Land cover dynamics in Panama: Local lessons addressing global challenges in the context of REDD+

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A Evelyn por todo su amor e incondicional apoyo y compañía,

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List of Abbreviations

AGB	Above-Ground Biomass
ANAM	Autoridad National del Ambiente
ANOVA	Analysis of Variance
BAU	Business-As-Usual
BEF	Biomass Expansion Factor
C	Carbon
CEF	Centre of Forest Research
CO_2	Carbon dioxide
COP	Conference of Parties
CV	Coefficient of Variation
DBH	Diameter at Breast Height
FAO	Food and Agriculture Organization
FCD	Forest Carbon Density
FRA	Forest Resource Assessment
GCP	Ground Control Point
GEC3	Global Environmental and Climate Change Centre
GHG	Greenhouse Gas
GOFC-GOLD	Global Observation for Forest and Land Cover Dynamics
GPS	Global Positioning System
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
KP	Kyoto Protocol
LUCC	Land-Use and Cover Change
MRV	Measuring, Reporting and Verifying
NA	Not Available
PES	Payment for Environmental Services
REDD+	Reducing Emissions from Deforestation and forest Degradation in developing countries (including conservation, sustainable management of forest and forest carbon enhancement)
REL	Reference Emission Levels
RL	Reference Level
RS	Remote Sensing
SBSTA	Subsidiary Body on Scientific and Technical Advice
SD	Standard Deviation
SINAP	Sistema Nacional de Áreas protegidas
STRI	Smithsonian Tropical Research Institute
UNFCCC	United Nations Framework Convention on Climate Change
UN-REDD	United Nations REDD Initiative
WB	World Bank
WD	Wood Density

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Preface

This is a manuscript-based thesis consisting of a collection of papers of which I am the primary author. Chapters 1 and 2 have already been published; Chapter 3 has been submitted for publication. All chapters have been formatted in the style of the scientific journal Conservation Letters. The manuscripts and associated journals are as follows:

Chapter 1

Vergara-Asenjo, G. & Potvin, C. 2014. Forest protection and tenure status: the key role of Indigenous Peoples and protected areas in Panama. Global Environmental Change 28, 205-215.

Chapter 2

Vergara-Asenjo, G., Sharma, D., Potvin, C. 2015. Engaging stakeholders: Assessing accuracy of participatory mapping of land cover in Panama. Conservation Letters, doi:10.1111/conl.12161

Chapter 3

Vergara-Asenjo, G., Potvin, C. 2016. Nothing to lose? The costs of deforestation and forest invasion in Eastern Panama. Submitted to Conservation Letters.

Contributions of Authors

I am the primary author of all the studies included in this thesis. I formulated the hypotheses, proposed the methods, collected all the data (sometimes in collaboration with others; see below), analyzed the data and wrote the manuscripts. Catherine Potvin supervised the conceptual framework, methods, interpretation of the results and writing of all manuscripts in this thesis.

For Chapter 1, Mr. German Hernandez collaborated in collecting information from COONAPIP, together with all of the regional and traditional indigenous authorities in Panama who authorized and participated in this study. Aerin Jacob (McGill University), Sebastien Jodoin (Yale University) and Dr. William F.J. Parsons (Centre d'étude de la forêt) also contributed useful comments and reviewed the manuscript.

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For Chapter 3, Javier Mateo-Vega provided me with forest plot information in the area of comarca Madungandi.

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Thesis Abstract

Forest ecosystems play a critical role as carbon sinks at the global scale. Rapid land use/cover change (LUCC) and the large source of greenhouse gas emissions associated with tropical deforestation established the argument to create the first global mechanism to combat climate change using the forestry sector. During the 16^{th} session of the Conference of the Parties (COP-16) to the UNFCCC in 2010 in Cancun, Mexico, the policy framework for Reducing Emissions from Deforestation and Forest Degradation (REDD+) was officially established. For the first time, developing countries might be compensated for their efforts in either reducing carbon dioxide emissions from the forestry sector or increasing forest carbon stocks. In the context of REDD+, land tenure and tenure security have emerged as critical concepts in achieving forest conservation and securing biodiversity and local livelihoods. Using Panama as a case study of complex land tenure dynamics in the context of forest conservation through REDD+ implementation, the present research improved the understanding of tenure as a key factor of land cover change. I determined the efficiency of protected areas and indigenous territories in avoiding deforestation, presenting a novel statistical method to demonstrate their additionality in reducing deforestation through the pairing of comparable areas with respect to remoteness, topography or other relevant characteristics. The results are of great relevance for REDD+ strategies, showing that matching is a scientifically-sound way to quantify the contribution of each jurisdiction to emissions avoidance. In order to improve forest classification in a way that promotes the full and effective participation of Indigenous Peoples and local communities in monitoring activities, I demonstrated that local knowledge can improve land cover classification and facilitate the identification of forest degradation. The UNFCCC's call for the full and effective participation of local and Indigenous Peoples could, therefore, improve the accuracy of monitoring in MRV systems of REDD+. These findings are complemented by an analysis of land invasions in indigenous territories as a key element of the rapid expansion of the agricultural frontier, which shows that tenure alone is not enough to guarantee rights over land and forests in indigenous territories. I stressed that REDD+ strategies among developing countries can improve forest conservation and secure livelihoods on-the-ground through conflict resolution mechanisms and mediation processes that provide an avenue to resolve long-standing land conflicts.

Résumé

Les écosystèmes forestiers jouent un rôle essentiel en tant que puits de carbone à l'échelle mondiale. Les changements rapides de l'occupation et de l'utilisation du territoire et la grande source d'émission de gaz à effet de serre liée à la déforestation tropicale ont mené à la création du premier mécanisme mondial pour lutter contre les changements climatiques par le biais du secteur forestier. Au cours de la seizième session de la Conférence des Parties à la CCNUCC en 2010 à Cancun, au Mexique, le cadre politique de la réduction des émissions issues de la déforestation et de la dégradation des forêts (REDD +) a été officiellement créé. Pour la première fois, les pays en voie de développement pourraient être compensés pour leurs efforts visant soit à réduire des émissions de dioxyde de carbone du secteur forestier ou à augmenter les stocks de carbone forestier. Dans le cadre de la REDD +, le régime foncier et la sécurité foncière sont devenus des concepts essentiels dans la réalisation de la conservation des forêts et de la protection de la biodiversité et des moyens de subsistance locaux. En utilisant le Panama comme une étude de cas des dynamiques foncières complexes dans le cadre de la conservation des forêts à travers la mise en œuvre de la REDD +, la présente recherche a amélioré la compréhension du régime foncier en tant que facteur clé du changement de l'occupation du sol. J'ai déterminé l'efficacité des aires protégées et des territoires autochtones dans la déforestation et j'ai présenté une nouvelle méthode statistique pour démontrer leur additionnalité dans la réduction de la déforestation à travers le jumelage de zones comparables en termes d'éloignement, de topographie et d'autres caractéristiques pertinentes. Les résultats sont d'une grande importance pour les stratégies REDD +, montrant que l'appariement est une manière scientifiquement solide pour quantifier la contribution de chaque juridiction pour éviter des émissions. Afin d'améliorer la classification des forêts d'une manière qui favorise la participation pleine et effective des peuples autochtones et des communautés locales dans les activités de surveillance, j'ai démontré que les connaissances locales peuvent améliorer la classification de l'occupation du territoire et faciliter l'identification de la dégradation des forêts. L'appel de la CCNUCC à la participation pleine et effective des populations locales et autochtones pourrait, par conséquent, améliorer la précision des systèmes de surveillance systèmes de surveillance, de suivi et vérification de REDD +. Ces résultats sont complétés par une analyse des invasions territoires autochtones comme un élément clé de l'expansion rapide de la frontière agricole, ce qui montre que le régime seul ne

suffit pas à garantir les droits sur les terres et les forêts dans les territoires autochtones. J'ai souligné que les stratégies REDD + entre les pays en voie de développement peuvent améliorer la conservation des forêts et protéger des moyens de subsistance sur le terrain par le biais de mécanismes de résolution des conflits et les processus de médiation qui fournissent un moyen de résoudre des conflits fonciers de longue date.

General Introduction

Forests are the dominant terrestrial ecosystem on Earth, covering 31% of global land area (Pan *et al*. 2015, Keenan *et al* 2015). They play a critical role in the global carbon cycle, are important refuges for terrestrial biodiversity and are a source of essential goods and ecosystem services for human well-being (Millennium Ecosystem Assessment 2005). According to the World Bank (2004), 1.6 billion people – more than 25 per cent of the world's population – rely on forest resources for their livelihoods.

Despite the fact that forests play a key role for humanity, the global extent of the world's forests has declined at alarming rates. Forests have completely disappeared in 25 countries, and another 29 countries have lost more than 90% of their forest cover (Millennium Ecosystem Assessment 2005). Overall, there was a net decrease in global forest area of 3% between 1990 and 2015, from 4,128 M ha to 3,999 M ha, with natural and human-induced deforestation, which represents an area equivalent to South Africa (Keenan *et al*. 2015). Notwithstanding, many countries have realized successful efforts to tackle deforestation in recent years. In Brazil, the net rate of forests lost between 2010 and 2015 decreased by 60% from the 1990s rate, while in Indonesia the net rate of loss has also dropped by two thirds, from 1.9 M ha y^{-1} in the 1990s to 684 K ha y^{-1} from 2010 to 2015. In Mexico the net rate of loss has halved from 190 K ha y^{-1} to 92 K ha y^{-1} between 2010 to 2015 (Keenan *et al.* 2015).

Because forests exchange carbon dioxide (CO_2) with the atmosphere, they contribute to climate change mitigation and are substantial carbon sinks (Pan *et al*. 2011). Five main carbon pools are typically identified in forests: aboveground and belowground living biomass, leaf litter, woody debris and soil organic carbon (IPCC 2003). Natural processes and human disturbances are responsible for CO_2 emissions from forest ecosystems. Land-use/cover change (LUCC) is the second largest anthropogenic source of CO_2 emissions worldwide after the burning of fossil fuels. LUCC accounts for roughly 15% of global greenhouse gas emissions (van der Werf *et al*. 2009, IPCC 2007). Deforestation is estimated responsible for some 90% of the emissions caused by LUCC; it will be necessary to reduce deforestation to stabilize climate change (IPCC 2001). Deforestation also affects climate, biodiversity and other ecosystem services. On average, tropical forests hold around 50% more carbon per hectare than forests outside the tropics

(Houghton 2005). Consequently, equal deforestation areas will cause more CO_2 emissions in tropical regions than in other regions of the world.

According to Houghton (2003), deforestation can be permanent through the conversion of forests to croplands and pastures, or temporary through the partial removal of forests for shifting cultivation and selective logging. However, under the United Nations Framework Convention on Climate Change (UNFCCC), only permanent removal is recognized as deforestation (UNFCCC 2001). Forest degradation represents a direct anthropogenic decrease in carbon stocks, with forest defined as measured canopy cover remaining above the threshold (GOFC-GOLD 2015).

The choice of forest definition can have a large impact on estimates of deforestation (GOFC-GOLD 2015). Forest definitions vary widely in terms of tree size, area and canopy density (Chazdon *et al.* 2016). The most common definition defines forest as a "land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10%" (FAO 2010). FAO's definition, agreed on by all its members, is the first to be used by all countries for harmonized reporting; the definition adopted by FAO remains the most widely used forest definition today (Grainger 2008). Here, I use FAO's definition of forests where non-native species in forest plantations can also be considered forests.

LUCC is driven by a combination of synergetic factors such as pressures on resources, opportunities created by markets, policy intervention, vulnerability and social organization (Lambin *et al.* 2003). Among these underlying causes, the effect of poverty, population growth, economic development, insecure land tenure and weak law enforcement, among others, have been analyzed (Sunderlin *et al*. 2008, Rudel *et al*. 2005, Stern 2006).

A wealth of studies forms a comprehensive understanding of drivers of deforestation and forest degradation in developing countries (Geist and Lambin 2001, Lambin *et al*. 2003, Rudel *et al.* 2005, Kissinger *et al.* 2012). These studies recognize proximate or direct causes of deforestation as specific human activities at the local level that originate from intended land use and directly impact forest cover (e.g. agricultural expansion). In addition, underlying or indirect driving forces are social processes that affect the proximate causes at the local level or have an indirect impact from the national or global level (demographic, economic, cultural, policy and institutions and technological factors) (Geist and Lambin 2002). Several studies identify agricultural expansion as the major proximate cause of tropical deforestation, particularly the production of commercial commodities such as rubber, palm oil, cattle, soybean, coffee and cocoa (Fearnside 2001, Gibbs *et al*. 2010). Together, these agricultural activities account for three-quarters of all tropical deforestation (IPCC 2007).

Forests and climate change mitigation

In 2006 the Stern Review (Stern 2006) proposed that reducing deforestation is the “single largest opportunity for cost-effective and immediate reductions of carbon emissions” in the context of climate change. It stimulated great international interest in promoting international negotiations under the UNFCCC. In 2005, international negotiations began to aim to create the first global mechanism to combat climate change using the forestry sector (Pistorius 2012). During the 16th UNFCCC Conference of the Parties (COP-16) in 2010 in Cancun, Mexico, the policy framework for Reducing Emissions from Deforestation and forest Degradation (REDD+) was officially established. In addition to activities to avoid deforestation and forest degradation, REDD+ also includes conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries.

REDD+ is essentially a way to financially reward developing countries for their verified efforts to reduce emissions and enhance removals of greenhouse gases. REDD+ could mobilize public and private finance and form part of an international carbon trading regime to achieve large emission reductions (Streck 2012). In 2009, developed countries pledged USD 3.5 billion in REDD+ fast-start finance and the pledge has increased throughout 2011 to USD 4.17 billion (Simula 2010). It is presumed that these unprecedented levels of funding towards forest conservation will promote biodiversity conservation as well as poverty alleviation of forest-dependent people, by means of carbon markets (Kindermann *et al*. 2008, Laurance 2007). Studies have emphasized risks for local communities and Indigenous Peoples posed by, among others, possible recentralization of forest management as well as the stimulus of corruption and elite capture (Clements 2010, Hansen *et al*. 2009, Phelps et al. 2010, Potvin *et al*. 2007). In response to concerns over the potential for misuse and misappropriation, REDD+ countries must provide information on ‘‘safeguards’’ to address a range of environmental and social issues, including respect for indigenous and local communities, public participation and protection of biodiversity (UNFCCC 2011).

Tenure and forest conservation

Land tenure and tenure security are critical in achieving forest conservation and ensuring potential success of REDD+ strategies (Cotula and Mayers 2009, Sunderlin *et al.* 2009, Robinson *et al*. 2011, Larson *et al.* 2013). Tenure security – defined as the certainty that a community's land rights will be recognized and protected if challenged (Gray *et al*. 2015) – is one of the most prominent underlying causes of deforestation, forest degradation and the spread of extensive ranching, and it is the dominant driver of land use change in Latin America (Fearnside 2001, Araujo *et al.* 2009, Larson *et al.* 2010). Land tenure also directly influences REDD+ outcomes by defining who is eligible to receive benefits while influencing the ability of recipients to enforce carbon contracts (Naughton-Treves and Wendland 2014).

Lessons from forest conservation can inform the REDD+ mechanism. This is particularly relevant for protected areas (PA) given that they are a widespread environmental policy tool that has been used to protect forests (Bertzky *et al*. 2012). Covering extensive areas at global scales, protected areas have been identified as being potentially efficient at preventing deforestation (Andam *et al*. 2008, Nelson and Chomitz 2011). Latin America, for example, has a higher percentage of terrestrial protected areas (20.4%) than either developed countries (11.6%) or other developing regions (13.3%) (United Nations 2012).

Protected areas embrace a vast variation in nomenclature and management objective for forest conservation at global scales. In order to provide a global standard for defining and standardizing descriptions across protected areas, the International Union for Conservation of Nature (IUCN) has developed a system of classifying protected areas according to the underlying management objectives and different levels of protection (Dudley 2008). In Panama, protected areas cover approx. 35% of the total country area with 17 different categories of protection and management objectives. In general, protected areas are more effective than other forms of land tenure in reducing deforestation (Nepstad *et al*. 2006, Clark *et al*. 2008, Joppa *et al*. 2008, Joppa and Pfaff 2009, Nelson and Chomitz 2011, Porter-Bolland *et al.* 2012). Their success, however, generally depends upon location, governance and budgets (Nelson and Chomitz 2011). In the context of this thesis and because of its broad-scale analysis, terrestrial protected areas are treated as a unique tenure category without changes in levels of protection.

While the creation of protected areas in Latin America and the Caribbean has been one of the most popular top-down instruments for protecting forests (Elbers 2011), most of their recent expansion (1990 and 2000) has been associated with some previous level of protection or the presence of indigenous areas (Nelson and Chomitz 2011). The underlying assumption is that indigenous territories can also play an important role in forest conservation (Bhagwat and Rutte 2006; Nepstad *et al.* 2006; Hayes and Murtinho 2008). In several Latin American countries, Indigenous Peoples possess extensive areas of land, as is the case in Brazil (135 million ha), Bolivia (12 million ha), Mexico (39 million ha) and Colombia (36 million ha) (Larson et al. 2010). In Latin America, studies have shown that when governments have recognized traditional local rights, Indigenous Peoples are better able to control deforestation than private land regimes and can successfully prevent incursions into their forested territories (Nepstad *et al*. 2006, Hayes and Murtinho 2008). Across the tropics, apart from protected areas, lands under the control of Indigenous Peoples also exhibit low deforestation rates and have shown a high potential for conserving forests (Hayes and Murtinho 2008, Lu *et al*. 2010, Porter-Bolland *et al*. 2012).

Unveiling the role of land tenure and local communities in Panama

Using Panama (a fascinating country due to its singular tenure arrangements and the presence of multiple Indigenous Peoples across the nation) as a case study at different scales, this thesis explores the links between tenure and forest cover, in particular the role of local communities in forest conservation in potential REDD+ initiatives. As such, it enriches discussions stressing the importance of local dwellers and tenure security as an essential precept of REDD+ strategies.

Chapters 1, 2 and 3 explore land cover and land tenure dynamics among main tenure dwellers using Panama as a case study. The Republic of Panama is a small Central American nation, covering about 74,000 km^2, that is working with two REDD+ multilateral readiness programs, viz., the Forest Carbon Partnership Facility (FCPF) of the World Bank and the REDD program of the United Nations, with the goal of developing a national strategy that could reverse deforestation (World Bank 2011, UNDP 2012). In Chapter 1, using the statistical technique of "matching", I analyze forest cover and assess effectiveness in avoiding deforestation in three main land tenure regimes in Panama, namely protected areas, indigenous territories and non-protected areas. All categories of protected areas of Panama were included in the analysis to explore their potential effectiveness in avoiding deforestation.

One of the findings of the first chapter is the importance of indigenous territories for forest conservation in the country. Chapter 2 demonstrates that local knowledge can improve land cover classification and facilitate the identification of forest degradation. Here, I produce and compare accuracy of two land cover maps using digital image processing with a land cover map using a participatory approach.

Finally, Chapter 3 evaluates the importance of tenure security and its relation to forest conservation, an issue identified in Chapter 1. In this final chapter, I use the Upper Bayano watershed in Eastern Panama as a case study of complex land tenure dynamics in the context of forest conservation through REDD+ implementation. Using and validating free forest cover data (2001-2014), I estimate the importance of land invasions in indigenous territories as a key element of the rapid expansion of the agricultural frontier, showing that tenure alone is insufficient to guarantee rights over land and forests in indigenous territories.

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Contributions to Knowledge

Chapter 1 provides an analysis of forest cover and effectiveness assessment in avoiding deforestation in three main land tenure regimes in Panama, namely protected areas, indigenous territories and non-protected areas, in the context of a climate change mitigation strategy known as Reducing Emissions from Deforestation and forest Degradation (REDD+). This study shows the promises of matching techniques as a potential tool for demonstrating and quantifying conservation efforts. We therefore propose that the statistical method of "matching" could be integrated into methodological approaches to allow for compensating forest protectors. Since conserving forest carbon stocks in forested areas of developing countries is an essential component of REDD+ and its future success, the discussion of our results is relevant to countries or jurisdictions with high forest cover and low deforestation rates.

Considering the results obtained from Chapter 1, which identified the key role of Indigenous Peoples in forest conservation, **Chapter 2** is novel research that provides an analysis of accuracy of participatory mapping and local communities' understanding of LUCC. The results are relevant given the requirement for full and effective participation of Indigenous Peoples and local communities and given that high accuracy estimates are necessary for the purposes of monitoring forests at the international level in the context of REDD+ strategies. I demonstrate that local knowledge can improve land cover classification and facilitate the identification of forest degradation. The UNFCCC's call for the full and effective participation of local and Indigenous Peoples could, therefore, improve the accuracy of monitoring.

Expanding the area of analysis to the Bayano watershed in Eastern Panama, **Chapter 3** advances the knowledge of land tenure and tenure security as key factors determining the success of largescale REDD+ programs. The Bayano watershed presents complex land tenure dynamics where land invasions by colonist settlers have occurred in indigenous territories regardless of their legal land tenure status. By using and validating free forest cover data (2001-2014) produced by Hansen *et al.* (2013), I estimate the importance of land invasions in indigenous territories as a key element of the rapid expansion of the agricultural frontier, showing that tenure alone is insufficient to guarantee rights over land and forests in indigenous territories. I advocate that REDD+ strategies among developing countries can improve forest conservation and secure livelihoods on-the-ground and that conflict resolution mechanisms and mediation processes might provide an avenue to resolve long-standing land conflicts.

Linking Statement 1

In the Introduction section, land tenure is identified as a key factor in achieving forest conservation and ensuring potential success of REDD+ strategies. In Chapter 1, an analysis of forest cover and effectiveness in avoiding deforestation in three main tenure regimes in Panama is effectuated. It includes a mosaic of tenure categories, i.e. protected areas, indigenous territories, the overlaps between protected areas and indigenous territories, and non-protected areas. Comparing tenure regimes and forest cover when controlling for covariate variables, namely distance to roads, distance to towns, slope and elevation, allowed us to determine the effectiveness of matching techniques as a potential tool for demonstrating and quantifying conservation efforts. The analysis brings to the discussion the importance of tenure in countries or jurisdictions with high forest cover and low deforestation rates where the REDD+ mechanism has difficulties in proving the additionality of forest conservation.

CHAPTER 1:

Forest protection and tenure status: The key role of Indigenous Peoples and protected areas in Panama

Status: Vergara-Asenjo, G. & Potvin, C. (2014) Forest protection and tenure status: the key role of Indigenous Peoples and protected areas in Panama. Global Environmental Change 28, 205-215.

ABSTRACT

Using recent land cover maps, we used matching techniques to analyze forest cover and assess effectiveness in avoiding deforestation in three main land tenure regimes in Panama, namely protected areas, indigenous territories and non-protected areas. We found that the tenure status of protected areas and indigenous territories (including comarcas and claimed lands) explains a higher rate of success in avoided deforestation than other land tenure categories, when controlling for covariate variables such as distance to roads, distance to towns, slope, and elevation. In 2008 protected areas and indigenous territories had the highest percentage of forest cover and together they hosted 77% of Panama's total mature forest area. Our study shows the promises of matching techniques as a potential tool for demonstrating and quantifying conservation efforts. We therefore propose that matching could be integrated to methodological approaches allowing compensating forests' protectors. Because conserving forest carbon stocks in forested areas of developing countries is an essential component of REDD+ and its future success, the discussion of our results is relevant to countries or jurisdictions with high forest cover and low deforestation rates.

INTRODUCTION

The proposal for reducing emissions from deforestation and forest degradation (REDD+), which was advanced by the United Nations Framework Convention on Climate Change (UNFCCC), is the first global mechanism to combat climate change using the forestry sector (Pistorius 2012). Since 2005, it has been subject to negotiation at successive Conferences of the Parties (COPs) of the UNFCCC. In addition to activities to avoid deforestation and forest degradation, REDD+ also includes conservation, sustainable management of forests, and enhancement of forest carbon stocks in developing countries.

Lessons from forest conservation might help REDD+ avoid reinventing the wheel. Designation of protected areas (PA) is a widespread environmental policy tool that has been used to protect forests (Bertzky *et al.* 2012). Covering extensive areas at global scales, protected areas have been identified as being potentially efficient for preventing deforestation (Andam *et al*. 2008, Nelson and Chomitz 2011). Latin America, for example, has a higher percentage of terrestrial protected areas (20.4%) than either developed countries (11.6%) or other developing regions (13.3%) (United Nations 2012). In general, protected areas are more effective than other forms of land tenure in reducing deforestation (Nepstad *et al.* 2006, Clark *et al*. 2008, Joppa *et al*. 2008, Nelson and Chomitz 2011, Porter-Bolland *et al.* 2012). Their success, however, generally depends upon location, governance, and budgets (Nelson and Chomitz 2011).

While the creation of protected areas in Latin America and the Caribbean has been one of the most popular top-down instruments for protecting forests (Elbers 2011), most of their recent expansion (1990 and 2000) has been associated with some previous level of protection or the presence of indigenous areas (Nelson and Chomitz 2011). The underlying assumption is that indigenous territories also can play an important role in forest conservation (Nepstad *et al.* 2006; Hayes and Murtinho 2008). In several Latin American countries, Indigenous Peoples possess extensive areas of land, as is the case in Brazil (135 million ha), Bolivia (12 million ha), Mexico (39 million ha), and Colombia (36 million ha) (Larson *et al*. 2010). In Latin America, studies have shown that when the governments have recognized traditional local rights, Indigenous Peoples are better able to control deforestation than private land regimes and can successfully prevent incursions into their forested territories (Nepstad *et al*. 2006; Hayes and Murtinho 2008). Across the tropics, apart from protected areas, lands under the control of Indigenous Peoples also exhibit low deforestation rates and have shown a high potential for conserving forests (Hayes and Murtinho 2008, Lu *et al*. 2010, Porter-Bolland *et al.* 2012).

Using Panama as a case study, we specifically tested the hypothesis that protected areas and indigenous territories ensure forest conservation. We considered indigenous territories as geographic areas that are legally recognized, that are in the process of recognition, or that are claimed by Indigenous Peoples. Our study addresses two main questions: (1) What is the extension of forests in indigenous territories of Panama and how it has changed through time? (2) Are protected areas and indigenous territories effective in reducing deforestation in Panama? To answer these questions, we first mapped indigenous claimed lands, then compared forest cover through time under three main land tenure regimes, viz., protected areas and indigenous territories versus non-protected areas. Evaluating the effects of forest conservation requires controlling for landscape characteristics (Joppa and Pfaff 2010). For example, factors that are associated with remoteness, topography and access, such as distance from roads, distance from populated areas, slope steepness and soil fertility (Joppa and Pfaff 2010, Nelson and Chomitz 2011). We devised an empirical test to support, or refute, the hypothesis that protected areas and indigenous territories are effective in reducing deforestation. To do so, we used matching methods (Rubin 1973), a statistical impact analysis technique that allowed pairing protected and indigenous territories with unprotected areas with similar landscape characteristics. We also discuss the implications of our findings for the Panamanian REDD+ strategy, together with potential positive incentives that could reward forest conservation in high forest cover/low deforestation rate countries or subnational initiatives.

Panama's national context

The Republic of Panama is a small Central American nation that covers about 74,000 km^2, and is officially divided into nine provinces and five legally established indigenous territories, which are referred to as comarcas. Panama is a country that is rich in biodiversity, with western Panama being considered part of the Mesoamerican hotspot and eastern Panama, a part of the Chocó/ Darién/Western Ecuador hotspot (Myers et al., 2000). The country is uniquely situated as a biological corridor between Central and South America. Panama's deforestation rate was about 413 km^2 yr^{-1} between 1992 and 2000, and 134 km^2 yr^{-1} between 2000 and 2008 (CATHALAC 2008). Over the last 20 years, forest cover in Panama has decreased from 36,951 km^2 (49.3% of the total land area) in 1992, to 33,507 km^2 in 2000, and to 32,433 km^2 in 2008 (CATHALAC, 2008). In 2008, Panama started to work with two REDD+ multilateral readiness programs, viz., the Forest Carbon Partnership Facility (FCPF) of the World Bank and the REDD program of the United Nations, with the goal of developing a national strategy that could reverse deforestation, while developing an economic framework to do so (World Bank 2011, UNDP 2012). Panama's REDD+ readiness proposal to the FCPF identified six main drivers of deforestation: traditional and mechanized agricultural practices; extensive cattle ranching practices; exploitation of forests in a disorderly and unsustainable manner; poorly planned urban development; inadequate practices for exploiting mineral resources; and low levels of education and environmental culture (World Bank 2008).

Since the creation of Altos de Campana National Park in 1966, protected areas have represented the Panamanian government's principal strategy for in situ forest conservation within the country (ANAM 2006). Protected areas have also played a role in preventing the loss of Panama's forests (Nelson *et al.* 2001; Oestreicher *et al.* 2009, Haruna 2010), which currently represent 35.8% of the total land area (ANAM 2009). However, many of Panama's protected areas overlap with indigenous territories, thereby creating a mosaic of different tenures and tenure overlap zones, which are a source of diverse land-use conflicts. Indigenous territories within the borders of Panama are constituted as legally recognized areas and as areas being claimed by indigenous groups who wish to obtain legal recognition. These areas are hereafter referred to as ''legally recognized territories or comarcas'' and ''claimed lands,'' respectively. Claimed lands in Panama are based on customary ownership. As defined by Sunderlin *et al*. (2008), customary ownership is determined at local level and based on oral agreements by the community itself rather than the state or state law (statutory land tenure). However, under Law 72 (Gaceta Oficial 2008), indigenous groups that are living outside of comarcas can request official recognition of their lands. According to official data, comarcas encompass 12% of the country and include 27% of national forests (CATHALAC 2008, ANAM 2009). Official statistics only report forest cover and deforestation for three of the five comarcas because only three comarcas have provincial-level status, while the other two only have sub-provincial status (corregimiento). As a result, the remaining two comarcas are merged with provinces in national reports (ANAM/ ITTO 2003, ANAM 2009). This situation prevents a complete understanding of the role that indigenous territories might play with respect to forest conservation in Panama.

The comarcas are located in the western and eastern sections of the country, and along the Caribbean coast. The first comarca, Guna Yala, was established in 1938, while the most recent one was legally recognized in 2000 (Velasquez *et al.* 2011). Outside of the comarcas, the precise location of most claimed lands in Panama had not been mapped prior to our study, and as a result, the extent and percentage of forests inside these claimed lands was unknown. Under the authority of the General Congresses of the Collective Lands of Alto Bayano, the General Congress of Emberá-Wounaan Collective Lands, and the National Congress of Wounaan People, which are located in eastern Panama, the claimed lands are currently in the process of legalization under the country's Law 72 (Gaceta Oficial 2008) and Decree 223 (Gaceta Oficial 2010). The three remaining claimed lands, which are attempting to gain official recognition as comarcas, include Dagarkunyala, which is in easternmost Panama, and the Bribri and Naso territories, which are in western Panama. Over the past two decades, many of these areas have experienced an increase in invasion by non-indigenous groups, which has generated greater deforestation and other environmental problems. Most of these invasions are related to the expansion of the agricultural frontier by cattle ranchers or farmers (colonos) from other areas of the country (Wali 1993, Peterson St- Laurent *et al.* 2012).

METHODS

1) Mapping indigenous claimed lands

The first step in our study was mapping the claimed lands of Panama to determine the location and size of these areas. We began by gathering existing documentation on GIS coverage of national administrative units (provinces and comarcas) and the national system of protected areas, together with land-use maps from 1992, 2000, and 2008. These data came from three Panamanian institutions: the National Authority for the Environment (ANAM), the National Land Program (PRONAT), and the National Geographic Institute Tommy Guardia. A detailed list of the information that we obtained can be found in the Table S1 of the Electronic supplementary material (ESM). Indigenous communities without a formal proposed boundary were not included in this study (e.g., the Chagres communities).

To identify the extent of indigenous claimed lands, we held meetings with the Coordinadora Nacional de los Pueblos Indígenas de Panamá (COONAPIP, National Assembly of Indigenous Chiefs of Panama) and the local traditional indigenous authorities of each claimed land. In June 2010, COONAPIP formally nominated their General Secretary, Mr. German Hernandez, to assist us in identifying the indigenous areas to be mapped and to help contact local traditional authorities. Thereafter, meetings were held with the COONAPIP Secretary and local traditional authorities to explain the mapping project and obtain their authorization to visit the territories. Verbal authorization was the common way in which traditional authorities accepted participation in this project. We followed McGill University's protocols that are related to research conducted in indigenous areas and with Indigenous Peoples in Panama. Respect for intellectual property and the right of Indigenous Peoples to free, prior and informed consent were an essential part of this process. When available, geo-referenced maps of the claimed lands were provided by the traditional authorities and served as a starting point for our work. We visited four of the six claimed lands to collect qualitative and quantitative information about local land-use dynamics from Indigenous Peoples' own point of view at a local scale. The visit allowed us to determine, with the aid of local traditional authorities, the boundaries of claimed lands on 1:250,000 official topographical maps. The other two territories were not visited. One of these was not easily accessible, while we were unable to contact traditional authorities in the other.

The aforementioned information was amassed in ArcGIS (ESRI 2011) to create a draft map of all of Panama's indigenous territories (including established comarcas and claimed lands). The draft map was validated by COONAPIP's traditional authorities and their technicians during a workshop, which was held in the Ngäbe Bugle comarca in August 2011. Representatives of ten of the 12 authorities of Indigenous Nations were present at this workshop; those of the Ngäbe and the Naso were not present. Printed maps with recognizable landscape features (e.g., rivers, roads, and coastlines) were given to participants, who carefully examined the limits of their own areas and made comments. We used these comments to correct the draft maps where necessary. The resulting map was finalized at the end of August 2011 and officially approved by COONAPIP in October 2011. Printed and digital versions of the map were presented to each of the 12 traditional indigenous authorities through COONAPIP.

2) Forest cover in Panama

To identify forest cover among land tenure regimes of Panama, we used ANAM land cover maps for the years 1992 and 2000. In the absence of a more recent official land cover map, we used the digital land cover map that had been produced for 2008 by the Centro del Agua del Trópico

Húmedo para America Latina y el Caribe (CATHALAC, Water Center for the Humid Tropics of Latin America and the Caribbean) (CATHALAC 2008). Following the ANAM-International Tropical Timber Organization (ITTO 2003) system of classification, we included all mature forests in the country, which consisted of mature, secondary mature, homogenous cativo (*Prioria copaifera Griseb*.), mixed cativo, homogenous orey (*Campnosperma panamensis Standl.),* mixed flooded, or mangrove forests. Note that this classification has been retained by Panama in the context of REDD+ readiness. All these categories are characterized as having more than 80% tree cover. Fallows, young secondary forest, and highly degraded lands were not included in the analysis.

We overlaid our GIS map of indigenous and non-indigenous territories with a digital coverage of ANAM's protected areas, together with the extent of mature forests, to determine forest cover in every sector. Nationally protected areas are the 89 areas that are included in the Sistema Nacional de Áreas Protegidas (SINAP, National System of Protected Areas), and which were centrally managed by the Panama's National Environmental Authority (ANAM) (ANAM 2010). This overlay analysis allowed us to estimate mature forest cover in: (1) legally established comarcas that did not overlap protected areas (C); (2) overlap between legally established comarcas and protected areas (C-Over); (3) claimed lands that did not overlap with protected areas (Cl); (4) overlap between claimed lands and protected areas (Cl-Over); (5) nationally protected areas that do not overlap with indigenous territories (PA); and (6) other lands without protection in the country (OL) (Fig. 1). The effects of land tenure regimes on the percentage of mature forest cover were analyzed using repeated measures ANOVA. The area that was covered by mature forest (log-transformed) in 1992, 2000 and 2008 were used as dependent variables. Tukey–Kramer post hoc multiple comparisons were used to test for significant differences among land tenure regimes and time. All statistical analyses were performed in the car package 2.0-16 of R (www.r-project.org).

3) Effectiveness of avoided deforestation: matching analysis

We assessed the effectiveness in avoiding deforestation by comparing protected areas and indigenous territories (the treated group) with non-protected areas (the control group) using matching methods (Rubin 1973, Stuart 2010). The response variable was the presence/absence of deforestation in at least one forested pixel (minimum area of 200 m x 200 m) during two evaluation periods. The country total area with approx. 1.8 million records was split in two separate categories to form the control and the treatment group for the analysis, including one record for every pixel. Two cohorts of pixels were prepared to compare the treated and control groups. The first cohort used 1992 as the base year, considering all protected areas that were created in or before 1992, with the response variable being the presence/absence of deforestation between 1992 and 2008. The second cohort used 2000 as the base year, including all protected areas that were created in or before 2000, with deforestation being estimated between 2000 and 2008 (Table S4 for details). We included all indigenous territories in both cohorts because Indigenous Peoples have permanently inhabited these areas, and we wanted to evaluate the potential effectiveness of these areas in avoiding deforestation against non-protected areas. Hence, the treated group considered five tenure categories (PA, C, Cl, C-Over, Cl-Over) to evaluate potential differences in avoided deforestation among these areas. The final control group was three or more times larger than the treatment group in the analysis, as a result, the 1992 cohort included 923,775 pixels with 642,840 pixels for the five tenure categories in the treatment group. The 2000 cohort was constituted of 837,675 pixels with 633,459 pixels for the same five tenure categories.

Matching allowed the pairing of the treated group with forested pixels that were similar in terms of topography and remoteness, but lacking ‘‘protection’’. Four matching covariates were selected to ensure the comparability of the treated and control groups. Elevation and Slope were used as proxies for topography, since agriculture and cattle ranching are mostly conducted on mild slopes and at lower elevations (Nelson and Chomitz 2011). We used the CGIAR-CSI version of the 90-m resolution STRM (Shuttle Radar Topography Mission; Jarvis et al., 2008) digital elevation model to derive elevation and slope for the entire study area. Distance to roads and Distance to towns were proxies for remoteness (Kaimowitz and Angelsen 1998, Geist and Lambin 2002). Distance from roads and town was estimated using ArcGIS 10.1, considering all towns or cities with more than 5000 inhabitants, together with road systems, in 2000 for cohort pre-1992, and in 2008 for cohort pre-2000. Digital and hardcopy maps were used to extract cities and road networks for the two cohorts.

We used nearest-neighbour covariate matching with replacement -using the Mahalanobis distance metric- as an evaluation method (Rubin 1973). Matching was applied without and with calipers (Rubin and Thomas 2000) using 0.5 and 0.25 standard deviations; calipers indicate a tolerance level for evaluating the quality of matches. However, we do not report results with calipers because they did not produce any reduction in the number of treated/control matched pairs. The analysis tested several matching techniques until an adequate before- and after-matching balance was reached. Matching balance was evaluated using the set of balance statistics included by default in the R package Matching (version 4.7-14, Sekhon 2011) and included the mean difference between control and treatment groups for each covariate before and after matching. Matching was considered satisfactory if the difference was 0. The matching balance achieved was tested with the t-test of difference of means, the Kolmogorov–Smirnov test and quantile–quantile plots, to identify potential differences in covariate balance. It was deemed adequate if the difference of means for a covariate was not statistically significant between the two groups.

RESULTS

Extension and forest cover of different land tenure in Panama

According to the GIS analysis, the indigenous territories that were legally established as comarcas, together with all of the claimed lands, represented 31.6% (23,470 km2) of Panama's total area (Fig. 2). With a total of 27 separated areas, the newly mapped claimed lands represented 9.2% (6850 km2) of the country's total area. Their sizes range from 231 km2 for Alto Bayano to 3030 km2 for the Collective Emberá Lands (more details in the ESM). With the exception of the Collective Embera´ Lands, claimed lands are smaller in area than legally established comarcas. Only two indigenous territories, the Guna Yala and Madungandi comarcas (Table S2, ESM), do not overlap with existing protected areas. Otherwise, overlap with protected areas is high for all indigenous territories. Close to 1 million ha (979,850 ha) of the indigenous territories consequently have dual tenure, i.e., they are simultaneously both indigenous territory (comarcas or claimed lands) and protected areas (Fig. 3). Aggregate tenure overlay is higher in claimed lands (78.7%) than in comarcas (22.4%). A detailed description of all of the mapped claimed lands can be found in the ESM.

Mapping indigenous areas helped understand the comparative importance of forests in the six tenure regimes. Together, PAs and indigenous territories hosted 77% of Panama's total mature forest area in 2008. Our GIS analysis showed that about 725,300 ha of mature forests remain in PA, while 1,754,000 ha remained in indigenous territories, considering the forested areas of both comarcas and claimed lands. Indigenous territories, as a whole, represented 54% of mature forest cover of Panama in 2008. At that time, only three of the indigenous territories (the claimed lands of Alto Bayano and Collective Wounaan Lands, and the comarca Ngäbe Bugle) had less than 80% mature forest cover (Fig. 4). About 903,000 ha of the indigenous territories (comarcas and claimed lands) overlapped with protected areas, representing 28% of the total mature forest cover in 2008.

To analyze forest cover within Panama's provinces, we estimated their net areas and discounted claimed lands while including protected areas. The province with highest mature forest cover in 2008 were Bocas del Toro (166,000 ha), Colon (245,000 ha), and Darien (325,000 ha). Those provinces that had the lowest mature forest cover were Herrera (8000 ha) and Los Santos (25,000 ha). In 1992, four provinces (Chiriquı, Herrera, Los Santos, Coclé) had less than 20% forest coverage. The provinces that lost the most mature forest cover between 1992 and 2008 were Darien (141,000 ha) and Panama (92,000 ha).

In 2008, the tenure regime with the lowest proportion of mature forest cover (20%), compared to its total area (730,000 ha) was other lands (OL). Overlaps between indigenous territories and protected areas, i.e., overlapped comarcas (C-Over) and overlapped claimed lands (Cl-Over), had the highest proportion of forest cover (>90%) over a total area of 906,000 ha (Fig. 5). Net areas of comarcas (C) with 760,730 ha, claimed lands (Cl) with 85,240 ha, and protected areas (PA) with 725,300 ha had an intermediate proportion of mature forest cover (70–80%).

ANOVA showed significant differences in mean forest cover percentage among tenure regimes ($F_{5,32}$ = 14.58, P = 0.001). Tukey post hoc tests (P < 0.05) showed that the other lands (OL) regime (20.7 ±17.8% forest cover) contained significantly less forest than all other tenure regimes. The percentage forest cover in comarcas (C) was 79.4% (±20.9% SD, P = 0.005), 76.7 (±23.3%, P = 0.005) in claimed lands (Cl), 93.7 (±5.35%, P = 0.005) in overlapped comarcas (C-Over), 93.8 (±10.2%, P = 0.005) in overlapped claimed lands (Cl-Over), and 71.6 (±24.2%, P = 0.005) in protected areas (PAs). In contrast, there were no significant differences among the tenure regimes of C, Cl, C-Over, Cl-Over and PA.

Across tenure regimes, the proportion of land that was covered by forests decreased significantly with time ($F_{2,64}$ = 7.62, P = 0.001). Across the six tenure regimes, the proportion of forest decreased, on average, from 76.6% ± 26.7% in 1992 to 71.7 ± 27.6% in 2000 and to 69.8 ± 27.3% in 2008. Our analysis further revealed a significant interaction between time and tenure regimes for forest cover ($F_{10,64} = 2.3$, $P = 0.02$). Forest cover for claimed lands (Cls) showed a pattern of change that distinguishes it from the other tenure regimes. The high forest cover observed in 1992 decreased abruptly in 2000 due to high deforestation rates in this last period.

Matching as a way to assessing the effectiveness of avoiding deforestation

The general characteristics of the covariates that were used in the matching analysis indicate that overlapped areas, i.e., Cl-Over and C-Over, were more remote, and along with PA, had steeper slopes and higher elevations compared to the other tenure regimes (Table S3 in the ESM). Other lands (OL) and claimed lands (Cl), in contrast, showed greater proximity to roads, were located at lower elevations, and had lower slope gradients. The categories of OL and PAs showed also greater proximity to towns.

To answer the question: ‘‘Are protected areas and indigenous territories effective in reducing deforestation in Panama?’’ we used matching analysis. This technique allow controlling for covariate variables such us distance to roads, distance to towns, slope, and elevation all of which could explain why protected areas and indigenous territories retain the highest forest cover of Panama. We found that the tenure status of protected areas and indigenous territories (including comarcas and claimed lands) explains a higher rate of success in avoided deforestation than other land tenure categories. Matching analysis proceeded in two steps, after separating pixels in two groups, treated and control, it compared the proportion of areas in both groups that lost forests during each of the two time interval considered. Hence it provided estimates of avoided deforestation as a proportion of forest in the control group, set to 100%, with negative signs indicating less forest loss (Table 1). Matching analysis indeed showed that the most effective tenure regime for avoiding deforestation in Panama was protected areas (PAs) with 18.1 and 10.8% less deforestation, respectively for the 1992 and 2000 cohorts, than control areas with similar covariate characteristics (Table 1). Amongst the indigenous territories, claimed lands (Cls) was the most efficient tenure regime, reducing deforestation by 9.2% and 8.3%, respectively, for the 1992 and 2000 cohorts, compared to the control group. This is an important result because of the proximity of Cl to roads (Table S3 in the ESM). On average, indigenous territories without overlap (C, Cl) performed better than overlapped indigenous territories (C-Over, Cl-Over). In non-overlapped indigenous territories, deforestation was reduced in 7.45% in the pre-1992 cohort and 7.2% in the pre- 2000 cohort, whilst clearing of overlapped indigenous territories was reduced in 5.65% and 2.6% for the respective cohorts.

DISCUSSION

Forest cover and tenure: insights from Panama

The question at the heart of our paper is: are protected areas and indigenous territories effective in reducing deforestation in Panama? We developed an empirical test to answer this question, and the result is unequivocal. Yes, protected areas and claimed indigenous territories of Panama are effective tenure regimes for avoiding deforestation. Furthermore, these areas also possess high forest cover. Several authors have indicated that tenure and forest tenure issues should become an essential part of REDD+ readiness programs (Cotula and Mayers 2009, Robinson *et al.* 2011, Angelsen *et al.* 2012, Holland *et al*. 2012). The evidence that is provided by this research on the effectiveness of protected areas and indigenous territories in avoiding deforestation certainly supports that recommendation.

Our results suggest that, because of their efficiency in conserving forests, both protected areas and indigenous territories could be part of the successful implementation of REDD+ in Panama. To date however, the relationship between Panama's Indigenous Peoples and REDD+ has been bumpy. In 2013, three different bodies representing traditional indigenous authorities, the Comarca Guna Yala, the Comarca Madungandi and the COONAPIP, indeed rejected REDD+ and the UN-REDD program, respectively (Potvin and Mateo-Vega 2013). In early 2014 however, the tensions between COONAPIP and ANAM apparently got resolved and a memorandum of collaboration was signed. This memorandum does not resolve the issues raised by the two Guna Comarcas where REDD+ remained, to date, banned. Carbon rights ownership was one of the demands that was established by COONAPIP when negotiating with the UNREDD program, stating: ''Determining carbon property rights, and consequently those over carbon credits that may be generated, is crucial and a matter where differences persist'' (COONAPIP 2011, p. 14; Cuellar *et al.* 2013). In Panama, Article 10 of the Forest Law 1 (1994) indicates that the ''forest patrimony of the state is constituted by all natural forests, the lands on which those forests are located, and state lands of preferably aptitude for forestry'' (República de Panamá 1994, article 10). This article of the Forest Law embodies the risk of exclusion that is feared by Indigenous Peoples in Panama and in Latin America (Griffiths 2009, Velásquez 2012).

While it is clear that attention should be given to resolving tenure conflicts and clarifying tenure rights in designing REDD+ strategies (Larson *et al*. 2013), many REDD+ strategies that are submitted by countries to the Forest Carbon Partnership Facility of the World Bank have not included a concrete proposal on how to implement or resolve tenure conflicts, even when the risks were mentioned in their proposals (Dooley *et al*. 2011, Westholm *et al*. 2011). Encouraging governments to clarify and resolve local forest tenure rights, and to remove the perverse legal incentives to deforest appears to be an unavoidable prerequisite for favouring Indigenous Peoples participation in REDD+ programs (Agrawal *et al.* 2008, CIRAD 2012). Furthermore, local governance has been proposed as a way of conserving forests and ensuring local livelihoods in a cost-effective way compared to centralized governance (Sandbrook *et al.* 2010), which again suggests that clarifying land tenure is a key step for REDD+ success (Sunderlin *et al.* 2009).

Rewarding forest protectors

Although REDD+ was originally positioned primarily as a carbon mitigation and offset mechanism, there is widespread expectation that REDD+ will also contribute to conserving and maintaining tropical forest biodiversity and other endangered forest ecosystems (Angelsen *et al*. 2012). To do so, REDD+ needs to be able to compensate for forest conservation keeping in mind the principles of additionality and permanence (as defined by Parker *et al*. 2009) to produce real and credible reductions (Vallatin 2011). The concern that a REDD+ mechanism largely structured to reward past deforestation has been plaguing the REDD+ discussion since its inception (Achard *et al*. 2005, Angelsen and Wertz-Kanounnikoff 2008). Griscom *et al*. (2009), using FAO (2006) data to classify 56 tropical countries, showed that countries with high forest cover and low deforestation rates, and those with high forest cover and medium deforestation rates harboured 10.5% and 63.7% of tropical carbon stocks, respectively. The majority of countries in these two categories are Latin American (65% of the total). Given the importance of these high forest cover countries with medium to low deforestation rates, their broad participation in REDD+ is a required step to maximize the mechanism's mitigation potential, minimize international leakage, and promote equity (Eliasch 2008). Over the past decade, diverse positive incentives to reward countries with high forest cover and low deforestation (e.g., Peru, Suriname, Belize) have been proposed (Achard *et al*. 2005, Mollicone *et al*. 2007, Prior *et al*. 2007, Gutman and Aguilar-Amuchastegui 2012).

The challenges inherent to rewarding past conservation effort are likewise present at a jurisdictional and sub-national scale for example in protected areas and indigenous territories. It remains unclear whether protected areas that have already been established would be eligible for REDD+ (Larson 2011) because their additionality is under question. Some would say that REDD+ funding should only apply to newly protected areas in areas where forests are at risk (Angelsen *et al*. 2012). In practice, deforestation frequently continues inside protected areas, particularly when funding, management capacity or political support is limited (Ricketts *et al*. 2010). If independent assessment shows that forest cover is being lost or degraded within an existing PA, and that additional resources could reduce these threats, PAs might present an attractive, previously overlooked opportunity to reduce emissions (Ricketts *et al.* 2010).

Consistent with our findings, several other studies have shown less deforestation within protected areas compared to other tenure regimes (Nagendra 2008, Oestreicher *et al*. 2009, Haruna 2010, Nelson and Chomitz 2011), suggesting that the creation of new PAs or enhanced support for existing PAs could be important for successfully implementing REDD+ (Ricketts *et al.* 2010). Our results suggest that Panama's National REDD+ strategy could rely upon the enforcement of laws pertaining to national protected areas, and upon close collaboration with its Indigenous Peoples. This condition could be applicable to many other Latin American countries, where PAs and indigenous territories contain extensive forest resources. However, several questions arise. How do we reconcile the fact that these areas could represent the core of REDD+ national strategies with the possibility of using future carbon markets? If performance-based payments are adopted, how can incentives be provided to tenure regimes that already protect forests and have low deforestation rates?

Our results reinforce the idea that rewarding forest protectors is a necessary step for the global success of REDD+ because it can contribute to climate mitigation, reduce national and international leakage, and also promote a fair distribution of costs and benefits among and within countries (i.e., equity) (Meridian Institute 2011). With the broadening of the scope of REDD+, three main approaches to performance-based payment for countries with medium to low deforestation rates have been suggested: (a) the strictly historical, (b) an intermediate historically based approach with adjustments, and (c) forward-looking approaches. The schemes could also apply at the jurisdictional level, e.g., provinces or indigenous territories.

(a) Strictly historical: Early proposals of performance-based payments are based on historical deforestation rates. One example of an historical proposal is ‘‘compensated reduction’’ (Santilli *et al.* 2005), which suggests that the strictly historical reference level (RL) be revised every 3 or 5 years to include more recent historical rates of deforestation. Joanneum Research *et al*. (2006) proposed the ‘‘corridor approach,’’ which uses minimum and maximum RL values (based on observed past deforestation levels) instead of considering the average value for the reference period. In both of the aforementioned proposals, it is hard to envision how forest protectors who are engaged in forest conservation either at the national or jurisdictional level would be compensated, since conservation does not translate into any rates of change in forest cover nor emissions. In response to this challenge, the Government of India proposed, ‘‘compensated conservation’’ in 2007. Its intention is to compensate countries for maintaining and increasing their forests as carbon pools as a result of conservation, with increases and improvements to forest cover backed by a verifiable monitoring system (UNFCCC 2011). India recommends measuring forest change with a previously set reference level, which could be fixed at 1990. This proposal intends to compensate countries for maintaining and increasing forest carbon stocks using a non-market based mechanism; however, it is not clear how this approach could resolve additionality issues regarding conservation payments.

(b) Historical adjusted: Concerns that compensation based on historical emissions would penalize forest protectors and favour past bad behaviour opened the door for the development of alternative proposals. The specific intention here is to address the issue of countries with high forest cover/low deforestation rates. Among proposals of this type is that of the Joint Research Centre (Achard *et al.* 2005), which presents a mechanism to account for preserved carbon if the country keeps its conversion rate below half of the global rate. This proposal also includes an accounting of forest degradation. Mollicone *et al*. (2007) proposed the ‘‘incentive accounting’’ approach, which recommends the establishment of a reference level for low deforestation countries at half of the global historical deforestation rate. In the same year, Prior *et al*. (2007) proposed an alternative, i.e., the ‘‘carbon stocks approach.’’ It establishes a trading mechanism that allows participating Non-Annex I (developing countries with emission reduction targets) countries to sell ‘‘Carbon Reserve Units.’’ These Carbon Reserve Units are linked to projects protecting forests that are under threat of degradation or deforestation, and which can be designed

to include net increases of carbon stocks in degraded forests. Strassburg *et al.* (2009) have suggested the idea of ''combined incentives,'' which explores the outcomes of establishing a universal benchmark that is equal to the global average rate of deforestation. This approach is intended to promote incentives that reduce deforestation and degradation, as well as stimulate forest conservation, while promoting reforestation and afforestation activities. Lastly, the ''stock and flow approach'' withholds a percentage of payments for emission reductions relative to historical deforestation levels to pay for conserving forest stocks (Cattaneo *et al*. 2010).

(c) Forward-looking (projected): The Terrestrial Carbon Group (2008) proposed that credit should be based upon the country's carbon stocks, but should differentiate between protected carbon areas and tradable carbon areas. In this proposal, protected areas that currently are not receiving compensation under REDD+ would be allowed to emit a certain quantity of tradable carbon stocks each year.

CONCLUSION

Matching analysis as a novel approach

REDD+ performance-based proposals are primarily based on deforestation rates or on changes in forest cover (Gutman and Aguilar-Amuchastegui 2012), making it difficult to apply them to forest conservation. Furthermore, the core of these performance-based proposals is also set at national scales while, in many developing countries, REDD+ actions are being developed at local or regional levels, including communities, civil society and regional governments. So-called nested approaches (Pedroni *et al.* 2009), for example, which support Indigenous Peoples' good forest stewardship, could facilitate early actions of local stakeholders. Standardized nested schemes have indeed been proposed by the Verified Carbon Standard (VCS) through the ''Jurisdictional and Nested REDD initiative (JNR)'' (VCS 2012) and the ''Nested REDD+ standard,'' which were created by the American Carbon Registry (ACR 2012). In nested schemes, countries could divide their territory according to biomes, political boundaries or tenure regimes, such as indigenous territories, protected areas and private lands, which could be considered sub-national jurisdictions. It has been suggested that sub-national jurisdictions could develop their own rewarding mechanisms (bottom- up approach) or, conversely, that countries could suggest standardized sub-national proposals for some regions (top-down approach) (Chagas *et al*. 2011).

We contend the matching analysis that we developed offers a statistically valid way of determining the effectiveness that is granted by protected areas and, therefore, their additionality. At jurisdictional levels, rewarding forest protectors could be resolved by matching the observed deforestation within a jurisdiction to land of similar area and characteristics outside the jurisdiction. A potential risk of stock-based payments, i.e., payments based upon the total carbon in the forest during a specific period, is that a part of these incentives could be made to areas that are under no ‘‘threat’’, which is also called passive conservation (Angelsen and Wertz-Kanounnikoff 2008). We contend that matching methods could help to resolve this issue through the pairing of comparable protected and non-protected areas with respect to remoteness, topography or other relevant characteristics. From this perspective, the technique could help to demonstrate forest protectors' conservation efforts in a quantitative manner by estimating the impact in avoided deforestation of conservation areas. It could also serve to separate protected areas with real contribution to reduce deforestation from ‘‘rock and ice’’ areas, i.e., remote and unattractive areas for agriculture and raising cattle without real potential of being deforested. This approach is particularly important for PAs or other conservation areas, which could be biased towards areas that prevent land conversion the least (Joppa and Pfaff 2009). Therefore, identifying areas under threat would also help us to compensate forest protectors in areas in which there are real pressures of forest clearing. Thus matching is a scientifically-sound, simple and elegant way to quantify the contribution of each jurisdiction to emissions avoidance.

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TABLES AND FIGURES

Table 1. Estimates of the proportion of treated pixels (in percentages) used in the matching analysis that retained forest cover between 1992-2008 and 2000-2008. The treated group included different forms of indigenous tenure as well as protected areas. By convention, negative signs reveal that the treatment resulted in avoided deforestation. All the estimates are statistically significant at $p < 0.01$ and standard error (SE) was derived from the repetitions of the analysis. The number of matched pairs is a measure of sample size, each matched pair including two pixels with similar covariates one from the treated and one from the control groups. For tenure: C-Over = overlap between *comarcas* and protected areas, Cl-Over = overlap between claimed lands and protected areas.

	Pre-1992 cohort		**Pre-2000 cohort**	
Tenure	**Estimates [SE]**	**Matched pairs**	**Estimate* [SE]**	**Matched pairs**
Comarca	-5.7 [0.0011]	24,921	-6.1 [0.0009]	21,384
C-Over	-6.5 [0.0011]	10,149	-3.1 [0.0008]	9,937
Claimed lands	-9.2 [0.0033]	3,175	-8.3 [0.0027]	2,361
Cl-Over	-4.8 [0.0008]	12,672	-2.1 [0.0007]	12,975
Protected areas	-18.1 [0.0013]	15,610	-10.8 [0.0010]	16,486

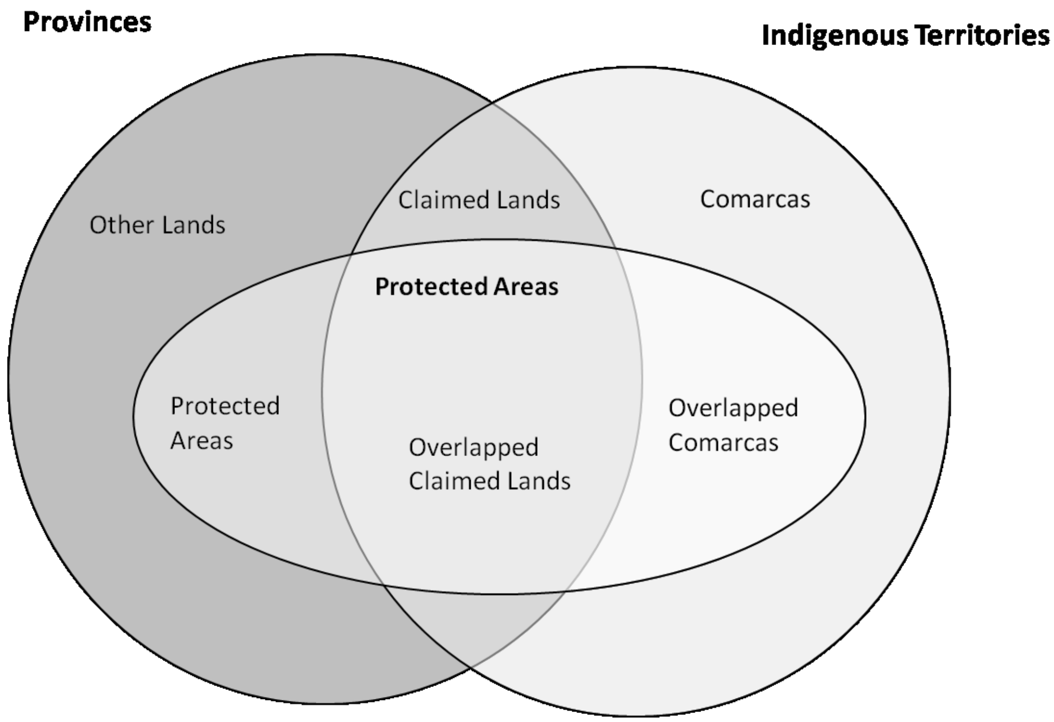

Figure 1. Provinces, indigenous territories, and protected areas and their respective overlaps.

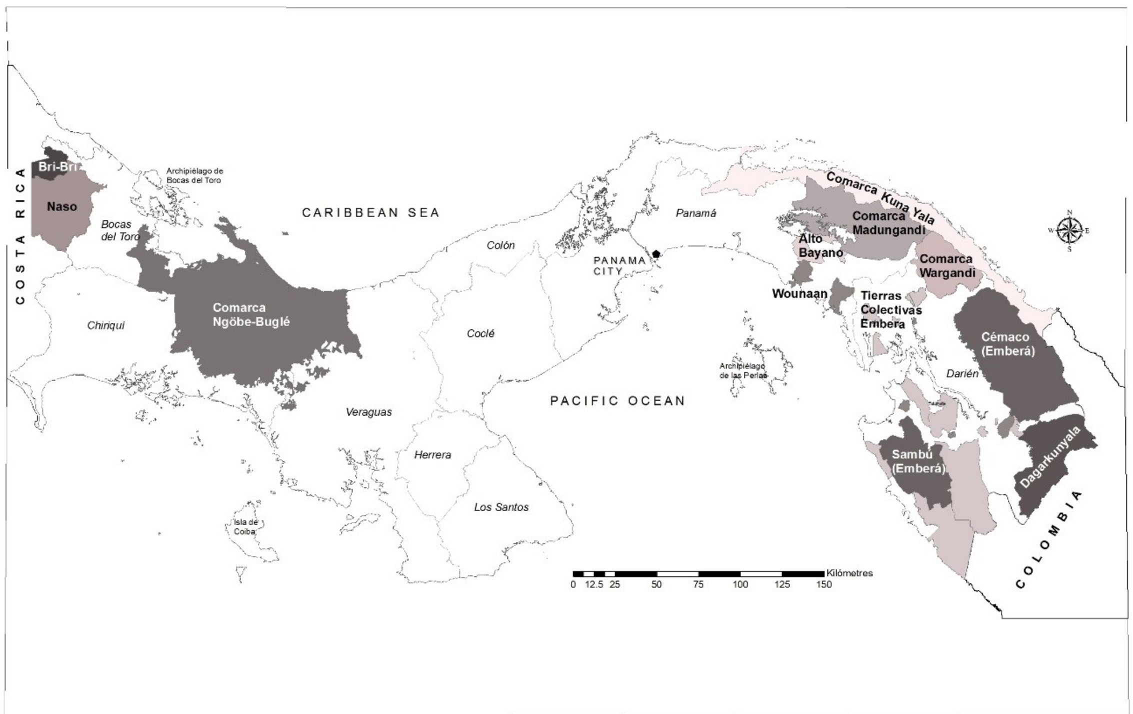

Figure 2. Map of indigenous territories across Panama showing the five comarcas and the six indigenous claimed lands.

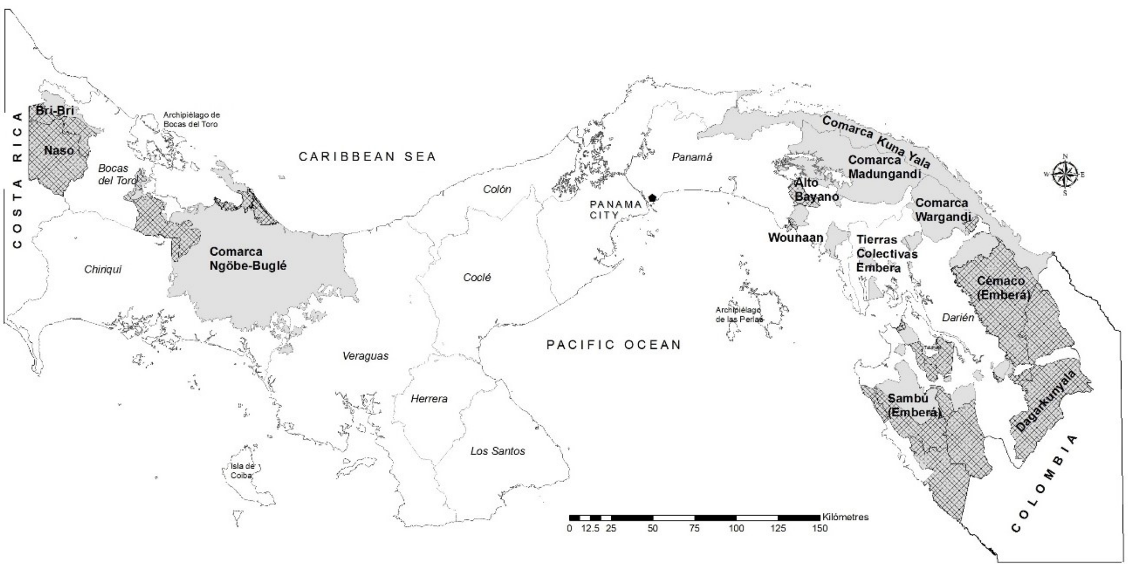

Figure 3. Overlap between indigenous territories and protected areas in Panama (shaded areas).

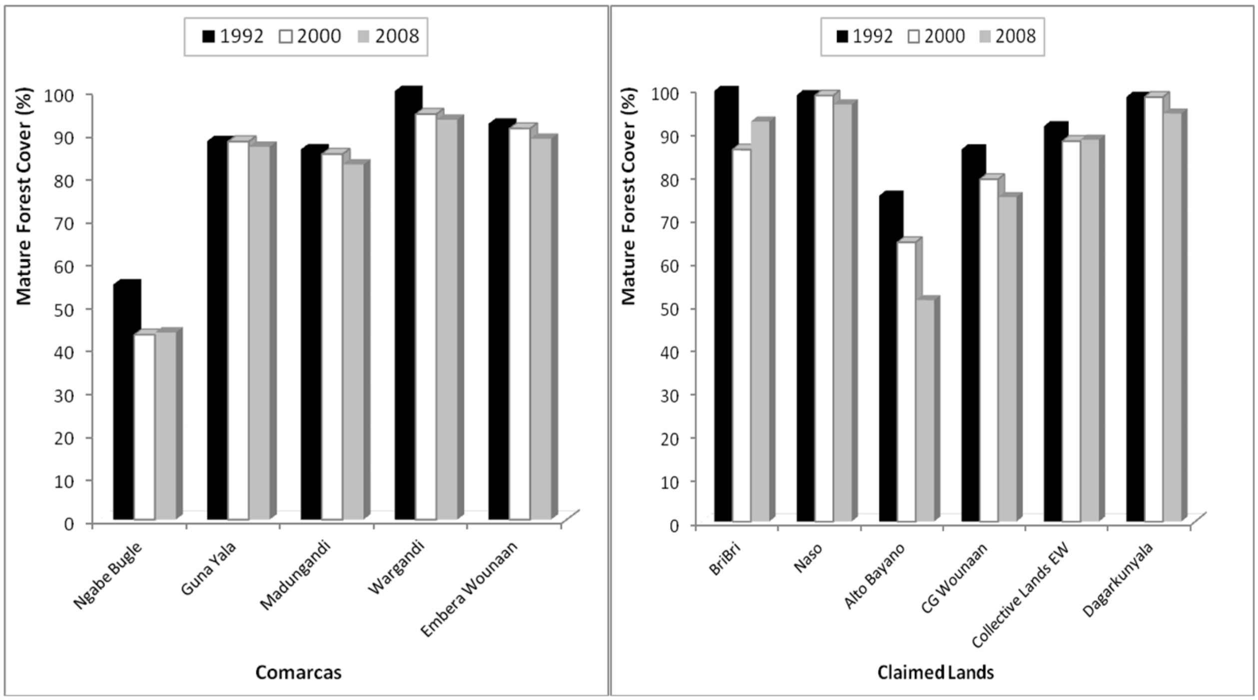

Figure 4. Variation in forest cover (%) within the indigenous territories of Panama. Year 1992 (black), 2000 (grey), and 2008 (white). Left panel: comarcas, including overlapped areas (C-Over). Right panel: claimed lands, including overlapped areas (CL-Over).

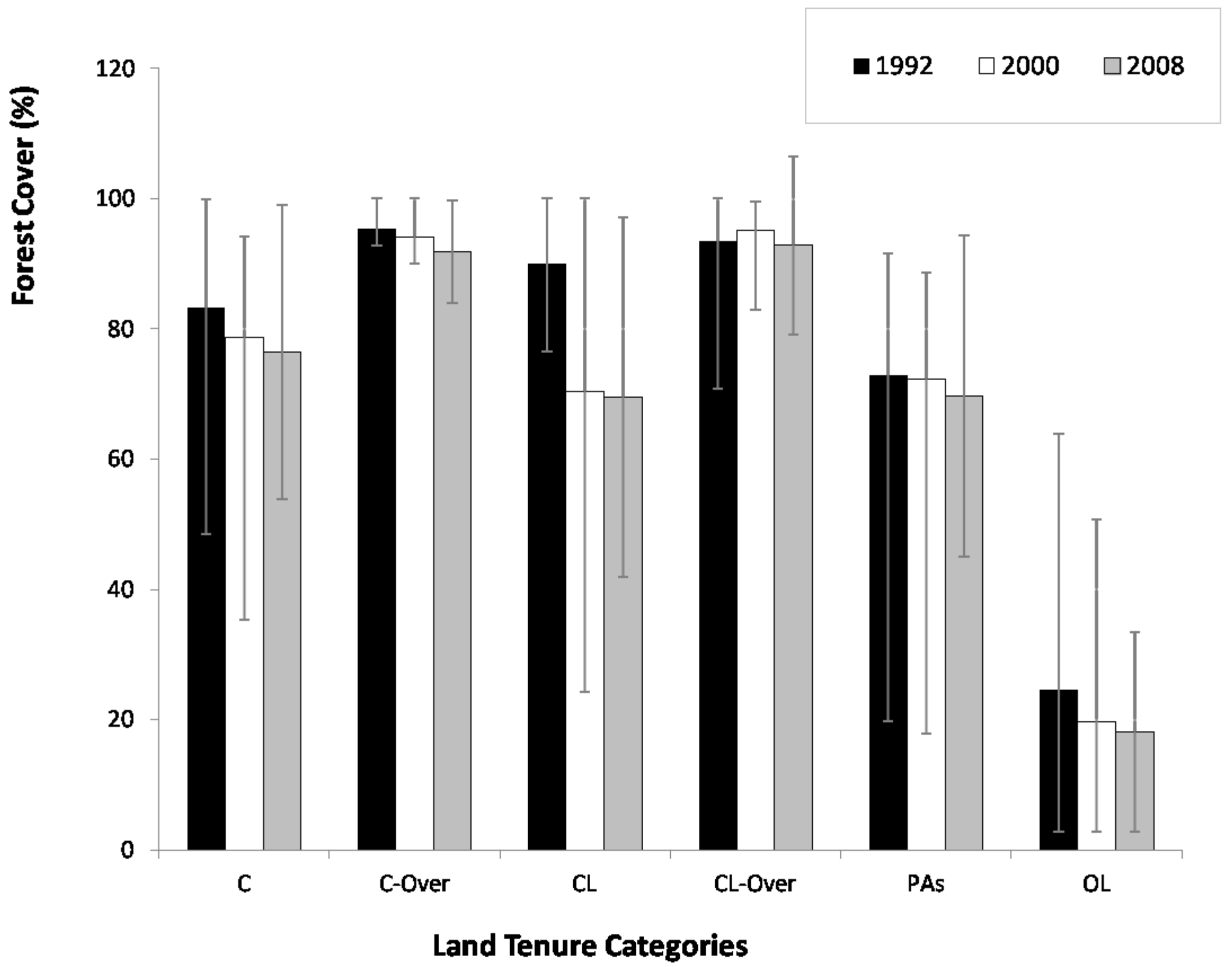

Figure 5. Average forest cover (%) within six land tenure regimes for the period 1992-2008 in Panama (grey lines are ranges). C: comarcas, C-Over: overlapped comarcas, CL: claimed lands, CL-Over: overlapped claimed lands, PAs: protected areas, OL: other lands.

Linking statement 2

While the results of Chapter 1 show Indigenous Peoples' key role in forest conservation, Chapter 2 provides novel research of accuracy of participatory mapping and local communities' understanding of LUCC. Here, I produce and compare land cover maps, based on digital image processing using remote sensing imagery, and a land cover participatory map for indigenous territories of eastern Panama. Using field data, I evaluate and analyze the accuracy of the produced maps in the context of the current requirements of monitoring forests at the international level in the context of REDD+ strategies.

CHAPTER 2:

Engaging stakeholders: Assessing accuracy of participatory mapping of land cover in Panama

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ABSTRACT

Full and effective participation of Indigenous Peoples and local communities, and high accuracy estimates are two current requirements for the purposes of monitoring forests at international level. We produced two land cover maps, both of which were based on digital image processing (decision trees) using RapidEye imagery, and a land cover participatory map, for indigenous territories of eastern Panama. Accuracy of the three maps was evaluated using field data. Classification that was based on participatory mapping gave best overall accuracy of 83.7 % (κ = 0.783), followed by the decision tree that included textural variables (DT2 - overall accuracy of 79.9 %, κ = 0.757). We have demonstrated for the first time that local knowledge can improve land cover classification and facilitate the identification of forest degradation. The plea of the UNFCCC for the full and effective participation of local and Indigenous Peoples could, therefore, improve the accuracy of monitoring.

INTRODUCTION

Reducing Emissions from Deforestation and Forest Degradation in developing countries (REDD+) is a mitigation mechanism that has now been agreed upon under the United Nations Framework Convention on Climate Change (UNFCCC). For the first time, developing countries might be compensated for their efforts in either reducing carbon dioxide emissions from the forestry sector or increasing forest carbon stocks. The integrity of such forest-based carbon-trading schemes will strongly depend upon the accuracy/precision of forest measuring/monitoring systems (Herold and Skutsch 2009). In the context of REDD+, the accuracy of actual carbon stock change estimates will be especially important for countries that are interested in claiming credits for their efforts in reducing deforestation.

Identifying, delineating and *mapping land cover* is the first critical task that is required for evaluating and monitoring changes in forest carbon stocks. While there are multiple approaches to classifying land cover, the mapping of land cover categories is never considered to be a perfect representation of the landscape (Lowry *et al.* 2007). Despite the evolution of remote sensing technologies over the last few decades, interpretation is still plagued by difficulties when the time comes to identifying specific land cover categories, in particular with medium to low resolution satellite imagery. For instance, Pelletier *et al.* (2011) identified the lack of understanding of fallow land dynamics as a significant source of uncertainty for Panama, given that fallows occupy a substantial fraction of the national territory. In many parts of the world, fallows and other successional stages of forest lands can cover large areas. Thus, methods for improving the classification success of areas that are in various forest successional stages, together with logging activities having reduced impacts, would contribute to reducing the uncertainties surrounding changes in forest carbon stocks. Classification of primary forest, selectively logged forest, and degraded forest is likewise prone to error (Herold *et al.* 2011, Bucki *et al.* 2012, GOFC-GOLD 2013).

The UNFCCC has repeatedly called for the full and effective participation of Indigenous Peoples and local communities in carbon measuring and monitoring, as described in paragraph 3 of Decision 4/CP.15, paragraph 72 of Decision 1/CP.16, paragraph 2 of Appendix I, and paragraph 71(d). The specific guidelines on how to engage Indigenous Peoples and local communities has been left to the discretion of the individual countries that are implementing REDD+ (SBSTA 2009; Skutsch and Trines 2011). Alternatively, it has been also highlighted that the evaluation of accuracy of locally based forest monitoring is a key task for monitoring REDD+ systems, for instance, Danielsen *et al*. 2011 have emphasized that "further quantitative assessments of the ability of locally based forest monitoring methods to detect changes in forest condition are needed".

This paper examines the extent to which local knowledge, through participatory mapping, could improve the accuracy of land cover classification. Participatory mapping is a powerful tool that allows the inclusion of key local knowledge about location, land cover and land use history of the landscape and serves *to help communities make land use decisions* (NOAA 2009; Coomes *et al.* 2011; Danielsen *et al.* 2013). During the past decade, participatory mapping has become widely popular in both developing and developed countries (Corbett 2009). While there are several participatory approaches, ranging from low-resources and low-cost activities to high – resources and high-costs and low input tasks, its application will depend on how the final map will be utilized, the expected impact of the resources to be utilized and its accuracy, and the resources available in the project (Corbett 2009; NOAA 2009). Different forms of technological support have been utilized in its implementation, including satellite images, aerial photographs, global positioning systems (GPS), and geographic information systems (GIS), among others (Corbett 2009). We used here a combination of local knowledge, training in image interpretation and technological tools (satellite images and GPS devices) as a way to increase accuracy of land cover classification.

In Tanzania and Nepal, Skutsch *et al.* (2011) have demonstrated that participatory carbon measurements can be reliable, given that they observed no more than a 5% difference in the estimates of mean carbon stocks between professionals and the community. We are not aware of similar evaluations for participatory mapping that employs digital image processing techniques. Here, we compare the accuracy of two land cover maps in this article: one that uses participatory methods and another that uses a digital image classification, which is based upon a decision tree. Our objective was to determine if locally produced maps could provide reliable information in the context of REDD+. The study took place in the complex landscape of the Emberá people in the Bayano area of eastern Panama, where multiple successional forest stages and forest structures are present.

METHODS

Study area

The study took place in indigenous territories that are located in the Province of Panama, close to the Pan-American Highway and Bayano Lake (78°30’ - 78°49’ W, 8°54’- 9°05’ N). These territories are under the authority of the General Congress of the Collective Lands of Alto Bayano (CLAB), and include the collective lands of Ipeti (3285 ha), Piriati (3869 ha) and Majé Emberá-Drua (18920 ha) (Figure 1). Elevations in the CLAB territories range from 60 to 1080 m above sea level, with the highest areas in Majé. The territories are covered by "tropical moist" and “premontane wet” forest, according to the Holdridge Life Zone system (Smithsonian Tropical Research Institute 2013). Average annual precipitation ranges between 2000 and 3000 mm at high altitude. Annual temperature averages 26º C in the lowlands and 22º C in highlands, with a pronounced dry season from December to April (Autoridad Nacional del Ambiente 2010). CLAB is inhabited by ~1,500 Emberá People, who constitute one of three indigenous groups in eastern Panama that migrated from Colombia to the Bayano region in the 1950s. Indigenous territories in Panama are constituted as legally recognized areas (comarcas) and as areas being claimed by indigenous groups who wish to obtain legal recognition (claimed lands). Claimed lands in Panama are based on customary ownership. CLAB correspond to a claimed land and is currently in the process of legalization under the country's Law 72 (Gaceta Oficial, 2008) and Decree 223 (Gaceta Oficial, 2010). Primary economic activities include subsistence cultivation, cattle ranching, day laboring, and handicraft production (Tschakert, Coomes and Potvin 2007) (Additional information in Appendix S1).

Land cover classification

The mapping was based on two preprocessed 5-meter resolution multi-spectral RapidEye® images (Appendix S2) that were taken on February 5th, 2012, where terrain images containing clouds and cloud shadows were excluded. This yielded total areas of 2685 ha, 14723 ha and 3083 ha, respectively, for Ipeti, Maje, and Piriati. These net areas were used as a reference for all subsequent analyses. Our methodology evaluates the accuracy of participatory mapping in terms of land cover classification in relation to satellite image classification that was based on a decision tree.

Land cover participatory mapping

The first step in this project was to obtain authorization to determine the land cover in the CLAB in a participatory manner. Therefore, we held meetings with local traditional indigenous authorities of each local congress of the CLAB to explain the purpose and objectives of the mapping and request the necessary authorization. During these meetings it was agreed to test to use a combination of local knowledge, image interpretation and satellite imagery to increase accuracy of the land cover maps in the CLAB. After receiving a written authorization for every local congress of the authorities of the CLAB, we also informed the *Coordinadora Nacional de los Pueblos Indígenas de Panamá* (COONAPIP, National Assembly of Indigenous Chiefs of Panama). We then carried out participatory land cover workshops in Ipeti, Piriati, and Maje in February 2012. The workshops were jointly coordinated with the local traditional authorities. A total of 95 participants attended the workshops (27 in Ipeti, 45 Piriati, and 22 in Maje). During the workshops, a printed RapidEye® satellite image of the territory, including borders and other geographic landmarks such as villages, roads and rivers, served as a base map.

We also brought a blank map where the satellite image had been extracted, but the aforementioned land-marks were included. At the onset of the workshop, the attendees (including local traditional authorities and landowners) discussed how to reach a consensus for the land cover classes in their territories. For all territories, the land cover classes that were adopted included primary forests, intervened forests (logged forests), tall fallows, short fallows, plantations, pastures, cultivation, bare soil, communities (villages), and water bodies. During the second part of the workshop, landowners were invited to identify their parcels and they assign the corresponding land cover categories that had been previously defined for that portion of territory. The satellite image was used to guide the classification; meanwhile, the blank map was used for drawing the interpreted areas of the satellite image. To complete the mapping exercise, we visited the landowners who were unable to attend the workshops. The exercise was explained and they then classified their plots using the same classes that had been adopted during the workshop. In addition to the workshop participants, over 80 landowners were visited and consulted at this stage: 48 in Ipeti, 21 in Arimae, and 14 in Maje. The final map was presented and validated by the attendants at a later meeting (Appendix S3).

Land cover mapping using remote sensing

Decision tree classification (DT) was used to create a second set of land cover maps that were based on the spectral and textural attributes of the RapidEye images. DT is a hierarchical, method that involves recursive partitioning of a training data set, which is separated into increasingly homogeneous subsets on the basis of tests that are applied to one or more of the feature values or attributes (Pal and Mather 2003) (Appendix S4). In this method, binary splits are performed according to maximum likelihood tests that are based on one (univariate) or several predictor variables (multivariate) or, in the case of other methods, are based on formal *t*-, *F*- or chi-square tests. DT belongs to the larger family of machine-learning approaches that include vector support machines, artificial neural networks, classification, and regression tree analysis.

Two variants of the DT method were employed in our analyses, with one correcting for reflectance values of the five bands in the RapidEye images (DT1) and the other (DT2) adding eight textural features to the input data (Appendix S5). Training areas (subset of the data) were selected in the RapidEye images using ENVI-5.0® software (https://www.exelisvis.com/envi-5/) for the same land cover categories that had been defined in the participatory maps to make the classifications comparable. All training areas were selected from "pure" spectral and homogenous areas so as to choose the most appropriate categories and, thereby improve classification (Lillesand and Kiefer 2009). The training areas are based on a priori knowledge of the region, including field knowledge and scientific sources. Training areas represented 4 % of the total study area.

Training areas were also used to define threshold values for the nodes and branches of the decision tree. Decision tree classification was performed using the Waikato Environment for Knowledge Analysis (WEKA; http://www.cs.waikato.ac.nz/~ml/weka/), which is an open-source data mining software suite that includes machine-learning algorithms for data mining tasks. The J48 decision tree algorithm of Quinlan (1993), which is available within WEKA, was used for training the RapidEye image dataset (Appendix S6). The resulting rules that were generated were implemented on the RapidEye satellite image data for classification. This work was also carried out in ENVI 5.0 software.

Ground Truthing and Accuracy Assessments

As defined by Foody (2002), classification accuracy is the degree of "correctness" of a map or classification. Field assessment of accuracy was carried out using two sets of validation data. The first data set consists of 56 randomly selected GPS points in the areas of Ipeti (20) and Piriati (36). These points were selected in ArcGIS 10.1® and visited on the ground in July 2013 by trained indigenous technicians, who identified the associated land cover. The second set of independent data consisted of 38 forest carbon plots that had been measured in the areas of Maje (16) and Ipeti (22) from July to August 2012 (Figure 1). These 25 m-radius plots were measured by local indigenous technicians, who had been trained in forest mensuration, and which were established in areas that covered a full range of elevational and human intervention gradients, where land cover of these points had also been registered (Appendix S7). The accuracy classification of the three final maps was then evaluated using a confusion matrix (Congalton and Green 2002), which estimates a classification error indicating a discrepancy between the situation that is depicted on the map and the reality that was observed in the field (Foody 2002). Kappa (κ) inter-rater reliability assessments that compared results of the three classifications were carried out (Cohen 1960), where a theoretical maximum value of 1 represents complete agreement between a given classification method and the field data. In addition, two tests were used to identify significant differences among methods, i.e., Cochran's Q (among all classification methods) and McNemar tests (McNemar 1947) (Appendix S8).

RESULTS AND DISCUSSION

Our analysis revealed substantial variation in the classification of land cover among the methods that were considered (Figure 2). Participatory mapping maximizes the areas of forests (4,878 ha) and of grasslands (4,667 ha) in which intervention had taken place, while DT1 maximizes primary forests (11,771 ha) (Figure 3). DT1 further yields the lowest coverage of tall fallows (981 ha) of all methods. Significant differences were found in the number of correct cover categories that were produced by the three classification methods (Cochran's Q test: χ^2 (2) = 20.26, $P < 0.05$), while pairwise comparisons using McNemar's test revealed significant differences between DT1 and DT2 ($P < 0.001$), and between DT1 and the participatory mapping ($P < 0.005$). Participatory mapping had the greatest overall accuracy (83.7 %, κ = 0.783), followed by DT2 (79.9 %, κ = 0.757).

Participatory mapping accuracy varies from 20% for bare soils to 100% for primary forests, grasslands, and water (Table 1). The bare soil category has the lowest accuracy, given that this category was apparently confounded with grasslands and short fallows. In forested areas, participatory classification was respectively 100% and 97% for primary and intervened forests, which was significantly higher than classification that was based on digital image processing. The two classification methods that were based on decision trees apparently overestimated primary forests while underestimating forests in which there had been intervention. Classification that was based only on remote sensing, however, had high accuracy for tall and short fallows (Table 1).

Indigenous Peoples who participated in this study demonstrated a high degree of knowledge regarding the land cover and historical land use of their territories, which we validated on the ground. We assume that the higher accuracy of the participatory approach -in identifying primary and intervened forest- is a result of this local knowledge that allows increasing land cover and forest degradation detection. A similar observation was made by Danielsen *et al.* (2013) for identifying forest strata in Indonesia, China, Laos, and Vietnam. In tropical countries where slash-and-burn agriculture is practiced, the development of vegetation from recently cleared forests to short fallows, then to tall fallows and more advanced second-growth forests makes the implementation of land cover classifications a challenging task (Pelletier *et al.* 2011). This is particularly relevant in areas where indigenous forms of agriculture produce a complex landscape mosaic of grasslands and annual crops that are interspersed with areas in different regrowth stages, as well as older forests in more inaccessible areas (Tschakert *et al.* 2007). Such a complex and highly dynamic land cover makes it difficult to achieve high accuracy solely through digital image classification that is based on decision trees. Our results show that in digital image processing, intervened forests are easily confounded with primary forest and that local knowledge could more efficiently contribute to differentiating these otherwise relatively similar forest types. According to GOFC-GOLD (2013), digital image processing is of limited use in identifying logged areas and human interventions that result in forest degradation. With gradual losses of biomass and the creation of small clearings in the canopy, forest degradation cannot be effectively measured using standard optical remote sensing methods, since their resolution is too coarse or the effects of logging too well-hidden to be detected either visually or by computer analysis (DeFries *et al.* 2007). The complexity that is involved in identifying more subtle changes in vegetation has triggered the identification of proxies (i.e., road proximity) for determining these potential impacts on the forest and simplifying the identification of forest degradation (Bucki *et al.* 2012). We have demonstrated for the first time that local knowledge can improve land cover classification and facilitate the identification of forest degradation. The plea of the UNFCCC for the full and effective participation of local and Indigenous Peoples could indeed improve the accuracy of monitoring.

One caveat needs to be kept in mind. Our results show that the accuracy of participatory maps varied according to the three territories in the CLAB, the Ipetí map had the highest level of accuracy (0.925, κ= 0.87) and the lowest one (*0.67,* κ =0.58) in Piriati. We have identified two main reasons behind these differences. Firstly, the areas with lower accuracies in the three territories present a greater extension of grassland and short fallows. Most landowners in these territories labeled bare soils short fallows as grasslands, suggesting that landowners in these areas tend to classify the parcels according to its land use instead of its land cover. Secondly, high accuracy in the Ipeti area is not surprising because many leaders and local dwellers have had an extensive experience in working with other land cover classification and carbon projects for more than ten years (Kirby and Potvin 2007, Potvin *et al*. 2007). While Danielsen *et al.* (2013) argued that even local stakeholders with limited education can measure forests with acceptable standards, the differences that they observed among villages suggest that prior training can help improve the detailed spatial knowledge of territories. If participatory mapping is to be successfully incorporated into the REDD+ tool-box, we propose that the preparation and training of local dwellers in interpreting basic aspects of aerial or satellite images becomes a fundamental step before any participatory mapping exercise takes place (Rambaldi 2010). In doing this, the trainers should avoid complex aspects and terminologies of conventional scientific methods, and keep the training stage as simple as possible (Fry 2011).

Finally, we must concur with Danielsen *et al.* (2013) that involvement of local communities could improve the capacity of many developing countries for monitoring forest emissions at a reasonable cost and within a short time-frame. It has shown that local knowledge is a valid option that complements satellite imagery, but participatory mapping could also be helpful in resolving issues that are related to cloudiness, a pervasive problem for many countries in the humid tropics. Complementarity between locally –based data and remote sensing data can also be valuable to identify land cover areas with similar spectral properties (training areas) for areas that are not under the control of local communities but where national governments can have satellite coverage to generate land cover maps.   Meanwhile different communities can propose different land cover classifications making difficult to manage in a REDD+ national context such as Measuring, Reporting and Verifying (MRV) systems, it could be translated to a more general or standardized system to be utilized in a national context. Given the importance of Indigenous Peoples as forest custodians in Panama and many other Latin American countries (Vergara-Asenjo and Potvin, 2014), engaging them in forest monitoring under REDD+ appears to be a win-win opportunity for improving mapping accuracy, while also unlocking the sometimes complex relationship between Indigenous Peoples and national REDD+ strategies. In moving away from the fear of REDD+ (Potvin and Mateo-Vega, 2013), Indigenous Peoples and local community participation in forest carbon assessment or in national forest inventories could establish a new starting point that is based on real collaboration and mutual benefits.

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TABLES AND FIGURES

Table 1. Accuracies (%) of predicting land cover categories and classification methods, relative to field observations. Classification methods include: participatory = participatory method, DT1= decision tree using regular RapidEye bands, DT2 = DT1 plus textural features.

Land class	Participatory	DT1	DT2
Water	100	99.9	100
Grasslands	100	86.3	88.4
Bare Soils	20	49.7	61.3
Short Fallows	53.8	98.0	97.8
Tall Fallows	83.3	97.8	97.6
Intervened	96.8	55.9	72.0
Primary	100	39.8	56.2

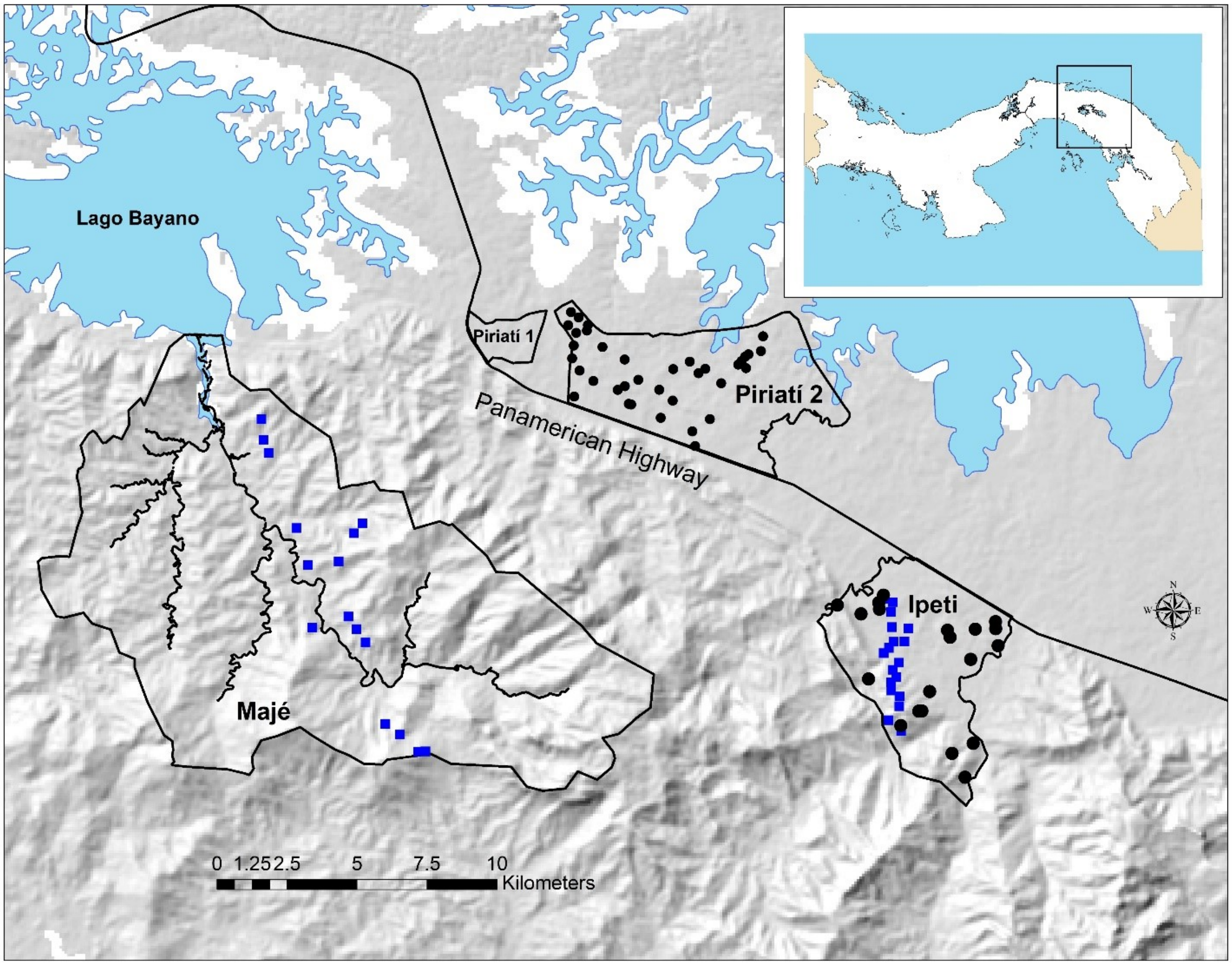

Figure 1. Bayano area showing the three indigenous territories that were included in the study. Datasets that were used for validation, including randomly selected points (circles) and forest inventory plots (squares), are also shown in the map.

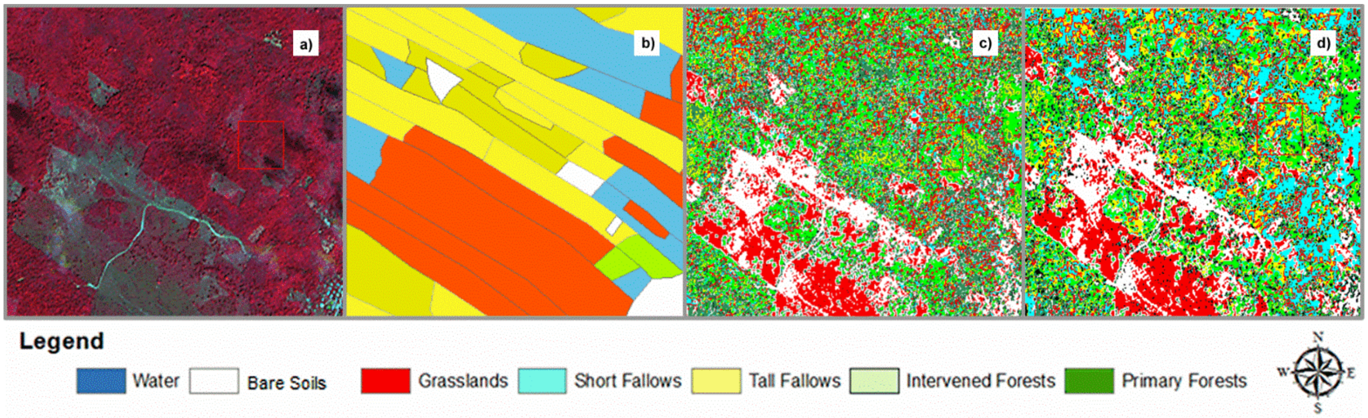

Figure 2. Satellite image and land cover classification maps for a section of the Ipeti territory (a) RapidEye image standard false-colour composite. (b) Participatory classification. (c) DT1 classification with the five spectral bands. (d) DT2 with five spectral bands and textural features.

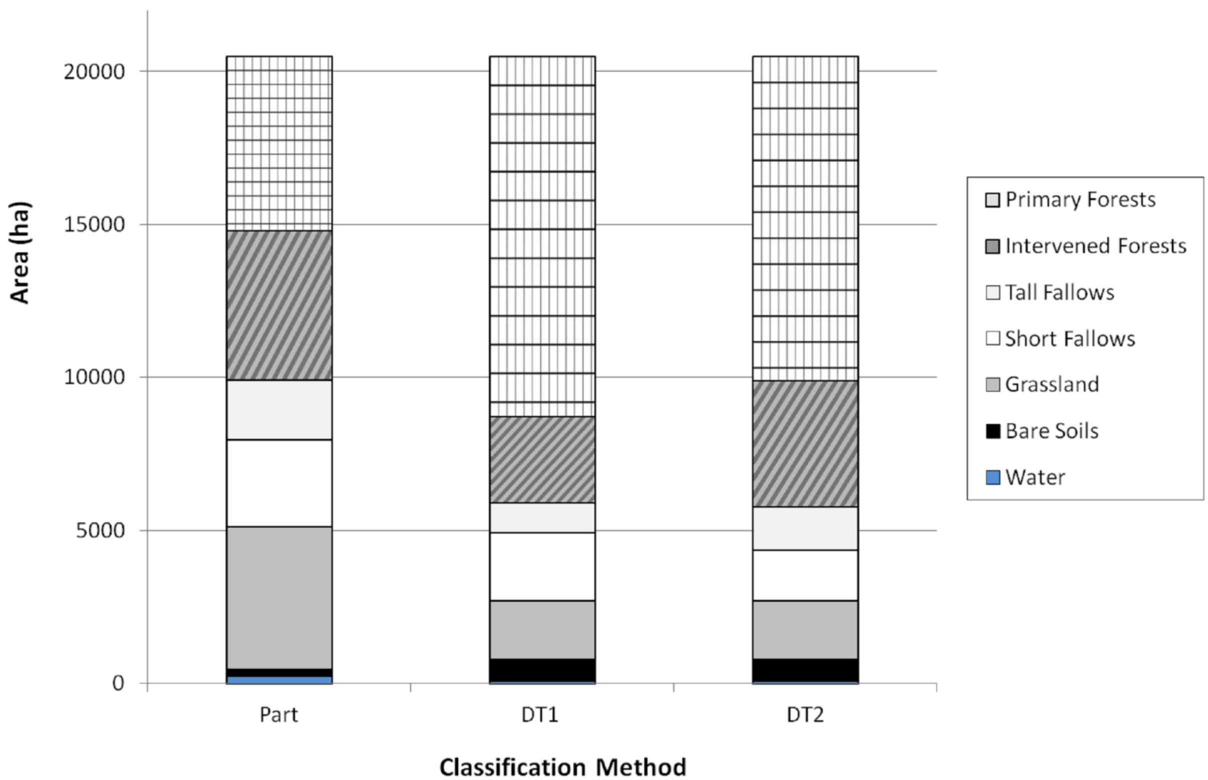

Figure 3. Area distribution using three different classification methods in the CLAB.

Linking statement 3

Among the information received from interviews with local communities and the revision of historical documents from Chapters 1 and 2, complex land tenure conflicts were identified across the country. In Chapter 3, I evaluate the importance of tenure security and its relation to forest conservation using the Upper Bayano watershed in Eastern Panama as a case study of complex land tenure dynamics in the context of forest conservation through REDD+ implementation. Here, I use and validate free forest cover data (2001-2014) to estimate the importance of land invasions in indigenous territories as a key element of the rapid expansion of the agricultural frontier. Using two spatial models, I also estimate potential costs of carbon emissions for 2015-2024 to explore potential future impacts of tenure and tenure conflicts in the watershed.

CHAPTER 3:

Nothing to lose? The impact of deforestation and invasions on forest conservation initiatives

Status: Vergara-Asenjo, G. & Potvin, C. Nothing to lose? The costs of deforestation and forest invasion in Eastern Panama. *In Preparation.*

ABSTRACT

Reducing emissions from deforestation and forest degradation (REDD+) can decrease emissions of greenhouse gases and provide considerable benefits for biodiversity and livelihoods. Land tenure and tenure security are among the most important factors determining the success of largescale REDD+ programs. In this paper, we use the Upper Bayano watershed in Eastern Panama as a case study of complex land tenure dynamics in the context of forest conservation through REDD+ implementation. Using and validating free forest cover data (2001-2014), I estimate the importance of land invasions in indigenous territories as a key element of the rapid expansion of the agricultural frontier, showing that tenure alone is insufficient to guarantee rights over land and forests in indigenous territories. We advocate that REDD+ strategies among developing countries can improve forest conservation and secure livelihoods on-the-ground and that conflict resolution mechanisms and mediation processes might provide an avenue to resolve long-standing land conflicts.

INTRODUCTION

Reducing emissions from deforestation and forest degradation (REDD+), an international mechanism of the United Nations Framework Convention on Climate Change (UNFCCC), promotes incentives to reduce, avoid or sequester carbon dioxide emissions from forested lands, offering a way to "internalize" the value of this ecosystem service the forests provide (Angelsen *et al*. 2009, Hamrick and Goldstein 2015). As REDD+ was being negotiated, it soon became apparent that reducing deforestation to conserve carbon sinks could entail important co-benefits, for example, with respect to biodiversity conservation and rural livelihoods (CBD 2011). It has indeed been shown that, at the global scale, there is a strong positive relationship between forest carbon stocks and tree species richness largely because tropical ecosystems are both species- and carbon-rich (Strassburg *et al*. 2009). In that context, REDD+ can add important new tools to the conservation agenda (Harvey *et al.* 2010).

Several authors have claimed that land tenure and tenure security are critical to achieving emission reductions and ensuring potential success of REDD+ strategies (Cotula and Mayers 2009, Sunderlin *et al.* 2009, Robinson *et al.* 2011, Larson *et al.* 2013). Tenure security – defined as the certainty that a community's land rights will be recognized and protected if challenged (Gray *et al.* 2015) – is one of the most prominent influences on deforestation, forest degradation and the spread of extensive ranching as the dominant driver of land use change in Latin America (Fearnside 2001, Araujo *et al.* 2009, Larson *et al.* 2010). Land tenure also directly influences REDD+ outcomes by defining who is eligible to receive benefits while influencing the ability of recipients to enforce carbon contracts (Naughton-Treves and Wendland 2014).

In this paper, we use the Upper Bayano watershed in Eastern Panama as a case study of complex land tenure dynamics in the context of forest conservation through REDD+ implementation. Panama, one of several developing countries engaged in REDD+, presents 54% of its mature forest cover in indigenous territories, meanwhile 65.7% of the Upper Bayano watershed is under indigenous tenure (Vergara-Asenjo and Potvin 2014). Notwithstanding this, Panama faces multiple challenges due to Indigenous Peoples' fear of threats to traditional land uses and rights (Potvin and Mateo-Vega 2013). Since the Panamanian Government constructed the Ascanio Villalaz hydroelectric dam in 1976, flooding lowlands to create the 350 km^2 artificial Bayano Lake and forcing Indigenous Peoples to resettle to other areas in the watershed, the region has been plagued with extended land tenure disputes and land invasions of colonist settlers in indigenous territories (Wali 1989).

In the Bayano watershed, land invasions have occurred in indigenous territories regardless of their legal land tenure status. Land invasions in recognized or claimed indigenous lands stem from colonist farmers' interests and visions about natural resource use and management, seeing the largely forested indigenous territories as "free and unused" (Peterson-St Laurent *et al*. 2012). This suggests that discussions of the role of land tenure in REDD+ implementation must be broadened to include enforcement of indigenous collective land titles (additional information in Appendix S1). Invasions of indigenous territories have indeed been the principal driver of deforestation and land use change in the Bayano watershed (Sloan 2008, Peterson-St Laurent *et al.* 2012).

The main objective of this study is therefore to determine deforestation associated with land conflicts in the Bayano watershed as well as the "carbon" cost of deforestation, to propose directly applicable solutions for conflict resolution and tenure security by policymakers as an integral part of REDD+ implementation. Evaluating the environmental costs of deforestation in relation to tenure regimes, in particular in areas of insecure or unenforced property rights, can help inform the impact of losing ecosystem goods and services from forests, as well as stimulate identification of ways to improve forest governance in practice (Cotula and Mayers 2009).

To meet our objective, we documented land cover dynamics for the Upper Bayano watershed, using forest cover change maps of Hansen *et al.* (2013). Accordingly, an ancillary objective is to verify the potential of these maps to develop REDD+ reference levels. The map of Hansen *et al.* (2013) is the first freely available global, wall-to-wall, annual change map, using time-series analysis of Landsat images at 30m spatial resolution, that can be translated to carbon emissions for REDD+ accounting. Thus, we think that Hansen's maps have the potential to enable the development of reference levels of deforestation for community groups that lack the financial means to pay for satellite imagery and the technical ability to analyze remote sensing. We propose that using Hansen's maps also offers the possibility of developing a global methodology, making the results comparable across countries or jurisdictions. We therefore used Hansen's maps to determine the extent of deforestation resulting in emissions in the Bayano region. We combined deforestation taking place during a given period of time, also known as activity data (GOFC-GOLD 2015), with estimates of aboveground biomass derived from field inventories carried out in the Bayano to develop an emissions reference level for the historical period 2001-2014. Projecting deforestation using a land change model for the period 2015-2024, we estimated resulting carbon dioxide emissions.

METHODS

Study area

The study took place in the Upper Bayano watershed, a 3,695 km^2 area in Eastern Panama (78°30' - 78°49' W, 8°54'- 9°05' N) (Figure 1). The gap-filled Shuttle Radar Topography Mission (SRTM) digital elevation model (DEM) version 4 (Jarvis *et al.* 2008) at 90 m resolution, from the CGIAR Consortium for Spatial Information (http://srtm.csi.cgiar.org/), was used to delineate the watershed boundaries.

According to the Holdridge Life Zone system, the Bayano area is covered by "tropical moist", "premontane wet" and "tropical wet" forests (STRI 2013). Elevations in the watershed range from 60 to 1,080 m above sea level, with the highest areas in Majé. Average annual precipitation ranges between 2,000 and 3,000 mm at high altitude. Annual temperature averages 26ºC in the lowlands and 22ºC in highlands, with a pronounced dry season from December to April (ANAM 2010).

Identification of invasion areas

One of our first tasks was to determine the importance of invasion as a causal agent of deforestation in the study area. The national map of Panama's indigenous territories that we previously created allowed us to determine the size of indigenous territories in the Bayano watershed (Vergara-Asenjo and Potvin 2014). Identification and mapping of areas of invasion inside indigenous territories (Figure 1) was undertaken using historical documents from government and non-governmental institutions, which offered descriptions, locations and other relevant details about land use conflicts. Local documents from indigenous organizations and 20 informal interviews with indigenous leaders in the Bayano area and government technicians complemented the archival search and helped document current and unresolved invasions. A 3-dimensional model of the Bayano watershed further helped us locate land use conflicts (Guillemette *et al.* 2016). Participatory validation of all the information obtained on invasion areas was then carried out with indigenous leaders and technicians in three workshops, one with the Guna and two with the Embera, in November 2015. The workshops were conducted in three different communities (Akua Yala, Piriati and Majé) to facilitate participation with a total of 25 participants. Finally, three specific areas identified as invasion sites were visited for field validation after the workshops, as participants felt that limits of invasions should be clarified.

Forest cover and forest cover change in the Bayano

To analyze forest cover and forest cover change in the Bayano watershed, we integrated remote sensing and GIS analysis. Hansen *et al*. (2013) derived their forest cover change data from Landsat imagery over the period 2000–2014 using multispectral satellite imagery from the Landsat 7 thematic mapper plus (ETM+) and Landsat 8 Operational Land Imager (OLI) sensors (Appendix S2).

High quality spatial data are scarce for the Bayano region, as extreme cloud cover precludes the availability of extensive time series of remote sensing data. As part of our evaluation of the accuracy of global forest cover and forest cover change classification maps (encoded in binary format, with 1 as loss or 0 as no loss) produced by Hansen *et al.* (2013) at the local level, we randomly selected a total of 173 points on Hansen's maps and assessed their accuracy in the field for both forest cover and forest cover change in 2013 and 2014 (Figure 2). The accuracy classification of Hansen's maps was conducted using a confusion matrix (Congalton and Green 2002, Vergara-Asenjo *et al*. 2015), which estimates a classification error that indicates a discrepancy between the situation depicted on the map and the reality observed in the field (Foody 2002). Producer and user accuracies – two measures of omission and commission errors, respectively – were also estimated to obtain an assessment of how well specific land categories in Hansen's maps were classified (Additional details in Appendix S3).

We combined activity data derived from Hansen's maps for 2001-2014 with emission factors to derive a reference level of deforestation among indigenous territories and colonist areas in the watershed and determine the impact of invasion on forest emissions. A predictive analysis of deforestation quantifying potential impacts on forests in incoming years was then carried out using the Land Cover Change (LCM) module of Terrset software (Eastmann 2015) and two different models for the period 2015-2024. To provide a rough estimate of the cost of land invasion in a carbon-constrained world subjected to carbon pricing, we performed a back-of-the-envelope calculation based on historical carbon prices (Appendix S4).

RESULTS AND DISCUSSION

Estimating activity data for land use change in the Bayano

Our assessment of Hansen's maps of forest cover and forest cover change in the Upper Bayano watershed shows a high overall accuracy of 84.9% and 85.1% for the years 2013 and 2014, respectively (Tables in Supplementary Information). Both user and producer accuracy are greater than 80%. We noted that Hansen's maps tend to overestimate forest areas in some zones due to inclusion of short fallows, shrublands and non-forested riparian vegetation (less than 5 m tall) around Lake Bayano as forests (user accuracy of 80%). However, overall, Hansen's maps meet the recommendation of Thomlinson *et al.* (1999) of an overall accuracy of 85% with no class less than 70% accurate. Accordingly, we consider it an adequate tool to monitor deforestation in the Upper Bayano watershed and suggest that Hansen's maps are appropriate wherever stakeholders need to develop reference levels of deforestation, and for monitoring forest loss at low or no cost.

After verifying the accuracy of Hansen's maps at the local scale, we therefore used them to analyze differences in gross deforestation in indigenous territories and colonist areas. The areas with highest forest cover in 2000 correspond to two indigenous territories, Madungandi and Majé, with 99.1% and 98.5%, respectively. Colonist areas conversely present the lowest forest cover in the Upper Bayano watershed, with 74.2% in 2000. For the period 2000-2014, 18,153 ha of forests were cleared in the Bayano watershed, that is, 5.9% of the total forest area in 2000. The three Embera territories, Majé, Ipeti and Piriati lost 14%, 13.8% and 13.7% of total forest area, respectively, while the Guna territory of Madungandi lost 4,515 ha of forest, a decrease of 2.28% of the total forest (Figure 3 and 4). During the same period, colonist lands lost 10,191 ha, a reduction of 8.9% of the total forest area. In contrast to an annual rate of forest loss from 0.18% in the early 1990s to 0.08% of the world's forests during the period 2010-2015 (FAO 2015), the Upper Bayano watershed shows a worrying net annual rate of forest loss of 0.42% for the period 2000-2014, higher than the 0.35% reported by Panama for the period 2000-2012 (FAO 2015b) (complementary results in Appendix S5).

In indigenous territories, deforestation was categorized as internal (produced locally) or external (invasions) to calculate the effect of colonist invasion. Our analysis shows that, in the entire Bayano watershed, 5,006 ha of forest clearing (27.6% of total deforestation) resulted from invasions of indigenous territories. This situation is the paramount concern in the Embera territory of Majé, with 95.4% of total deforestation (14% of total forests in the territory) caused by invaders. In the Comarca of Madungandi, colonist invasions account for 61.2% of deforestation for the period 2001-2014. Total deforestation produced by invaders in Ipeti is low at 7%, and there is no deforestation associated with invaders in Piriati. Deforestation in indigenous territories occurs in non-habited forested zones, where monitoring and control by communities to restrict access to invaders is more complex.

These results enrich discussions stressing the importance of tenure security as an essential tenet of REDD+ implementation (Cotula and Mayers 2009). While the highest proportion of deforestation due to invasion is in Majé, the only indigenous territory in the Bayano without secured land tenure, the impact of invasions in the Comarca Madungandi accounted for more than half of total deforestation in the territory despite legal rights having been granted 20 years ago. Titling or granting of legal rights is clearly insufficient to fully protect forests in indigenous territories in Panama. We suggest that the broad legal framework of Panama could explain this situation. As in many Latin American countries, in Article 123 the Panamanian Constitution indicates that the State will not tolerate the existence of land that is uncultivated, unproductive or idle ("incultas, improductivas u ociosas"), adding that the State will provide titles to those who make the land "productive" (Foro y Observatorio de Sostenibilidad 2015). Colonist farmers that invade forested indigenous lands therefore consider that they are doing so in the context of their constitutional rights (Amado *et al*. 2014). This perception contrasts starkly with the recommendation that local monitoring of forests and sanctioning of illegal activities such as deforestation are essential for forest conservation and governance for sustainable livelihoods (Ostrom and Nagendra 2007). We suggest that REDD+ readiness therefore demands fundamental reforms of the legal system, resolving not only land tenure but also the broader policy framework to remove perverse incentives.

Projecting deforestation and estimating carbon emissions costs

We projected deforestation for the period 2015-2024 to understand potential impacts of land invasion in the Bayano watershed. Projected deforestation using Model 1 estimated an average of 1,296 ha yr^{-1} of deforestation for the period 2015-2024 – a rate similar to the reference period.

Projected deforestation in colonist areas is 743 ha yr^{-1}, that due to land invasion is 469 ha yr^{-1}, while internal deforestation in indigenous areas is only 84 ha yr^{-1}. Model 2 averaged 5,576 ha yr^{-1} of areas vulnerable to deforestation for the period 2015-2024 and a potential of 2,017 ha yr^{-1} of deforestation from land invasions. Combining emissions factor with the estimated projected loss in forest area allowed us to develop a historical reference level for emissions and to make a projection. Model 1 estimated 160,000 and 157,000 Mg CO_2 eq. yr^{-1} for colonist areas and Comarca Madungandi respectively. An average of 75,000 Mg CO_2 eq. yr^{-1} was estimated for Maje, meanwhile projected emissions in Piriati and Ipeti were low with 10,000 and 6,500 Mg CO_2 eq. yr^{-1} respectively (Figure 5).

A back-of-the-envelope cost analysis indicates that historical costs of deforestation are ~US$11.3 million, with ~US$2.7 million corresponding to deforestation resulting from land invasions. If Panama implements REDD+ or a similar scheme of payment for ecosystem services, the lost revenues from a hypothetical carbon market (at US$5/tonne) for combined historical and projected (2015-2024) deforestation – using scenario Model 1 – would be US$56.1 ±27.7 million, with US$14.1± 6.29 million associated with land invasions. Using Model 2, estimated potential income loss rises to US$157.7 ± 74.46 million, with US$21.1 ± 6.13 million due to land invasions (Table 1). A useful exercise is to compare the foregone income from a hypothetical carbon market to the cost of forest protection. Oestreicher *et al.* (2009) estimated US$889,922 as the necessary yearly funding for effective protection of the Chagres National Park, a 129,000 ha protected area representing 80% of the Comarca Madungandi, one of the most effective protected areas in avoiding deforestation in Panama. Clearly, a strategy for reducing deforestation based on REDD+ payments that turns the income lost from deforestation into a revenue for the communities could support protection and make a strong contribution to forest conservation in areas under conflict. The Juma project in the Brazilian Amazon provides an interesting initial case study where the government invested not only in direct payment for conservation but also in education and health to support communities' reduced impact on the forest (Viana *et al*. 2009).

The costs calculated above provide a proxy value for conserving standing forest carbon stocks (Grabowski and Chazdon 2012) and constitute, according to our best knowledge, the first evaluation of the impact of land invasions on carbon storage as an ecosystem service. These estimated costs are conservative, as they include but a small portion of all goods and ecosystem services provided by forests.

Indigenous Peoples own extensive areas of land in Latin America (Larson *et al*. 2010), for example 25.3% of the Amazon region is under Indigenous Peoples' custody (Benavides 2009). Since indigenous tenure has demonstrated greater efficiency in controlling deforestation than private tenure (Nepstad *et al*., 2006), Indigenous Peoples should be considered essential partners in REDD+ and forest conservation programs. Regardless of whether rights have been recognized or not, the vast extent of tropical forests is threatened by colonists, illegal loggers, mining and oil extraction, and more, endangering not only the forests but also the existence of indigenous territory as a whole (Larson *et al*. 2010). Tenure is indeed conditioned by governance, as Cotula and Mayers (2009) indicated: "Effective tenure is both impossible to achieve without supportive policy and institutional systems, and rather useless without broader institutional capacity to do something with it".

An important question to highlight is: Why do conservation areas, which can include indigenous territories, need REDD+? The answer could be provided via the transition curve, a concept introduced by Mather (1992), which offers patterns that could apply to a forested country as it progresses on its developmental curve. In the transition curve, those countries characterized by high forest cover and low deforestation (HFLD), such as Guyana, could shrink their forest cover as they develop and as pressures on natural resources increase. The curve could also be applied to high forest cover indigenous territories where external pressures over forests, including land invasions, can increase deforestation if no measures are implemented. Considering these aspects of the curve, any potential reduction in emissions and increase in emissions offsets could support forest conservation efforts led by local communities and Indigenous Peoples, making REDD+ a potential mechanism to strength local capabilities in high forest cover areas.

Our findings also stress the importance of national policies and governance on land rights, in particular the application of effective measures to stop deforestation and land invasions. There are many avenues to resolve such a problem. The Republic of Ecuador, for example, paved the way for fundamental shifts in legislation when, in 2008 it approved a new Constitution granting rights to Nature. The application and real impact of those changes remain to be evaluated (Campaña 2013). Radical changes such as those made in Ecuador do not appear within reach in Panama in the short- to middle-term timeframe. As an alternative and given the necessity of resolving invasions as a key driver of deforestation, we propose conflict resolution mechanisms through mediation processes as a viable option. Amado *et al.* (2014) advocated the establishment of a conflict resolution process for REDD+ that would include relevant actors at the national and local scale, and which, supported by an independent monitoring commission, could offer a way to improve forest conservation in developing countries working under REDD+ approaches. We also hope that our estimates of the "carbon" cost of forest change in areas under conflict will contribute to stimulating adjustment of current policies and improve forest governance in order to apply more effective REDD+ strategies on-the-ground.

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TABLES AND FIGURES

Table 1. Carbon emission costs for historical and projected deforestation in the Bayano watershed.

Categories	Cost (US$)	SD (US$)
Historical Deforestation -Total	42,233,699	20,118,858
Historical Deforestation - Invasions	10,924,413	4,836,193
Model 1 - Total	13,868,606	6,613,164
Model 1 - Invasions	3,289,237	1,492,406
Model 2 - Total	113,543,451	54,360,774
Model 2 - Invasions	10,234,045	4,643,432

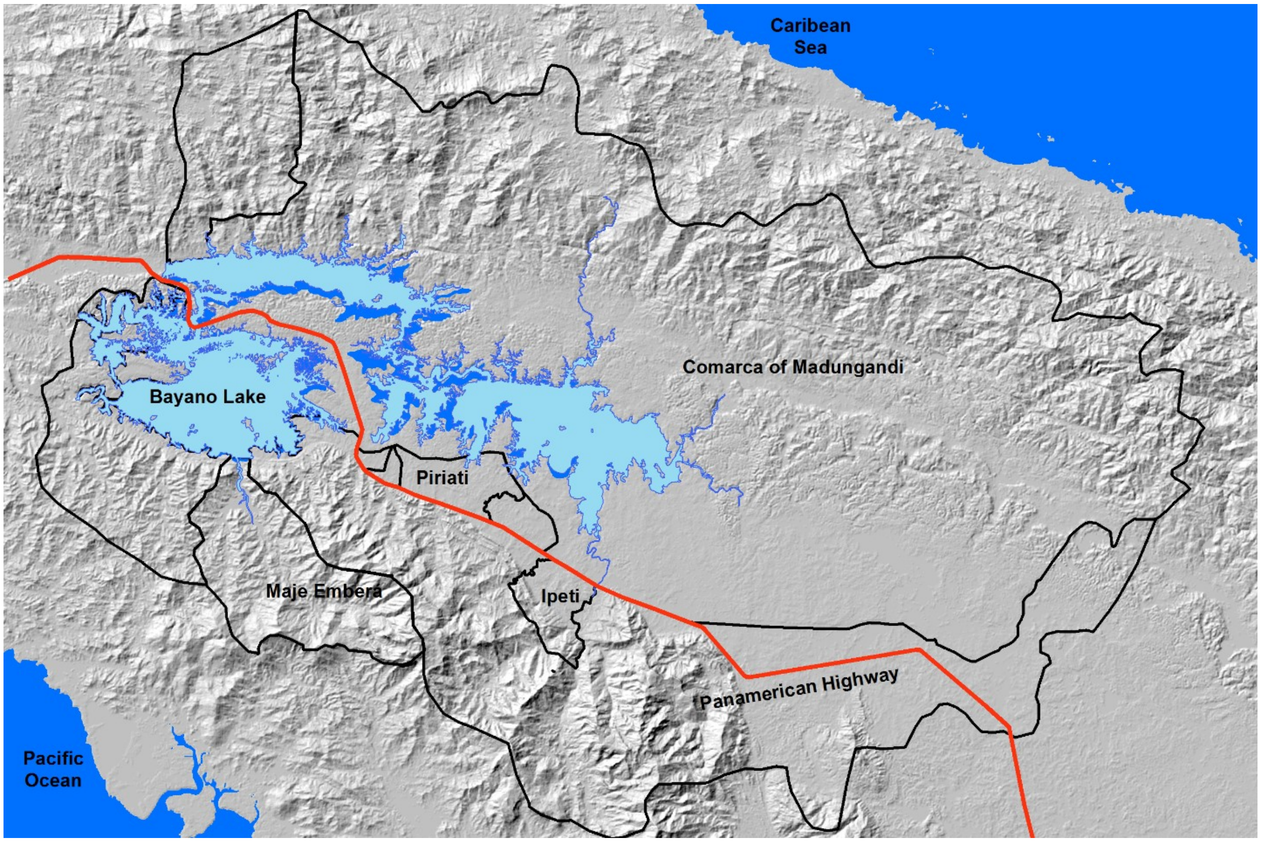


Figure 1. The Upper Bayano watershed in Eastern Panama with indigenous territories identified.

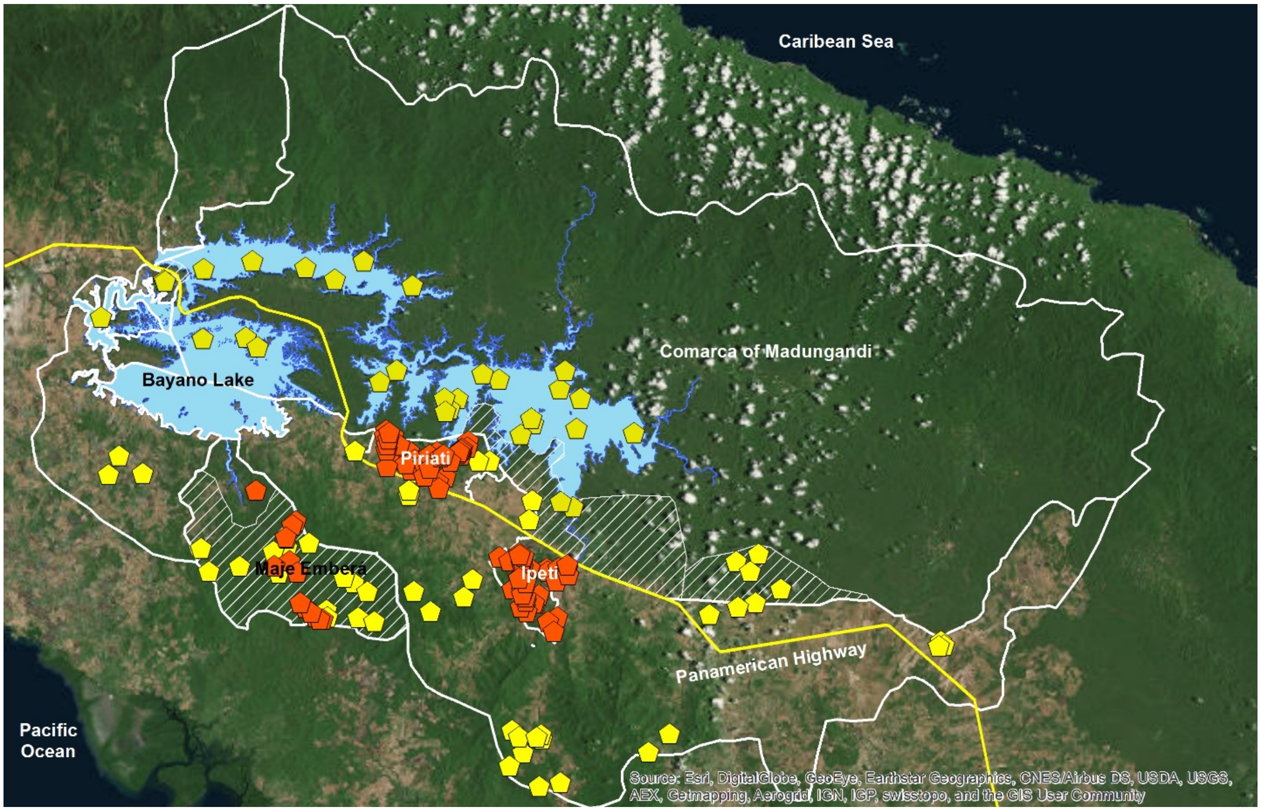
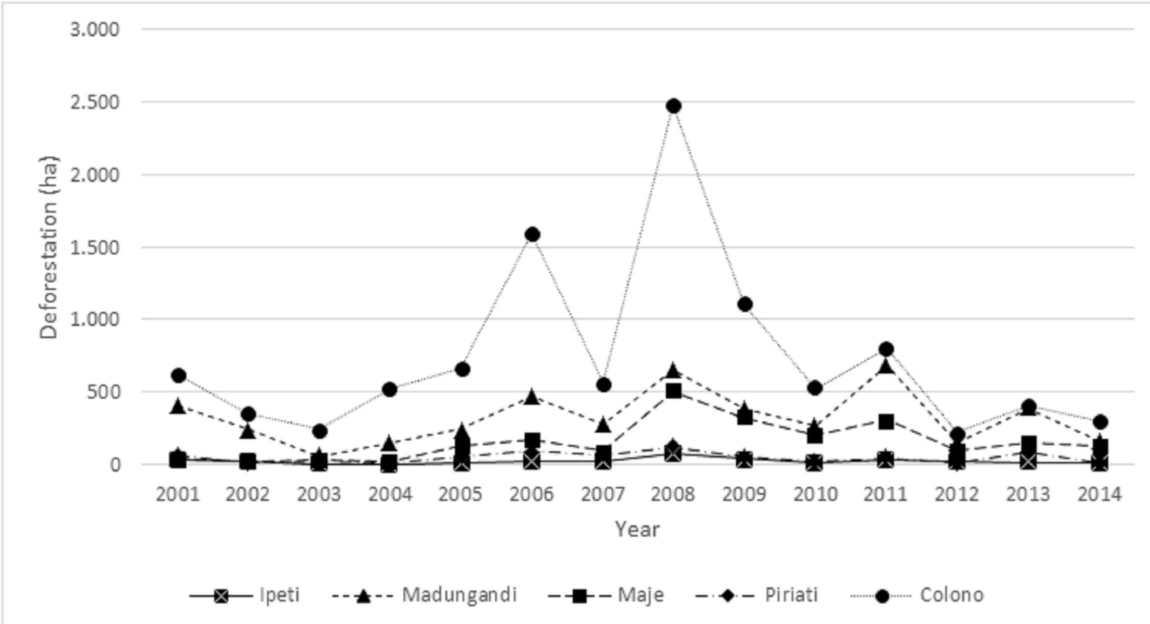


Figure 2. Ground truth data to assess Hansen's maps in the Upper Bayano watershed with Esri basemaps in the background. Forest inventory data are represented in red; validation points are represented in yellow. Areas under invasion influence are identified with hatched patterns.

Area	Forest Area- Year 2000 (ha)	Forest Area -Year 2014 (ha)
Colono	85,327	75,136
Ipeti	2,804	2,353
Madungandi	196,204	191,689
Maje	15,643	13,419
Piriati	2,997	2,477
Total	**302,975**	**285,074**

Figure 3. Change in forest cover between 2000 and 2014 in indigenous territories and colonist areas (graph in percentages and table in hectares).



Indicator	Colono	Ipeti	Madungandi	Maje	Piriati
Deforested area (%)	9.1	10.0	2.3	14.0	17.4
Average Deforestation per Total (ha yr^{-1})	743.3	23.4	323.7	159.0	47.3
Average Deforestation - Internal(ha yr^{-1})	743.3	22.0	123.6	7.3	47.3
Average Deforestation - Invasions (ha yr^{-1})	0.0	1.4	200.1	151.7	0.0

Figure 4. Graph of total deforestation (ha) and indicators of deforestation for the period 2001-2014 in indigenous territories and colonist areas.

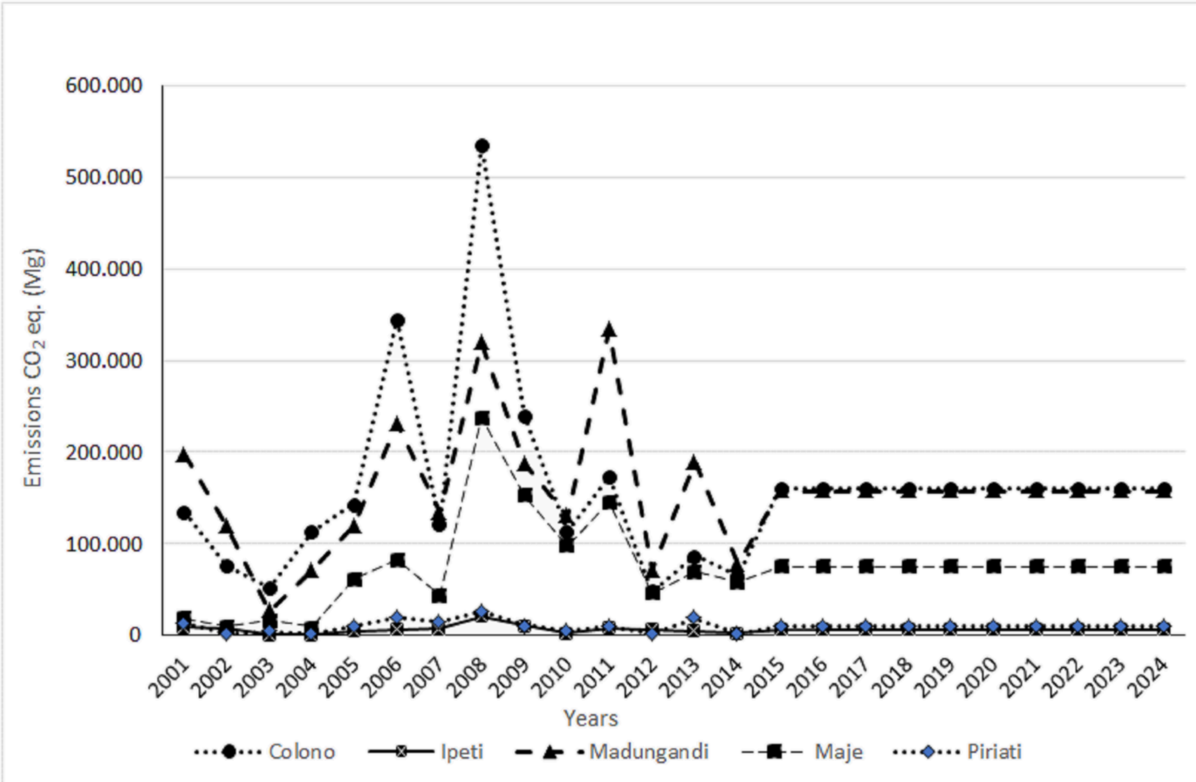


Figure 5. Historical and projected emissions of CO2 eq (Mg) in the Bayano watershed for 2001-2024 using Model 1.

Final summary and conclusions

Deforestation is a main cause of land-use and land-cover change and a source of carbon emissions at the global scale, threatening climate stability, biodiversity and local livelihoods (Foley *et al*. 2005, MEA 2005). The implementation of a potential REDD+ global strategy has enriched discussions of the conservation agenda, offering a space to explore new solutions to old environmental and social problems. Among the multiple factors for the success of REDD+ implementation, governance and land tenure have emerged as key elements of any strategy searching for forest carbon emission reductions in developing countries (Cotula and Mayers 2009, Larson *et al.* 2013). These ideas are also reinforced by Ongolo (2015), who showed that fragmented governance – the result of competing interests and objectives that lead to power games in policymaking between public and private actor groups – influences land policies and forest conservation.

In this thesis, I have used Panama as a case study to explore the links between tenure and forest cover, in particular the role of local communities in forest conservation in potential REDD+ initiatives. The results of this research are informative to REDD+ strategies in many developing countries. In **Chapter 1**, I found that the tenure status of protected areas and indigenous territories (including comarcas and claimed lands) explains a higher rate of success in avoided deforestation than other land tenure categories in Panama. The results suggest that, because of their efficiency in conserving forests, both protected areas and indigenous territories could be part of the successful implementation of REDD+ in Panama. Chapter 1 also reinforced the idea that rewarding forest protectors – and not only those with high rates of deforestation – is a necessary step for the global success of REDD+, as it can contribute to climate mitigation, reduce national and international leakage, and also promote a fair distribution of costs and benefits among and within countries (i.e., equity) (Meridian Institute, 2011). I contend that the matching analysis offers a statistically valid way to resolve the issue of demonstrating the contribution to and effectiveness in reducing deforestation, through the pairing of comparable areas with respect to remoteness, topography or other relevant characteristics. The results are of great relevance for REDD+ strategies, showing that matching is a scientifically-sound way to quantify the contribution of each jurisdiction to emissions avoidance.

In **Chapter 2**, I demonstrated that local knowledge can improve land cover classification and facilitate the identification of forest degradation. The UNFCCC's call for the full and effective participation of local and Indigenous Peoples could, therefore, improve the accuracy of monitoring in MRV systems of REDD+. The results also show that local knowledge is a valid option that complements other technical classifications (i.e., satellite image classification). They reinforce the idea of promoting the full and effective participation of Indigenous Peoples and local communities in monitoring activities, which are generally considered too technical to involve local dwellers. Finally, in **Chapter 3,** I estimated the importance of land invasions in indigenous territories as a key element of the rapid expansion of the agricultural frontier, showing that tenure alone is insufficient to guarantee rights over land and forests in indigenous territories. Using and validating free forest cover data (2001-2014) produced by Hansen *et al.* (2013), I advocate that REDD+ strategies among developing countries can improve forest conservation and secure livelihoods on-the-ground and that conflict resolution mechanisms and mediation processes might provide an avenue to resolve long-standing land conflicts.

The insights and conclusions of this thesis suggest that additional studies on environmental rights will be required to guarantee the appropriate implementation of REDD+ performance incentives, either from public or private sources. Based on the conclusions, research will be necessary to clarify forest carbon rights, as they define the right to benefit from the sequestered forest carbon – especially if REDD+ incorporates a trading component. Carbon rights are a form of property right that 'commoditizes' carbon and allows such trading (Cotula and Mayers 2009). Even clear tenure and absence of land conflicts cannot guarantee benefits or who should be supported under REDD+ schemes, e.g., who should get payments if forest carbon rights are contested. Across the tropics, forests are mostly owned by governments (Boucher 2013). As REDD+ is based on conditional rewards for reducing carbon emissions, I believe that more research will be necessary to identify benefit-sharing mechanisms among developing countries to clarify who gets rewarded, why, under what conditions, in what proportions and for how long.

The Nagoya Protocol, a supplementary agreement to the Convention on Biological Diversity (CBD), addresses traditional knowledge associated with genetic resources, with provisions on access, benefit-sharing and compliance (CBD, 2016), and could offer measures to ensure communities' free prior and informed consent, and fair and equitable benefit-sharing in

REDD+ implementation in developing countries. As demonstrated in this thesis, threats to land tenure and tenure rights, and the potential impacts of a global REDD+ strategy have been evident in recent years among Indigenous Peoples and forest-dependent communities. Indigenous Peoples' fear with respect to REDD+ is often that by conferring new value on forest lands, REDD+ could create incentives for government and commercial interests to actively deny or passively ignore the rights of indigenous and other forest-dependent communities to access and control forest resources (Angelsen 2008). In fact, during the United Nations climate negotiations in Durban in 2011, a global organization of Indigenous Peoples and local communities against REDD+ was formed to call attention to the lack of full recognition of indigenous rights in the texts of the UN climate negotiations.

Over the past 25 years, developing countries have transitioned toward decentralized forest management that allows local actors increased rights and responsibilities – a trend that could be interrupted if governments justify centralization by showing themselves as more capable and reliable than local communities at protecting national interests (Colfer 2005, Phelps *et al*. 2010). According to Phelps *et al.* (2010), new research is necessary to optimize REDD+ effectiveness through a combination of decentralized and centralized forest governance. The fact that this necessity contrasts with the premise that 'benefits should go to low-emitting forest stewards' is of relatively little concern for both government and REDD+ project developers. However, it is often treated as high priority in international debates and discourses on REDD+ benefit-sharing (Pham *et al*. 2013). This issue should be brought to the attention of decision-makers – in particular in jurisdictions or countries where Indigenous Peoples and local communities are important stakeholders – to identify and promote REDD+ benefit-sharing mechanisms designed to 1) maximize equity among the actors responsible for the reduction of deforestation and forest degradation, 2) improve the effectiveness of forest management and 3) increase the efficiency of national and subnational programmes (Brockhaus *et al*. 2013). According to Pham *et al*. (2013), REDD+ benefits should be shared with the forest actors that are essential for REDD+ implementation, whether they are private sector, civil society or central or local government. Due to the importance of the benefit distribution of potential policy interventions, the future of REDD+ will depend on how countries design their benefit-sharing mechanisms and how they address their potential impacts on communities and different beneficiary groups on the ground.

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APPENDICES

SUPPLEMENTARY INFORMATION CHAPTER 1

Identification of claimed indigenous lands of Panama

Table S1. Information collected from claimed indigenous lands of Panama.

Claimed land/Ethnic group	**Visit to the field/Date**	**Collected Information**	**Source of information**
Collective Lands of Alto Upper Bayano/Emberá	yes/ June 2010	Claimed territory (paper/geo-referenced)	Local authorities
Collective Emberá Lands /Emberá	yes/ July 2010	Claimed territory (digital/geo-referenced)	Local authorities
Collective Wounaan Lands/Wounaan	yes	Claimed territory (paper/ geo-referenced)	Local authorities
Dagarkunyala/Guna	no	Claimed territory (participatory mapping)	Local authorities
Naso/Naso	yes/ July 2010	Claimed territory (paper/ geo-referenced)	Local authorities/PRONAT
Bribri/Bribri	no	Claimed territory (paper/ geo-referenced)	Local authorities

PRONAT= National Land Program

S2. Supplementary results.

Description of claimed indigenous lands of Panama

Most claimed lands are fragmented and consist of several polygons (areas), where each polygon corresponds to one indigenous community of the same ethnic group. The different polygons of the same claimed lands are embedded in a matrix of non-indigenous private properties or public lands and are thus separated from one another. This spatial separation was most apparent for the Collective Emberá Lands, where the polygons for Jaque (close to the Colombian border) and Arimae (close to the Wargandi comarca) are separated by > 110 km (Figure 2). The fragmented nature of the newly mapped indigenous territories contrasts with those of the comarcas, which constitute one or at most two (Comarca Emberá-Wounaan) continuous territories where several indigenous communities share connected land.

The claimed lands in Panama (from west to east) are:

Bri-Bri. The Bri-Bri People live (~300 peoples) in the northwestern area of Bocas del Toro Province close to the Costa Rican border, mainly along the Sixaola and Yorkin Rivers, the main access route for communities in the area. The claimed area is 28,204 ha and 13,600 ha of this territory overlaps with La Amistad National Park.

Naso. The Naso or Teribe (Spanish) or Tjër Di (in Naso) People are located in northwestern Panama, in Bocas del Toro Province close to the Costa Rican border. Some 3,500 peoples live along the Teribe River. This river is also the main access route for the communities in the area. Almost the entire Naso territory overlaps two protected areas, viz., La Amistad National Park (78%) and the Palo Seco Forest Reserve (15.2%) (Table S2).

Collective Lands of the Upper Bayano. This claimed territory consists of four separate areas that are inhabited by ~1,500 Emberá People. The collective lands of the Upper Bayano are located in the Province of Panama close to the Pan-American Highway, and include Ipeti (3,191 ha), Piriati (3,754 ha, with two polygons) and Majé Emberá-Drua (16,155 ha), which is located south of Bayano Lake. This last area overlaps completely with the Maje Hydrologic Reserve. Since construction of the Pan-American highway, the collective lands of the Upper Bayano have faced external pressure that has been caused by the expansion of the agricultural frontier by colonists.

Collective Wounaan Lands. A total of ~400 Wounaan peoples live within these claimed lands in Eastern Panama, which are composed of five separate areas: three in Darien Province and two in Panama Province. These last two areas suffer from invasions by colonists due to the presence of valuable forest tree species and the areas' proximity to the Pan-American Highway. Some 4,200 ha overlap with protected areas (Darien National Park and the Bahia de Panama swamp).

Collective Emberá Lands. The largest of the claimed indigenous lands, the collective Emberá lands consist of 14 polygons, which range between 479 and 114,250 ha. The total area claimed represents 4% of Panama's national territory. The collective Emberá lands are located in Darien Province and are inhabited by about 3,000 Emberás. These areas were not included in the Emberá-Wounaan Comarca when it was created in 1983, because the landscape is a mosaic of different settlements, which include both indigenous and non-indigenous peoples. A large portion of this territory, ~ 226,000 ha, overlaps with Darien National Park and the Alto Darien Reserve on the western side of the Province of Darién.

Dagarkunyala. This Guna-claimed land includes the districts of Pucuro and Paya in Darién Province and covers a total area of ~144,000 ha. It is located close to the border with the Republic of Colombia and is inhabited by about 200 people. The main access to the communities is the Tuira River in the western part of the territory. The total area of this claimed land completely overlaps with protected areas – in this case, Darién National Park and the Alto Darien reserve.

Table S2. Geographic characteristics of comarcas and claimed indigenous lands of Panama, including their total area, the number of polygons (areas) that form each territory and the percentage of each territory that overlaps protected areas (PA).

Comarcas	**Polygons**	**Total Area (ha)**	**Overlap PA (%)**
Ngäbe Bugle	7	680,567	14.4
GunaYala	1	237,164	0
Madungandi	1	208,550	0
Wargandi	1	95,890	5.1
Embera Wounaan	2	439,204	76.8
Indigenous Claimed Territories			
BriBri	1	28,204	48.1
Naso	1	145,402	93.2
Upper Bayano	4	23,101	69.9
Collective Wounaan Lands	6	41,786	10.2
Collective Embera Lands	14	303,031	74.5
Dagarkunyala	1	144,136	100

Table S3. General statistics for six tenure categories in Panama (Part 1).

	CLOVER			**CL**			**COVER**		
Variable	**Mean**	**SD**	**Median**	**Mean**	**SD**	**Median**	**Mean**	**SD**	**Median**
Elevation (m)	687.2	676.5	431	204.3	223.3	126	451.9	412.7	341
Dist. Towns (m)	165,079.6	85,642.6	21,2957	96,410.0	62,723.6	74,331	162,742.2	58,269.2	183,401
Dist. Roads (m)	15,986.7	9,649.5	15,358	2,453.1	2,318.8	1,887	11,717.2	8,759.4	9,916
Slopes (deg.)	11.6	8.1	11	6.6	5.7	5	10.1	8.2	9

Table S3. General statistics for six tenure categories in Panama (Part 2).

	C			**PA**			**OL**		
Variable	**Mean**	**SD**	**Median**	**Mean**	**SD**	**Median**	**Mean**	**SD**	**Median**
Elevation (m)	385.8	423.2	229	637.5	590.1	472	262.1	331.2	133
Dist. Towns (m)	88,364.1	42,466.8	76,591	73,167.7	75,541.8	32,517	59,959	48,978.2	48,374
Dist. Roads (m)	6,458.5	6,202.4	4,386	7,931.4	11,999.8	2,631	2,661.1	6,859.3	825
Slopes (deg.)	8.5	7.1	7	11.2	7.5	10	6.7	6.3	5

Table S3. Panama's protected areas considered in each cohort.

Protected Area	**Base year 1992**	**Base year 2000**
Altos de Campana	✓	✓
Cerro Hoya	✓	✓
Chagres	✓	✓
Coiba	✓	✓
Darien	✓	✓

Golfo de Chiriqui		✓
Isla Bastimentos	✓	✓
La Amistad	✓	✓
Omar Torrijos	✓	✓
Camino de Cruces	✓	✓
Soberania	✓	✓
Metropolitano	✓	✓
Portobelo	✓	✓
Sarigua	✓	✓
VolcanBaru	✓	✓
Alto Darien	✓	✓
Palo Seco	✓	✓
San Lorenzo		✓
CorredorBagre		✓
Golfo de Montijo	✓	✓
Lagunas del Volcan		✓
Patino		✓
San San Pond Sank		✓
Barro Colorado	✓	✓
Cerro Gaital	✓	✓
Los Pozos de Calobre		✓
Canglon	✓	✓
Chepigana	✓	✓

La Tronosa	✓	✓
La Yeguada	✓	✓
Tonosi	✓	✓
Filo del Tallo		✓
Isla Maje		✓
Cerro Guacamaya		✓
Cerro Cerrezuela		✓
Cenegon del Mangle	✓	✓
Isla Canas		✓
Isla de Taboga yUraba	✓	✓
Playa Boca Vieja		✓
Playa la Barqueta		✓
Nargana		✓
Isla Iguana	✓	✓
Cienaga de las Macanas	✓	✓
Isla Galeta		✓
Lago Gatun	✓	✓
Las Palmas		✓
Tapagra		✓

SUPPLEMENTARY INFORMATION CHAPTER 2

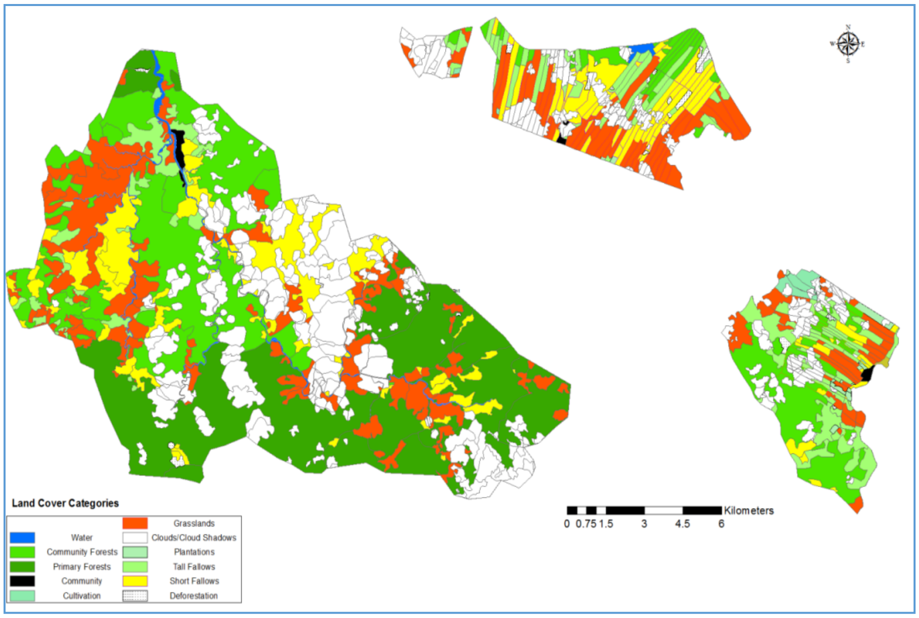
Appendix S1.

The Emberá People in the CLAB typically use the terms short fallow (1- to 4-year-long fallows), and tall fallow (≥ 5 years of fallow) to describe initial forest successional stages. Most of the collective properties in Ipeti and Piriati are elongated parcels, ranging between 30 and 100 ha in size. According to Tschakert *et al.* (2007), individual parcels in these collective lands are allocated to households by the traditional community authority, but decisions regarding land use and management of the parcel are taken entirely at the level of households or, in some cases, close family groups. The plot is left in fallow after a cultivation cycle (generally of 2–3 years' duration), whereupon locals move to another site within their parcel to reinitiate a new planting cycle. The duration of a fallow period is variable, ranging from 2 to 31 years before the same plot is used again for cultivation. The most deforested area of the CLAB is Piriati, where the majority of the land is dedicated to agricultural crops and cattle raising. Over the past two decades, many areas in the CLAB have experienced an increase in invasion by non-indigenous groups, which has generated greater deforestation and other environmental problems. Most of these invasions are related to the expansion of the agricultural frontier by cattle ranchers or farmers (*colonos*) from other areas of the country, and because these areas have not been legally recognized by the Panamanian government (Wali, 1993; Peterson St-Laurent *et al*., 2012).

Appendix S2. Satellite data and pre-processing

RapidEye images include five spectral bands: blue (440-510 nm); green (520-590 nm); red (630-690 nm); red-edge (690-730 nm); and near-infrared (760-880 nm). The scenes were delivered as level 3A products which include standard radiometric and geometric corrections. The images were pre-processed with Environment for Visualizing Images (ENVI® software) 5.0 by ITT VIS before being used in decision tree classification. Prior to their use, the satellite images in the decision tree classifier were atmospherically corrected using the FLAASH model, preprocessed to Top-of-Atmosphere-Reflectance and mosaicked in ENVI 5.0 to form one single image for the whole territory.

Appendix S3. Participatory land cover map of the CLAB territories.



Appendix S4. Decision tree classification

DT is represented by nodes. Labels are assigned to terminal (leaf) nodes by means of an allocation strategy that is based on spectral traits or integration of ancillary data (Pal and Mather, 2003). Unlike conventional classifiers that use all available features simultaneously and make a single membership decision for each pixel, DT uses a multi-stage or sequential approach to the problem of label or land cover classification. The classification process is considered to be a chain of simple decisions that are based on the results of sequential tests rather than a single, complex decision (Friedl and Broadley 1997). Branches of DT represent outcomes of a particular test (classification based on attributes) and nodes represent tests on an attribute of the tree.

Appendix S5. Land cover classification using decision trees

DT1 was executed with corrected reflectance values of the five RapidEye spectral bands. In the case of DT2, we also added eight textural features, which provide information about the spatial distribution of tonal variations within a channel, or its combinations. The principal elements in human interpretation of colour images, such as those obtained from satellites, are their spectral, textural and contextual features (Haralick *et al.* 1973). Adding selected textural measures can increase forest classification accuracy, as shown by Chan *et al.* (2003). The near-infrared band was used from the RapidEye images to obtain eight textural variables (mean, variance, homogeneity, contrast, dissimilarity, entropy, second moment, and correlation). These textural variables were based on a co-occurrence matrix for capturing the pixel textures that resulted from the different land cover categories. Vegetation indices and textural features were extracted from RapidEye images using ENVI 5.0.

Appendix S6. Pruned decision trees that were generated using WEKA software.

<table>
<tr><th>DT1</th><th>DT2</th></tr>
<tr><td>B3 <= 632
| B5 <= 425: Water (762.0/1.0)
| B5 > 425
| | B1 <= 192: Primary (1427.0/206.0)
| | B1 > 192
| | | B3 <= 260: Primary (560.0/208.0)
| | | B3 > 260
| | | | B5 <= 3385
| | | | | B2 <= 585
| | | | | | B5 <= 2975: Intervened (462.0/108.0)
| | | | | | B5 > 2975
| | | | | | | B4 <= 1020: Intervened (288.0/145.0)
| | | | | | | B4 > 1020: Tall Fallows (305.0/127.0)
| | | | | B2 > 585: Intervened (232.0/89.0)
| | | | B5 > 3385
| | | | | B2 <= 466: Primary (176.0/67.0)
| | | | | B2 > 466
| | | | | | B5 <= 4321
| | | | | | | B2 <= 548
| | | | | | | | B5 <= 3650: Tall Fallows (200.0/89.0)
| | | | | | | | B5 > 3650: Short Fallow (260.0/160.0)
| | | | | | | B2 > 548
| | | | | | | | B4 <= 1082: Tall Fallows (149.0/95.0)</td><td>Mean <= 4.2222: Water (761.0)
Mean > 4.2222
| B3 <= 632
| | B1 <= 192
| | | Contrast <= 10.2222
| | | | Mean <= 19.6667
| | | | | B1 <= 151: Primary (73.0/16.0)
| | | | | B1 > 151
| | | | | | B5 <= 3151: Intervened (60.0/21.0)
| | | | | | B5 > 3151: Tall Fallows (60.0/35.0)
| | | | Mean > 19.6667: Primary (185.0/26.0)
| | | Contrast > 10.2222: Primary (1050.0/71.0)
| | B1 > 192
| | | Contrast <= 10
| | | | Mean <= 19.5556
| | | | | Mean <= 17.4444: Intervened (431.0/61.0)
| | | | | Mean > 17.4444
| | | | | | B2 <= 623
| | | | | | | Var <= 1.5802: Tall Fallows (459.0/119.0)
| | | | | | | Var > 1.5802
| | | | | | | | Mean <= 18.6667: Intervened (192.0/93.0)
| | | | | | | | Mean > 18.6667: Tall Fallows (180.0/95.0)
| | | | | | B2 > 623: Short Fallow (65.0/38.0)
| | | | Mean > 19.5556
| | | | | B3 <= 260: Primary (111.0/58.0)
| | | | | B3 > 260</td></tr>
</table>

| | | | | | | | B4 > 1082: Short Fallow (699.0/298.0) | | | | | | B5 > 4321: Primary (213.0/98.0) B3 > 632 | B3 <= 937: Grassland (593.0/45.0) | B3 > 937: Bare soils (621.0/5.0	| | | | | | B2 <= 549 | | | | | | | Mean <= 20: Tall Fallows (88.0/36.0) | | | | | | | Mean > 20: Short Fallow (282.0/133.0) | | | | | | B2 > 549: Short Fallow (630.0/169.0) | | | Contrast > 10 | | | | Mean <= 20.3333 | | | | | B3 <= 222: Primary (114.0/13.0) | | | | | B3 > 222 | | | | | | B2 <= 386: Primary (82.0/32.0) | | | | | | B2 > 386: Intervened (477.0/140.0) | | | | Mean > 20.3333: Primary (433.0/104.0) | B3 > 632 | | B3 <= 937: Grassland (593.0/45.0) | | B3 > 937: Deforestation (621.0/5.0)

Appendix S7. Control points

Dataset that was used for accuracy classification, including randomly selected points (sampling) and inventory plots.

ID_Total	X_coord	Y_coord	Area	Source	Land class_Field
1	768149	994365	Ipeti	sampling	Grassland
2	769793	994717	Ipeti	sampling	Tall fallow
3	769624	994452	Ipeti	sampling	Grassland
4	769647	994222	Ipeti	sampling	Tall fallow
5	768994	994048	Ipeti	sampling	Tall fallow
6	769260	991729	Ipeti	sampling	Intervened forest
7	772082	993466	Ipeti	sampling	Tall fallow
8	772178	993218	Ipeti	sampling	Tall fallow

9	771166	990586	Ipeti	sampling	Tall fallow
10	770418	990065	Ipeti	sampling	Intervened forest
11	773075	993504	Ipeti	sampling	Short fallow
12	772911	992427	Ipeti	sampling	Short fallow
13	771446	991280	Ipeti	sampling	Intervened forest
14	771059	990579	Ipeti	sampling	Intervened forest
15	773808	993521	Ipeti	sampling	Grassland
16	773895	992913	Ipeti	sampling	Short fallow
17	773797	993770	Ipeti	sampling	Grassland
18	772998	989430	Ipeti	sampling	Intervened forest
19	772246	989072	Ipeti	sampling	Intervened forest
20	772710	988220	Ipeti	sampling	Tall fallow
21	762876	1003051	Piriati	sampling	Tall fallow
22	763595	1001000	Piriati	sampling	Grassland
23	762967	1000567	Piriati	sampling	Grassland
24	760781	1001525	Piriati	sampling	Grassland
25	760556	1002175	Piriati	sampling	Grassland
26	760551	1003126	Piriati	sampling	Tall fallow
27	759433	1002365	Piriati	sampling	Grassland
28	758938	1002731	Piriati	sampling	Grassland
29	758634	1004819	Piriati	sampling	Short fallow
30	761040	1002405	Piriati	sampling	Short fallow
31	765496	1003955	Piriati	sampling	Intervened forest
32	765421	1003429	Piriati	sampling	Intervened forest
33	763065	1000038	Piriati	sampling	Short fallow

34	764885	1002820	Piriati	sampling	Intervened forest
35	764975	1003312	Piriati	sampling	Intervened forest
36	764849	1003225	Piriati	sampling	Short fallow
37	764605	1002947	Piriati	sampling	Short fallow
38	764730	1003077	Piriati	sampling	Intervened forest
39	763996	1002279	Piriati	sampling	Tall fallow
40	760697	1001546	Piriati	sampling	Short fallow
41	760304	1002041	Piriati	sampling	Grassland
42	758750	1001805	Piriati	sampling	Grassland
43	759202	1004160	Piriati	sampling	Bare soil
44	758534	1004353	Piriati	sampling	Intervened forest
45	759235	1004357	Piriati	sampling	Tall fallow
46	761788	1002049	Piriati	sampling	Short fallow
47	762282	1002785	Piriati	sampling	Grassland
48	759758	1003576	Piriati	sampling	Intervened forest
49	761839	1001046	Piriati	sampling	Grassland
50	762264	1001662	Piriati	sampling	Tall fallow
51	763187	1002639	Piriati	sampling	Short fallow
52	763432	1002795	Piriati	sampling	Tall fallow
53	758674	1003166	Piriati	sampling	Tall fallow
54	758731	1003625	Piriati	sampling	Intervened forest
55	758819	1004078	Piriati	sampling	Tall fallow
56	758905	1004626	Piriati	sampling	Intervened forest
57	770276	990169	Ipeti	inventory	Intervened forest
58	770544	0993041	Ipeti	inventory	Intervened forest

59	769987	0992816	Ipeti	inventory	Intervened forest
60	770683	0993502	Ipeti	inventory	Intervened forest
61	770161	0993034	Ipeti	inventory	Intervened forest
62	770082	0993555	Ipeti	inventory	Short fallow
63	770010	993854	Ipeti	inventory	Tall fallow
64	770044	0994144	Ipeti	inventory	Tall fallow
65	770130	0994450	Ipeti	inventory	Intervened forest
66	769801	0994564	Ipeti	inventory	Tall fallow
67	770123	0992034	Ipeti	inventory	Intervened forest
68	770420	0989845	Ipeti	inventory	Intervened forest
69	770065	0991586	Ipeti	inventory	Intervened forest
70	747565	1000978	Maje	inventory	Intervened forest
71	747646	1000244	Maje	inventory	Tall fallow
72	747821	0999775	Maje	inventory	Primary forest
73	750673	0993933	Maje	inventory	Intervened forest
74	750958	0993487	Maje	inventory	Primary forest
75	751289	0993005	Maje	inventory	Intervened forest
76	751177	0997260	Maje	inventory	Primary forest
77	750875	0996909	Maje	inventory	Intervened forest
78	750327	0995904	Maje	inventory	Intervened forest
79	769974	0990237	Ipeti	inventory	Intervened forest
80	749233	0995774	Maje	inventory	Primary forest
81	749389	0993536	Maje	inventory	Intervened forest
82	748815	0997090	Maje	inventory	Primary forest
83	752862	989323	Maje	inventory	Primary forest

84	753429	0989121	Maje	inventory	Primary forest
85	753174	0989117	Maje	inventory	Primary forest
86	751994	0990097	Maje	inventory	Primary forest
87	752503	0989717	Maje	inventory	Primary forest
88	751573	990484	Maje	inventory	Primary forest
89	770357	0990734	Ipeti	inventory	Intervened forest
90	770363	0991082	Ipeti	inventory	Intervened forest
91	770256	0991765	Ipeti	inventory	Intervened forest
92	770071	0991295	Ipeti	inventory	Intervened forest
93	770345	0992287	Ipeti	inventory	Intervened forest
94	769799	0992627	Ipeti	inventory	Tall fallow

Appendix S8. Accuracy assessment

Confusion or error matrices identify key differences between producer and user accuracy (also called errors of omission and commission, respectively). Producer and user accuracies are measures of individual class performance within a classification. Producer accuracy (or omission error) is determined by dividing the total number of correctly identified pixels in a category by the total number of pixels that are present in that category of the test dataset (Congalton 1991). Overall accuracy of a classification is found by simply dividing the total number of correct cover categories by the total number of pixels in the confusion matrix. Generally, values greater than 80% represent strong agreement between the classification and reference data, while values between 0.4 and 0.8 represent moderate agreement.

The kappa coefficient (κ) was calculated from the confusion matrix following Cohen (1960):

$$\kappa = \frac{N\sum_{i=1}^{r} x_{ii} - \sum_{i=1}^{r}(x_{i+} * x_{+i})}{N^2 - \sum^{r}(x_{i+} * x_{+i})}$$

where r is the number of rows, x_{ii} is the number of observations in row i and column i, x_{i+} is the total observations in row i, x_{+i} is the total observations in column I, and N is the total number of observations in the matrix. The closer κ is to 1, the better the classification result, with κ > 0.8 being deemed acceptable (Foody 2008). Because κ is defined as a ratio, it is useful in normalizing accuracy results. Kappa is used in inter-rater comparisons of classifications involving differing numbers of categories being classified (Congalton 1991).

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SUPPLEMENTARY INFORMATION CHAPTER 3

Appendix 1. Study area

Currently, the watershed is home to two Indigenous Peoples, the Gunas and the Emberas, and it is also inhabited by around 8,000 Latino colonists (Instituto Nacional de Estadística y Censo (INEC) 2010). The Guna Comarca Madungandi is the indigenous territory with the longest tenure in the Bayano watershed, since it was officially recognized by the Panamanian Government under Law 24 on January 12th, 1996. Total population in the Comarca of Madungandi is 4,271 inhabitants. Emberas instead inhabit the so-called Collective Lands of Alto Bayano (CLAB), a group of three separated pieces of land, Ipeti, Piriati and Majé, located in the south of the watershed. Ipeti and Piriati were legally recognized by the Panamanian government under the country's Law 72 (Gaceta Oficial, 2008) after a long legal dispute in 2015, but land title has not yet been recognized for Majé. The CLAB is inhabited by some 1,500 people.

Primary economic activities among Indigenous Peoples include subsistence cultivation, cattle ranching, day labouring and handicraft production (Tschakert, Coomes and Potvin 2007). Meanwhile cattle ranching and slash-and-burn agriculture are the two primary land-use activities among colonists (Peterson-Saint Laurent *et al*., 2012).

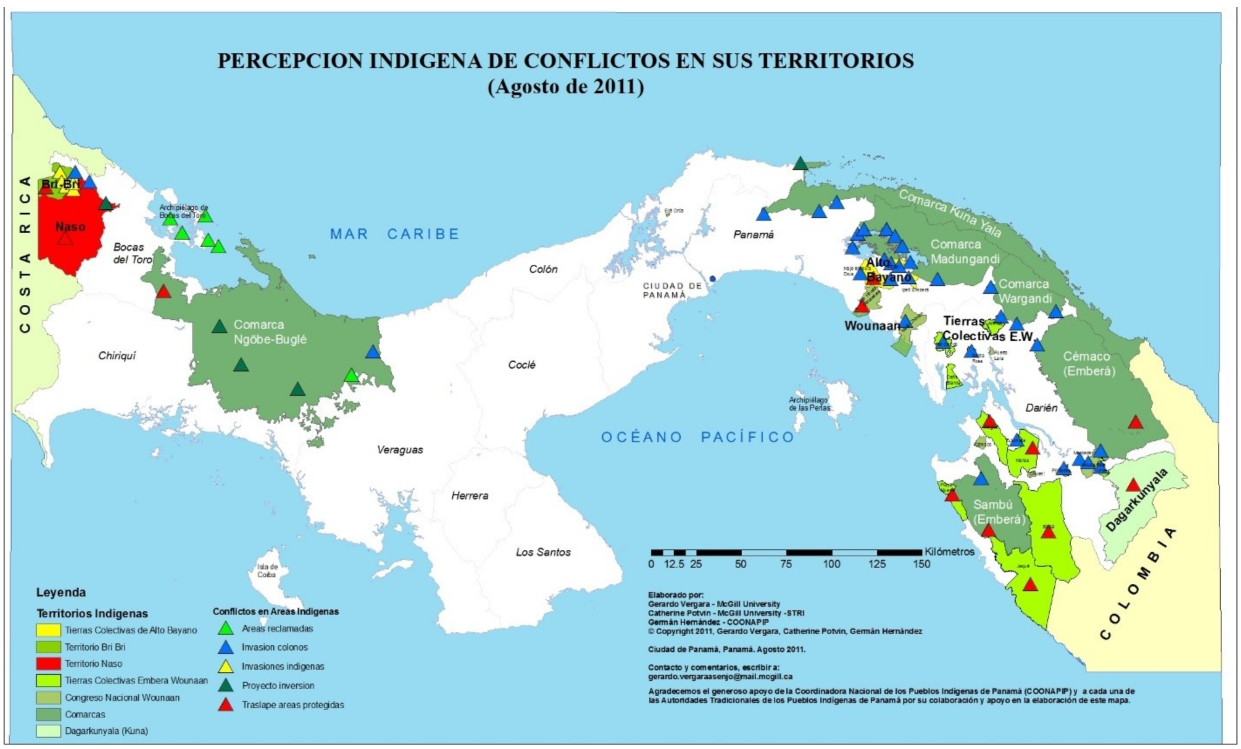

Figure 1. Conflict perception in indigenous communities in Panama. Triangles identify conflict perception in indigenous areas.

Appendix 2. Hansen's Forest Cover Map

Hansen's forest cover map was launched to enable interactive, online forest monitoring with data from over a decade (2000-2014) by the World Resource Institute through an initiative known as Global Forest Watch (http://www.globalforestwatch.org). Previously, the highest resolution for global land cover and forest maps was 250 meters (Bicheron *et al*. 2008, Hansen *et al.* 2008).

For the purpose of this study, “tree cover” was defined as all vegetation taller than 5 meters in height. “Tree cover” is the biophysical presence of trees and may take the form of natural forests or plantations existing over a range of canopy densities. “Loss” indicates the removal or mortality of tree canopy cover and can be due to a variety of factors, including mechanical harvesting, fire, disease or storm damage. As such, “loss” does not equate to deforestation.

Tree cover loss is defined as “stand replacement disturbance,” or the complete removal of tree cover canopy at the Landsat pixel scale. Tree cover loss may be the result of human activities, including forestry practices such as timber harvesting or deforestation (the conversion of natural forest to other land uses), as well as natural causes such as disease or storm damage. Fire is another widespread cause of tree cover loss, and can be either natural or human-induced.

Appendix 3. Accuracy assessment

The first dataset consists of 90 field plots established in forested areas in 2013 to estimate forest carbon stocks. The geographic coordinates of these plots were positioned on Hansen’s maps to validate forest cover in non-change areas. The second dataset consisted of field verification of 83 points randomly selected from Hansen’s map and validated on-the-ground.

For the first dataset, used to both validate Hansen’s maps and estimate emission factors or greenhouse gases per unit area, e.g. tonnes carbon dioxide emitted per hectare of deforestation for the area, we included 90 forest inventory plots established in 2012-2013 across the watershed in areas representing a range of elevational and human intervention gradients (Vergara-Asenjo *et al*. 2015, Mateo-Vega *et al.* 2016). These 25 m-radius plots were measured under the supervision of GVA by local indigenous technicians who had been trained in forest mensuration, included recording tree height, diameter at breast height (DBH) and species identification for carbon density estimation. Above ground biomass (AGB) was estimated using allometric equations produced by Chave *et al*. (2005), with wood density taken from the database created by Wright *et al*. (2010).

In the second dataset, a total of 53 out of the 83 points were visited to assess accuracy of forest cover change. Thirty points corresponded to water or shallow waters, as Hansen’s map appears to have identified some areas in shallow waters around Lake Bayano as forest cover or forest cover change. The 53 random points in forest cover change areas were restricted to changes that occurred in 2013 and 2014 (21 and 32 points respectively) and that could therefore easily be identified in the field. Land cover at specific points was verified in the field using a Garmin GPSMAP 60CSx by local indigenous technicians who were previously trained in operating GPS devices, locating coordinates of deforested areas and identifying current land cover. Informal consultations were carried out by the technicians with local inhabitants, when possible, to enquire about clear-cutting dates. Together, the whole set of 173 validation points served to verify forest classification as well as forest cover change in the study area.

User and producer accuracy are two widely used measures of class accuracy. Producer accuracy refers to the probability that a certain land-cover of an area on the ground is classified as such, while user accuracy refers to the probability that a pixel labeled as a certain land-cover class in the map really is this class.

Table 1. Confusion Matrix for Hansen’s map of forest cover and forest cover change in 2013.

		Reference				
		Forest	**Non-Forest**	**Water**	**Total**	**User´s Accuracy**
	Forest	64	8	8	80	80.0
	Non-Forest	6	44	1	51	86.3
Map	**Water**	0	0	21	21	100.0
	Total	70	52	30	152	-
	Producer´s Accuracy	91.4	84.6	70.0	Overall Accuracy =	84.9

Table 2. Confusion Matrix for Hansen's map of forest cover and forest cover change in 2014.

		Reference				
		Forest	**Non-Forest**	**water**	**Total**	**User´s Accuracy**
	Forest	64	8	8	80	80.0
	Non-Forest	4	35	1	40	87.5
Map	**Water**	0	0	21	21	100.0
	Total	68	43	30	141	-
	Producer´s Accuracy	94.1	81.4	70.0	Overall Accuracy =	85.1

Degraded forests and growing vegetation affect classification accuracy, in particular in areas around the limit of 30% forest cover – a threshold used for many countries. The area of forests can be overestimated or underestimated depending on the algorithm's ability to discriminate forest from these other land uses. For 2013 and 2014, user accuracy for the water category was 100% but producer accuracy was only 70%. Most misclassifications in the water category were related to inclusion of shallow waters, sand bars and non-forested riparian vegetation.

Appendix 4. Geospatial Reference Level

For emissions factors, we calculated an average aboveground biomass (AGB) per hectare according to ecozone (Holdrigde life zones) in the study area. Average AGB per ecozone was then transformed to carbon fraction using a factor of 0.47 (McGroddy *et al*. 2004) and then converted to tonne of carbon dioxide equivalent (tCO_2e) using the standard conversion factor of 3.67. Total CO_2 emissions produced by deforestation were estimated multiplying the average carbon per ecozone by the deforested areas identified in Hansen's maps for 2001-2014. Deforestation due to invasions was differentiated from other causes to determine its impact on the watershed.

A predictive analysis of deforestation quantifying potential impacts on forests in incoming years was then carried out using the Land Cover Change (LCM) module of Terrset software (Eastmann 2015) for the period 2015-2024. Slope, elevation, distance from previous deforestation and distance from roads were included as potential drivers in the model. The prediction process was based on Markov model. Two potential models were performed using both hard and soft predictors (Eastmann 2015). In both cases, the models assume no improvement in governance and a continued pattern of invasion.

Land change prediction in Terrset's Land Change Modeler (LCM) is an empirically driven process that moves in a stepwise fashion from 1) Change Analysis, 2) Transition Potential Modeling, to 3) Change Prediction. It is based on the historical change from Time 1 to Time 2 land cover maps to project future scenarios.

Model 1 uses a “hard” prediction that projects future land cover of each pixel based on the model’s drivers (slope, etc.) but considers only one possible outcome. As such, the hard prediction is a single realization of a future scenario chosen from many equally plausible ones (Eastman 2009). "Soft" prediction, in contrast, identifies vulnerability to change based on probabilities. Model 2 uses soft output, continuously mapping vulnerability to change, i.e. the degree to which the areas have the right conditions to precipitate deforestation. To do so, each pixel is assigned a probability of deforestation ranging from 0.0 to 1.0 based on the model’s drivers (slope, etc.) (Eastmann 2015). We selected only those pixels with higher probability than 0.5 to explore potential impacts. Activity data were combined with emissions factors to estimate emissions from the projected land use scenarios under both Models 1 and 2 for the period 2015-2024.

To provide a rough estimate of the cost of land invasion in a carbon-constrained world subjected to carbon pricing, we performed a back-of-the-envelope calculation based on historical carbon prices from voluntary markets using US$5/tonne, a price used by many early REDD buyers in bilateral agreements (Hamrick and Goldstein 2015, Kossoy *et al.* 2015). Projected deforestation costs for the period 2015-2024 were converted to US$ in 2015 using net present values at an 8% discount rate (Griess and Knoke 2011).

Appendix 5. Complementary Results

Carbon variation

Our forest inventory data reveal differences in carbon stocks among forests types, with tropical moist forest having twice (151.4 ± 73.2 Mg C ha^{-1}) as much carbon than premontane wet (58.8 ± 21.7 Mg C ha^{-1}) or premontane rain forests (55.1 ± 7.2 Mg C ha^{-1}). Carbon stocks of premontane wet and tropical moist forests are highly variable ranging 27.4-212 Mg C ha^{-1} and 72.2 -280.5 Mg C ha^{-1}, respectively. The high coefficients of variation of 60.4% and 48.6% for these forests can be explained by the fact that they grow in lowland areas of the Bayano watershed, and are located close to the Pan-American Highway and villages where anthropogenic activities have resulted in different levels of forest degradation (Sharma *et al*. 2016). In fact, the presence of short and tall fallow among regrowth vegetation forms from the forest inventory data is a clear indicator of previous human-induced forest disturbances in the area.

Carbon emissions

If the price of carbon were internalized in Panama’s economy, historical deforestation in the Bayano watershed between 2001-2014 would be equivalent to the foregone cost of 742 social houses in Panama (at a cost of US$50,000 each) (Ministerio de Vivienda, Republic of Panama, 2016) or approx. 59,000 monthly salaries (at US$624 per month).

Strictly regarding carbon, above ground biomass, considered in this study, represents only 67% of the total carbon distributed among different pools in tropical forests of Central America (FAO 2005). Additional forest goods and ecosystem services of significant subsistence and sociocultural importance for indigenous dwellers include biodiversity, food, craft, medicine or housing – values that are generally ignored by policymakers (Shanley *et al.* 2015). Furthermore, the Bayano watershed supports the Ascanio Villalaz hydroelectric dam that generates ~11% of total electricity in Panama (Sec. Nac. Energía, Rep. Panama 2016); forest ecosystems also contribute to generation of the important ecosystem service of water flow regulation for this facility.

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