	Sapphire	No	Pardieu (2013)	Location of town. Evidence of mining visible on Google Earth
Soamiakatra	Ruby	Yes	Rakotosamizanany (2009); https://www.mindat.org/loc-191507.html	Co-ordinates from Mindat, near village of Soamiakatra. Figure IV-4 in Rakotosamizanany (2009) suggests primary Morarano deposit is located to East, but within 5km of village.

Tananarive	Sapphire	No	Pardieu et al (2017); Perkins (2017)	
Vatomandry	Ruby	No	Rakotosamizanany et al (2014), Rakotosamizanany (2009)	Identified based on map in Rakotosamizanany (2009)
Vohimena Mahafala	Sapphire	No	Site visited by VP, location provided.	
Vohimena Vovo Village	Sapphire	No	Site visited by VP, location provided.	
Zahamena NP 1	Ruby	No	Pardieu *et al* (2015); Giuliani *et al* (2020)	
Zahamena NP 2	Ruby	No	Pardieu *et al* (2015); Giuliani *et al* (2020)	
Zazafotsy Quarry	Sapphire	Yes	Rakotondrazafy et al (2008), https://www.mindat.org/loc-27844.html	

Spatial overlay analysis

To enable the raster overlay analysis, we converted our polygon layer of gem potential to a raster with cell size 100m x 100m. Our polygon biodiversity data (KBAs and PAs) were converted to raster layers of the same resolution and snapped to the gem potential raster to align cells, resulting in a maximum spatial error of 50m. Biodiversity data originally in raster form (forests and priority areas) were not resampled to the same resolution to avoid unnecessary error.

We then used raster overlay to combine each biodiversity raster with the gem potential layer to produce a new raster showing the area of overlap, with a resolution equal to the finest resolution input data. This was used to calculate what percentage of the total area of KBAs, Priority Areas, PAs and forests is potentially prospective for gems. Following Eklund et al (2022) we disaggregated the results for forest by forest type (using the biome classification from the Resolve Ecoregions project; Dinerstein *et al.*, 2017), to evaluate whether certain types of forest (humid, dry or spiny) are more likely to be threatened by gemstone mining than others.

We also calculated the percentage of each individual locality (PA/KBA/Priority Area or forest block) which is potentially prospective for gems using Tabulate Intersection on the polygon data. To do so, the raster biodiversity layers (Priority Areas and forests) were converted to polygon. This produced > 900,000 forest polygons. To speed processing we removed forest polygons smaller than 84ha (the size of the smallest polygon in the other biodiversity datasets), which are too small be visible in the resulting maps and whose inclusion could locate prospective sites at too fine a scale.

Supplementary Results 2

Validating our map of gem potential against the locations of gem deposits

Supplementary Table 2.4: Distance of known gem deposits from the nearest potentially prospective zone

Name	Distance (km)	Deposit type
Morafeno	0	Primary Emerald
Ambodibakoly	0	Primary Emerald
Ianapera	0	Primary Emerald
Zahamena NP 1	0	Secondary Ruby
Maniry	0	Primary Ruby
Vatomandry	0	Secondary Ruby
Ampasimamitaka	0	Secondary Sapphire
Lovakadabo	0	Secondary Sapphire
Antsiermene	0	Primary Sapphire
Beforona	0	Primary Sapphire
Zazafotsy Quarry	0	Primary Sapphire
Befotaka, Nosy Be	0	Secondary Sapphire
Andrebabe	0	Secondary Sapphire
Ankazoabo	0	Primary Sapphire
Andranondambo	0	Primary Sapphire
Ankadilalana	0	Primary Emerald
Didy	0	Secondary Ruby and Sapphire
Ambondromifey	0.31	Secondary Sapphire
Sakabe	0.49	Secondary Sapphire
Anavoha	0.60	Primary Ruby
Antsirabe	0.77	Secondary Sapphire
Zahamena NP 2	0.80	Secondary Ruby
Anakondro	0.83	Secondary Sapphire
Sakalama	1.00	Secondary Sapphire

Soamiakatra	1.45	Primary Ruby
Andilamena	1.58	Primary Ruby
Tananarive	1.79	Secondary Sapphire
Ambodipaiso	4.33	Secondary Sapphire
Ankotika (Andampy)	7.15	Secondary Sapphire
Belamoty (Rush 2018)	8.74	Secondary Sapphire
Antsimobohitra	9.48	Secondary Sapphire
Ilakaka	14.88	Secondary Sapphire
Banque Suisse	17.63	Secondary Sapphire
Ampandamisivaly	20.86	Secondary Sapphire
Vohimena Mahafala	22.43	Secondary Sapphire
Madama Pauline (Vohimena)	25.10	Secondary Sapphire
Bepeha	26.13	Secondary Sapphire
Andohasilaka	27.01	Secondary Sapphire
Manombo Voavoa	27.43	Secondary Sapphire
VOHIMENA Vovo Village	27.62	Secondary Sapphire
Anduharano	27.71	Secondary Sapphire
Manombo – Misereno	27.90	Secondary Sapphire
Manombo Kel	28.98	Secondary Sapphire
Sakameloka	30.44	Secondary Sapphire
Limit	31.77	Secondary Sapphire
Ankilitelo	39.28	Secondary Sapphire
Ankilimasy	40.01	Secondary Sapphire
Ambalmasi	42.49	Secondary Sapphire
Ambarinakoho	43.05	Secondary Sapphire
Analalava Village	45.40	Secondary Sapphire
Betsingaly	47.17	Secondary Sapphire
Analasoa Village	48.92	Secondary Sapphire
Bekily	53.96	Secondary Sapphire
Mahasoa village	54.13	Secondary Sapphire

Ambatomianty	55.75	Secondary Sapphire
Ambalavihy mines	55.78	Secondary Sapphire
Ambalavihy (Village)	55.82	Secondary Sapphire
Ankaranduha (Antsoa)	56.51	Secondary Sapphire
Mangatuka (Antsoa)	56.55	Secondary Sapphire
Bevilany	58.55	Secondary Sapphire
Antsoa village	60.60	Secondary Sapphire
Anena	61.51	Secondary Sapphire
Maromiandry	61.62	Secondary Sapphire
Ambarinakoho (Rush 2018)	62.48	Secondary Sapphire
Esoki	62.63	Secondary Sapphire
Old Thai sapphire Mine Ankaboka Ambinany	68.46	Secondary Sapphire
Manamboay ZV	76.20	Secondary Sapphire
Sakaraha	81.69	Secondary Sapphire
Antaralava	87.75	Secondary Sapphire

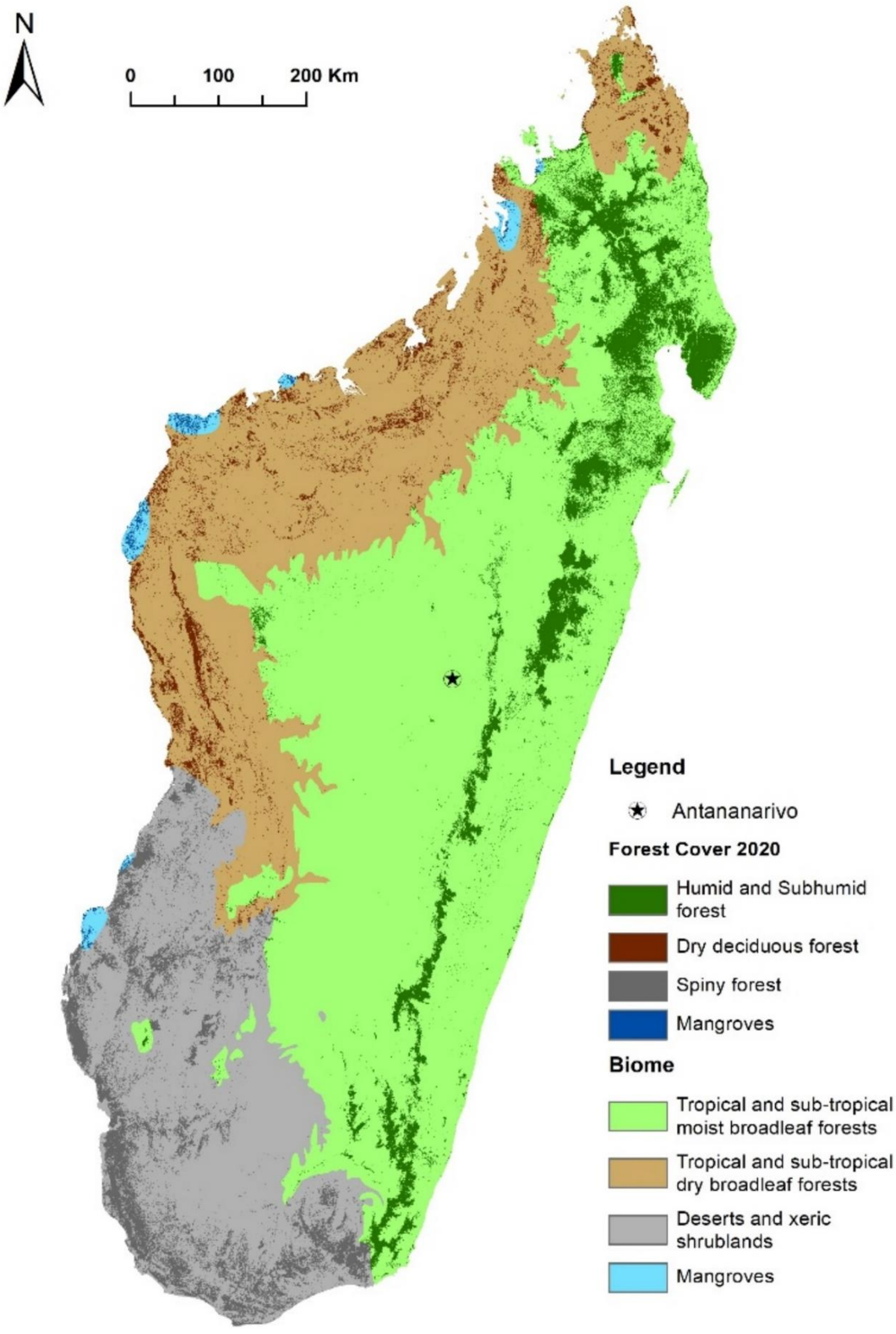
Raw results from the raster overlay

Supplementary Table 2.5: The area and percentage of each biodiversity layer which is potentially prospective for primary ruby, sapphire, or emerald deposits. The area and percentage of non-protected parts of Key Biodiversity Areas (KBAs), Priority Areas, and forests which are potentially prospective is also reported. Non-protected refers to the areas outside the formal protected area network.

	Area with gem potential (ha)	**Total area (ha)**	**Percentage with gem potential (%)**	**Non-protected area with gem potential (ha)**	**Total non-protected area (ha)**	**Percentage of non-protected area with gem potential (%)**	**Percentage of potentially prospective land unprotected (%)**
Protected Areas	742,000	6,758,000	11.0	-	-	-	-
KBAs	1,018,000	9,134,000	11.1	414,086	4,096,458	10.1	40.7
Priority Areas	839,000	5,927,000	14.2	559,928	3,810,900	14.7	66.7
Forest cover (2020)	992,000	8,163,000	12.1	466,479	4,557,603	10.2	47.0
Humid forest	709,000	4,028,000	17.6	-	-	-	-
Dry forest	45,000	1,428,000	3.1	-	-	-	-
Spiny forest	237,000	2,578,000	9.2	-	-	-	-
Mangroves	1,000	123,000	1.0	-	-	-	-

Additional results disaggregating forest by forest type

Disaggregating forest by type (using the biome classification from the Resolve Ecoregions project; Dinerstein *et al.*, 2017) shows that humid and sub-humid forests are the most potentially prospective forest type in terms of area (18% have gem potential) and are consequently more likely to host gem mining in future than other forest types (Supplementary Figure 2.2). Only 9% of spiny forests, 3% of dry forests and 1% of mangrove forests are potentially prospective. This is not surprising as dry forests and mangroves are mostly found in Western Madagascar where the underlying geology comprises relatively recent sedimentary sequences. In contrast, humid and spiny forests are concentrated in the northern, southern, and eastern parts of the island which experienced regional metamorphism and magmatism during the East African and Kuungan Orogenies. In certain areas, with the right lithologies, these large-scale processes provided the required temperature and pressure conditions to enable the formation of gemstones.



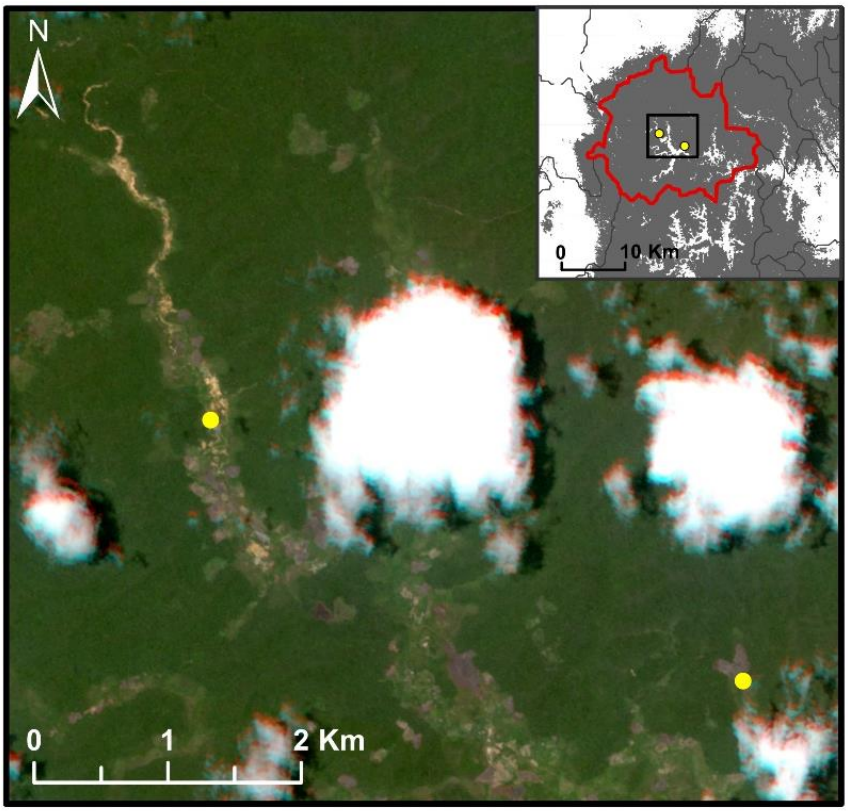
Supplementary Figure 2.2: Map of forest cover in 2020 categorised by forest type. The four biomes of Madagascar are also shown (Dinerstein *et al.*, 2017).

Appendix 3:

Chapter 4: No evidence of increased forest loss from a mining rush in a biodiversity hotspot

Supplementary Methods 3

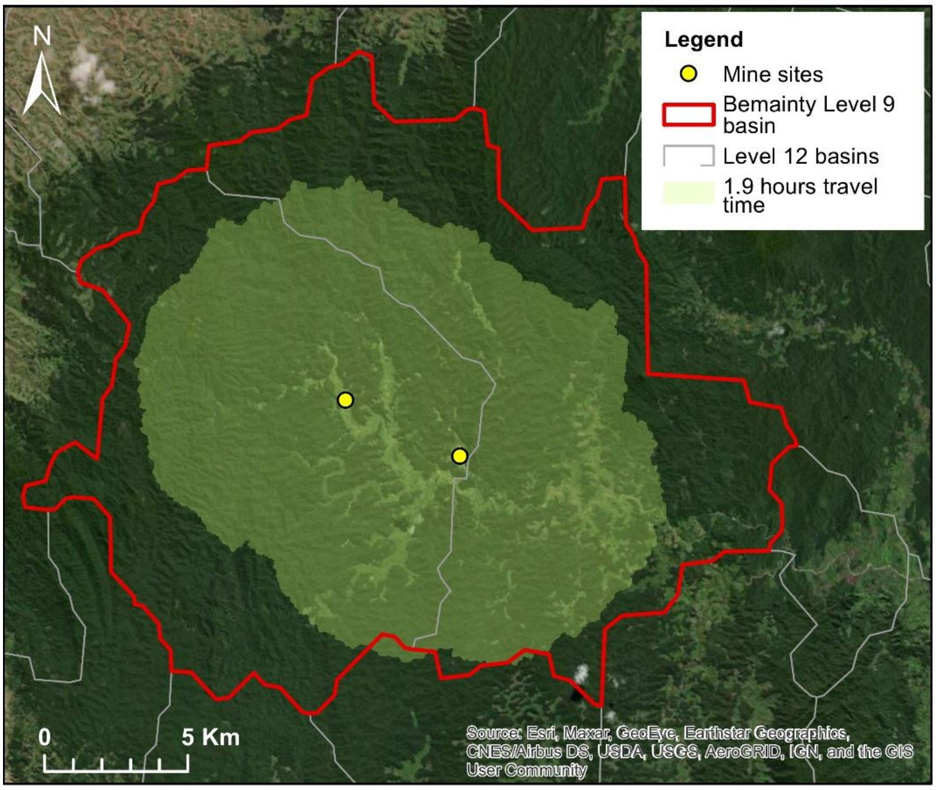
Study area



Supplementary Figure 3.1: Satellite image of the northern part of the Bemainty valley on 17th November 2013 taken from the RapidEye sensor (Planet Team, 2017). The image shows artisanal gem mining activity in the Ambodipaiso valley (left yellow dot), visible as disturbed yellowish sediments stretching several kilometres along the river in the upper part of the valley. The image also shows recent *tavy* (where forest is burned and then cleared for shifting cultivation) in the Antananarivo valley (right yellow dot) where the mining rush would later occur.

Choosing the unit of analysis

To choose which scale of drainage basin from the HydroBASINS data to use, we mapped the potential impact zone around the two mining valleys. To define the size of this zone we drew on survey data from a study area in north-east Madagascar which found that, on average local people would travel up to 1.9 hours to collect forest products (Allnutt *et al.*, 2013). Following Allnutt et al (2013) we converted travel time to distance using the Path Distance function in ArcGIS and Tobler's function to account for the effect of slope on distance covered. We then mapped the resulting impact area (Supplementary Figure 2). This is likely an overestimate as the Path Distance function did not incorporate the difficulties of moving through forested terrain. Short-term migrant miners may also be especially unlikely to venture far from the mine site to access materials. However, we wanted to ensure we captured all potential impacts within our treated unit and avoided spillovers into neighbouring control units. Figure S2 maps the Level 12 and Level 9 drainage basins compared to the potential impact zone. The boundary of the Level 12 basin is extremely close to the Antananarivo valley meaning it may not capture the full spread of impacts. Therefore, we chose the Level 9 basins as our unit of analysis as this best matches the potential impact zone.



Supplementary Figure 3.2: The estimated area within 1.9 hours walking distance from the Antananarivo (right yellow dot) or Ambodipaiso (left yellow dot) mining valleys is shaded light green. This represents the potential spread of forest impacts from mining. The Level 9 drainage basin encompassing Bemainty is shown in red and the smaller Level 12 basins (which we decided not to use in this analysis) are shown for comparison in light grey.

Covariates used in synthetic control matching

Supplementary Table 3.1: Details of the covariates used in the synthetic control matching, including the hypothesized mechanism through which they influence deforestation and degradation, data sources, and any subsequent manipulation. References are all specific to Madagascar. All data were reprojected to WGS 1984 UTM Zone 38S.

Covariate	**Hypothesised causal mechanism**	**Data source**	**Data manipulation**	**References**
Population density 2011	Demand	WorldPop unconstrained, UN-adjusted	Summed population count per sub-basin.	Elmqvist et al (2007), Brinkmann et al

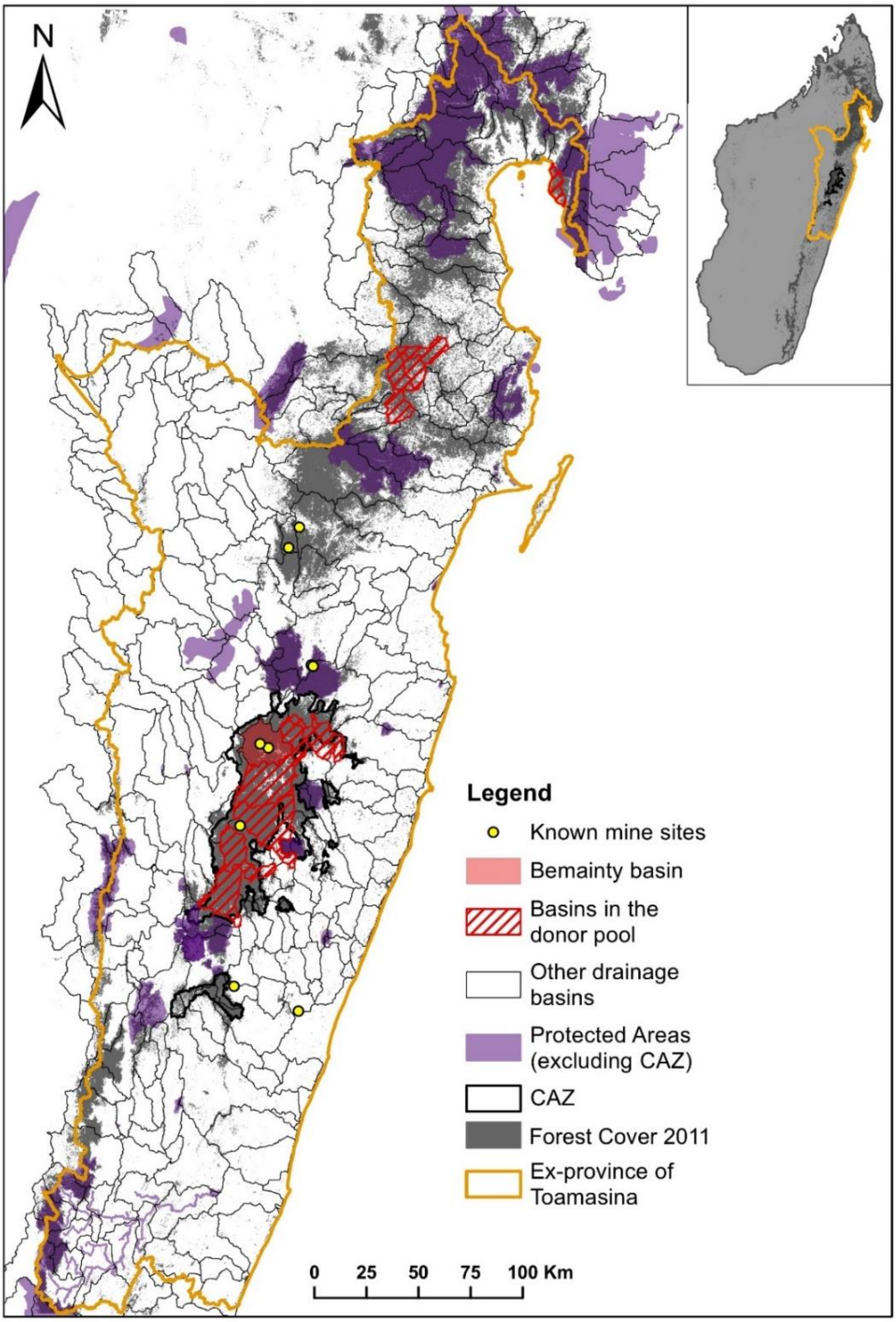
(people per km^2)		population counts 100m resolution	Divided by the area of the sub-basin	(2014), McConnell et al (2004), Agarwal et al (2005), WorldPop (2018)
Population growth 2001-2011	Demand, Migration	WorldPop unconstrained, UN-adjusted population counts 100m resolution	Calculated the percentage population growth between 2001 and 2011 using population counts per sub-basin in 2001 and 2011	Kull (2000), Vagen (2006), WorldPop (2018)
Mean distance to settlement (m)	Accessibility, Demand	NGA OCHA-ROSA Populated Places	Produced a distance raster using the Euclidean Distance tool in ArcMap 10.7. Calculated the mean distance to settlement per sub-basin using Zonal Statistics	Brinkmann et al (2014), McConnell et al (2004), Vagen (2006), NGA OCHA ROSA (2007)
Mean elevation (m)	Accessibility, Suitability for agriculture	SRTM 30m Digital Elevation Model	Calculated the mean elevation per sub-basin	Agarwal et al (2005), McConnell et al (2004), Vagen (2006)
Mean slope (°)	Accessibility, Suitability for agriculture,	SRTM 30m Digital	Derived from the DEM using the Slope tool in	Andriatsitohaina et al (2020),

	Vulnerability to natural hazards	Elevation Model	ArcMap 10.7. Calculated the mean slope per sub-basin	Burivalova et al (2015)
Mean annual precipitation 1970-2000 (mm)	Suitability for agriculture	WorldClim v.2	Calculated the mean annual precipitation per sub-basin	Fick and Hijmans, (2017), Andriatsitohaina et al (2020), Eklund et al (2016)
Mean distance to cart track (m)	Accessibility	FTM (Foiben Taosarintanin 'I Madagasikara)	Produced a Euclidean distance raster and calculated mean distance to cart track per sub-basin	Rasolofoson et al (2015)
Mean distance to road (m)	Accessibility	FTM (Foiben Taosarintanin 'I Madagasikara)	Produced a Euclidean distance raster and calculated mean distance to road per sub-basin	Brinkmann et al (2014), Elmqvist et al (2007), Vagen (2006)
Mean distance to river (m)	Accessibility, Vulnerability to natural hazards	HydroSHEDS	Produced a Euclidean distance raster and calculated mean distance to river per sub-basin	Burivalova et al (2015), Allnutt *et al.,* (2013)
Percentage forest cover 2011	Availability	Tropical Moist Forests Annual	Reclassified to remove non-forest pixels. Remaining	Vieilledent *et al.,* (2018),

		Change dataset 2011	classes are undisturbed, degraded and regrowing tropical moist forests. Masked to a forest cover map for Madagascar in 1990 (Vieilledent *et al.*, 2018). The resulting layer represents the proportion of tree cover at any successional stage available to be deforested post-2011.	Vancutsem *et al.*, (2021)
Mean distance to forest edge 2011 (m)	Accessibility, Suitability for agriculture	Vieilledent *et al.*, (2018)	Calculated mean distance to forest edge per sub-basin	McConnell et al (2004)
Basin area (ha)	Availability	WaterSHEDS		(Lehner and Grill, 2013)

Selection of control units to the donor pool – secondary analysis

As a robustness check and to increase the number of control basins for the placebo tests we ran a secondary analysis sampling control units from a wider area - the ex-province of Toamasina (shown in yellow in Figure S3). This resulted in 13 control drainage basins being selected into the donor pool. This includes the original eight control basins from the CAZ plus an additional five basins encompassing unprotected forests in the north of the province.



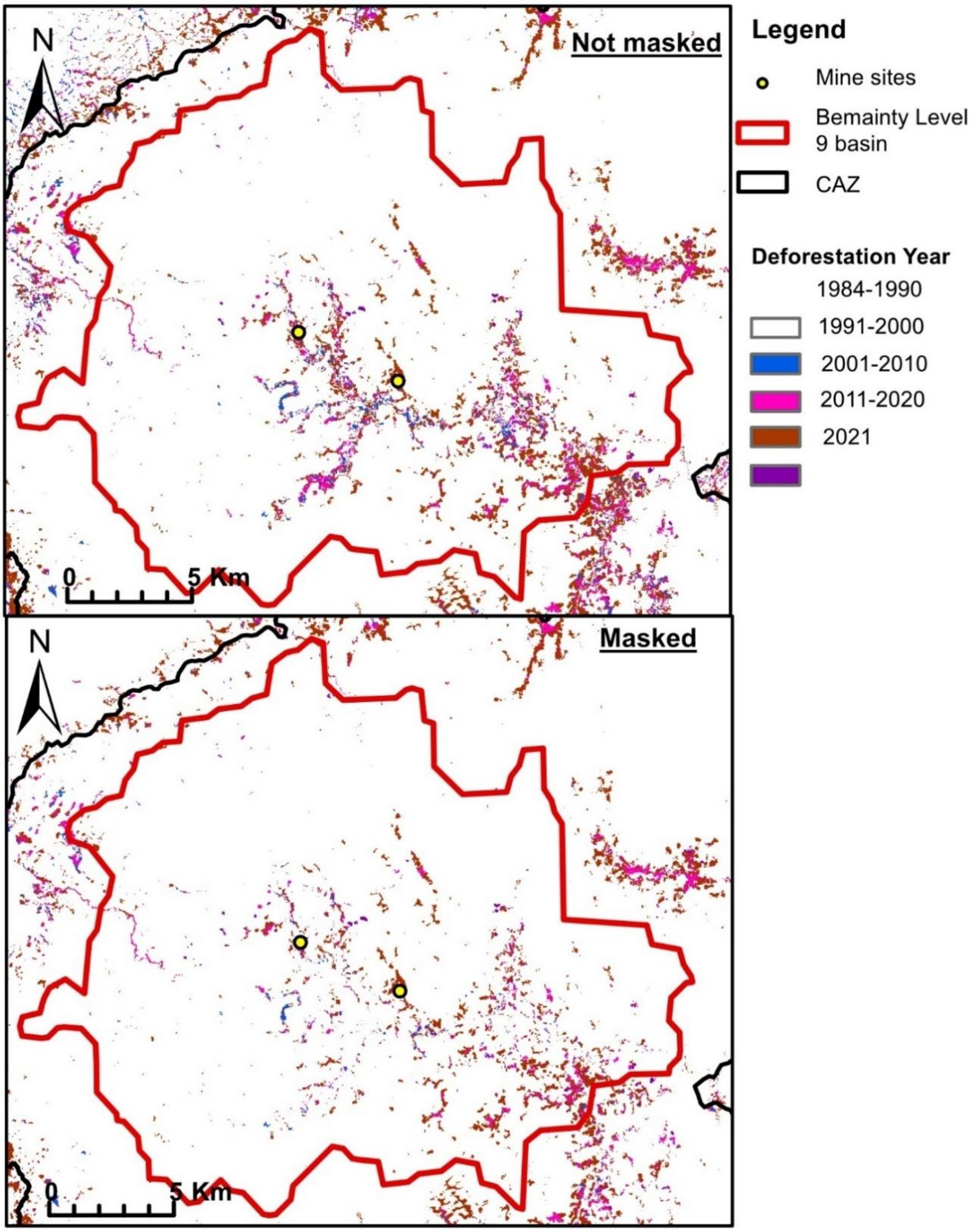
Supplementary Figure 3.3: Map shows the treated Bemainty basin (shaded red) and the 13 drainage basins included in the donor pool (red hashed) for the wider secondary analysis, where units were sampled from the ex-Province of Toamasina (yellow). Only drainage basins with over 70% forest cover in 2011 were included. Drainage basins which overlap with Protected Areas or biodiversity offsets (purple) or which contain other gem mining sites (yellow points) were excluded.

Steps to improve the accuracy of the outcome variable

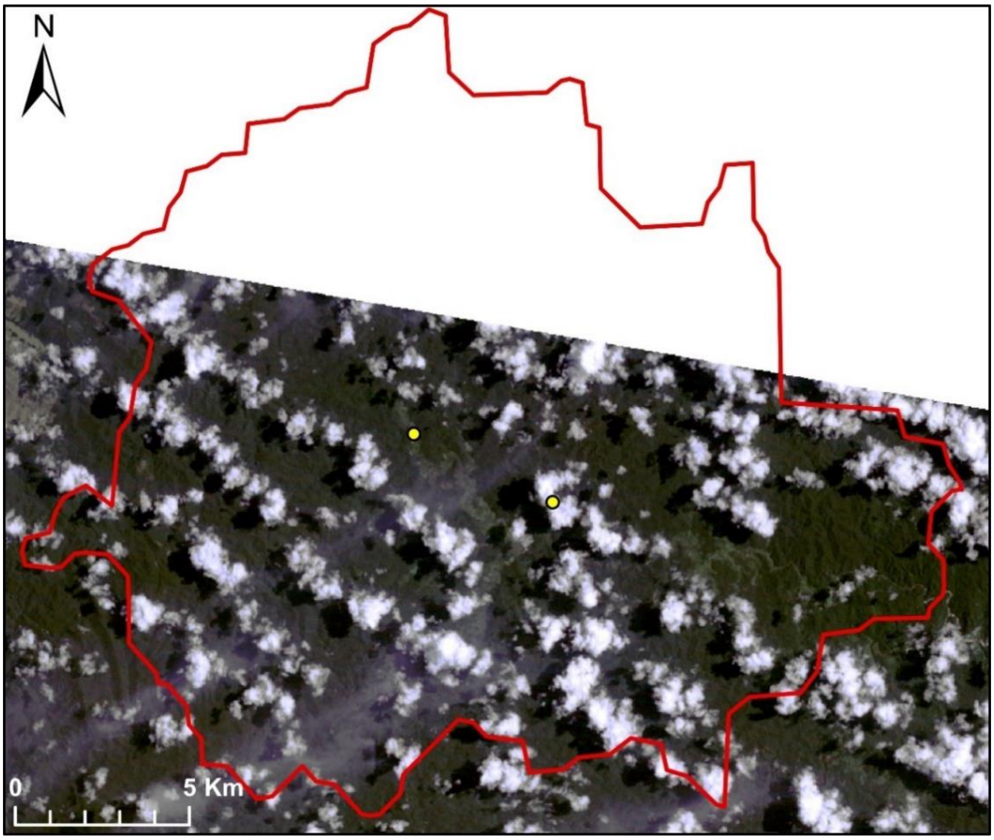
We masked the TMF data products used in our analysis to a national-scale forest cover map of Madagascar in 1990, the start of our study period (Harper *et al.*, 2007; Vieilledent *et al.*, 2018). We considered the national-scale study, designed to capture local specificities, is likely to be more effective at distinguishing forests from other land cover types in Madagascar than a global study.

To investigate this, we compared the original and the masked TMF data and cross-referred with other sources of data, focussing on the Bemainty basin (Supplementary Figure 4). The original TMF deforestation data indicates that the Bemainty valley was cleared during the study period, mostly between 2001 and 2010 (Supplementary Figure 4, top). However, a LANDSAT satellite image from 1989 shows that the valley had already been cleared long before (Supplementary Figure 5). This means that there are false positives in the TMF data where forest loss is identified in pixels which were not forested, perhaps due to clearance of agricultural/fallow land being mis-identified as deforestation. In contrast, the national-scale study, which maps forest change from 1953 – 2000, aligns with the satellite imagery and indicates the valley was cleared in the 1970s (Harper *et al.*, 2007). Therefore, we consider the masked map to be a more accurate representation of forest loss in Madagascar than the original global data.

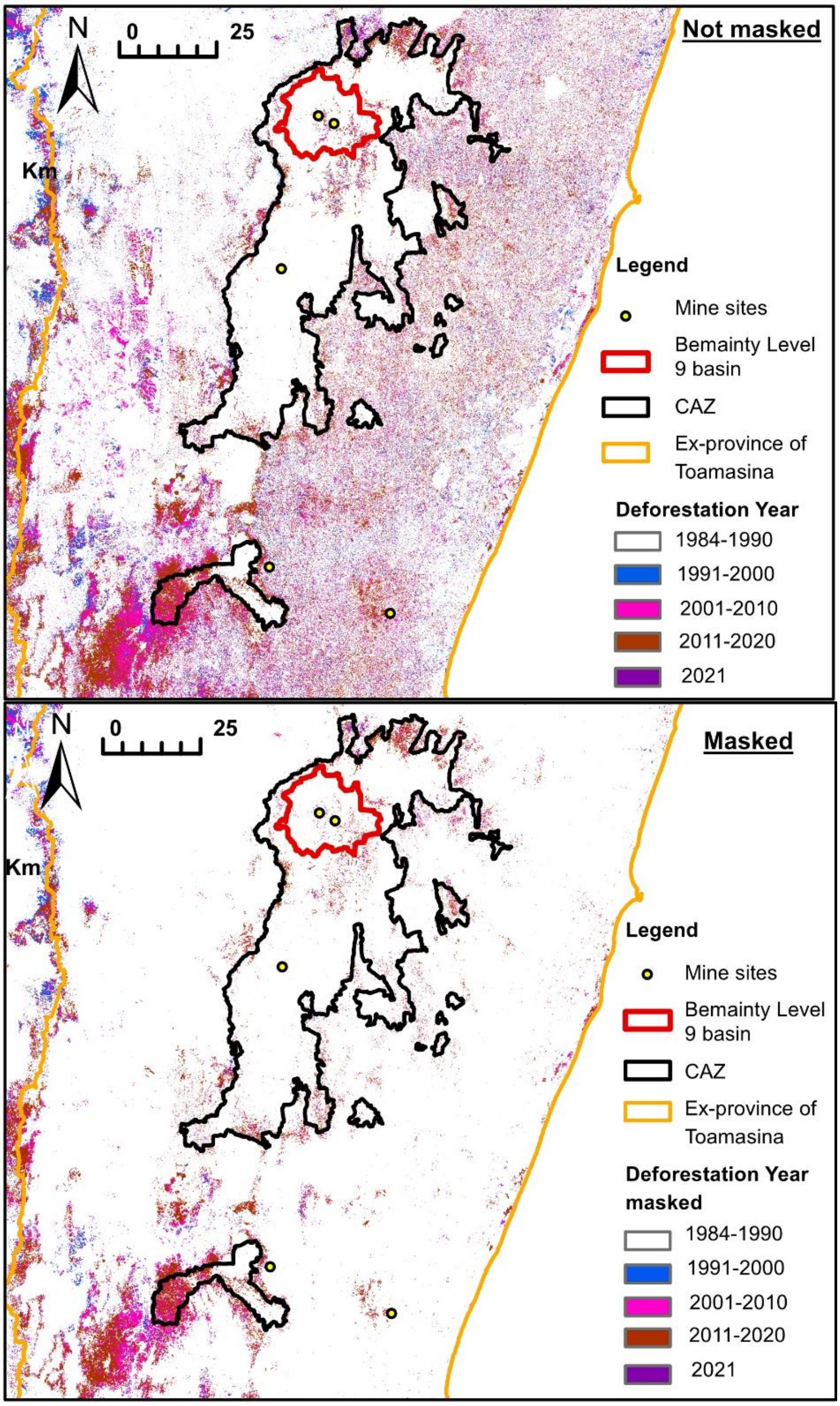
Masking the TMF data to the national map of forest cover in 1990 substantially alters the data, resulting in a large reduction in estimated deforestation, particularly in the lowlands east of the CAZ (Supplementary Figure 6). There the original TMF data detects a large amount of deforestation on land which is not classed as forest in the 1990 map and is most likely agricultural land.



Supplementary Figure 3.4: Comparison of the original Tropical Moist Forests Deforestation Year data (top) to the data masked to a map of forest cover in 1990 from Harper et al (2007; bottom) for the Bemainty basin (outlined in red). Masking the data in this way removes many false positives, where deforestation is identified on land which is not forest. For example, the original data suggests the Bemainty valley was cleared between 2001 and 2020 while satellite imagery shows that it had already been cleared by 1989 (Figure S6). The yellow dots represent the Antananarivo (right) and Ambodipaiso (left) mining valleys.



Supplementary Figure 3.5: LANDSAT image of the Bemainty basin from 12th March 1989. This shows that the Bemainty valley was already cleared by this date.



Supplementary Figure 3.6: Comparison of the original Tropical Moist Forests Deforestation Year data (top) to the data masked to a national map of forest cover in 1990 from Harper et al (2007; bottom). This highlights the substantial differences between the original and masked datasets. The Bemainty drainage basin is outlined in red and the two mining valleys shown by yellow dots within.

Ground-truthing the Tropical Moist Forests data

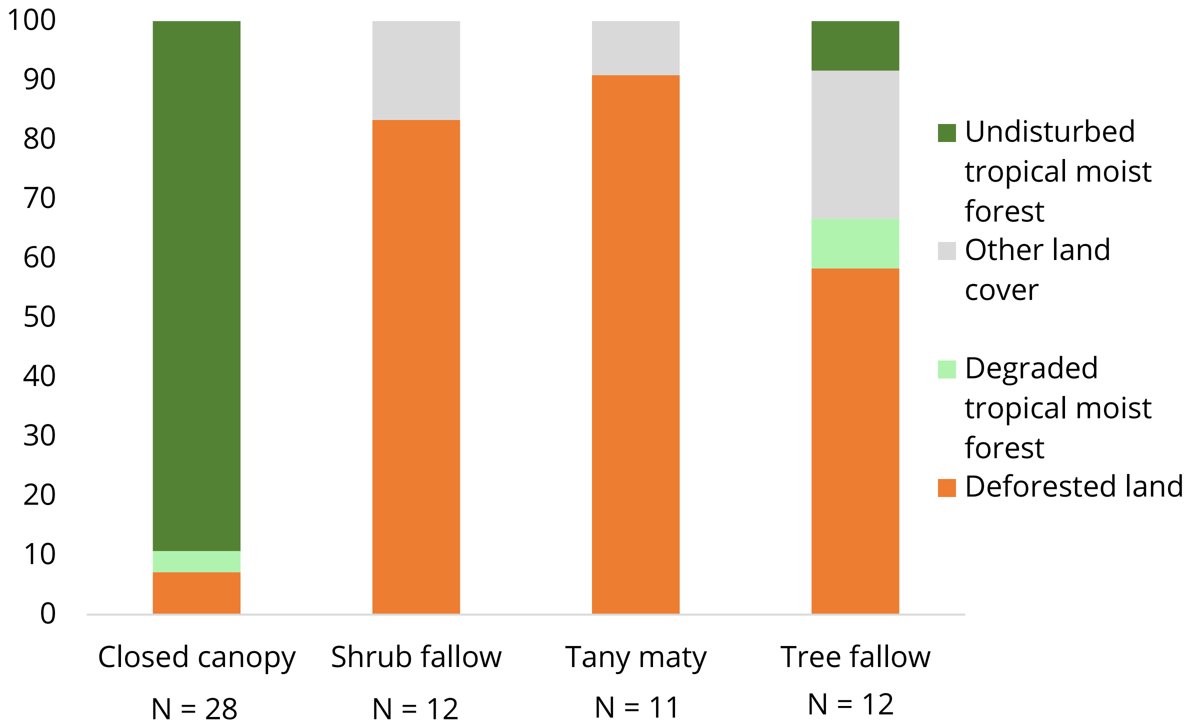
We use land cover data from sites surveyed as part of the Payments for Global Ecosystem Services (P4GES) project to ground-truth the masked TMF data (Razakamanarivo *et al.*, 2017). This project included a carbon stock assessment of the forests of the CAZ based on soil and vegetation data collected from 132 representative sample sites. Sample sites were located within four zones of interest, two in the north and two in the southern part of the CAZ. Land cover data was needed to inform the sampling design, so the initial classification was done using satellite imagery. The land cover at each site was then validated (and revised if incorrect) during the field surveys. Additional information on land use history at each site was obtained from interviews. Land cover was classified into six categories (Razakamanarivo *et al.*, 2017), drawing on Styger *et al* (2007):

- Closed canopy forest
- Eucalyptus plantation
- Reforestation
- Tree fallow (the first fallow period after deforestation where vegetation is dominated by a few tree species)
- Shrub fallow (subsequent fallow periods where the site is dominated by shrub species)
- *Tany maty* (degraded land. This is treeless land at the end of the fallow-cropping cycle where the land has become so degraded it is no longer suitable for agriculture.)

We use this field data to test the accuracy of the masked TMF data at distinguishing different land cover types in the CAZ (Supplementary Table 3.2, Supplementary Figure 7). Field surveys were conducted between April 2014 and June 2015, so we compare to the TMF Annual Change data for 2015. There were 63 surveyed sites which overlapped with our masked TMF data (masked to the area of forest in 1990).

Supplementary Table 3.2: Correspondence between land cover at 63 sites in the CAZ identified through field surveys (conducted April 2014- June 2015) and land cover classification in the TMF Annual Change dataset for 2015. E.g., 89% of sites identified as closed canopy forest in field surveys were classed as undisturbed tropical moist forest in the TMF data.

Ground-truthed land cover 2014-2015	**TMF land cover classification 2015**				
	Deforested (%)	Degraded (%)	Other land cover (%)	Undisturbed tropical moist forest (%)	Total sites
Closed canopy	7.1	3.6	0.0	89.3	28
Shrub fallow	83.3	0	16.7	0.0	12
Tany maty	90.9	0	9.1	0.0	11
Tree fallow	58.3	8.3	25.0	8.3	12
Total sites					63



Supplementary Figure 3.7: TMF classification of 63 sample sites, by land cover category identified during field survey. N = 63.

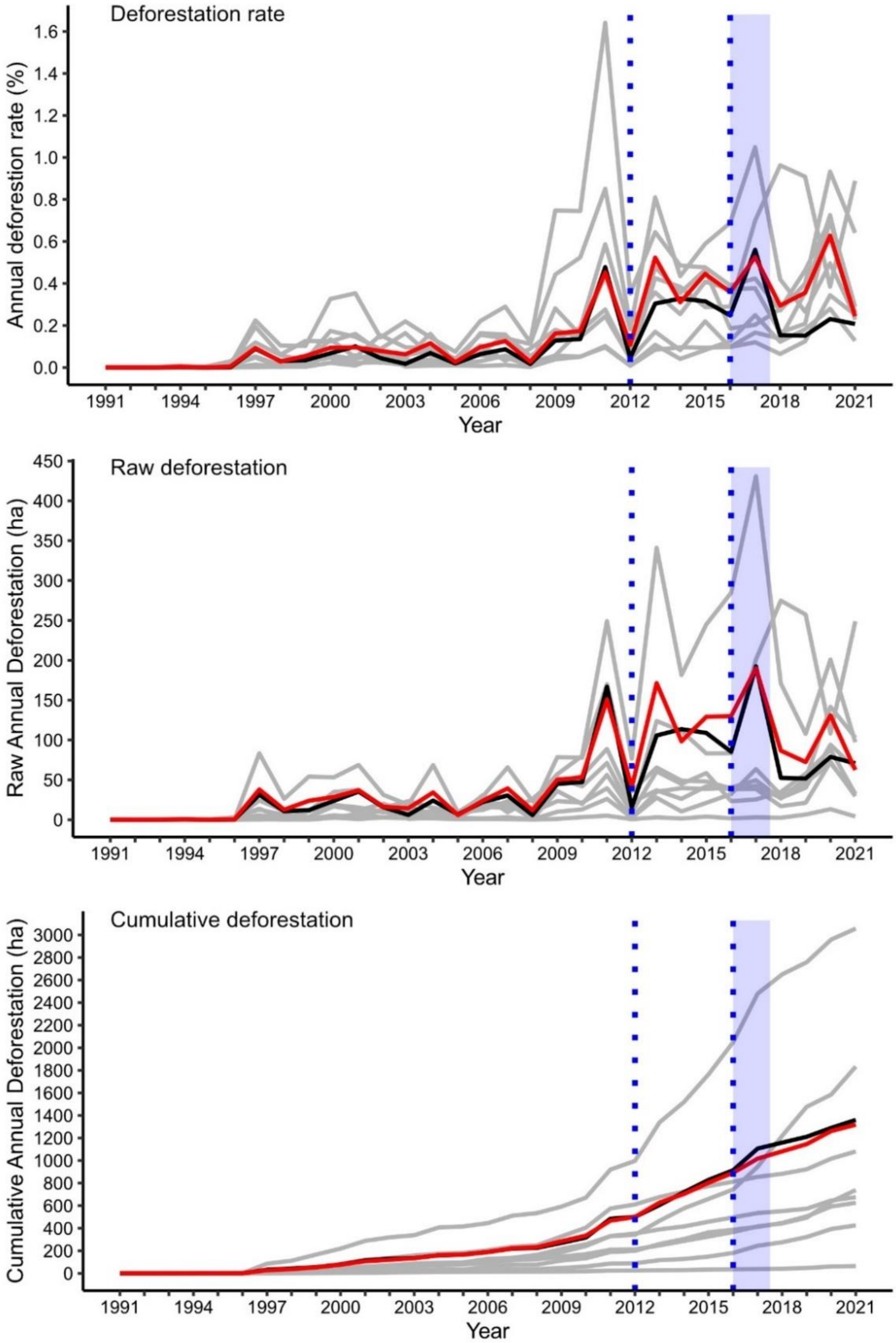
This data indicates that for a sample of 63 sites in and around the CAZ, the TMF data effectively identified closed canopy forest (which most closely aligns to undisturbed tropical moist forest in the TMF classification) in 89% of cases. With the addition of the closed canopy site classed as degraded in the TMF data, which may have been degraded earlier and recovered to closed canopy by the time of the surveys, this increases to 93%. Shrub fallow, tany maty, and tree fallow were mostly classified as deforested land (i.e., pixels forested in 1990 which were cleared during the study period and remained without canopy cover in 2015), or other land cover (which includes agricultural land). Tree fallow was only mistakenly identified as undisturbed forest in one case. While the small sample size (concentrated in four areas) limits the interpretation of these results, it provides some reassuring first evidence of the effectiveness of the masked TMF data at classifying land cover in the study area.

Deriving annual forest cover layers from the Tropical Moist Forests data

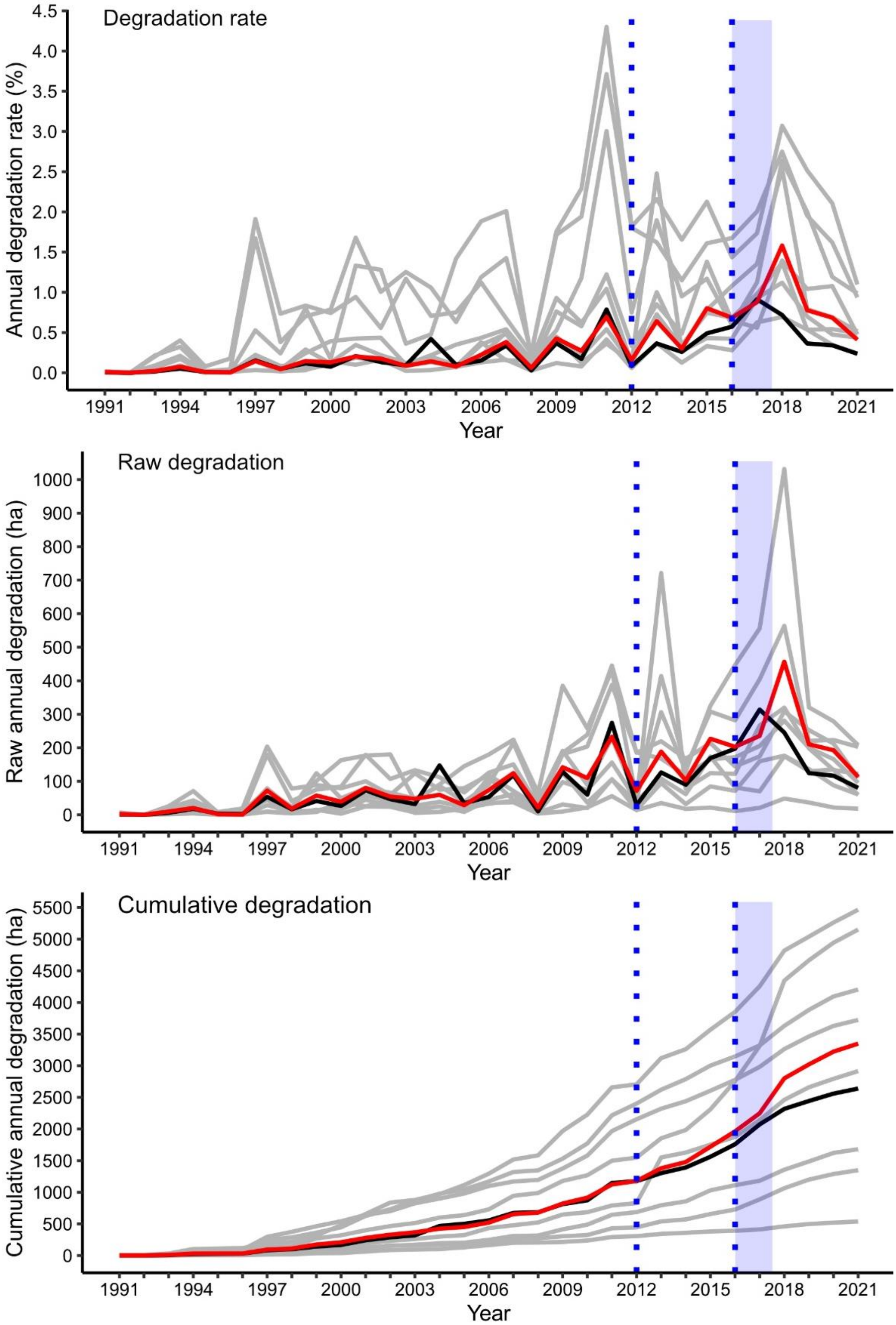
We obtained annual forest cover layers by reclassifying each of the Tropical Moist Forests Annual Change layers. Undisturbed tropical moist forest, degraded tropical moist forest and forest regrowth pixels were reclassified as forest (1) and all other classes were removed.

Degraded tropical moist forest pixels are those which were tropical moist forest at the start of the time series but experienced a loss of tree cover (termed disruption), *lasting less than 2.5 years,* at some point between the start of the time series and the year in question. This means that some recently degraded pixels without tree cover will have been included in our measure of forest cover for each year. However, Vancutsem et al show that most tree loss (50%) within degraded forest pixels lasts less than six months, after which some vegetation recovery is observed (i.e. the disruption is no longer observed). According to this, a degraded pixel which has experienced tree loss in June 1999, for example, will likely have some vegetation recovery by January 2000 and is consequently available to be cleared again. We will be using forest cover at the end of the previous year (*t-1)* to calculate the deforestation rate in year *t.* This allows time for vegetation recovery to begin in pixels degraded in year *t-1.* Tropical forest does not regrow in a few months or years, it can take decades for tropical forest to recover to closed canopy state following disturbance. However, we include degraded pixels in our forest cover layers because this data is intended to represent *the land available to be deforested*, including forest at any successional stage. Furthermore, excluding degraded pixels from our forest cover layer would miss potentially large areas of secondary forest which had been degraded long ago, (e.g. 6% of the Bemainty basin was classed as degraded forest in 2021).

Supplementary Results 3



Supplementary Figure 3.8: Deforestation outcomes in Bemainty (black), the synthetic control (red), and the eight control drainage basins within the CAZ donor pool (light grey), for each of the three outcome measures. The dotted blue lines indicate the onset of mining in 2012 (left) and the start of the mining rush in 2016 (right). The light blue shaded area indicates the duration of the peak mining rush. These results are from our primary analysis focussed on the CAZ.



Supplementary Figure 3.9: Degradation outcomes in Bemainty (black), Synthetic Bemainty (red) and the eight control drainage basins within the CAZ donor pool (light grey), for each of the three outcome measures. The dotted blue lines indicate the onset of mining in 2012 (left) and the start of the mining rush in 2016 (right). The light blue shaded area indicates the duration of the peak mining rush. These results are from our primary analysis focussed on the CAZ.

Supplementary Table 3.3: Percentage change in deforestation between 2016 and 2017 in the Bemainty basin, Synthetic Bemainty and the eight drainage basins in the CAZ donor pool.

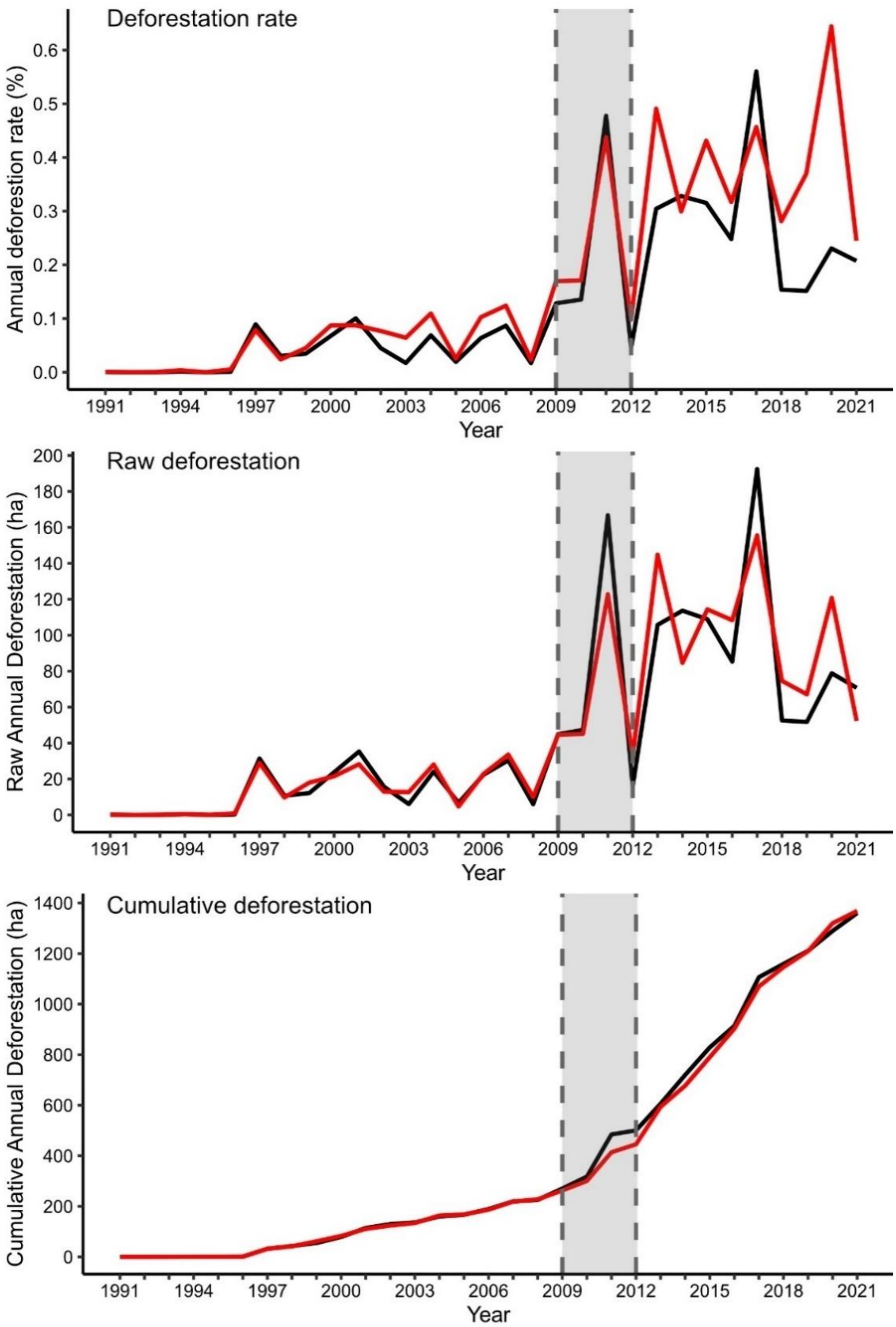
Unit	Raw hectares of deforestation (ha)		Percentage change
	2016	2017	
Bemainty	85.3	192.5	125.6
Synthetic Bemainty	130.1	190.2	46.2
1812	40.2	49.6	23.3
1839	83.4	201.1	141.0
1902	38.3	38.6	0.9
1903	39.3	42.6	8.2
1904	1.8	3.1	70.0
1905	23.3	25.5	9.3
1906	284.7	430.7	51.3
1938	32.3	63.6	96.9

In-time placebo tests

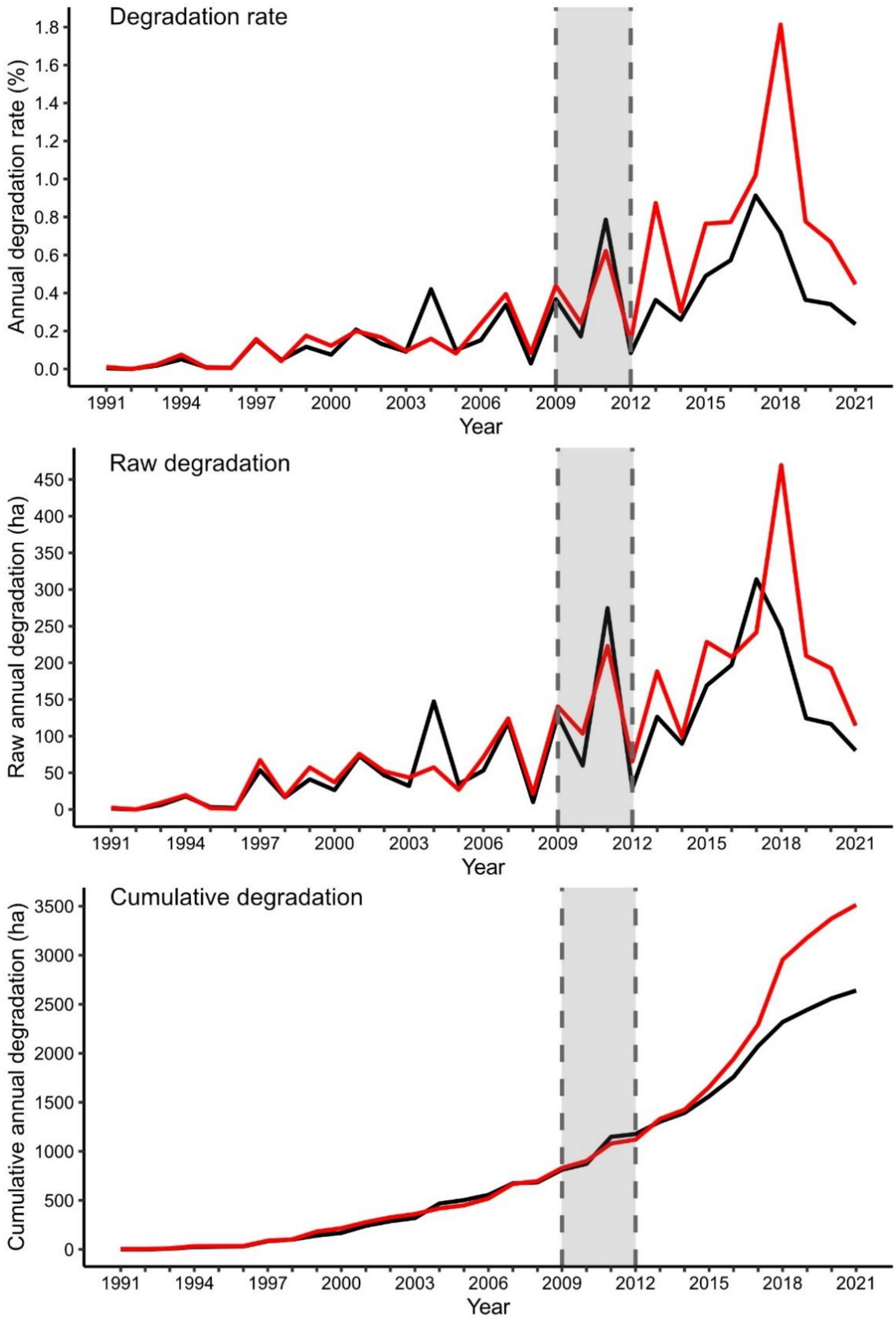
In-time placebo tests were conducted to validate the method and to test the robustness of results to an alternative temporal specification (see Methods). In these placebo tests we falsely-assigned treatment to 2009 and constructed a synthetic control using forest change outcomes from 1991-2008.

The ability of the synthetic control to closely reproduce outcomes in Bemainty between 2009-2012 (i.e., a period without mining), indicates that the method can produce credible estimates of forest change in Bemainty without mining in the real post-intervention period (i.e., the counterfactual). Visually, we can see that outcomes in the synthetic control mostly track outcomes in Bemainty over the validation period (shown in grey). There is a difference in raw (and consequently cumulative) deforestation between Bemainty and its synthetic control in 2011. However, this difference (44 ha) is within the threshold of 0.5% of project area (which is 36,618 ha) considered by West et al (2023) to be acceptable.

In contrast to the main results, the in-time placebo tests indicate higher deforestation in Bemainty in 2017, although this is a very similar magnitude (37 ha) to the difference in 2011, and therefore cannot be robustly attributed to the mining rush. Degradation is mostly lower in Bemainty than the synthetic control at the height of the mining rush (2016 and 2017).

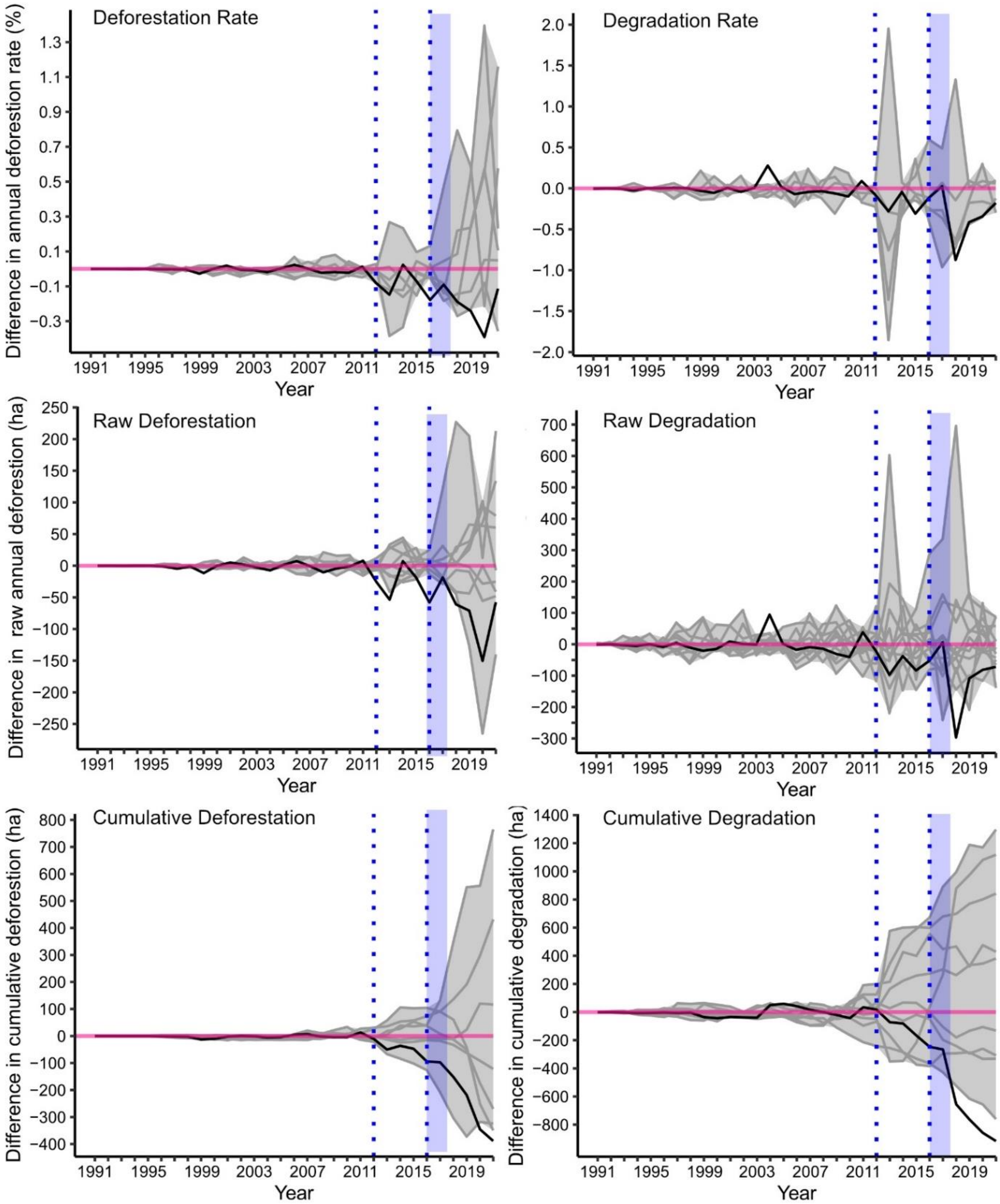


Supplementary Figure 3.10: Results from the in-time placebo tests for the three deforestation outcomes. Treatment was falsely-assigned in 2009 (left grey dashed line). The black line shows deforestation outcomes in Bemainty while the red line shows outcomes in the synthetic control. The shaded grey area indicates the validation period between false-treatment and actual treatment (i.e., the start of mining in 2012). The difference between the red and black lines in this period reflects the ability of the synthetic control to reproduce outcomes in Bemainty in the absence of mining.



Supplementary Figure 3.11: Results from the in-time placebo tests for the three forest degradation outcomes. Treatment was falsely-assigned in 2009 (left grey dashed line). The black line shows degradation outcomes in Bemainty while the red line shows outcomes in the synthetic control. The shaded grey area indicates the validation period between false-treatment and actual treatment (i.e., the start of mining in 2012). The difference between the red and black lines in this period reflects the ability of the synthetic control to reproduce outcomes in Bemainty in the absence of mining.

Placebo tests to assess the significance of results from the wider analysis



Supplementary Figure 3.12: Assessing the significance of results from the wider analysis using placebo tests to quantify the range of noise in the estimation method. Grey lines represent the difference in outcomes between each false-treated (control) basin and its synthetic control (only pairs where the synthetic control is an acceptable match to the false-treated unit are included, see Methods). The range of values from the placebo tests represents the statistical noise in estimation post-intervention (shaded in light grey). The difference in forest change outcomes between Bemainty and it's synthetic control is shown in black. A strong significant effect is indicated where the black line falls outside the shaded grey area. The dotted blue lines indicate the onset of mining in 2012 (left) and the start of the mining rush in 2016 (right). The light blue shaded area indicates the duration of the peak mining rush. Results are from the wider analysis sampling control basins from the ex-province of Toamasina.

Interview data

Supplementary Table 3.4: Characteristics of four villages in the Bemainty basin and estimates of the population in 2019. Population estimates were provided by the Chief of the Fokotany of Antsevabe within which Bemainty is located. Antananarivo is the temporary mining settlement created during the mining rush. The other three villages existed prior to mining.

Village	Number of houses	Number of grocery shops	Number of houses with metal roofs	Estimates of the adult population	School
Antananarivo	133	11	0	500	No
Sahamatra	23	0	0	120	No
Sahananto	22	0	1	150	Yes -1
Bemainty	49	3	4	220-300	

Evidence from interviews revealed that the mining rush brought serious socio-economic costs for the local community. Farmers (who can be considered local residents) reported that the mining rush caused declines in water availability and quality, which impacted rice production.

"When the mining extraction started, rice production became lower and we noticed that the water become scarce in the ricefield." (Local farmer, Bemainty).

Lack of sanitation in the densely populated mining valleys led to pollution and increased the risk of disease:

"The environment was great few years ago. With the mining, the water became scarce and dirty. Some people use the river as a toilet so it is not good for the environment and our health."

Others reported that the sudden increase in demand from thousands of migrant miners caused high inflation in the price of food.

"...during the sapphire rush, the bandits attacked people and insecurity increased. And the price of rice increased. Before it was 200 ariary per cup and now the price is 500ariary per cup" (Local farmer, Bemainty).

The impacts of the mining rush on rice production and food prices likely reduced food security in local villages. Miners may also have struggled with the high food prices.

Many farmers (35%) reported that the economic benefits of the mining rush were not equally shared among the local community, and mostly accrued to the migrant miners:

"Migrant people got lot of money compared to us. Now, there are few miners, but the negative impact of sapphire rush remains because there is less water for crops" (Local farmer, Bemainty)

Most miners (55%) mentioned the money which could potentially be earnt through artisanal mining, which could be used to fund investments in land, housing, or children's education:

"I have kids who are studying at the university and I pay their fees using money from sapphire mining. This is our job and we know that sapphires can change our life." (Miner, Antananarivo).

Several respondents (miners and farmers) recounted stories of lucky miners who had found large stones and made a lot of money. Yet many miners only reported finding small stones themselves, or no stones at all.

*"I've only found small pieces of sapphire (0.5g or 0.6g). I have been in Milliard 2 years now. Some people got 16g, even 30g". (*Miner, Milliard).

Many miners stated that it had become harder to find sapphires so many people had left. Others emphasized the hard work required to find sapphires and the need to abide by taboos (*fady).*

"Working in sapphire is great and it brings money. However, it is hard to find sapphires in Antananarivo nowadays." (Miner, Antananarivo).

"Sapphire is easy money but you need to be a hard-worker to get it and you have to follow the rules and taboos"

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