Impact Evaluation in a Landscape: Protected Natural Forests, Anthropized Forested Lands and Deforestation Leakages in Madagascar's Rainforests

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Études et Documents n° 13

June 2016

To cite this document:

Desbureaux S., Kéré E., Combes Motel P. (2016) "Impact Evaluation in a Landscape: Protected Natural Forests, Anthropized Forested Lands and Deforestation Leakages in Madagascar's Rainforests", *Études et Documents,* n° 13, CERDI.
http://cerdi.org/production/show/id/1817/type_production_id/1

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This work was supported by the LABEX IDGM+ (ANR-10-LABX-14-01) within the program "Investissements d'Avenir" operated by the French National Research Agency (ANR).

Études et Documents are available online at: http://www.cerdi.org/ed

Director of Publication: Vianney Dequiedt
Editor: Catherine Araujo Bonjean
Publisher: Mariannick Cornec
ISSN: 2114 - 7957

Abstract

This paper analyzes deforestation leakages from natural rainforests to anthropized habitats following the creation of Protected Areas in Madagascar. A simple theoretical framework highlights that a conservation constraint does not necessarily create deforestation leakages on secondary forests. An original dataset is built combining fine scale vegetation cover images and spatialized census data over the period 2000 to 2012. Cover images allow us to distinguish a mosaic of landscapes. Multilevel panel regressions and matching techniques indicate a causal effect of Protected Areas on deforestation leakages. Though Protected Areas reduce deforestation in protected natural forests, forest clearing is mostly reported on other types of anthropized forests. Our results demonstrate the limitations of Porter-like mechanism in agricultural innovation. They also support the hypothesis of a conservation dilemma: protecting biodiversity may come at the expense of the welfare of locals who rely on local (provisioning) ecosystem services.

Keywords

Land use patterns, Deforestation, Environmental policies, Agricultural innovation.

JEL codes

O12, O13, Q15, Q23, R14

Acknowledgment

Sébastien Desbureaux thanks the *DP Forêts et Biodiversité* for its institutional support in Madagascar and Emilson Rakotorisoa for his research assistance; and acknowledges the financial support of the National Geographic Society through the Young Explorer Program (#C293-15). Eric Kéré was supported by the French government's *Agence Nationale de la Recherche* through the "*Investissements d'avenir*" program (ANR-10-LABX-14-01). This project started while he was a Research Fellow at CERDI. Viewpoints and any errors are our own.

1 Introduction

Human beings have been modifying tropical forests for tens of thousands of years. Between 2000 to 2012 alone, an estimated 2.3 million square kilometers of forests have been lost worldwide (Hansen et al. 2013). These forests have been transformed mainly into agricultural lands (Kissinger and Herold 2012) but also into anthropized forested landscapes that supply direct ecosystem services to locals (construction wood, fueldwood, charcoal etc), resulting in the development of many human-modified mosaic landscapes. Today 410 million people to depend highly on the access to natural and anthropized forests to support their livelihoods (WRI 2005).

International efforts to tackle tropical deforestation have increased since the early 1990s, partly in the name of the global ecosystem services forests provide to human beings, such as biodiversity conservation, carbon storage and sequestration (Costanza, d'Arge, et al. 1997; MEA 2005; Costanza, Groot, et al. 2014). Protected Areas (PAs) remain the dominant policy answer to stopping deforestation of natural forests and have proven to be effective at attaining this objective (Miteva, Pattanayak, and Ferraro 2012). Yet locals may simply displace deforestation from the created PA to elsewhere in the landscape. This phenomenon is known in the literature as deforestation leakage [1].

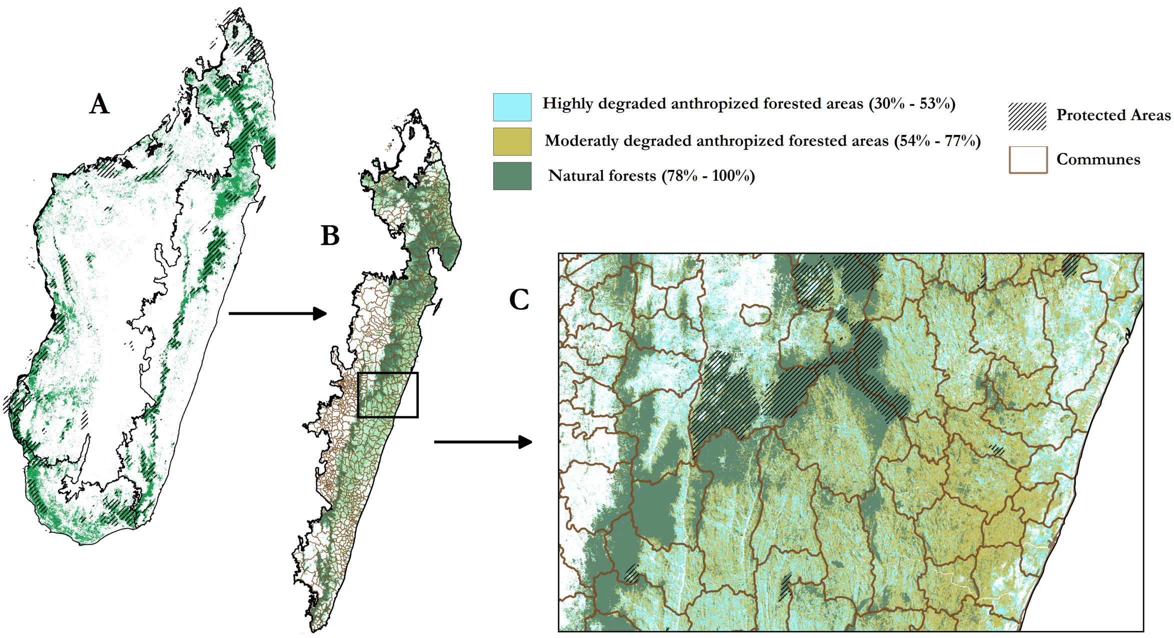
This paper examines the possibility of deforestation leakages from natural forests to anthropized forested areas following the creation of PAs in the context of Madagascar's rainforests. Madagascar is one of the top three countries worldwide in terms of mega-biodiversity (Goodman and Benstead 2003). Most of its terrestrial biodiversity is concentrated in natural forests. Over the last 60 years, the country has probably lost around 50% of its natural forest cover (Harper et al. 2007; McConnell and Kull 2014), making Madagascar one of the hottest spots for global biodiversity conservation (Myers et al. 2000). Approximately 9 million hectares of natural forests remain in Madagascar, of which about 5.6 million are rainforest, located in the eastern ecoregion. According to our calculations, around 1.5 million people live in highly forested areas in Madagascar. Family agriculture is the principal economic activity of about 90% of these forest inhabitants, with slash and burn being the dominant agricultural technique for rice production (ed, the staple food in Madagascar). In a context of rapid population growth (2.9% average according to the World Bank), of an observed sharp decrease of fallow lengths (Styger et al. 2007)and of limited adoptions of agricultural innovations (Minten

[1] The possibility of leakages also appears for other types of conservation instruments such as Payments for Environmental Services (Alix-Garcia, Shapiro, and Sims 2012; Le Velly, Sauquet, and Cortina-Villar 2015) ; reforestation policies (Meyfroidt and Lambin 2009, forest concessions in Brazil (Oliveira et al. 2007) or REDD+ initiatives (Aukland, Costa, and Brown 2003)

and Barrett 2008), slash and burn is the primary driver of deforestation, despite its practice on natural forests being prohibited since 1881 (Jarosz 1993). As a result, a growing network of PAs now covers around 40% of the remaining forests (3.6 million hectares) and appear to have been effective in slightly reducing (but not in halting) deforestation on natural forests (Gimenez 2012; Desbureaux, Aubert, et al. 2015).

In addition to natural forests, vegetation data from satellite images clearly illustrate that a large part of the Malagasy landscape is now covered by anthropized forested areas (Figure 1). These can be degraded remains of past natural forests, eucalyptus and pine plantations, and agro-forestry systems such as shaded cocoa or coffee groves. In the eastern ecoregion alone, the dataset we constructed reveals the presence of about 6.5 million hectares of anthropized forested areas with moderate or highly degraded tree cover. These areas are not of primary interest for biodiversity conservation or for carbon sequestration. They do however supply many provisioning ecosystem services to the local population. As soils' fertility in these degraded forests tends to be poor (particularly after eucalyptus plantation), most of these areas are often unsuitable for agriculture.

Figure 1: Vegetation Mosaic in Eastern Madagascar



Note: This figure represents the landscape mosaic in eastern Madagascar. *A* displays natural forests in 2000 and PAs for the whole country. *B* and *C* represent the whole vegetation mosaic for the eastern ecoregion. *Data:* Natural forest cover in 2000 from BioSceneMada. Vegetation data in 2000 from Hansen et al. 2013, v.1.0. Protected Areas from SAPM (MEF-CIRAD database). Municipalities (ndlr, *Communes*) boudaries from INSTAT. *Reference System:* WGS84-UTM 38s. Source: Authors.

However, anecdotal evidence from our field visits reveals a significant transformation of anthropized forests – namely the conversion of eucalyptus plantations into rice plantations at the edge of some PAs where we have previously worked (see Figure A1 in the Appendix). Some villagers repeatedly stated during informal discussions that they had decided to cultivate rice on eucalyptus parcels because clearing natural [ed, protected] forests was illegal. Thus, they simply displace rice production and deforestation elsewhere in the landscape rather than investing in more intensive agricultural techniques as predicted in a Porter-like hypothesis (Porter and Van der Linde 1995). By contrast, a recent study by geographers has reported notable intensification of agricultural practices in the north-east part of Madagascar where large PAs have been established (Zaehringer et al. 2016). The authors hypothesize that this intensification may have been propelled by the establishment of PAs. Our deforestation dataset indicates that 57,000 hectares of highly degraded forested areas and 260,000 hectares of moderately degraded forested in the eastern ecoregion were lost lost between 2000 and 2012. Part of these losses may correspond to deforestation leakages from natural protected forests. If so, the protection of natural forests to conserve global ecosystem services may negatively impact local inhabitants who would experienced a weakening of provisioning ecosystem services provided by antrhopized areas.

The paper begins by theoretically highlighting via an agricultural household model taking into account ecosystem services how and under what conditions leakages from natural habitats to anthropized forests can occur after the establishment of PAs. In our model, the household can produce an agricultural commodity (rice) by clearing natural forests or by clearing anthropized forested areas (e.g., plantations). Standing anthropized forests provide the household provisioning ecosystem services (e.g. charcoal) when associated with some labor time. The household thus faces a land use conflict over anthropized forests: either it converts plantations in agricultural lands, harvests rice but loses direct ecosystem services, or it keeps the plantation and enjoys the ecosystem services. We show that conservation constraints on natural forests such as the creation of PAs if effective, release labor time for the household. Whether the household reallocate this time to agricultural production (the leakage effect), or to the production of ecosystem service will in the end depend on differences in marginal labor productivities for the two activities. Second, we empirically test for the presence of leakages from natural to human impacted forests between 2000 and 2012 for the entire Malagasy rainforest. We use a panel of 13 years of fine scale Landsat vegetation cover images to distinguish between three types of anthropized habitats: natural forests, a slightly degraded habitat and a highly degraded one. We spatialize census data then use multi-level panel regressions and matching techniques to exhibit a potential causal leakage effect of PAs.

This study contributes to the impact evaluation literature regarding nature conservation policies. Recent studies converge to show that PAs have allowed for a reduction in deforestation of about 10% to 20%[2]. These studies have mainly focused on a small number of countries in Latin America and South East Asia, but few concern Africa with the exception of Gimenez (2012) and Desbureaux, Aubert, et al. (2015) that look at Madagascar. We add to this literature by focusing on leakages. Previous studies find both close to no leakages (Macedo et al. 2012; Soares-Filho et al. 2010), particularly negative leakages (Ewers and A. S. Rodrigues 2008) or positive leakages (Honey-Roseés, Baylis, and Ramirez 2011; Gaveau et al. 2009; Robalino and Pfaff 2012. Our findings suggest that PAs have allowed for decreased deforestation in Madagascar within the protected natural forests themselves but have caused negative leakages: after the creation of PAs, deforestation in adjacent areas has increased.

To our knowledge, this papers is the first to examine leakages in a landscape, i.e. leakages from natural habitats to already anthropized areas instead of leakages from protected natural forests to other natural yet unprotected forests. In Madagascar, PAs are large in sizes. New PAs cover areas of up to more than 600,000ha. Thus, leakages from protected natural forests to other unprotected natural forests would involve population migrations and resettlement, and would be a mid- to long-term phenomenon. By studying leakages from the protected natural forests to anthropized forests within a municipality, our approach emphasizes a more short-term adaptation of the behaviors of farmers. We show that the leakages on anhropized areas have surpassed the observed decrease in deforestation within protected natural forests during the period studied.

Additionaly, the idea of negative leakages is closely related to the literature focused on the adoption of agricultural innovations: instead of innovating to comply with new land use restrictions as in a Porter-like hypothesis of a forced innovation (Porter and Van der Linde 1995), farmers stick with the same land-extensive agricultural practice and fulfill their needs for fertile lands by displacing deforestation to unprotected areas. The hypothesis of contextual-driven innovation has been proposed for long in the case of agrarian transitions (Boserup 1965). This hypothesis of adjustment remains debated today, as in Madagascar (Zaehringer et al. 2016). We discuss our results in the light of this literature to provide thoughts on why an agricultural transition has not yet emerged for Malagasy family farmers.

The paper proceeds as follows: Section 2 provides the theoretical foundations for our

[2]Examples of recent studies are Andam et al. 2008; Pfaff et al. 2009; Arriagada et al. 2012 in Costa Rica ; Gaveau et al. 2009 in Indonesia ; Nolte et al. 2013 in Brazil ; Sims 2010 in Thaïland ; Alix-Garcia, Sims, and Yañez-Pagans 2015; Blackman, Pfaff, and Robalino 2015 in Mexico and Nelson and Chomitz 2011 for a set of tropical countries but using fire data instead of direct deforestation data.

analysis, Section 3 describes the data, Section 4 describes the empirical strategy and Section 5 presents the empirical results of leakages for Madagascar's rainforests. We discuss our results in Section 6 and then conclude.

2 Theoretical Framework

In Madagascar, 80% of the population live in rural areas and rely on subsistence agriculture for their living (Instat - EPM 2010) Rice is the staple food and the dominant crop in the country: 87% of agricultural households grow rice as the dominant commodity (Instat – EPM 2010). They are probably about 1.5 million individuals living directly in highly forested areas. These forests provide them with a stock of potential fertile lands for their agricultural activities with slash and burn being the dominant agricultural technique.

Households' welfare also depends on local forest ecosystem services (ES hereafter) whether they come from natural forests or anthropized forested areas. Local ES do not obviously embed global ES such as forest carbon storage which can represent the main part of the value of standing forests (e.g. Bulte et al. 2002). Local ES are rather privately appropriated by households. They mainly consist in the provision of charcoal and construction woods. All subsistence generating activities depend on land and labor inputs.

2.1 The Setup

We model the production choice of a welfare maximizing agricultural household in the presence of multiple types of lands that provide different ES. Motivated by the Malagasy context and following several authors (e.g. De Janvry, Fafchamps, and Sadoulet 1991), it is assumed that the labor market is shallow (i.e. households are self-sufficient in labor), and that labor cannot be traded on the market. There is neither off-farm labor nor rented labor. We denote L the stock of labor held by the household.

We assume that the household produces a single agricultural commodity, let's say rice, for subsistence purposes. Subsistence cropping is mainly conducted on cleared land, through slash-and-burn agriculture (only 12% of agricultural lands are irrigated rice parcels in the eastern eco-region, Table 1). Land can be cleared either on natural forests (practiced known as *tavy* in Madagascar) or on secondary forested areas (*teviala*). We call A_1 the amount of converted lands by the household on natural forests. When no conservation effort is implemented, natural forests are *de facto* freely accessible. The implementation of protected areas is modeled as an upper constraint $\bar{A}$ on land converted from natural forests. Secondary forests are constituted for instance by eucalyptus plantations. Since they stem from previously converted forests, it is assumed that their surfaces $\bar{A}_2$ is fixed. Two types of activities can take place in secondary forests: A_2 can be cleared once again for subsistence purposes (rice production) while $\bar{A}_2 - A_2$ can be kept in place for the provision of ES (e.g., charcoal production). The household therefore faces a double trade-off: *(i)* Producing rice on land plots encroached either on natural or secondary forests; and *(ii)* Allocating secondary forested areas to rice or ES provision.

We call $Y^1(A_1, L_1)$ the return on rice grown on A_1 units of cleared natural forests with L_1 units of labor. $Y^2(A_2, L_2)$ is the return on rice grown on A_2 units of cleared secondary forests with L_2 units of labor. $Y^3(\bar{A}_2 - A_2, L - L_1 - L_2)$ stands for the return on the provision of local ES which rely on remained secondary forested areas $\bar{A}_2 - A_2$ and on remained labor $L - L_1 - L_2$. Y^i are net of labor and other input costs.

The question is whether an increase i, conservation effort such as the implementation of PAs will induce deforestation leakages i.e., an increase in deforestation on secondary forests A_2?.

Our model shares several features with the agricultural household models' literature initiated by Singh, Squire, and Strauss (1986). It is however different in several ways. First, production and consumption decisions are separable. Consequently labor allocation decisions do not depend on household composition (see e.g. Benjamin 1992). Second, the model does not address the question of market failures (e.g. De Janvry, Fafchamps, and Sadoulet 1991). There exists several examples of models of land conversion (among several authors: Angelsen and Kaimowitz 1999; Barbier 2007; Delacote and Angelsen 2015) but to the best of our knowledge, none explicitly modeled deforestation leakages induced by the implementation of PAs.

2.2 The household maximization program

Once the Y^i are aggregated, they constitute the net return from agricultural production and local ES provision, denoted π. π is supposed to be strictly quasi-concave. Cross derivatives are assumed to be strictly positive. More generally, it is supposed to be well-behaved so that all solutions are interior. Formally, the problem is:

$$max_{A_1,A_2,L_1,L_2}\pi \equiv Y^1(A_1, L_1) + Y^2(A_2, L_2) + Y^3(\bar{A}_2 - A_2, L - L_1 - L_2) \text{ such that } A_1 < \bar{A}$$

The inequality constraint $A_1 < \bar{A}$ corresponds to the conservation constraint: a decrease in $\bar{A}$ denotes an increase in the conservation effort i.e. an increase in natural forests included in PAs. The corresponding Lagrangean function writes as:

$$L_{A_1,A_2,L_1,L_2,\mu;\bar{A},\bar{A}_2,L} \equiv Y^1(A_1,L_1) + Y^2(A_2,L_2) + Y^3(\bar{A}_2 - A_2, L - L_1 - L_2) - \mu(A_1 - \bar{A})$$

where μ is the non-negative multiplier. It represents the implicit additional cost born by households when PAs are implemented. This theoretical setting allows taking into account local benefits extracted from standing forests but without internalizing global benefits such as carbon storage. The potential conflict between local and global benefits from conservation initiatives is not taken into account (Perrings and Gadgil 2003).

First order necessary conditions (FONC) for an interior solution are:

$$FONC: \begin{cases} \frac{\partial L^\star}{\partial A_1} = 0 \Leftrightarrow \frac{\partial Y^1}{\partial A_1} - \mu = 0 \\ \frac{\partial L^\star}{\partial A_2} = 0 \Leftrightarrow \frac{\partial Y^2}{\partial A_2} - \frac{\partial Y^3}{\partial A} = 0 \\ \frac{\partial L^\star}{\partial L_1} = 0 \Leftrightarrow \frac{\partial Y^1}{\partial L_1} - \frac{\partial Y^3}{\partial L} = 0 \\ \frac{\partial L^\star}{\partial L_2} = 0 \Leftrightarrow \frac{\partial Y^2}{\partial L_2} - \frac{\partial Y^3}{\partial L} = 0 \frac{\partial L^\star}{\partial \mu} \geq 0, \mu \geq 0 \\ \quad \text{and } \mu \frac{\partial L^\star}{\partial \mu} = 0 \text{ i.e. } -A_1 + \bar{A} \geq 0, \mu\left(A_1 - \bar{A}\right) = 0 \end{cases}$$

From the envelope theorem, the overall effect of the conservation effort is shown to have a non-positive impact on the maximized profit: $\frac{\partial \pi^\star}{\partial \bar{A}} = \frac{\partial L^*}{\partial \bar{A}} = \mu > 0$. The quasi-concavity of π ensures the existence of a maximum (Appendix B1). As a consequence, FONC allows defining A_1, A_2, L_1 and L_2 as implicit functions of exogenous parameters. In the following, we are particularly interested in the effect of $\bar{A}$ on the demands for agricultural factors.

Remark 1: For the sake of simplicity it is assumed that conservation effort is linear in $\bar{A}$. However, one hectare of PA often does not mean one hectare effectively protected because of imperfect law enforcement in tropical countries. See in the appendix the discussion on that.

Remark 2: It would be possible to refine the maximization problem so as to consider subsistence requirements. This can be made by explicitly introducing a minimum level for rice production. The Lagrangean function would be:

$$\begin{aligned} L_{A_1,A_2,L_1,L_2,\mu;\bar{A},\bar{A}_2,L,\underline{Y}} &\equiv Y^1(A_1,L_1) + Y^2(A_2,L_2) + Y^3(\bar{A}_2 - A_2, \bar{L} - L_1 - L_2) \\ &- \mu(A_1 - \bar{A}) - \lambda(\underline{Y} - Y^1 - Y^2) \end{aligned}$$

where $\underline{Y}$ is a minimum level of rice production return. It would deliver an additional

FONC written as: $\frac{\partial L^\star}{\partial \lambda} \geq 0, \lambda \geq 0$ and $\lambda \frac{\partial L^\star}{\partial \lambda} = 0$.

Another possibility would be to maximize the indirect utility function V of the returns from rice production and local SE provision: $V(Y^1(A_1, L_1) + Y^2(A_2, L_2) + Y^3(\bar{A}_2 - A_2, L - L_1 - L_2))$ under the additional constraint $V(Y^1(A_1, L_1) + Y^2(A_2, L_2) + Y^3(\bar{A}_2 - A_2, L - L_1 - L_2)) \geq V_0$. This idea was explored by Pagiola (1995) in a dynamic setting.

Remark 3: We present here a static model. As highlighted by Porter, in a dynamic model the establishment of environmental regulations might incite the household to invest in new and more intensive agricultural practice to comply with the law, and in our case, thus ensure the provision of ES. However in Madagascar, we observe a limited adoption of agricultural practices in general (Minten and Barrett 2008). We thus believe that our static framework is sufficient to capture insights in the Malagasy context.

2.3 Comparative Statics

The comparative statics exercise allows establishing two propositions highlighting how and when leakages can occur. We denote $|H|$ the positive determinant of the bordered hessian matrix. Following the FONC, we first study how $L_1^\star$ varies for a change in the amount of protected forests $\bar{A}$:

$$|H|\frac{\partial L_1}{\partial \bar{A}} = \frac{\partial^2 Y^1}{\partial A_1 \partial L_1}\left(\left(\frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^3}{\partial A^2}\right)\left(\frac{(\partial^2 Y^2}{\partial L_2^2} + \frac{\partial^2 Y^3}{(\partial L^2}\right) - \left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)^2\right) \tag{1}$$

Because of the concavity of production functions, Equation 1 will be positive if the direct second order derivatives dominates cross derivatives i.e. when "own-effects dominates cross effects" (Silberberg, Wing Suen, and WC Suen 1990):

$$|H|\frac{\partial L_1}{\partial \bar{A}} > 0 \text{ iif } \left(\frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^3}{\partial A^2}\right)\left(\frac{\partial^2 Y^2}{\partial L_2^2} + \frac{\partial^2 Y^3}{\partial L^2}\right) > \left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)^2$$

In this condition, a decrease in $\bar{A}$ (ndlr, a tightening of conservation efforts) results in a decrease in L_1. This result holds even if the conservation constraint is not slack. When PAs are implemented, land conversion from natural forests is relatively costlier and induces household labor reallocation towards other activities located in secondary forests. This result is summarized in the following proposition:

Proposition 1: An increase in PAs incites households to reallocate labor towards other activities under rapidly decreasing marginal productivity of labor on secondary forests.

Regarding the effect of a change on conservation efforts on deforestation leakages on $\bar{A}_2$, it follows from FONC that:

$$|H|\frac{\partial A_2}{\partial \bar{A}} = -\frac{\partial^2 Y^1}{\partial A_1 \partial L_1}\left(\frac{\partial^2 Y^3}{\partial A \partial L}\frac{\partial^2 Y^2}{\partial L_2^2} - \frac{\partial^2 Y^3}{\partial L^2}\frac{\partial^2 Y^2}{\partial A_2 \partial L}\right) \quad (2)$$

The sign of Equation 2 is *a priori* unknown without additional assumptions. Following Proposition 1, conservation efforts reallocate labor in favor of Y^2 and Y^3 which are obtained on secondary forested areas. The household is therefore facing a dilemma: either it chooses to increase rice production Y^2 which is detrimental to the the provision of local ES, or it chooses to increase the local ES provision which positively depends on secondary forest. The net effect of $\bar{A}$ on A_2 is thus dependent on the relative magnitude of decreasing marginal labor productivities. For Equation 2 to be negative, we indeed need to have:

$$|H|\frac{\partial A_2}{\partial \bar{A}} < 0 \text{ iff } \frac{\frac{\partial^2 Y^3}{\partial L^2}}{\frac{\partial^2 Y^2}{\partial L_2^2}} > \frac{\frac{\partial^2 Y^3}{\partial A \partial L}}{\frac{\partial^2 Y^2}{\partial A_2 \partial L}}$$

We synthesized this result in Proposition 2:

Proposition 2: An increase in surfaces under PAs does not necessarily induce deforestation leakages on secondary forests. This will occur only when the provision of local ES services is subjected to more rapidly decreasing marginal productivities of labor than rice production $(-\frac{\partial^2 Y^3}{\partial L^2} \gg -\frac{\partial^2 Y^2}{\partial L_2^2})$

The full derivations of propositions 1 and 2 as well as additional results are provided in Appendix B. In the following sections, we are going to test the empirical validity of the propositions related to the possibility of leakages from natural forests to anthropized forested areas summarized in propositions 1 and 2.

3 Data

To test our predictions, we build a spatially explicit dataset combining vegetation data, the last PAs' spatial census and socio-economic census surveys at the municipality level.

3.1 Vegetation Data: Defining Forests

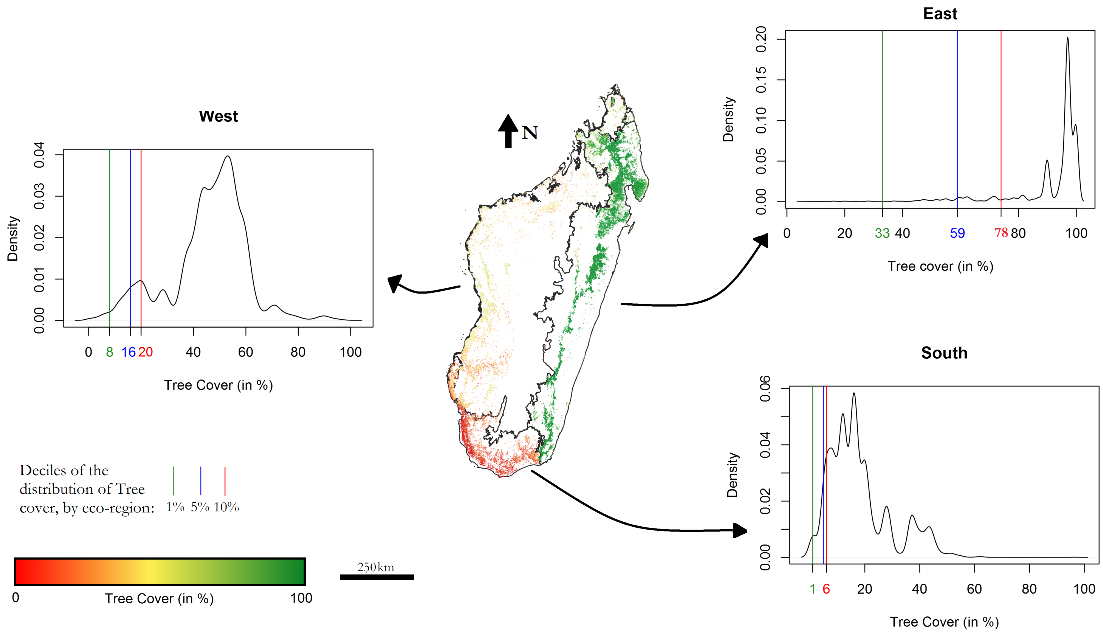
We want to track deforestation dynamics over different type of forested landscapes. We define three types of forested areas using vegetation data from Hansen et al. (2013) v1.0. We rely on two spatial layers: tree cover in 2000 and annual tree cover losses between 2001 and 2012. The Tree cover layer indicates for each pixel of 30m $\times$ 30m in 2000 the density of tree cover of a potential size higher than 5m. Values range from 0% to 100% of vegetation density cover for each pixel. The annual tree cover losses layer indicates whether a pixel has been deforested and if so, in what year.

Madagascar is divided into three ecologically diverse areas, each with its own forest and vegetation types. There are rainforests in the eastern ecoregion, savannah and dry forests in the western ecoregion and spiny forests in the southern ecoregion (Figure 2). Each natural habitat presents different level of tree cover density. To distinguish between natural habitats and anthropized habitats pixels, we overlap our vegetation layer with a map from the BioSceneMada project delimitating natural forests (Figure 2). This us to identify the minimal vegetation density within natural habitats. We find that southern spiny natural forests are dominantly composed of areas with a tree cover density of at least 6%, western dry natural forests are characterized by tree cover density of at least 20%, and eastern natural rainforests are composed of areas with tree cover density of at least 78%. From these thresholds, we define natural forest as the pixels that present at least this minimal tree cover density. We then define anthropized areas as the pixels with less than this minimal tree cover density.

For the purpose of this study, we focus on the eastern ecoregion in order to get a large enough spectrum of anthropized vegetation (< 78% of tree cover density per pixel). We define natural forests as the pixels with a tree cover density larger than or equal to 78%. We consider two strata of degraded forests: a first strata of highly degraded forest with a tree cover density between 30% and 53%, and a second strata of moderately degraded forest with a tree cover density between 54% and 77% (Figure 2). We do not include pixels with a tree cover density lower than 30% as doing so would require us to define the whole eastern ecoregion as forests, which is not the case[3]. Our geographical unit of analysis is municipalities. We thus aggregate pixel data within municipalities and compute the deforestation rate within the forest strata j in municipality i at date t as:

[3]The FAO for example may consider as forest a plot with a tree cover density from 10%. In such case, the plot should not have another land use, which would not be the case in the context of eastern Madagascar. Despite being arbitrary, 30% is also used in other studies, e.g. the Global Forest Watch platform.

Figure 2: Defining Forests for Each Ecoregion



Source: Authors. Forest Map from 2005 using Conservation International reference map of forests. Vegetation data from Hansen et al. 2013. Reference System: WGS84 - UTM38s.

$$Def_{ijt} = \left| \frac{Forest_{i,j,t} - Forest_{i,j,t-1}}{Forest_{i,j,t-1}} \right|$$

3.2 Treatment: Protected Areas

PAs aim to protect natural habitats of rich biodiversity and important carbon sinks in the context of REDD+. As such, PAs cover natural habitats but not degraded forests. PAs began to be established in Madagascar in 1927 and covered 1.7 million hectares in 2000. In 2003, an ambitious plan to triple the network of PAs was launched so that PAs now cover about 40% of the remaining natural forests in Madagascar. PAs and New PAs (NPAs) are all included in Madagascar's network of protected areas (*SAPM - Système des Aires Protégées de Madagascar*). Currently, there are 138 PAs in Madagascar. Fifty of them are the "historic" PAs created between 1927 and 1999. They are managed by the Madagascar National Parks public agency. The other 88 are NPAs.

In this study, we consider all PAs and NPAs from the eastern ecoregion that were officially included in the SAPM in 2012. Our database includes 24 historic PAs and 31 NPAs impacting 109 and 126 municipalities, respectively.

3.3 Control Variables

Finally, we construct a set of control covariates. First, we use census data from the ILO Cornell project (2001). The census includes information on economic, social and political characteristics for 1,385 of the country's 1,392 municipalities, known in Madagascar as *communes*. Particularly, we use information regarding the size of the population, the share of irrigated rice parcels (i.e., the alternative to slash and burn) and the share of impoverished people (people who face recurrent food shortage every years). These three variables will have a direct impact on deforestation. We spatialize the census and match it with official municipality' boundaries.

Second, we construct a set of bio-physical covariates for each municipality: average slope, average elevation, average Euclidean distance to nearest road (Deininger and Minten 2002). Within each municipality, we determine the same bio-physical measures for each of the three vegetation strata. To compute them, we draw a random sample of 60,000 pixels for each strata and average the values inside each municipality.

At the end, we have a sample of 688 municipalities of which 561 have a natural forest cover larger than 50 hectares[4]. Data sources and summary statistics are provided in Table 1.

4 Empirical Strategy

For each municipality, we consider the three main categories of forest cover: natural forest (more than 78% forest cover density), slightly degraded anthropized forest (forest cover between 54 and 78% density) and highly degraded anthropized forest (between 30 and 54% of forest cover density). We seek to determine if the establishment of a PA within a given municipality has decreased deforestation on natural forests but has caused an increase in deforestation on anthropized areas in the same municipality.

To avoid confounding effects, we control for other determinants of deforestation. These include both the location characteristics of forest cover (altitude, slope and distance to the nearest road of each type of forest within a municipality) and the characteristics of the municipality itself. For example, two natural forests with similar characteristics may have very different rates of deforestation if the municipal context is not the same. The context can be captured here by including demographic or agricultural pressures. The characteristics of the municipality allow for capturing contextual effects (Manski 1993). The hierarchical nature

[4]We drop municipalities with less than 50 hectares of natural forest cover as PAs aim at protecting large paths of remaining natural forests. Hence, including localities with almost no forest would bias estimates as these municipalities are structurally different from those still covered by important areas of natural forests.

of the determinants (forest strata and municipal) allows us to estimate a multilevel model. Multilevel models provide a natural framework for accounting for the correlation of observations within the municipalities. This correlation refers to correlated effects (Manski 1993), and according to Anselin (2002) and Wendland et al. (2011) they allow for taking into account the spatial correlation of the error term in municipalities. However, multilevel models assume that the explanatory variables and random effects are independent. If this assumption is violated the estimation results may be biased. This problem of correlation can be dealt with by a within (-group) transformation of the explanatory variables.

The econometric model has the following specification:

$$\begin{aligned} Def_{ijt} &= \beta_0 + \beta_1 Cover54_{ijt} + \beta_2 Cover30_{ijt} + \beta_3 PA_{jt} + \beta_4 Cover54_{ijt} \times PA_{jt} + \beta_5 Cover30_{ijt} \times PA_{jt} \\ &+ \beta_6 \left(X_{ijt} - \bar{X}_{ijt} \right) + \beta_7 \bar{X}_{ijt} + \beta_8 Z_j + \nu_t + \mu_j + \epsilon_i jt \end{aligned}$$

where index i represents the forest cover category, j refers to the municipality and t is a time index. Def_{ijt} is the deforestation rate of each forest cover category i in municipality j in year t, $Cover54_{ij}$ and $Cover30_{ij}$ are the fixed effects associated with forest cover. They are set to 1 when the forest cover is between 54 and 78% and between 30 and 54% respectively. PA_{jt} is a binary variable equal to 1 if the natural forest is protected in the municipality j in year t. X_{ijt} are location variables measured for each forest cover category i in municipality j at year t and Z_j are contextual variables. We crossed the protected area variable with forest cover categories ($Cover54_{ijt} \times PA_{jt}$ and $Cover30_{ijt} \times PA_{jt}$) in order to capture the leakage effect of deforestation of protected natural forests protected on secondary forests within municipalities. Parameters $\beta_0, \beta_1, \beta_2, \beta_3, \beta_4, \beta_5, \beta_6, \beta_7$ and β_8 are the regression coefficients to be estimated.

The error term is broken down into three parts. The first component $\nu_t \sim N(0, \sigma_{\nu^2})$ is specific to the year and constant between forest cover categories and municipalities. The second component is $\mu_j \sim N(0, \sigma_{\mu^2})$ which is specific to the municipality and constant between forest cover categories of the same municipality. The last component varies between year, forest cover categories and municipalities, and is designated as $\epsilon_{ijt} \sim N(0, \sigma_{\epsilon^2})$.

5 Empirical Results

Compared to natural forests, anthropized forests are found at a lower altitude and on flatter areas, closer to roads. In other words, Anthropized forests are in areas more prone to economic activities (Angelsen and Kaimowitz 1999) and consequently were the first to be converted into productive areas. However, the average annual deforestation rate on natural forests is significantly higher than the ones on already antrhopized forests as natural forests offer more fertile soils and provide less direct ES to locals. 18% of the observations were impacted by a PA over the period (Table 1). We now turn to the impact of PAs on deforestation and leakages.

Table 1: Descriptive Statistics

Forest strata regressors (level one, Obs= 5,886 /strata)	**Natural Forests**		**Slightly Degraded Forests**		**Highly Degraded Forests**	
	Mean	S.D.	Mean	S.D.	Mean	S.D.
Deforestation rate	0.0234	0.0683	0.00695***	0.0147	0.00217***	0.00560
Tree Cover	0.0795	0.175	0.0638***	0.0695	0.0402***	0.0406
Elevation	0.585	0.542	0.554***	0.552	0.546***	0.549
Slope	9.955	5.610	8.676***	4.348	7.884***	3.773
Distance to Nearest Road	5.504	5.422	5.205***	5.068	5.033***	5.034
Municipal level regressors (level two, Obs: 561)						
Share of municipalities with a PA at Year t	18.3	38.7				
Population in 2001	13,600	8730				
Share of Irrigated Paddy Rice	13.52	24.58				
Share of Impovrished People	8.847	12.96				
Average Elevation	0.562	0.520				
Average Slope	8.839	3.994				
Average Dist to Nearest Road	5.248	5.048				

Note: Superscripts *, ** and *** correspond to 10%, 5% and 1% levels of significance. Student test measuring differences between natural forests and slightly degraded forests, and between natural forests and highly degraded forests are presented. In addition, deforestation rates, tree cover and slope are statistically different at 1% between slightly and highly degraded forests, distance to nearest road is significant at 10% level and elevation is not statistcally different

The relevance of the multilevel model and the existence of contextual effects are first assessed with a Hausman test. The χ^2 statistic (Table 2) does not allow us to reject the null hypothesis of independence between errors and explanatory variables. It therefore favors multilevel modeling. Second, we implement a likelihood ratio test between the multilevel model without contextual effects (restricted model) and the full model with contextual effects (unrestricted model). The statistics of the likelihood ratio test allows us to reject the joint null hypothesis of contextual effects at the 1% level for all multilevel models. The portion of the total variability accounted for at the municipal level calculated as the intra-class correlation coefficient $\rho = {}^{\sigma^2_\mu} / {}_{\sigma^2_\mu + \sigma^2_\epsilon}$ is at least equal to 7%. In other words, at least 7% of the total variance in the deforestation rate is explained at the municipal level. These results justify the inclusion of both contextual and correlated effects.

Table 2 shows the results of the estimations. Three separate estimates have been run: the fixed effects model, the multilevel model with years' fixed effects and the full multilevel model with municipality and year random effects. All of these results show that the establishment of a protected natural forest contributes to a reduction in deforestation by at least 1.1%. According to Proposition 1, the effectiveness of PAs in reducing deforestation will lead to a reallocation of household labor time.Proposition 2 stipulates that a leakage effect of deforestation is not automatic. Here, the coefficients associated with multiplicative variables ($Cover30_{ijt} \times PA_{jt}$ and $Cover54_{ijt} \times PA_{jt}$) are positive and significant. This indicates that deforestation increases by about 1% in each of the two categories of anthropized forests that are located in the municipalities where the natural forest is protected. In other words, there is a leakage of deforestation from protected natural forests to anthropized forests in the same municipality. Based on Proposition 2, this leakage effect can partly be explained by more rapidly decreasing marginal productivity of labor in the provision of local ES over rice production.

Taken globally, because of the leakage effect, PAs seem ineffective or even counterproductive in terms of vegetation cover changes. Indeed, one can define the net impact of PAs on cover changes is the sum of avoided deforestation and the induced leakages. Tests on the sum of the coefficients ($\beta_3 + \beta_4 + \beta_5$) presented in Table 1 show that PAs generate a net increase of 1% in vegetation clearing. Indeed, as the anthropized forests are less fertile, households must clear more land for the same level of agricultural productivity. Therefore, the leakage of deforestation from natural forests to anthropized forests exceeds avoided deforestation. We would, however, need to convert these physical land use changes in economically more relevant values (ecosystem services, rice production) to get a broader view of the welfare impact of PAs (Vincent 2015).

Finally, our results show that forest cover, slope, elevation, distance from roads and the percentage of land devoted to rice production are associated with lower deforestation. For better efficiency, conservation policies must be supported by rural development policies in particular the promotion of agricultural intensification techniques (use of fertilizers or irrigation land), processing of agricultural products and the development of non-agricultural income generating activities.

5.1 Robustness Check: Evidence from Matching Techniques

The regression results may be biased if treated and non-treated municipalities do not have the same average deforestation rate. For example, if treated municipalities are intrinsically exposed to lower deforestation than those from the control group, traditional models would overestimate the impact of PAs (Ferraro 2009; Joppa and Pfaff 2009). Therefore, we need to find an acceptable counterfactual group to estimate an unbiased effect of protected areas and their leakage effects. We use a reweighting technique, entropy balancing (Hainmueller 2012), in order to ensure that the distribution of covariates is the same in the control group and in the treated group. Entropy balancing attains more effective results than propensity score or mahanolobis methods when reducing covariate imbalances. In practice in a first step, we

Table 2: Deforestation, Protected Areas and Leakages

	(1)		**(2)**		(3)	
	Full fixed effect		**Multilevel model with year fixed effects**		**Full multilevel model**	
	Estimates	Std. Err.	Estimates	Std. Err.	Estimates	Std. Err.
Level one regressors						
PA			-0.01265***	(0.00189)	-0.01167***	(0.00188)
Cover 30	-0.02557***	(0.00174)	-0.02557***	(0.00082)	-0.02558***	(0.00080)
Cover 54	-0.01978***	(0.00157)	-0.01978***	(0.00079)	-0.01977***	(0.00077)
Cover 30 x PA	0.01183***	(0.00183)	0.01183***	(0.00193)	0.01173***	(0.00189)
Cover 54 x PA	0.01062***	(0.00156)	0.01062***	(0.00191)	0.01053***	(0.00188)
Forest-C	-0.01048**	(0.00424)	-0.01047***	(0.00394)	-0.01122***	(0.00387)
Elevation-C	-0.01795***	(0.00674)	-0.01795***	(0.00214)	-0.01792***	(0.00210)
Slope-C	-0.00028	(0.00029)	-0.00028*	(0.00015)	-0.00028*	(0.00015)
Road-C	-0.00104*	(0.00058)	-0.00104***	(0.00028)	-0.00103***	(0.00028)
Municipal level regressors						
Forest-M			-0.01703**	(0.00828)	-0.02031**	(0.00837)
Elevation-M			0.01262***	(0.00121)	0.01263***	(0.00123)
Slope-M			-0.00049***	(0.00016)	-0.00054***	(0.00016)
Road-M			-0.00022*	(0.00013)	-0.00022*	(0.00013)
Population 2001			0.10156	(0.06344)	0.11338*	(0.06412)
Irrigated rice paddy			-0.00005**	(0.00002)	-0.00005**	(0.00002)
Impoverished people			0.00005	(0.00004)	0.00005	(0.00004)
σ_μ			0.01034***	(0.00043)	0.01023***	(0.00044)
σ_ν					0.00882***	(0.00067)
σ_ϵ			0.03776***	(0.00020)	0.03705***	(0.00024)
Municipality fixed effects	Yes					
Year fixed effects	Yes		Yes			
Constant	0.06385***	(0.00427)	0.23002***	(0.01278)	0.23002***	(0.01278)
Observations	17,658		17,658		17,658	
Hausman test						
χ^2 statistic			2.95		0.87	
p-value			1.000		0.999	
Net deforestation						
$\beta_3 + \beta_4 + \beta_5$			0.01005***		0.00979***	
Std. Err.			(0.00269)		(0.00272)	

Note: Superscripts *, ** and *** correspond to 10%, 5% and 1% levels of significance respectively. Standard errors in Model 1 are clustered within each municipality. Standard errors in the multilevel models account for potential correlations within municipalities. The difference between Model 2 and Model 3 consists in the way yearly variations are taken into account: Model 2 relies on year fixed effects, Model 3 treats years as a third level.

determine a set of entropy weights that allows us to match the means of the covariates in the treatment group with those in the reweighted control group. In a second step, we perform the multilevel model by weighting this regression with the entropy weights. The coefficient associated with the treatment variable can be considered as the average treatment effect. Table C.1 in Appendix C presents the mean of the distribution of covariates before and after the reweighting. This table shows that after the reweighting procedure, the control and treated group are very similar in terms of the observed characteristics. The estimation results after reweighting remain qualitatively unchanged (Table 3). We still observe a negative and significant effect of protected natural forest on deforestation and leakage of deforestation displaced to anthropized forests. The net effect of protected areas on deforestation is approximately 0.7% and positive.

6 Discussion

The literature has highlighted the possibility of leakages following the implementation of conservation restrictions and these leakages have been empirically found in a few studies. In this present work, we have extended the conception of leakages by studying deforestation displacement from protected natural forests to already anthropized forested areas.

A large body of literature is devoted to studying the opportunity cost of conservation policies for local inhabitants (Shyamsundar and Kramer 1996; Ferraro 2002). Studying leakages with an ES framework allows us to cast a new look at this question by highlighting the possibilities of farmers' coping strategies in response to the creation of PAs. In our approach, the local cost of conservation does not correspond to the raw income losses on now protected lands but rather is a loss net of leakages. Leakages themselves have a dual composition: on the one hand, locals minimize the raw income loss following PAs' creation by harvesting agricultural commodities on previously anthropized lands, but on the other hand, they lose the direct provisioning ES these lands provided. Our results, nonetheless, confirm a salient point of the literature through an ES approach: an important part of the cost of protecting the provision of global ES is borne by locals and subsequent conservation policies may displace deforestation onto lands that provide locals important provisioning ES.

The types of leakages we capture in our study probably corresponds to a short-term adjustment of farmers' behavior. Farmers first reap economic profit in their neighborhoods by continuing to clear some (but less) natural forest despite prohibitions, and by reporting some deforestation on anthropized forested areas. Once these economic opportunities are exploited, we can expect mid- to long-term adjustments possibly involving migration or the

Table 3: Matching with reweighting

	(1) Multilevel model with year fixed effects		(2) Full multilevel model	
	Estimates	Std. Err.	Estimates	Std. Err.
Level one regressors				
PA	-0.01071***	(0.00150)	-0.00960***	(0.00145)
Cover 30	-0.02083***	(0.00157)	-0.02087***	(0.00270)
Cover 54	-0.01530***	(0.00132)	-0.01533***	(0.00213)
Cover 30 x PA	0.00981***	(0.00160)	0.00982***	(0.00149)
Cover 54 x PA	0.00825***	(0.00133)	0.00825***	(0.00137)
Forest-C	-0.01125***	(0.00389)	-0.01148***	(0.00157)
Elevation-C	-0.01604***	(0.00488)	-0.01600***	(0.00536)
Slope-C	-0.00010	(0.00020)	-0.00011	(0.00012)
Road-C	-0.00000	(0.00035)	0.00000	(0.00009)
Municipal level regressors				
Forest-M	-0.00929**	(0.00450)	-0.00846***	(0.00097)
Elevation-M	0.00674***	(0.00169)	0.00439***	(0.00082)
Slope-M	-0.00027***	(0.00010)	-0.00016***	(0.00005)
Road-M	-0.00022***	(0.00008)	-0.00018***	(0.00005)
Population 2001	-0.01495	(0.03857)	-0.01875**	(0.00847)
Irrigated rice paddy	-0.00003*	(0.00002)	-0.00003***	(0.00000)
Impoverished people	0.00005*	(0.00003)	0.00003***	(0.00001)
σ_μ	0.00001	(0.00001)	0.07446	(0.08406)
σ_ν			0.05945**	(0.02130)
σ_ϵ	0.00047**	(0.00021)	0.00037***	(0.00016)
Year fixed effect	Yes			
Constant	0.02228***	(0.00228)	0.03810	(0.10549)
Observations	17658		17658	
Net deforestation				
$\beta_3 + \beta_4 + \beta_5$	0.00734***		0.00846***	
Std. Err.	(0.00152)		(0.00145)	

Note: Superscripts *, ** and *** correspond to 10%, 5% and 1% levels of significance. The difference between Model 1 and Model 2 consists in the way yearly variations are taken into account: Model 1 relies on year fixed effects, Model 2 treats Years as a third level.

intensification of current agricultural techniques (*eg*, terrace cultivation).

6.1 Distinguishing between Anthropized Forests

In order to study the displacement of deforestation from protected natural forests to anthropized habitats, we use vegetation data and distinguish the two types of habitats based on differences in their tree cover density. In our case, natural rainforests present a higher percentage of tree cover density than anthropized habitats. We set up a threshold between natural and anthropized areas (*ed*, 78% of tree cover density) using official forest maps. Then, in order to differentiate between slightly degraded and highly degraded habitats, we cut our anthropized forested areas (30% to 77%) into two equal parts (30% to 53% and 54% to 77%). However, in doing that, we can't precisely determine the composition of the two strata of anthropized forested areas, i.e. eucalyptus or pine plantation, agri-forestry systems, old fallows etc. Official land use maps would have allowed us to distinguish between each land use. Yet no such recent and precise maps are available for Madagascar.

An alternative to official land use maps would have been to conduct some ground checking for the vegetation data that we have: for each category of tree cover density, we could go observe to which they correspond. Nonetheless, Hansen et al. (2013) vegetation data provide percent of tree cover density only for the year 2000. Field observation 15 years later would clearly not informative. Therefore, we believe there is still room for future research to better understand the interaction between conflicting land use with the help of land use maps, perhaps in the context of a country with richer data than Madagascar.

6.2 Understanding the Absence of Agricultural Intensification in Madagascar

Whether forests are protected or not, slash-and-burn agriculture on natural forests in Madagascar is illegal[5]. Yet it remains the principal agricultural technology used and few farmers have invested in new practices to comply with the law. In conservationists' theory of change relative to the creation of PAs, a Porter-like hypothesis (Porter and Van der Linde 1995) is implicit: new legal restrictions will incite farmers to invest in order to comply with the rule (Phalan et al. 2011). Yet, the adoption of innovations has remained extremely limited in Madagascar (Moser and Barrett 2003) and our results show that people have chosen to displace deforestation and continue the practice of slash and burn even on anthropized forested areas, despite the fact that it it might impede the provision of local ES and despite lower

[5]Actually, some permits may be accorded to farmers to clear forests. However, permits are never requested by locals making the clearing of forest illegal.

yields. Understanding the reasons why little agricultural innovation has occurred in Madagascar remains a key policy challenge.

The economic literature has pointed out several factors that limit the adoption of new practices in the agricultural domain. They are the lack of access to credits and information, insufficient human capital and aversion to risk among others (Feder, Just, and Zilberman 1985; Abdulai and Huffman 2005; Moser and Barrett 2006; Foster and Rosenzweig 2010). Among these factors, risk aversion appears as a key explanation in the specific context of Madagascar's eastern ecoregion (Brimont et al. 2015). Madagascar is among the countries most vulnerable to climate risks and high levels of poverty make Malagasy farmers particularly risk-averse (Nielsen 2001). Even if it provides low yields, slash and burn on uplands is an efficient risk-management strategy to cope with climatic vulnerability (Aubert, Razafiarison, and Bertrand 2003; Delille 2011) as opposed to many alternative practices.

Recent findings have also highlighted the role of social interactions in the adoption of new agricultural practices (Bandiera and Rasul 2006; Conley and Udry 2010). This stream of literature somehow echoes the findings of Moser and Barrett (2006) that underline the research of social conformity as a determinant of technological adoption (or non-adoption) of agricultural innovations in Madagascar. The authors argue that people may adopt (or not) an innovation so as to comply with the group's social norms. As slash and burn is at the basis of the social organization of forests community in Madagascar, it serves as a mechanism to establish the elders' authority in collective decision processes and as a support of cultural and religious rites (Aubert, Razafiarison, and Bertrand 2003; Desbureaux and Brimont 2015). Conformity with the dominant social norm then requires local farmers to continue the practice of slash and burn instead of adopting alternative practices.

In order to reduce deforestation leakages from protected natural forests to anthropized ones, it might be useful to put in place policies for intensification of agricultural production oriented toward peers (elderly, traditional and religious authorities) and based on the access to credit, training and insurance to manage risks. Indeed, a policy supported by the peers would help to break down social barriers. Access to credit and training would enable populations to switch to more intensive techniques.

7 Concluding Remarks

This paper presents evidence that deforestation leakages have constituted a significant adaptation strategy of Malagasy farmers to the creation of PAs in the eastern rainforest. We use fine scale vegetation satellite images to distinguish natural forests from already anthropized forested areas and construct a panel dataset of annual deforestation between 2000 and 2012 for each vegetation type. We found that PAs contributed to decrease deforestation of natural forests in Madagascar. This decrease, however, has resulted in an increase in deforestation of already anthropized forested areas despite the fact that anthropized forested areas provide crucial provisioning ecosystem services to locals' inhabitants. This result confirms the idea through a ecosystem services approach that locals bear an important cost of conservation policies. In addition, this deforestation leakage contradicts a Porter hypothesis reasoning which suggests that a change in institutional rules, herethe creation of PAs, should have increased the adoption of new more land-intensive agriculture practices. Risk aversion in a context of a highly risky environment and collective action dynamics appear as a possible explanation for the absence of adoption of alternate agricultural practices.

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8 Appendix

A: Rice Plantation After Eucalyptus

Figure 3: Rice cultivation after eucalyptus plantation

Source: S. Desbureaux. Eucalyptus is known for impoverishing and acidifying soils making them unsuitable for agriculture. Yet, our field visits suggest that some Malagasy farmers clear eucalyptus plantations to grow rice.

B: Maths Appendices

B1: Second Order Necessary Conditions

The second order sufficient condition for a maximization problem with four decision variables subjected to one constraint depends on the properties of the bordered hessian matrix H. From FONC, H is such that:

$$H = \begin{pmatrix} \pi_{11} & 0 & \pi_{13} & 0 & -1 \\ 0 & \pi_{22} & \pi_{23} & \pi_{24} & 0 \\ \pi_{13} & \pi_{23} & \pi_{33} & \pi_{34} & 0 \\ 0 & \pi_{24} & \pi_{34} & \pi_{44} & 0 \\ -1 & 0 & 0 & 0 & 0 \end{pmatrix}$$

where $\pi_{11} = \frac{\partial^2 Y^1}{\partial A_1^2} < 0$; $\pi_{22} = \frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^3}{\partial A^2} < 0$; $\pi_{33} = \frac{\partial^2 Y^1}{\partial L_1^2} + \frac{\partial^2 Y^3}{\partial L^2} < 0$; $\pi_{34} = \frac{\partial^2 Y^3}{\partial L^2} < 0$;

$\pi_{44} = \frac{\partial^2 Y^2}{\partial L_2^2} + \frac{\partial^2 Y^3}{\partial L^2} < 0$. Moreover, $\pi_{13} = \frac{\partial^2 Y^1}{\partial A_1 \partial L_1} > 0$; $\pi_{23} = \frac{\partial^2 Y^3}{\partial A \partial L} > 0$ and $\pi_{24} = \frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L} < 0$.

The sufficient condition for a maximum is that the principal minors M_k of order k (which have $k+1$ rows and columns), $k = 2, 3$ and 4, alternate in sign with $M_4 = |H| > 0$. Under the assumption that π is quasi-concave the sufficient second order condition for a maximization is fulfilled (Arrow & Enthoven 1961).

B2: Full derivation of propositions 1 and 2 and additional results

Generally speaking the effect of $\bar{A}$ on all endogenous variables is obtained while solving the following system:

$$H \begin{pmatrix} \frac{\partial A_1}{\partial \bar{A}} \\ \frac{\partial A_2}{\partial \bar{A}} \\ \frac{\partial L_1}{\partial \bar{A}} \\ \frac{\partial L_2}{\partial \bar{A}} \\ \frac{\partial \mu}{\partial \bar{A}} \end{pmatrix} = \begin{pmatrix} 0 \\ 0 \\ 0 \\ 0 \\ 0 \\ -1 \end{pmatrix}$$

It can be calculated that $|H| = - \begin{vmatrix} f_{22} & f_{23} & f_{24} \\ f_{23} & f_{33} & f_{34} \\ f_{24} & f_{34} & f_{44} \end{vmatrix} > 0.$

B2-1: Effect of an increase in $\bar{A}$ on A_1

The effect is obtained while calculating:

$$\frac{\partial A_1}{\partial \bar{A}} = \frac{\begin{vmatrix} 0 & 0 & \pi_{13} & 0 & -1 \\ 0 & f_{22} & \pi_{23} & \pi_{24} & 0 \\ \pi_{13} & f_{23} & \pi_{33} & \pi_{34} & 0 \\ 0 & \pi_{24} & \pi_{34} & \pi_{44} & 0 \\ -1 & 0 & 0 & 0 & 0 \end{vmatrix}}{|H|}$$

It can be established: $\frac{\partial A_1}{\partial \bar{A}} = 1$.

This result is quite straightforward. An increase in the conservation effort induces a 100% decrease in cleared natural forests even if the conservation constraint is slack. This results holds under the proviso that protected areas are effectively implemented. This is obviously not the case in Madagascar. This institutional feature may be easily handled while slightly modifying the conservation constraint as $A_1 \leq g(\bar{A})$ with $g^{'} > 0$ and $g^{'} \leq 0$ as suggested in

(Delacote & Angelsen 2015).

B2-2: Effect of an increase in $\bar{A}$ on L_1 (Proposition 1)

The effect is obtained while calculating:

$$\frac{\partial L_1}{\partial \bar{A}} = \frac{\begin{vmatrix} \pi_{11} & 0 & 0 & 0 & -1 \\ 0 & \pi_{22} & 0 & \pi_{24} & 0 \\ \pi_{13} & \pi_{23} & 0 & \pi_{34} & 0 \\ 0 & \pi_{24} & 0 & \pi_{44} & 0 \\ -1 & 0 & -1 & 0 & 0 \end{vmatrix}}{|H|}$$

It comes that $|H|\frac{\partial L_1}{\partial \bar{A}} = \frac{\partial^2 Y^1}{\partial A_1 \partial L_1}\left(\left(\frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^3}{\partial A^2}\right)\left(\frac{\partial^2 Y^2}{\partial L_2^2} + \frac{\partial^2 Y^3}{\partial L^2}\right) - \left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)^2\right) > 0$ iif $\left(\frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^3}{\partial A^2}\right)\left(\frac{\partial^2 Y^2}{\partial L_2^2} + \frac{\partial^2 Y^3}{\partial L^2}\right) > \left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)^2$. This property reads as when "own-effects dominate cross effects" (Silberberg 1990, p.164), then the expression in parentheses is positive.

B2-3: Effect of an increase in $\bar{A}$ on A_2 (Proposition 2)

$$|H|\frac{\partial A_2}{\partial \bar{A}} = \frac{\partial^2 Y^1}{\partial A_1 \partial L_1}\left(\frac{\partial^2 Y^3}{\partial A \partial L}\frac{\partial^2 Y^2}{\partial L^2} - \frac{\partial^2 Y^3}{\partial L^2}\frac{\partial^2 Y^2}{\partial A_2 \partial L}\right)$$

$\frac{\partial A_2}{\partial \bar{A}} < 0$ (leakage) iif $\frac{\partial^2 Y^3}{\partial A \partial L}\frac{\partial^2 Y^2}{\partial L^2} - \frac{\partial^2 Y^3}{\partial L^2}\frac{\partial^2 Y^2}{\partial A_2 \partial L} > 0$ that is $\frac{\partial^2 Y^3}{\partial L^2} / \frac{\partial^2 Y^2}{\partial L_2^2} > \frac{\partial^2 Y^3}{\partial A \partial L} / \frac{\partial^2 Y^2}{\partial A_2 \partial L}$. In other words, providing $-\frac{\partial^2 Y^3}{\partial L^2}$ is "sufficiently" high with respect to $-\frac{\partial^2 Y^2}{\partial L^2}$, i.e., $-\frac{\partial^2 Y^3}{\partial L^2} \gg -\frac{\partial^2 Y^2}{\partial L^2}$, such that $\frac{\partial^2 Y^3}{\partial L^2} / \frac{\partial^2 Y^2}{\partial L_2^2} > \frac{\partial^2 Y^3}{\partial A \partial L} / \frac{\partial^2 Y^2}{\partial A_2 \partial L}$. It follows that an increase in the conservation effort represented by a decrease in $\bar{A}$ induces an increase in land converted from secondary forests A_2.

B2-4: Effect of an increase in $\bar{A}$ on L_2

The intuitive result holds:

$$sign \frac{\partial L_2}{\partial \bar{A}} = sign - \frac{\partial L_1}{\partial \bar{A}}$$

with:

$$\frac{\partial L_2}{\partial \bar{A}} = -\frac{\partial^2 Y^1}{\partial A_1 \partial L_1}\left(\left(\frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^2}{\partial A^2}\right)\frac{\partial^2 Y^3}{\partial L^2} - \frac{\partial^2 Y^3}{\partial A \partial L}\left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)\right)$$

Using the same kind of arguments than for the effect of $\bar{A}$ on L_1, it can be established that

$\left(\frac{\partial^2 Y^2}{\partial A_2^2} + \frac{\partial^2 Y^2}{\partial A^2}\right)\frac{\partial^2 Y^3}{\partial L^2} > \frac{\partial^2 Y^3}{\partial A \partial L}\left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)$ and therefore $\frac{\partial L_2}{\partial \bar{A}} < 0$ [6].

B2-5: Effect of an increase in $\bar{A}$ on μ

At last, it is shown that:

$$\frac{\partial \mu}{\partial \bar{A}} = \frac{\begin{vmatrix} \pi_{11} & 0 & \pi_{13} & 0 \\ 0 & \pi_{22} & \pi_{23} & \pi_{24} \\ \pi_{13} & \pi_{23} & \pi_{33} & \pi_{34} \\ 0 & \pi_{24} & \pi_{34} & \pi_{44} \end{vmatrix}}{|H|}$$

The numerator is the determinant of the hessian of the net return function. If π is assumed to be strictly concave then the numerator is negative and therefore $\frac{\partial \mu}{\partial \bar{A}} < 0$. The net agricultural return function is concave in $\bar{A}$.

C: Empirical Additional Details

Table 4: Entropy Matching Sample Mean

	Before-Reweighting		After-Reweighting	
	Treat	Control	Treat	Control
Level one regressors				
Forest-C	3.38E-10	0.0002504	3.38E-10	3.38E-10
Elevation-C	1.84E-09	-1.03E-09	1.84E-09	1.84E-09
Slope-C	-1.77E-09	-3.91E-09	-1.77E-09	-1.77E-09
Road-C	1.51E-08	7.13E-09	1.51E-08	1.51E-08
Municipal level regressors				
Forest-M	0.1089	0.0509	0.1089	0.1089
Elevation-M	0.6480	0.5430	0.6480	0.6480
Slope-M	10.9500	8.3850	10.9500	10.9500
Road-M	6.6260	4.9570	6.6260	6.6260
Population 2001	0.0116	0.0141	0.0116	0.0116
Irrigated rice paddy	15.0900	13.1700	15.0900	15.0900
Destitute people	6.1500	9.4500	6.1500	6.1500

[6] Again, the same argument allows establishing the ancillary result: $\frac{\partial L_2}{\partial \bar{A}} + \frac{\partial L_1}{\partial \bar{A}} = \frac{1}{|H|}\frac{\partial^2 Y^1}{\partial A_1 \partial L_1}\left(\left(\frac{\partial^2 Y^2}{\partial A^2} + \frac{\partial^2 Y^3}{\partial A^2}\right)\frac{\partial^2 Y^2}{\partial L_2^2} - \frac{\partial^2 Y^2}{\partial A_2 \partial L}\left(\frac{\partial^2 Y^2}{\partial A_2 \partial L} + \frac{\partial^2 Y^3}{\partial A \partial L}\right)\right) > 0$. It evidences the reallocation of labor, say $\frac{\partial L_1}{\partial \bar{A}} > -\frac{\partial L_2}{\partial \bar{A}}$. The intensity of decreasing marginal productivities on land and labor for Y^2 and Y^3 allows gauging the magnitude of labor reallocation towards Y^2 and Y^3.