Institutional and Fiscal Policies for Forest Conservation

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Abstract

Tropical deforestation contributes to climate change and entails further environmental externalities such as biodiversity loss and soil erosion. Consequently, the preservation of the world's remaining tropical forests is the objective of international and national policies.

Preventing deforestation requires identifying the structural causes of deforestation. Reforms of political institutions and fiscal policies bear the potential to reduce deforestation more structurally than localized protected areas. Such efforts are particularly relevant in countries with weak political institutions, low levels of economic development, and an economic specialization in the agricultural sector.

The first part of this dissertation contributes to the understanding of the role of political institutions in deforestation processes with chapter 2 and 3. Chapter 2 presents the results of a meta-analysis of the empirical cross-country literature on the effect of governance improvements on deforestation. The analysis finds that the choice of the governance measure is the main source of variation that explains diverging study outcomes. In particular, it can be shown that variables that measure the specific effects of environmental governance increase the likelihood of a study to find a deforestation reducing outcome, whereas the general governance variables democracy and civic rights decrease the likelihood. Furthermore, it can be shown that additional elements of study design, notably the choice of the control variable and the estimation technique, significantly influence the study outcome. Chapter 3 uses a forest model in order to analyze whether the model's ability to correctly predict past deforestation trends can be improved by taking differences in the quality of political institutions across countries into account. An index measuring the ability of a country to guarantee the sustainable management of natural resources is constructed. Subsequently, it is tested empirically whether the index can explain the gap between modelled and observed deforestation trends. The results show that building the indicator into the model can reduce this gap and thus improve the model.

The second part of this dissertation analyzes forest conservation policy options for countries with weak political institutions with chapter 4 and 5. Chapter 4 uses content analysis to analyze how African policy makers perceive deforestation drivers. It can be shown that policy makers emphasize the role of institutional and policy drivers of deforestation. Furthermore, it can be shown that these problems correspond to concrete opportunities for interventions, such as increased funding for forest sector administrations, improved alignment of different policies, or land tenure right reforms. Chapter 5 uses a theoretical model to analyze the effects of a policy mix that combines export tariffs on agricultural commodities with public investments. Public invest-

ments are defined as investments that increase agricultural productivity. The model shows that export tariffs and public investments can be combined, such that the output level in the agricultural sector remains constant, while deforestation and domestic food prices are reduced.

Common insights and the broader significance of the research conducted in the context of this dissertation are discussed in the last chapter. In particular, it is discussed how international forest conservation programs can support institutional and fiscal reforms for forest conservation.

Zusammenfassung

Die Abholzung tropischer Wälder trägt zum Klimawandel bei und bringt weitere Externalitäten wie Bodenerosion und den Verlust von Artenvielfalt mit sich. Der Schutz bestehender Waldflächen ist darum international und in vielen Ländern ein politisches Ziel geworden. Um den Verlust der verbleibenden tropischen Wälder zu verhindern, ist es notwendig die strukturellen Ursachen von Entwaldung zu identifizieren. Reformen von politischen Institutionen und Fiskalpolitiken bergen ein größeres Potential, Entwaldung strukturell zu reduzieren, als eine Fokussierung auf lokal begrenzte Naturschutzgebiete. Solche Reformen sind besonders relevant für Länder mit schwachen politischen Institutionen, niedrigen Einkommensniveaus und einem stark ausgeprägten landwirtschaftlichen Sektor.

Der erste Teil dieser Dissertation trägt mit Kapitel 2 und 3 zum Verständnis der Rolle von politischen Institutionen in Entwaldungsprozessen bei. Kapitel 2 beinhaltet die Ergebnisse einer Metastudie der empirischen Literatur zum Effekt von politischen Institutionen auf Entwaldung. Die Ergebnisse zeigen, dass die Wahl des Maßes für Governance die Hauptursache für divergierende Studienergebnisse ist. Studien, die spezifisch den Effekt von Umweltgovernance untersuchen, finden mit einer höheren Wahrscheinlichkeit, dass Entwaldung durch verbesserte Governance reduziert wird. Die Nutzung von allgemeineren Governancemaßen hingegen senkt die Wahrscheinlichkeit, einen positiven Effekt von Governance auf Entwaldung zu finden. Die Studie zeigt darüber hinaus, dass weitere Elemente des Studiendesigns, so wie die Auswahl der Kontrollvariablen und des Schätzverfahrens die Ergebnisse signifikant beeinflussen. In Kapitel 3 wird ein Forstmodell genutzt um zu untersuchen, ob beobachtete Entwaldungstrends besser vorhergesagt werden können, wenn Unterschiede in der Qualität von politischen Institutionen berücksichtigt werden. Hierfür wird ein Index konstruiert, mit dem die Fähigkeit eines Landes gemessen wird, natürliche Ressourcen nachhaltig zu verwalten. Es wird empirisch untersucht, ob durch den Index die Diskrepanz zwischen Modellschätzung und beobachteter Entwaldung reduziert werden kann. Die Ergebnisse zeigen, dass diese Diskrepanz reduziert werden kann, wenn der Indikator in das Modell eingebaut wird.

Der zweite Teil der Dissertation untersucht politische Handlungsoptionen für Waldschutz in Ländern mit schwachen Institutionen in Kapitel 4 und 5. In Kapitel 4 wird mit einer Inhaltsanalyse untersucht, wie afrikanische Entscheidungsträger Entwaldungstreiber wahrnehmen. Es kann gezeigt werden, dass die Entscheidungsträger die Schwäche von politischen Institutionen als eine zentrale Ursache von Entwaldung identifizieren. Die Analyse zeigt konkrete Möglichkeiten auf, institutionelle Probleme zu adressieren, zum Beispiel durch eine gestärkte Finanzierung von Forstverwaltungen, bessere Koordination von

unterschiedlichen Politiken, oder Landrechtsreformen. Kapitel 5 nutzt ein theoretisches Modell, um die Effekte eines Politikmixes zu untersuchen, der Exportzölle auf landwirtschaftliche Güter mit öffentlichen Investitionen kombiniert. Öffentliche Investitionen sind in diesem Kontext als Investitionen definiert, die zu einem Anstieg in landwirtschaftlicher Produktivität führen. Das Modell zeigt, dass Exportzölle und öffentliche Investitionen so kombiniert werden können, dass das Produktionsniveau im Exportsektor konstant bleibt, während Entwaldung und inländische Nahrungsmittelpreise gesenkt werden.

Kapitelübergreifende Schlussfolgerungen und die allgemeinere Bedeutung der Ergebnisse werden im letzten Kapitel diskutiert. Es wird insbesondere thematisiert, inwiefern institutionelle und fiskalpolitische Reformen im Kontext internationaler Waldschutzprogramme umgesetzt werden können.

Contents

Chapter 1

Introduction

Two countries' experiences have dominated the discussions on forest conservation over the last decade. In 2015 in Indonesia, 669 thousand hectares of forest were lost in less than five months due to peatland fires (World Bank, 2016), causing more daily emissions than the whole US economy. This created a severe health risk for the entire region, and threatened the survival of a variety of endemic species (Tacconi, 2016). On the other side of the globe, markedly different events occurred: fueled by civil society demands, the Brazilian government reduced primary forest cover loss in the Amazon by more than 70% between 2004 and 2016 (INPE, 2017). One of the main differences in the response to these countries' respective deforestation crises can be found in their institutional capacity to enforce forest conservation. When peatland fires ravaged Indonesia in 2015, the country did not have sufficient monitoring capacities or sanctioning mechanisms, let alone a unified national registry of land tenure rights to ensure deforestation control (Gaveau et al., 2016). In this environment, the government did not manage to prevent peatland burning and illegal deforestation (Cattau et al., 2016). In contrast, the historic success in reducing deforestation in Brazil, in particular between 2004 and 2012, can in large parts be attributed to the forest cover monitoring and deforestation sanctioning efforts of the Brazilian government (Cisneros et al., 2015; Hargrave and Kis-Katos, 2013).[1] While other factors have also influenced the diverging deforestation outcomes in both of these countries, political institutions are thought to have played a pivotal role.

Forests - in particular tropical forests - play a central role in stabilizing the global climate, conserving biodiversity, and providing a variety of further ecosystem services such as soil quality and water cycles (Smith et al., 2014, 2016) (for details consider section 1.1). Acknowledging the role of forests for climate stability, parties to the United Nations Framework Convention on Climate Change (UNFCCC) introduced a mechanism through which governments of developing countries can be financially compensated for activities to reduce emissions from deforestation and forest degradation, as well as, activities related to forest conservation, sustainable forest management, and the enhancement of forest carbon stocks (decision 4/CP.15 UNFCCC, 2009). The program is called REDD+. The REDD+ program and all the possible forms of payments to forest-rich countries it might entail, triggered a wide range

[1] In spite of this success, deforestation rates in the Brazilian Amazon increased again by 29% between 2015 and 2016 (INPE, 2017).

of activities in potential recipient countries, ranging from pilot-protected area projects (Caplow et al., 2011; Sills et al., 2009) to plans for the structural transformation of entire economies (e.g. DR Congo, (CN-REDD, 2015)). Following these varied early experiences with REDD+, most countries have converged around the idea that a jurisdictional or national-level approach is necessary to develop effective strategies to guarantee a reduction, instead of a displacement of, deforestation and forest degradation (UNEP, 2015). In the academic literature, this concept is also referred to as the landscape approach (Hett et al., 2012; Sayer et al., 2013; Seymour and Angelsen, 2012). Parties to the UNFCCC agreed at the 2013 conference in Warsaw that countries should provide national forest monitoring systems (decision 11/CP.19 UNFCCC, 2013a) and invited parties to set up national entities for the management of REDD+ programs (decision 10/CP.19 UNFCCC, 2013b). Furthermore, different UNFCCC decisions underline that countries should have absolute sovereignty in their REDD+ strategy-making (e.g. decision 12/CP.17 UNFCCC, 2011) and acknowledge that different activities are necessary to reduce deforestation according to national circumstances (for more details on the international framework for REDD+, consider section 1.2).

In the early phase of REDD+, policy and academic discussions mostly focused on protected area projects and targeted local conservation efforts, such as the Kasigau corridor project in Kenya (Bernard et al., 2014) or the Makira project in Madagascar (Brimont et al., 2015). The increasing focus on jurisdictional and national levels opens up a different perspective and invites for a more systematic analysis of deforestation causes. While it still includes the option for countries to implement targeted local conservation projects (i.e. a so-called nested REDD+ approach (Angelsen, 2008; Hayes and Persha, 2010)), REDD+ is also planned to involve national-level coordination and implementation authorities and could thereby turn forest conservation into a potentially much more politicized issue compared to localized conservation projects (Skutsch et al., 2009; Wertz-Kanounnikoff and Angelsen, 2009).

Designing effective forest conservation strategies on a jurisdictional or national level is facing fundamental challenges in countries that (i) lack the political institutions to enforce such strategies, (ii) heavily rely on deforested land as a relatively low-cost input to agricultural production, and (iii) are furthermore low-income countries with high levels of food insecurity. Examining the countries where REDD+ programs are under preparation (FCPF, 2015; UN-REDD, 2015)[2] reveals that one third of them are low and lower-middle income countries (World Bank, 2013b), endowed with relatively weak political institutions (World Bank, 2014), and with a strong economic specialization in the agricultural sector (World Bank, 2013a) while at the same time experiencing food insecurity (FAO, 2013).

In its fourth assessment report, the IPCC notes that national forest policies, designed to slow deforestation, have had limited success in developing countries with insufficient institutional and regulatory capacities (Nabuurs et al., 2007). In light of this situation, some authors (e.g. Chagas et al., 2011; Neeff et al., 2014) point to the risk of overburdening REDD+ and advocate an approach, in which the focus of REDD+ activities is on decentralized con-

[2]Status: countries that formed part of the UN-REDD or FCPF program as of December 2015 (FCPF, 2015; UN-REDD, 2015).

servation projects. In such proposals, the role of national level authorities is reduced to guaranteeing basic frameworks such as a common emission accounting system.

In contrast to this approach, in this dissertation, I depart from acknowledging the structural deficits of these countries and treat them as limiting factors, rather than absolute limits to national level policy options for forest conservation. Central to the analysis is the idea that a more nuanced understanding of how political institutions affect deforestation processes can help identify targeted national level policy measures that address the systematic causes of deforestation. Following Kaufmann and Kraay (2008), the concepts of institutional quality, institutional capacity, and governance are used interchangeably throughout this dissertation and defined as the traditions and mechanisms through which political authority is exercised in a country.

An inquiry into political institutions in the context of forest conservation is of fundamental importance. Still, the academic literature on deforestation has had a tendency not to choose such an analytical lens. This can be illustrated by the two-dimensionality that has shaped the discussions of deforestation drivers. In particular, Geist and Lambin (2002) have introduced a commonly cited distinction between proximate and underlying causes of deforestation (Hosonuma et al., 2012; Kanninen et al., 2007; Pineda Jaimes et al., 2010): infrastructure, agriculture, wood extraction, and other factors are listed as proximate causes, whereas demographic factors, economic factors, technological factors, policy and institutional factors, and cultural factors are listed as underlying drivers of deforestation. This standard categorization lists political institutions as a deforestation driver among a variety of other underlying causes. It thereby conceptually dissociates political institutions from decision-making and ignores, for instance, that infrastructure projects or economic policy are in fact a result of political decisions made within political institutions rather than spontaneous, individual decisions emerging in a political vacuum.

In this dissertation, I assume that political institutions - even malfunctioning ones - generate a certain degree of homogeneity in human decision-making within the respective political system (Aoki, 2001). Thus, in order to identify systematic causes of deforestation and strategies to reduce deforestation, it is necessary to analyze the functioning of a political system in itself as well as the rules and procedures it generates, in order to understand how incentives for deforestation arise systematically as an outcome. This perspective suggests two central priorities for the forest conservation research agenda.

First, it encourages an investigation of the elements of political institutions, which are necessary to guarantee sustainable forest governance. The FAO and PROFOR (2011) propose a conceptual framework with three pillars for such an analysis including (i) the policy, legal, institutional, and regulatory frameworks, (ii) decision-making and planning processes, and (iii) the implementation, and enforcement of and compliance with such regulatory frameworks. In the first part of this dissertation, it will be demonstrated that both the empirical literature, but also the literature using global forest cover models for the analysis of forest cover trends, still offer considerable room for improvement to understand and represent the role of political institutions in deforestation processes.

Second, choosing to analyze deforestation drivers through the analytical lens of political institutions encourages a reflection on forest conservation strategies in contexts where systematic constraints to effective policymaking are particularly pronounced. These systematic constraints are especially weak political institutions and weak levels of economic development. As summarized in the detailed discussion on propositions for REDD+ in the UNEP Emissions Gap Report (UNEP, 2012), the majority of current propositions for REDD+ policies ranges from establishing protected areas and command-and-control measures to using economic instruments (taxes, subsidies etc.) and creating policies that change systematic drivers of deforestation (like the institutional framework). In the second part of this dissertation, I discuss such policy options in the context of countries with very weak political institutions and low levels of economic development.

The dissertation's analysis of policies for tropical forest conservation is mainly based on two fields of economic research: First, research on collective action problems, as discussed in in the context of Climate Change Economics (as well as in more general terms in Public Economics) and second, research in the field of Development Economics.

Collective action problems arise in situations, where individually rational behavior, such as the maximization of utility, leads to collectively irrational outcomes (Campbell and Sowden, 1985). Especially game theory is concerned with understanding how individually rational strategies lead to cooperative and non-cooperative equilibria (Nash, 1951). Many environmental goods are characterized by common pool resources properties, which means that they are rivalrous and non-excludable at the same time (Ostrom, 1990). Climate Change Economics aims to explain global, environmental collective action problems related to the emissions disposal space in the atmosphere and seeks to identify policy options that offer ways out of non-cooperative equilibria, taking into account that the costs and benefits of different policy options are unequally distributed within and among countries, as well as across generations (Goulder and Pizer, 2006; Stern, 2006). It furthermore analyzes the economic impacts of climate change (Dell et al., 2014). Forests are not always common pool resources in the strict sense that forest would always be perfectly rivalrous and non-excludable. In many cases, forest lands are actually excludable, because they are for example privately owned, or owned by states. Nevertheless, deforestation problems tend to occur in settings, where this excludability cannot be guaranteed. For example in settings with unclear or overlapping forest land right claims, or contexts, where forest monitoring is technically and practically limited. In contexts where the *de facto* excludability does not hold any more, forests are prone to collective action problems such as the overuse of forest resources. Therefore important parallels with common pool resource problems that are perfectly rivalrous and non-excludable, such as the atmosphere exist and much can be learnt from this literature, for the analysis of forest conservation policies.

While much of the literature concludes that non-cooperative equilibria are an inevitable outcome in collective action problem settings, Nobel Prize laureate Elinor Ostrom (1990) provides an alternative analysis to the otherwise "grim" perspectives on the human ability to cooperate. By conducting numerous in-depth case study analyses of common pool resource governance systems in different countries, she identifies design principles of local institutions that allow resource users to cooperate and manage common pool resources sustainably. Even though Ostrom's work focuses on local common pool resources, the idea that the quality and design of institutions are a central determinant of sustainable resource management is also central for economic research on larger common pool resources (e.g. at international or national scales).

Environmental externalities are another form of conceptualizing collective action problems in the economic literature. An externality is the secondary negative or positive effect of an economic activity on a third party (Buchanan and Stubblebine, 1962). Coase's (1960) seminal work on the problem of social costs analyzes solutions to externality problems. He argues that direct bargaining between the parties causing the externality and those affected by the externality can solve externality problems, if property rights are well-established and the costs of bargaining are low. However, he further argues that in cases where a large number of people is affected by an externality problem, where property rights are unclear and the costs of bargaining are high, government regulation can be a more efficient solution. He highlights that choosing the most efficient institutional arrangement in order to deal with harmful effects is an important question for economic research. The REDD+ concept is motivated by the idea that industrialized countries pay developing countries to reduce the deforestation externality of their economic activity. However, even though industrialized countries gain a right to determine that forests have to be conserved, they still gain no material and exclusive right over the resource use through the payments. Yet, the details of these agreements and the limits to which national sovereignty can be challenged after a REDD+ contract, in order to guarantee actual forest conservation, are central issues of discussions in the REDD+ context. The new institutional economics literature (Williamson, 1975) is heavily influenced by the externality concept and is rich in reflections on efficient institutional arrangements and the role that political institutions can play in order to reduce economic externalities.

Development Economics studies the reasons behind diverging patterns in economic growth (Ray, 2008), as well as causes for the economic diversification of some economies and the absence of such diversification patterns in others (Hidalgo and Hausmann, 2009). Given that tropical forests are predominantly located in low and lower middle income countries, effective forest conservation policies have to be in alignment with the economic development goals of the respective countries.

Development Economics has long recognized the importance of political institutions in explaining diverging patterns of economic growth. In a Hobbesian tradition, Weber (1947) discusses the role of legitimate authority and bureaucracy for economic development. As a reaction to the high rates of economic growth in Asian countries in the late 20th century, political institutions are viewed as a central explanatory factor in the literature (Amsden et al., 1996; Evans, 1995; Wade, 1990). Similarly, Herbst (2000) links the absence of sustained high rates of economic growth in many African countries to weak political institutions. Gennaioli and Rainer (2007) analyze the role of political institutions for African countries empirically, and their findings support the institutions hypothesis. Acemoglu et al. (2015) define state capacity as an analysis of the state functionaries and agencies. Furthermore, Acemoglu et al.(2001) use settler mortality rates and thereby confirm that the quality of political institutions is a fundamental determinant of long-run economic growth. Moreover, according to Acemoglu et al. (2011), societies with limited state capacity underprovide public goods.

Beyond this, within the field of Economic Development, the bureaucracy literature (Cingolani et al., 2015; Rauch and Evans, 2000) aims at understanding which specific attributes of political institutions are important for economic growth. Nunn and Trefler (2013) for instance, find that the complexity of contracting institutions is a predictor for the level of economic development; specifically, countries with weak contracting institutions tend to be involved in less complex production processes. Weak institutional systems prevent countries from transitioning towards more complex economic activities. Consequently, they prevent the diversification of the economy away from agricultural activities. Michalopoulos and Papaioannou (2013) and Osafo-Kwaako and Robinson (2013) analyze the specific effect of the role of political centralization on economic development. Others analyze the role of democratic legitimacy and salaries of public servants (Acemoglu et al., 2010; Finan et al., 2015). Besley and Persson (2013) furthermore investigate public budgets and aim at understanding why developing countries tax relatively less than developed economies. Other studies examine the effects of weak property rights protection on economic development (Besley and Ghatak, 2009). The research field also generates new creative measures of the quality of political institutions. Chong et al. (2014), for instance, mail letters to fictive businesses and use the time the letters take to be returned as a measure of bureaucratic efficiency. Such studies within the bureaucracy literature can provide entry points for reflections upon possible causes of deforestation found in political systems.

My dissertation builds on these general fields of economic literature and contributes in particular to the literature on deforestation and forest conservation policies. A short review of the state of the art literature on the (i) effect of political institutions on deforestation, (ii) representation of institutions in forest cover models, and (iii) policy options for forest conservation present the context and aims at motivating the choices of the research questions that are addressed in the four chapters of this dissertation.

First of all, the effects of specific aspects of forest governance on deforestation are largely unknown (Kishor and Belle, 2004). The way political institutions are described in the current economic literature on deforestation could be compared to the uncertainty principle of particles (Heisenberg, 1927), which stipulates that there is a limit to the precision by which complementary variables such as position and momentum of a particle can be measured. In studies analyzing the role of political institutions in deforestation processes, there is either a detailed description of the types of institutional challenges (Hett et al., 2012) or a very aggregated empirical analysis of the effects of weak institutions on deforestation, such as the role of democracy (Buitenzorgy and Mol, 2011; Ehrhardt-Martinez et al., 2002; Li and Reuveny, 2006; Midlarsky, 1998), perceived corruption (Barbier et al., 2005; Wolfersberger et al., 2015), or weak levels of the overall rule of law (Deacon, 1994; Ferreira, 2004). However, in contrast to the uncertainty principle, improvements in measurements could actually resolve the problem here. Yet, this has not happened so far. The literature does not allow drawing broader conclusions on the specific effects of the institutional processes that concern decisions relevant to the management of forest sector resources in different countries. It also does not provide specific conclusions on the role of bribery and corruption within the forest sector across countries, or on how levels of enforcement against illegal deforestation vary across countries and in how far this affects the overall forest stock of a country.

More recently, some efforts have been undertaken to improve the understanding of such channels. Galinato and Galinato (2016) analyze the effects of government spending for forest conservation on deforestation across countries. Barbier and Tesfaw (2015) examine the effect of the presence of specific forest sector policies on the likelihood for a country to reach a point, where it does not experience net forest cover loss any more. However, such studies remain an exception.

Second, given the sparse analysis of governance factors in the empirical literature on deforestation, it is not surprising that global models account for the role of political institutions to a very limited extent. De Vos et al. (2013) observe that assessment models of global environmental change mainly focus on the representation of biophysical and techno-economic processes and rarely include knowledge on environmental regimes. Reid et al (2010) make a similar observation in an article on the grand challenges for Earth System Modelling. In particular, most land use change models that attempt to incorporate a more complex representation of human decision-making either focus on agent-based modelling (An, 2012) or analyze the effect of institutions in a local or regional context (Janssen et al., 2000; Lobianco et al., 2015). Global models simulating land use and representing stock and flow dynamics - e.g. the Patuxent Landscape Model (PLM) (Voinov et al., 1999), the CLUE Model (Conversion of Land Use and Its Effects) (Veldkamp and Fresco, 1996) or the NELUP model (O'Callaghan, 1995) - either represent human decision-making systems in a very simplified manner, or do not attribute any role to them at all. In a review of state-of-the-art land use change models, Rounsevell et al. (2014) conclude that such models need to better conceptualize the representation of governance structures and institutional arrangements. More recent work relying on economic global land use models, which mostly maximize social welfare, has acknowledged this shortcoming and attempts to provide a more explicit representation of the role of human decision-making systems and political institutions in land use change models. Wang et al. (2016) directly adjust discount rates in a partial equilibrium model of global land use with endogenous technical change to account for the quality of governance. This disincentivizes investment in technologies and consequently leads to the expansion of croplands. Using a global forest model, Benítez et al. (2007) employ country risk ratings to adjust discount rates in order to capture the impact on afforestation investments. Some research thus moves into the direction of a more explicit representation of human decision-making and the quality of political systems, but still rather remains an exception.

Third, Deacon (1995) notes that the empirical basis for identifying deforestation drivers and linking them to government policies is "*very meager at present.*" Since policy debates on REDD+ have moved towards the implementation phase, governance issues are repeatedly brought up as central problems in the academic literature. However, Corbera and Schröder (2011) conclude that while the majority of publications on deforestation stresses the need to focus on governance issues, "*most of the available literature does not get into the subject of governance improvement in depth.*"

Furthermore, a range of academic discussions on REDD+ policy options is to be located in the realm of localized and context-specific conservation science (see section 1.4 for a detailed discussion of policy options for forest conservation in the literature). In this context, Hett et al. (2012) observe that "*researchers are often tempted to narrow their focus to provide detailed answers on a tiny aspect of REDD+ in a small case study area.*" They subsequently argue that "*despite the merit of these initiatives, the mismatch between the scale of knowledge production and currently relevant decision-making processes is reinforced.*" The literature is, for example, rich in case study evaluations of conservation policies. In particular, studies examine the effectiveness of protected areas (Arriagada et al., 2016; Spracklen et al., 2015; Miteva et al., 2012; Nelson and Chomitz, 2011; Ferraro et al., 2013; Holland et al., 2014; Ferraro et al., 2013; Joppa and Pfaff, 2009), integrated conservation and development projects (ICDP) (Garnett et al., 2007; Weber et al., 2011), and payments for ecosystem services (PES) (Alix-Garcia et al., 2009; Schomers and Matzdorf, 2013; Clements et al., 2010; Costedoat et al., 2015; Pagiola et al., 2016). However, Börner et al. (2016), for example, observe that in the conservation literature, there is an insufficient understanding of the causal mechanisms that determine the effectiveness of different forest conservation programs and approaches.

There is furthermore a wide range of empirical analysis of the indirect effects of other policies on the forest cover. Empirical studies evaluate, for instance, the effects of infrastructure projects (Nelson and Hellerstein, 1997), land tenure rights security (Liscow, 2013; Robinson et al., 2014), indirect effects caused by agricultural policies (Andersen, 1997; Fisher and Shively, 2007), effects of decentralization policies (Somanathan et al., 2009; Wright et al., 2016), or indirect effects of trade liberalization on forest cover (Combes et al., 2015; Diarrassouba and Boubacar, 2009; Ehrhardt-Martinez et al., 2002; Guerineau et al., 2008).

Ex-ante studies on the indirect effects of policies on deforestation furthermore analyze the role of trade liberalization (Copeland and Taylor, 2009; Brander and Taylor, 1997; Chichilnisky, 1994), real exchange rates (Arcand et al., 2008), timber trade restrictions and taxation (Deacon, 1995; Maestad, 2001), concessional timber logging systems (Walker and Smith, 1993), the role of land tenure rights (Mendelsohn, 1994), agricultural intensification, and technological change (Byerlee et al., 2014; Villoria et al., 2014; Ceddia et al., 2013; Ehui et al., 1990; Hertel, 2012), agricultural policies (Takasaki, 2007), proximity of settlements and the agricultural frontier to forests, on deforestation (Angelsen, 2007; Jones and O'Neill, 1995; Southgate, 1990).

In addition, empirical evaluations of forest conservation policy experiences in countries like Costa Rica and Brazil offer a more in-depth analysis of the causal mechanisms underlying a policy intervention that lead to forest conservation. In Costa Rica, a national program distributing payments for ecosystem services was combined with a ban on deforestation. Payments for sustainable forest management are funded through water and fossil fuel taxes as well as carbon credit payments. A variety of studies examine the success of the policy and criticize that the program does not sufficiently incentivize additionality of conservation efforts or provide additional benefits to participants (Andam et al., 2008; Arriagada et al., 2015; Joppa and Pfaff, 2009; Sierra and Russman, 2006).

The success in the reduction of deforestation in Brazil, mainly from 2004 until 2012, is attributed to enhanced political forest law enforcement efforts by the government (Assunção et al., 2015; Cisneros et al., 2015; Hargrave and Kis-Katos, 2013). Other elements of the policy mix, such as the restricted access to agricultural credits for 'blacklisted' municipalities, are also mentioned as success factors.

However, while the analysis of these two success cases offers interesting academic insights and policy lessons for countries with relatively strong political institutions and relatively high levels of economic diversification, they offer few policy lessons for countries with weak political institutions and low levels of economic development.

The research conducted in this dissertation attempts to contribute to both, the discussions on the role of political institutions in deforestation processes, as well as the discussions on policy options for forest conservation in the literature.

The first part of this dissertation aims at contributing to the understanding of political institutions in deforestation processes, both in the empirical literature and in forest models. Central research questions for the first part of this dissertation are: what does the existing literature tell us about the effect of different elements of political institutions on deforestation? Which role do political institutions play in deforestation processes? How could the role of political institutions be better reflected in forest cover models?

In Chapter 2, the results of a meta-regression analysis of empirical studies examining the relationship between political institutions (national level governance) and deforestation are reported. In view of the multitude of studies - coming to different conclusions concerning the impact of different governance variables - the analysis investigates the main source of variation across studies. The analysis reveals that the choice of the governance variable, rather than general study design, or a specific type of control variable for example, is the main source of variation in study outcomes.

Chapter 3 aims at contributing to the literature on forest cover change models. An indicator on the quality of political institutions for the sustainable management of environmental resources is integrated into a global forest model. The results presented in Chapter 3 show that this procedure can significantly reduce previously unexplained variations in observed deforestation trends.

The second part of this dissertation aims at analyzing policy options for forest conservation for countries with weak political institutions. Central research questions are: how do African policy makers perceive deforestation drivers? Which role do institutional and policy issues play in their perception and which role could they play in possible policy responses to deforestation?

Which fiscal policy instruments could be used in order to reduce deforestation, while maintaining current economic output levels?

Chapter 4 focuses on Sub-Saharan Africa, a region in which many countries with weak political institutions are located. Using content analysis the perception of deforestation drivers as expressed by African policy makers in national REDD+ documents is analyzed. The analysis reveals that institutional and policy drivers of deforestation receive considerable attention in national policy documents and that furthermore regularities in such deforestation drivers can be observed across Sub-Sahara African countries. The chapter shows that beyond more general governance problems, such as corruption, policy makers are able to identify a range of concrete institutional problems specific to the forest sector. This list of specific problems makes the governance challenges for forest conservation much more tangible and provides possible policy options for deforestation reduction through governance improvements.

Chapter 5 explores a fiscal policy for forest conservation in countries with weak political institutions, high levels of food insecurity, and a high level of dependence on the agricultural sector as driver of GDP. The proposed policy mix combines export tariffs and public investments in order to provide a systematic disincentive to deforestation, without constraining agricultural output levels.

The remainder of this introduction provides background information and empirical motivations for the research questions analyzed in this dissertation. Section 1.1 presents the biophysical background information on the importance of forest conservation for ecological sustainability. Section 1.2 provides background information on the emergence of the REDD+ mechanism and the evolution towards a jurisdictional approach. Section 1.3 discusses the relationship between structural change and deforestation. Section 1.4 presents an overview of discussions on policy options for forest conservation on a national level.

1.1 The importance of forests for global ecosystem stability

The world hosts close to 40 million km^2 of forests, of which between 0.33 million km^2 are lost each year to deforestation (FAO, 2016). Even though the rate of net forest cover loss has decreased by half since the 1990s (still 7.3 million ha loss per year), the world net forest cover loss in the last 25 years corresponds to the size of South Africa (FAO, 2016). The bulk of deforestation takes place in the global South, with the largest forest loss having occurred since the 1990s in Brazil, Indonesia, and the Congo Basin countries (FAO, 2016).

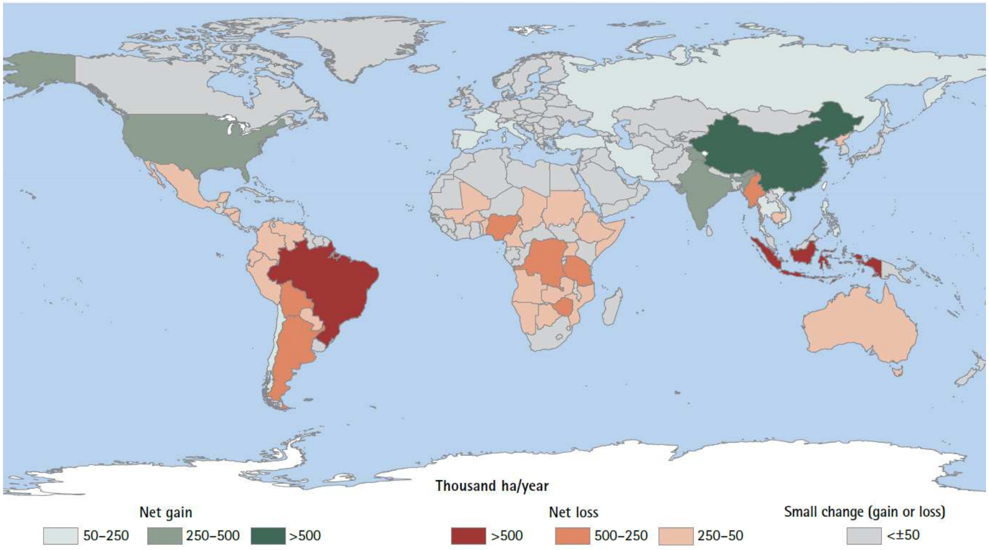

Figure 1.1: Forest area annual net change 1990-2015, (Source: FAO, 2016, reproduced with permission)

Tropical regions are also particularly affected by a decrease in density and an increase in disturbances of forests (Nabuurs et al., 2007). The FAO (2016) shows that in 2015, partial canopy cover loss was observed for 9% of tropical forest areas. Forests are vital to key ecosystem services, such as regulation services (climate, noise, pollution, disease and pest regulation, and pollination), supporting services (primary production, soil fermentation, decomposition, water cycling, and weathering), provisioning services (food, timber, water, energy, biodiversity) and cultural services (recreation, tourism, spiritual, or religious) (Smith et al., 2014). Deforestation undermines these services and notable consequences include climate change, biodiversity loss, soil degradation, changes to local water cycles, and reduced air quality.

In 2015, forests accumulated 296 Gt of carbon in above and below ground biomass, with a 0.15% loss rate per year since 1990 (FAO, 2016). The IPCC estimates that CO_2 emissions from forestry and other land use changes (FOLU/LULUCF) accounted for one third of CO_2 emissions from 1750 to 2011 and still represented 12% of global emissions from 2000 to 2009 (Smith et al., 2014). Global emissions from forestry and other land uses are based on a variety of sources (Houghton et al., 2012; Pongratz et al., 2013; Le Quéré et al., 2013), which explains the range of different estimates of CO_2 emissions. Figure 2 illustrates the ranges corresponding to different models for different world regions. It shows that the highest emissions from forests and other land uses occurred in Latin America (LAM) and Asia (ASIA) between 1980 and 1989, whereas past forest and other emissions related to land use change in the Middle East and Africa (MAF) have been comparably lower, but are currently rising.

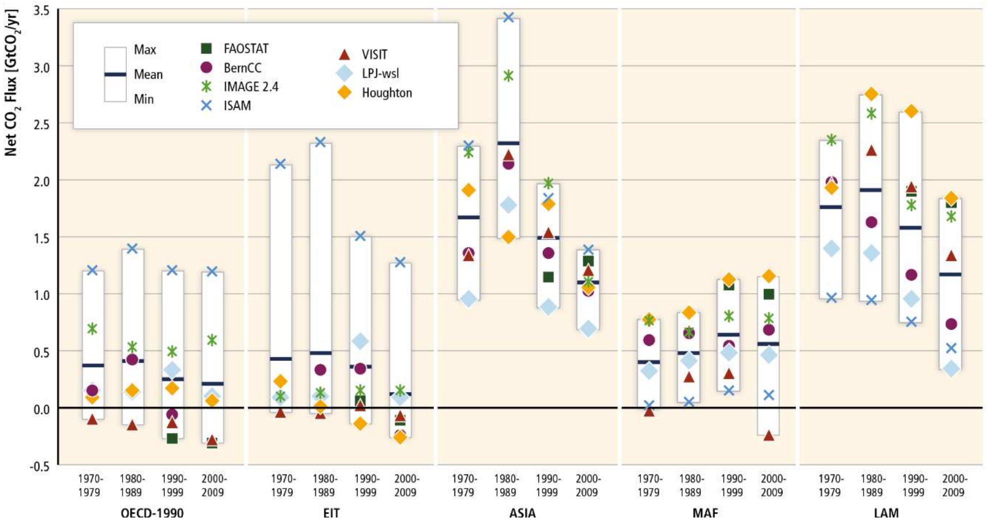

Figure 1.2: : Regional trends in net CO_2 fluxes from forests and other land uses, (Source: Figure 11.7 from Smith P., M. Bustamante, H. Ahammad, H. Clark, H. Dong, E.A. Elsiddig, H. Haberl, R. Harper, J. House, M. Jafari, O. Masera, C. Mbow, N.H. Ravindranath, C.W. Rice, C. Robledo Abad, A. Romanovskaya, F. Sperling, and F. Tubiello, 2014: Agriculture, Forestry and Other Land Use (AFOLU). In: Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel and J.C. Minx (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., (reproduced with permission)

Tropical forests, furthermore, host 15 of the 25 worldwide biodiversity hotspots with the highest concentration of endemic species, while experiencing exceptional habitat loss simultaneously (Myers et al., 2000; Mantyka-Pringle et al., 2015). Deforestation and land use change also leads to soil degradation, a reduced availability of fresh water, and air pollution problems (Foley et al., 2005). If forest stocks are degraded beyond an irreversible threshold, they will not only fail to provision critical services (Perman et al., 2011). A continued rise in CO_2 emissions and thus temperature is also likely to trigger a non-linear response from climate tipping elements and could lead to the dieback of the Amazon forest. This could trigger an exponential increase in emissions and entail irreversible and dangerous climate change (Lenton, 2011).

1.2 The international negotiations on REDD+

In the context of the United Nations Framework Convention of Climate Change (UNFCCC), deforestation has long been recognized as a central problem. However, the Kyoto Protocol only provided a policy measure for afforestation and reforestation under the Clean Development Mechanism in developing countries (UNFCCC, 2013). In 2005, Costa Rica and Papua New Guinea proposed that industrialized countries should be included by financially compensating developing countries for the reduction of emissions from deforestation (UNFCCC, 2005). At the conference in Bali in 2007, (decision 2/CP.13 UNFCCC, 2007) the proposal became a formalized part of the UNFCCC negotiations.

The climate negotiations in Copenhagen in 2009, concluded that the mechanism should comprise not only emission reductions from avoided deforestation and forest degradation, but also the role of forest conservation, sustainable management of forests and the enhancement of forest carbon stocks in developing countries (decision 4/CP.15 UNFCCC, 2009). During the negotiations in Cancun in 2010, parties agreed that countries should implement REDD+ in a phased approach, beginning with (i) the development of national strategies or action plans, followed by (ii) the implementation of these strategies, eventually followed by (iii) result-based payments (decision 1/CP.16, paragraph 73 UNFCCC, 2010).

These diplomatic decisions triggered a variety of different REDD+ readiness activities in developing countries. Due to uncertainty on the sources and the availability of funding for REDD+ projects, the activities ranged from pilot projects to more fundamental national reform ambitions. In 2011 in Durban, the parties agreed that both market-based approaches and non-market based approaches could support the result-based actions and that funding may come from a variety of sources (*"public and private, bilateral and multilateral, including alternative sources"* (decision 2/CP.17 UNFCCC, 2011).

Further negotiations in Warsaw in 2013 provided clarity on most of the methodological issues related to REDD+ emission reductions. Parties decided that forest cover monitoring should be reported at the national level, in order to prevent the displacement of emissions within one jurisdiction (decision 11/CP.19 UNFCCC, 2013b). The Paris Agreement encouraged the parties to take action to conserve and enhance forests under article 5 (UNFCCC, 2015). Furthermore, 82 developing countries included forest-related emission reductions or the enhancement of forest carbon stocks into their Nationally Determined Contributions (NDCs) (UNEP, 2015).

A study by Ecosystem Marketplace (2016) summarizes the main types of REDD+ payments across the globe in 2016. It shows that most transactions are currently voluntary. Transactions at the jurisdictional level are non-market result-based payments, usually provided by international donors or through bilateral agreements. Compliance market transactions mainly include domestic forest carbon offsets, although Ecosystem Marketplace (2016) suggests that this might change in the future.

1.3 Structural change and forest conservation

The challenge to design a global institutional framework that allows making forest compensation payment contracts requires an understanding of macroeconomic forces that systematically drive deforestation. Such an analysis is particularly helpful given the diversity of tropical countries.

Forest transition theory (Mather, 1992) aims at identifying such regularities. The forest transition occurs at the point when a country moves from a period of net forest cover loss to a period of forest cover stabilization or forest cover gain, a trajectory often visualized through a U-shaped graph and often referred to as a Kuznets curve for deforestation (Choumert et al., 2013). This forest transition is attributed to the emergence of labor- and land-saving agricultural technologies in combination with rural-urban migration (Mather and Needle, 2000; Walker and Smith, 1993).

In some countries, such a structural, economic transformation, associated with a decreased dependence on forest land as an input for economic activity has indeed been observed. Aside from high-income countries such as Denmark (Mather et al., 1998), France (Mather et al., 1999), Greece (Walker and Smith, 1993), Switzerland (Mather and Fairbairn, 2000), the United States, Canada, or the United Kingdom (Walker and Smith, 1993), emerging economies such as the Dominican Republic (Aide and Grau, 2004), El Salvador (Hecht et al., 2006), Puerto Rico (Rudel et al., 2000), Vietnam (Meyfroidt and Lambin, 2008), as well as some regions of Brazil (Baptista and Rudel, 2006), Ecuador (Rudel et al., 2002) and Mexico (Bray and Klepeis, 2005) have experienced a forest transition.

The existence of a homogenous pattern with systematic and generalizable macroeconomic causes, however, is disputed in the academic literature. Some authors, for instance Chiu (2012) associate forest transition with economic development. Culas (2012) goes as far as calculating a level of per capita income at which forest transition is likely to set in.

Other authors are more critical and discuss the need for conceptual refinements of the forest transition theory. Barbier et al. (2010), for instance, highlight the role of forest land values relative to other competing land values in their contribution. They find that due to market, policy and institutional failures, forest land values appear to be distorted at times, which can prevent the forest transition. Walker and Smith (1993) furthermore argue that the forest transition experience of some developed countries yields insufficient evidence to draw predictive conclusions for other countries. They argue that the conditions for the forest transition were different in the past, due to lower levels of international trade and more domestic control over forests. Meyfroidt et al. (2010) argue along similar lines. They find that leakage from countries that experience a forest transition to other countries is likely to prevent a global forest transition.

Rudel et al. (2005) propose two different channels as main causes for forest transition. They distinguish between an economic development and a forest scarcity path to forest transition. In contrast to the more commonly discussed economic development path, they find that forest transition can also be attributed to changing patterns of valuing forest products and ecosystem services (such as the prevention of soil erosion) derived from forests along a forest transition trajectory. Others express more fundamental skepticism: Meyfroidt and Lambin (2010) for instance find that countries do not necessarily experience a regular pattern of forest cover change over time or corresponding to a certain level of economic development.

Other authors reject the existence of a unifying theory on structural causes of forest transition altogether. Choumert (2013) conducts a meta-analysis of studies examining the existence of an environmental Kuznets curve in the context of forests. They find that while the number of publications on the topic has grown over the past two decades, the number of studies finding supportive evidence has declined. Along similar lines, Perz (2007) criticizes the effort of seeking macroeconomic "*grand theories*" on forest cover change patterns and argues that such a unifying pattern may not exist at all.

Due to the disagreement in the literature on the existence of homogenous (macroeconomic) forces for forest transition across countries, it seems pragmatic to choose a different analytical angle in order to identify policy options to tackle deforestation. Rather than aiming at understanding which fundamental driving forces cause forest transition in an almost deterministic way, it can be helpful to understand which structural features coincide today in forest-rich countries that restrain forest conservation from happening. Although this approach is not a causal analysis, it can contribute to clarifying the diversity of country contexts and consequently, the diversity of policy approaches required to make forest conservation in combination with international payments successful.

While there are certainly many more structural characteristics that complicate forest conservation in a range of countries, three characteristics emerge in particular. First of all, as discussed earlier in this introduction, the quality of political institutions plays a central role in whether a government is able to monitor and enforce forest conservation laws. In addition to the direct ability to enforce forest conservation, weak institutions - in particular weak contracting institutions - also hinder the emergence of more complex and more diversified economic activities (Nunn and Trefler, 2013). Consequently, countries with weak institutions can be locked into sectors that require low-quality contracting institutions, such as the extraction of natural resources, or agricultural activities.

Second, the sectoral composition of an economy plays a central role in whether a government is likely to be able to enforce forest conservation. Countries with a strong specialization in the production of agricultural goods and rather low levels of complexity in their economic production processes tend to have lower levels of income (Hidalgo and Hausmann, 2009). Consequently, a transition to areas of economic activity that rely less on deforestation is more complicated in such countries. Furthermore, in such contexts, REDD+ payments risk to not fully compensate the economic losses induced by the forgone agricultural income (Ollivier, 2012).

Third, low levels of gross national income reduce tax income and therefore the federal budget and consequently also the government's ability to engage in forest conservation efforts. More fundamentally, low income levels tend to coincide with food insecurity (Rahman et al., 2013). The prevalence of undernourishment is higher in low-income countries than in any other income group (FAO, 2013) and low-income households experience an inelastic demand for food products. Kongsamut et al. (1997) observe that as a household's income increases, the income fraction spent on agricultural goods declines. Low-income countries experiencing food insecurity are likely to face a sensitive reaction to stricter deforestation control rules and enforcement, because low-income household in particular are unable to substitute subsistence farming with other sources of food supply. Furthermore, expanding protected areas decreases the total amount of available agricultural land and raises land rents (Kalkuhl and Edenhofer, 2016). This situation can result in higher food prices and resistance, especially in densely populated areas (Brockington and Igoe, 2006; Oldekop et al., 2016).

In order to understand for which contexts, research on forest conservation policy options is most needed, it is crucial to understand, where - within the space of these three fundamental constraints - the most forest-rich countries are located. Figure 3 highlights countries with high forest cover (here identified with a threshold of having more than 50% of the national land area covered with forests in 2015 (FAO, 2016)) that have experienced forest cover loss between 2000 and 2015 in light green (FAO, 2016). It shows that, at least through a snapshot of 2015, countries with high levels of forest cover that are also experiencing forest cover loss do not fit a uniform pattern of socio-economic development. Rather, they are located on a socio-economic spectrum.

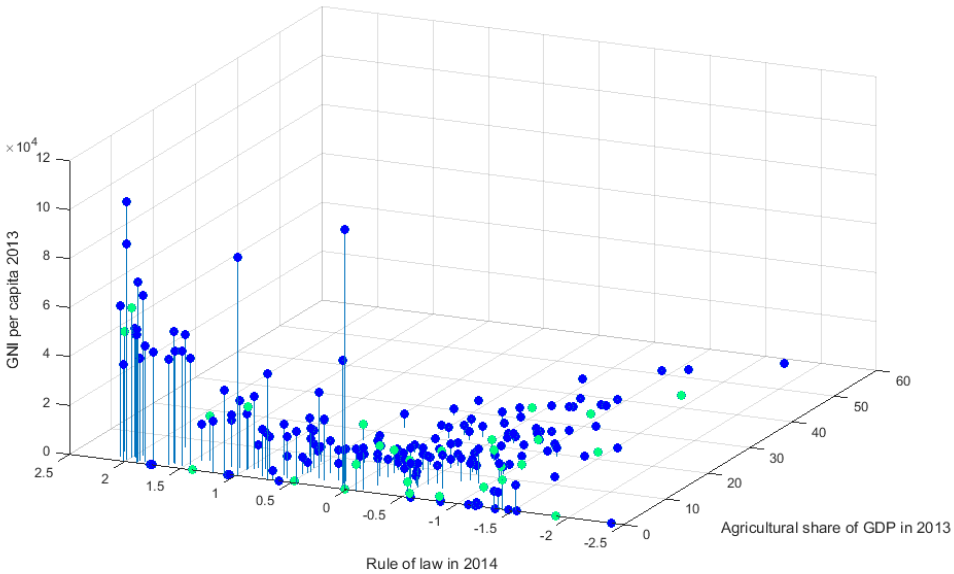

Figure 1.3: Shows all countries (in blue) and highlights countries with high forest cover (above 50%) that are experiencing deforestation in light green. Data sources: Forest cover and land area (FAO, 2016); Rule of law (World Bank, 2014); Agriculture as a contribution to GDP (World Bank, 2013a); Gross national income per capita (World Bank, 2013b).

Countries with relatively robust political institutions (with a rule of law indicator above 0), high levels of gross national income and low contributions of the agricultural sector to their GDP, located in the upper left front area of the graph, are likely to halt net forest cover loss with direct policy interventions, if there is political willingness to do so. Countries in the right back corner of the plot have weak political institutions, low levels of income and a strong contribution of the agricultural sector to their gross domestic product. In this context, it appears much more difficult to halt forest cover loss.

Karsenty and Ongolo (2012) make specific reference to low levels of territorial control and institutional quality when they argue that REDD+ policy discussions need to more actively take structural weaknesses of these countries into account.

In the next section, I discuss policy options for REDD+ that are currently debated in the academic literature and highlight that there is a research gap when it comes to the identification and analysis of policies for institutionally weak countries.

1.4 Policy options mapping for forest conservation

Edenhofer and Kowarsch (2015) suggest conducting cartographies of policy pathways for environmental policy making in order to provide scientific advice to identify policy options for sustainable natural resource governance. A similar idea is employed by Parry et al. (2016) who map, examine, and evaluate the likely effectiveness of a range of fiscal and regulatory policy options for CO_2 emission reductions in China. Pfaff et al. (2013) note that past policies have rarely addressed the underlying incentive structures that drive deforestation and that forest conservation policies in the context of REDD+ need to be tailored to the specific existing incentive structures in the corresponding countries.

In light of the new opportunities that arise through the focus on jurisdictional or national levels in the REDD+ context, it is useful to follow the logic of policy pathways mapping and explore existing knowledge on the implications of different policy options for forest conservation at jurisdictional and national levels. In contrast to exclusively focusing on localized conservation projects, a focus on jurisdictions allows thinking about how the existing regulatory framework and fiscal policies could be used to disincentivize deforestation.

1.4.1 Empirical analyses of policy options for forest conservation

First, a wide range of ex-post evaluations of the indirect effects of other policies on the forest cover have been conducted. Empirical studies assess, for instance, the effects of infrastructure projects, land tenure rights security, indirect effects of agricultural policies, the effects of decentralization policies, timber concessions, or the indirect effects of trade liberalization on forest cover.

There are several evaluations of the impact of the construction of road infrastructure on deforestation. Nelson and Hellerstein (1997), for instance, show that in Mexico the deforestation impact after road construction is higher if roads cut into previously undisturbed forests. Barber et al. (2014) analyze the effects of informal roads on deforestation in the Amazon. Busch and Ferretti-Gallon (2017) conduct a meta-analysis of 117 spatially explicit empirical studies on deforestation and conclude that one of the most promising approaches to stop deforestation is to reduce the intrusion of road networks into forests.

A variety of studies discuss the effects of land reforms and weak land tenure rights in specific contexts. Liscow (2013) evaluates the role of property rights allocation in Nicaragua after the agrarian reform in 2001 and concludes that it increases deforestation. Robinson (2014) conducts a meta-analysis of the literature and concludes that across different case studies, better land tenure rights tend to reduce deforestation.

Indirect effects of agricultural policies on deforestation are also discussed in the literature. For example, Andersen (1997) evaluates the effect of subsidized credits for agricultural development in the Brazilian Amazon on forest cover. Fisher and Shively (2007) find that agricultural subsidies in form of free maize seed allocation reduced forest extraction in Malawi.

In addition, a variety of empirical studies analyze the effects of decentralization on deforestation trends. While some studies conclude that decentralization is likely to reduce deforestation (for example Somanathan et al. (2009) in India, or Wright et al. (2016) in Bolivia), others find that decentralization leads to an increase in deforestation (e.g. Burgess et al. (2012) in Indonesia).

Furthermore, there are many empirical studies discussing the indirect consequences of macroeconomic policies on forest cover. Combes et al. (2015) find that a tighter monetary policy tends to hasten deforestation. Ehrhardt-Martinez et al. (2002) find that foreign debt and export dependency have had no effect on deforestation. López et al.(2005) show that trade openness is likely to increase deforestation. Guerineau et al. (2008) find that deforestation occurred as a response to interest rate increases in Latin American countries. Diarrassouba et al. (2009) find that countries with floating exchange rate regimes have the highest rate of deforestation.

Second, there is a range of studies examining the effectiveness of targeted and local conservation policies, such as protected areas, integrated conservation and development projects (ICDP), and payments for ecosystem services (PES). Miteva et al. (2012) conduct a literature review on protected areas and conclude that improved impact evaluations are needed.

Holland et al. (2014) use a random coefficient model to find that protected areas are associated with less deforestation in the Ecuadorian context. There is also a more critical strand in the literature that questions the effectiveness of protected areas. Spracklen et al. (2015) conduct a global analysis of protected areas for subtropical moist forests and find that 73% of protected areas experience substantial deforestation pressure. The pressure is strongest in Asian countries. Joppa and Pfaff (2009) show that across countries, protected areas tend to be biased towards remote locations with high elevations and steep slopes that are unlikely to face land conversion pressure, even in the absence of protection. Nelson and Chomitz (2011) find that multiple-use protected areas are more effective than strictly protected areas in Latin America. Ferraro et al. (2013) find in an analysis of Bolivia, Costa Rica, Indonesia, and Thailand that more strictly protected areas are more effective, but that this effect is not equally strong across countries.

There are also studies evaluating the effectiveness of integrated conservation and development projects (ICDP). Garnett et al. (2007) and Bauch et al. (2014) find little evidence for their effectiveness. Weber et al. (2011) evaluate ICDPs in the Brazilian Amazon and find no evidence for their conservation impact.

Furthermore, a range of studies examine the effectiveness of payments for ecosystem service (PES) programs. Schomers and Matzdorf (2013) conduct a review and show that the number of publications on PES has significantly grown since the late 1990s. They also note that the majority of studies focus on Latin American countries. The authors distinguish between PES schemes following a purely Coasian (private resource users contract) logic that restricts the task of the government to the initial allocation of property rights and a Pigouvian conceptualization, where resource users cannot be excluded and the government uses financial incentives to regulate their behavior.

Examples of the former are the Paso de Caballo river basin in Nicaragua, where upstream landowners are paid by downstream households for forest conservation efforts, and ecotourism programs in Cambodia, where community members are paid for not hunting rare bird species (Clements et al., 2010).

An example of the latter is the Mexican PES program, where the government allocates payments for the conservation of forests relevant for watershed protection to private land owners and *ejidos* (communities) (Alix-Garcia et al., 2009). Costedoat et al.(2015) empirically evaluate the effectiveness of the program for the state of Chiapas and find that the program had the effect of additional conservation compared to control areas. Another example is the Colombian PES program in Quindío, where Pagiola et al. (2016) examine to what extent PES-related conservation efforts are permanent and conclude that PES-induced land conservation is sustained beyond the payment cycle.

Finally, Costa Rica and Brazil are two cases of integrated national forest conservation policy that have been successfully implemented and extensively examined and evaluated in the past.

Costa Rica's national PES scheme for forest conservation also follows a Pigouvian conceptualization, but at a larger scale than many other programs (2016), which has been extensively studied in the literature. In contrast to more targeted and localized conservation efforts, Costa Rica developed a genuine national conservation strategy deploying payments for ecosystem services as the main policy instrument in combination with a deforestation ban and a phase-out of fiscal incentives to land clearing UNEP (2012). Payments are made for reforestation and sustainable forest management (2008) and are financed through earmarked water (US$ 3.6 million between 2007 and 2010 (Porras et al., 2013) and fossil fuel tax revenues (on average US$ 11.3 million per year (Murillo et al., 2011)) and carbon payments sold through the Chicago Climate Exchange as well as international loans and agreements.

Various empirical studies have evaluated the efficiency of the Costa Rican program. Some authors criticize the fact that payments do not imply long-term obligations for farmers to restore degraded land (Sierra and Russman, 2006) and that controlling for location (land that is unattractive for settlement and land conversion) actually reduces the conservation impact of the program (Andam et al., 2008). Pfaff et al. (2009) confirm these findings for a later period. Arriagada et al. (2015) examine whether PES recipients encounter spill-overs from the PES program and find that they experience neither income increases, nor increases in well-being.

Brazil's national forest conservation enforcement program for the Amazon forest that started in 2004 is not a PES scheme (Schomers and Matzdorf, 2013), but is more generally described as a field-based forest law enforcement policy (Börner et al., 2015). It has been successful during both of its implementation periods. Deforestation decreased, while the country experienced economic growth and a rapid recovery from the global economic crisis (World Bank, 2015). Several empirical studies attribute the political impetus to civil society pressure and the effectiveness in deforestation reduction to the government's enforcement strategy (Assunção et al., 2015; Cisneros et al., 2015; Hargrave and Kis-Katos, 2013). At the same time, during a period of relative political instability in Brazil between 2015 and 2016, deforestation rates have increased again by 29% (INPE, 2017).

However, forest conservation in both Costa Rica and Brazil were supported by a considerable national impetus for conservation and did not rely on international compensation payments alone (in particular in the case of Brazil).

1.4.2 Economic modeling literature on policy options for forest conservation

Due to the fact that REDD+ is a relatively recent program, there is only a limited number of empirical analyses of national level policy strategies (e.g. in form of fiscal policies) that are incentivized by international payments. It is therefore interesting to consider the (ex-ante) economic modelling literature in this regard, in order to see which lessons can be learnt for policy design in the REDD+ context.

A review of economic models on deforestation reveals that (i) most such models published in earlier periods analyze the indirect impacts of other policies on forest cover and (ii) that only recently, in the academic discussions on PES and REDD+, studies began to analyze optimal policy designs of jurisdictional and national level forest conservation policies in combination with REDD+ policies with economic models.

A range of publications analyze the effects of trade liberalization on forest cover. In her seminal paper, Chichilniskly (1994) shows that increasing trade openness in developing countries with weak property rights enforcement capacities is likely to lead to an unsustainable overuse of natural resources. Brander and Taylor (1997), draw a similar conclusion, in particular for small, open economies. Copeland and Taylor (2009) use a model, where the capacity to manage resources sustainably is endogenously determined. They find that enforcement capacity, harvesting capacity, and returns of alternative exports determine whether trade leads to an unsustainable over-extraction of resources. Arcand et al. (2008) find that depreciation in the real exchange rate is likely to increase deforestation. Several authors furthermore examine the effects of timber prices and timber trade policies. Maestad (2001) analyzes the effect of timber trade restrictions on tropical deforestation and shows that logging is reduced when timber trade restrictions reduce log prices equally for all tree species. Walker and Smith (1993) develop a sequential decision model to analyze the effectiveness of concessional timber logging systems with partial inspections of compliance. Deacon (1995) assesses the role of taxes and royalties on timber harvest in a general equilibrium framework.

There is also a range of models analyzing how land tenure rights affect deforestation processes. Angelsen (1999) examines the role of land titling programs and finds that deforestation can become an investment strategy for the farmer, when land rights are allocated. In contrast, Mendelsohn (1994) finds that insecure tenure rights are likely to drive deforestation.

Intensive theoretical discussions also exist on the role of agricultural intensification (Byerlee et al., 2014; Ceddia et al., 2013; Villoria et al., 2014). In line with the Jevons paradox, some scholars find that agricultural intensification policies are likely to increase deforestation. Others find - in line with the Borlaug hypothesis - that agricultural intensification is likely to be associated with less deforestation. Hertel (2012) finds that the dominating effect - Jevons or Borlaug - of technological change (agricultural productivity) depends on the elasticities of demand in the different consuming (or exporting) sectors. Takasaki (2007) examines the effect of agricultural policies in different land and labor market settings and finds that poor farmers are bought out of their newly cleared land. He therefore recommends that policies target farmers at forest frontiers to transfer agricultural technologies and improve their productivity to reduce deforestation as well as policies to reduce land price distortions. Theoretical discussions also use von Thünen-models to examine the role of the spatial dynamics and proximity to forests (Angelsen, 2007). Southgate (1990) analyzes the causes of forest degradation along agricultural frontiers. Jones and O'Neill (1995) also examine the role of spatial heterogeneity in economic activities in an urban-rural two-sector framework and assess the effect of different policies (taxation, infrastructure investments) on deforestation trends. They show that deforestation can be reduced through tax revenue reinvestments in infrastructure that facilitates the development of manufacturing industries.

With the emergence of more large-scale PES and REDD+ policy projects, economic models are increasingly used to examine national and fiscal policy strategies to reduce deforestation in combination with international transfer payments. Using a von Thünen-model, Angelsen (2010) analyzes policies to reduce the rents of extensive agriculture, policies to increase protective forest rents, and policies to establish protected areas in combination with possible REDD+ payments. He concludes that policies to stimulate agricultural intensification (improved technologies, roads, or land tenure) do not always lead to reduced deforestation and highlights that agricultural policies should promote technologies that are unlikely to lead to increased deforestation. Kalkuhl and Edenhofer (2016) combine a von Thünen model with a Ramsey-type model to discuss the forest conserving property of land taxes in the REDD+ context and highlight that in contrast to protected areas, they do not create windfall profits in the form of increased land rents (due to increased land scarcity) to land owners, but instead allow to generate public revenue that can be reinvested for the provision of public goods. They argue that international REDD+ payments (that function like a subsidy), should be combined with equally high taxes on non-primary forest land in order to internalize the global benefit of forest conservation. Ollivier (2012), uses a two-sector model to assess the effects of international REDD+ payments and economic growth and forest conservation trade-offs. She finds that for low transfer schemes, land sparing is observed, but for higher transfer schemes the increase in REDD+ transfers does not offset the decrease in agricultural output and the economy experiences a net welfare loss. Barua et al. (2014) use an infinite horizon dynamic model in order to derive optimal levels of land income taxation in combination with carbon compensation payments. The authors find that the government can use an optimal land income taxation level and pass on REDD+ payments from the international community to consumers to ensure a socially optimal level of forest conservation.

In summary, this literature review - even though focused on the role of institutions and thus not reflecting the entire REDD+ and forest conservation literature - on policy options reveals that while a range of studies have analyzed indirect effects of other policies on forest cover, the literature examining dedicated jurisdictional and national level forest conservation policies in combination with possible international transfer payments is only emerging and does not provide an in-depth analysis and discussion of policy options for countries with weak political institutions. My goal is to contribute to this research agenda with this dissertation.

References

Acemoglu, D., Egorov, G., and Sonin, K. (2010). Political selection and persistence of bad governments. *The Quarterly Journal of Economics*, 125(4):1511–1575.

Acemoglu, D., Garcia-Jimeno, C., and Robinson, J. A. (2015). State capacity and economic development: a network approach. *The American Economic Review*, 105(8):2364–2409.

Acemoglu, D., Johnson, S., and Robinson, J. A. (2001). The colonial origins of comparative development: an empirical investigation. *American Economic Review*, 91(5):1369–1401.

Acemoglu, D., Ticchi, D., and Vindigni, A. (2011). Emergence and persistence of inefficient states. *Journal of the European Economic Association*, 9(2):177–208.

Aide, T. M. and Grau, H. R. (2004). Globalization, migration, and Latin American ecosystems. *Science*, 305(5692):1915–1916.

Alix-Garcia, J., Janvry, Alain Sadoulet, E., and Manuel, J. (2009). Lessons learned from Mexico's payment for environmental services program. In Lipper, L., Sakuyama, T., Stringer, R., and Zilberman, D., editors, *Payment for Environmental Services in Agricultural Landscapes*, number 31 in Natural Resource Management and Policy, pages 163–188. Springer New York, New York, USA.

Amsden, A. H., Dongyi, L., and Xiaoming, Z. (1996). China's macroeconomy, environment, and alternative transition model. *World Development*, 24(2):273–286.

An, L. (2012). Modeling human decisions in coupled human and natural systems: review of agent-based models. *Ecological Modelling*, 229(2012):25–36.

Andam, K. S., Ferraro, P. J., Pfaff, A., Sanchez-Azofeifa, G. A., and Robalino, J. A. (2008). Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences*, 105(42):16089–16094.

Andersen, L. E. (1997). Modelling the relationship between government policy, economic growth, and deforestation in the Brazilian Amazon. *Social Science Research Network (SSRN) Working Paper*, 1997-2(54986):1–36.

Angelsen, A. (2007). Forest cover change in space and time: combining the von Thünen and forest transition theories. *World Bank Policy Research Working Paper*, 2007(4117):1–43.

Angelsen, A. (2008). *Moving ahead with REDD: issues, options and implications.* CIFOR, Bogor, Indonesia.

Angelsen, A. (2010). Policies for reduced deforestation and their impact on agricultural production. *Proceedings of the National Academy of Sciences*, 107(46):19639–19644.

Angelsen, A. and Kaimowitz, D. (1999). Rethinking the causes of deforestation: lessons from economic models. *The World Bank Research Observer*, 14(1):73–98.

Aoki, M. (2001). *Toward a comparative institutional analysis.* Massachusetts Institute of Technology MIT Press, Cambridge, USA.

Arcand, J.-L., Guillaumont, P., and Jeanneney, S. G. (2008). Deforestation and the real exchange rate. *Journal of Development Economics*, 86(2):242 – 262.

Arriagada, R. A., Echeverria, C. M., and Moya, D. E. (2016). Creating protected areas on public lands: is there room for additional conservation? *PLOS ONE*, 11(2):e0148094.

Arriagada, R. A., Sills, E. O., Ferraro, P. J., and Pattanayak, S. K. (2015). Do payments pay off? Evidence from participation in Costa Rica's PES program. *PLOS ONE*, 10(7):e0131544.

Assunção, J., Gandour, C., and Rocha, R. (2015). Deforestation slowdown in the Brazilian Amazon: prices or policies? *Environment and Development Economics*, 20(6):697–722.

Baptista, S. R. and Rudel, T. K. (2006). A reemerging Atlantic forest? Urbanization, industrialization and the forest transition in Santa Catarina, Southern Brazil. *Environmental Conservation*, 33(3):195–202.

Barber, C. P., Cochrane, M. A., Souza, C. M., and Laurance, W. F. (2014). Roads, deforestation, and the mitigating effect of protected areas in the Amazon. *Biological Conservation*, 177(2014):203–209.

Barbier, E. B., Burgess, J. C., and Grainger, A. (2010). The forest transition: towards a more comprehensive theoretical framework. *Land Use Policy*, 27(2):98–107.

Barbier, E. B., Damania, R., and Léonard, D. (2005). Corruption, trade and resource conversion. *Journal of Environmental Economics and Management*, 50(2):276–299.

Barbier, E. B. and Tesfaw, A. (2015). Explaining forest transitions: the role of governance. *Ecological Economics*, 119(2015):252–261.

Barua, S. K., Lintunen, J., Uusivuori, J., and Kuuluvainen, J. (2014). On the economics of tropical deforestation: carbon credit markets and national policies. *Forest Policy and Economics*, 47(2014):36–45.

Bauch, S. C., Sills, E. O., and Pattanayak, S. K. (2014). Have we managed to integrate conservation and development? ICDP impacts in the Brazilian Amazon. *World Development*, 64(Supplement 1):S135–S148.

Benítez, P. C., McCallum, I., Obersteiner, M., and Yamagata, Y. (2007). Global potential for carbon sequestration: geographical distribution, country risk and policy implications. *Ecological Economics*, 60(3):572–583.

Bernard, F., Minang, P. A., Adkins, B., and Freund, J. T. (2014). REDD+ projects and national-level readiness processes: a case study from Kenya. *Climate Policy*, 14(6):788–800.

Besley, T. J. and Ghatak, M. (2009). Property rights and economic development. In *Handbook of Development Economics*, volume 5, pages 4039–5061. North Holland, Amsterdam, Netherlands.

Besley, T. J. and Persson, T. (2013). Taxation and development. *Social Science Research Network (SSRN) Working Paper*, 2013(2210278):1–81.

Börner, J., Baylis, K., Corbera, E., Ezzine-de Blas, D., Ferraro, P. J., Honey-Rosés, J., Lapeyre, R., Persson, U. M., and Wunder, S. (2016). Emerging evidence on the effectiveness of tropical forest conservation. *PLOS ONE*, 11(11):e0159152.

Börner, J., Kis-Katos, K., Hargrave, J., and König, K. (2015). Post-crackdown effectiveness of field-based forest law enforcement in the Brazilian Amazon. *PLOS ONE*, 10(4):e0121544.

Brander, J. A. and Taylor, M. S. (1997). International trade and open-access renewable resources: the small open economy case. *The Canadian Journal of Economics*, 30(3):526–552.

Bray, D. B. and Klepeis, P. (2005). Deforestation, forest transitions, and institutions for sustainability in Southeastern Mexico, 1900 - 2000. *Environment and History*, 11(2):195–223.

Brimont, L., Ezzine-de Blas, D., Karsenty, A., and Toulon, A. (2015). Achieving conservation and equity amidst extreme poverty and climate risk: the Makira REDD+ project in Madagascar. *Forests*, 6(3):748–768.

Brockington, D. and Igoe, J. (2006). Eviction for conservation: a global overview. *Conservation and Society*, 4(3):424–470.

Buchanan, J. M. and Stubblebine, W. C. (1962). Externality. *Economica*, 29(116):371–384.

Buitenzorgy, M. and Mol, A. P. (2011). Does democracy lead to a better environment? Deforestation and the democratic transition peak. *Environmental and Resource Economics*, 48(1):59–70.

Burgess, R., Hansen, M., Olken, B. A., Potapov, P., and Sieber, S. (2012). The political economy of deforestation in the tropics. *The Quarterly Journal of Economics*, 127(4):1707–1754.

Busch, J. and Ferretti-Gallon, K. (2017). What drives deforestation and what stops it? A meta-analysis. *Review of Environmental Economics and Policy*, 11(1):3–23.

Byerlee, D., Stevenson, J., and Villoria, N. (2014). Does intensification slow crop land expansion or encourage deforestation? *Global Food Security*, 3(2):92–98.

Campbell, R. and Sowden, L. (1985). *Paradoxes of rationality and cooperation: prisoner's dilemma and Newcomb's problem.* University of British Columbia (UBC) Press, British Columbia, Canada.

Caplow, S., Jagger, P., Lawlor, K., and Sills, E. (2011). Evaluating land use and livelihood impacts of early forest carbon projects: lessons for learning about REDD+. *Environmental Science & Policy*, 14(2):152–167.

Cattau, M. E., Harrison, M. E., Shinyo, I., Tungau, S., Uriarte, M., and DeFries, R. (2016). Sources of anthropogenic fire ignitions on the peat-swamp landscape in Kalimantan, Indonesia. *Global Environmental Change*, 39(2016):205–219.

Ceddia, M. G., Sedlacek, S., Bardsley, N. O., and Gomez-y Paloma, S. (2013). Sustainable agricultural intensification or Jevons paradox? The role of public governance in tropical South America. *Global Environmental Change*, 23(5):1052–1063.

Chagas, T., Streck, C., O'Sullivan, R., Olander, J., and Seifert-Granzin, J. (2011). *Nested Approaches to REDD+: an overview of issues and options.* Climate Focus, Amsterdam, Netherlands.

Chichilnisky, G. (1994). North-South trade and the global environment. *The American Economic Review*, 84(4):851–874.

Chiu, Y.-B. (2012). Deforestation and the environmental Kuznets curve in developing countries: a panel smooth transition regression approach. *Canadian Journal of Agricultural Economics*, 60(2):177–194.

Chong, A., La Porta, R., Lopez-de Silanes, F., and Shleifer, A. (2014). Letter grading government efficiency. *Journal of the European Economic Association*, 12(2):277–299.

Choumert, J., Motel, P. C., and Dakpo, H. K. (2013). Is the environmental Kuznets curve for deforestation a threatened theory? A meta-analysis of the literature. *Ecological Economics*, 90(2013):19–28.

Cingolani, L., Thomsson, K., and de Crombrugghe, D. (2015). Minding Weber more than ever? The impacts of state capacity and bureaucratic autonomy on development goals. *World Development*, 72(2015):191–207.

Cisneros, E., Zhou, S. L., and Börner, J. (2015). Naming and shaming for conservation: evidence from the Brazilian Amazon. *PLOS ONE*, 10(9):e0136402.

Clements, T., John, A., Nielsen, K., An, D., Tan, S., and Milner-Gulland, E. J. (2010). Payments for biodiversity conservation in the context of weak institutions: comparison of three programs from Cambodia. *Ecological Economics*, 69(6):1283–1291.

CN-REDD (2015). *Stratégie-cadre nationale REDD de la République Démocratique du Congo*, volume 3. Coordination nationale REDD, Forest Carbon Partnership Facility, UN-REDD, Kinshasa, Democratic Republic of the Congo.

Coase, R. H. (1960). The problem of social cost. *Journal of Law and Economics*, 3(1):1–44.

Combes, J.-L., Motel, P. C., Minea, A., and Villieu, P. (2015). Deforestation and seigniorage in developing countries: a tradeoff? *Ecological Economics*, 116(2015):220–230.

Copeland, B. R. and Taylor, M. S. (2009). Trade, tragedy, and the commons. *American Economic Review*, 99(3):725–749.

Corbera, E. and Schroeder, H. (2011). Governing and implementing REDD+. *Environmental Science & Policy*, 14(2):89–99.

Costedoat, S., Corbera, E., Ezzine-de Blas, D., Honey-Rosés, J., Baylis, K., and Castillo-Santiago, M. A. (2015). How effective are biodiversity conservation payments in Mexico? *PLOS ONE*, 10(3):e0119881.

Culas, R. J. (2012). REDD and forest transition: tunneling through the environmental Kuznets curve. *Ecological Economics*, 79(2012):44–51.

De Vos, M. G., Janssen, P. H., Kok, M. T., Frantzi, S., Dellas, E., Pattberg, P., Petersen, A. C., and Biermann, F. (2013). Formalizing knowledge on international environmental regimes: a first step towards integrating political science in integrated assessments of global environmental change. *Environmental Modelling & Software*, 44(2013):101–112.

Deacon, R. T. (1994). Deforestation and the rule of law in a cross-section of countries. *Land Economics*, 70(4):414–430.

Deacon, R. T. (1995). Assessing the relationship between government policy and deforestation. *Journal of Environmental Economics and Management*, 28(1):1–18.

Dell, M., Jones, B. F., and Olken, B. A. (2014). What do we learn from the weather? The new climate - economy literature. *Journal of Economic Literature*, 52(3):740–798.

Diarrassouba, M. and Boubacar, I. (2009). Deforestation in Sub-Saharan Africa. In *Southern Agricultural Economics Association Annual Meeting, Atlanta, USA*.

EcosystemMarketplace (2016). View from the understory - state of forest carbon finance 2016. URL: http://foresttrends.org/releases/p/sofcf2016 (accessed 21/12/2016).

Edenhofer, O. and Kowarsch, M. (2015). Cartography of pathways: a new model for environmental policy assessments. *Environmental Science & Policy*, 51(2015):56–64.

Ehrhardt-Martinez, K., Crenshaw, E. M., and Jenkins, J. C. (2002). Deforestation and the environmental Kuznets curve: a cross-national investigation of intervening mechanisms. *Social Science Quarterly*, 83(1):226–243.

Ehui, S. K., Hertel, T. W., and Preckel, P. V. (1990). Forest resource depletion, soil dynamics, and agricultural productivity in the tropics. *Journal of Environmental Economics and Management*, 18(2):136–154.

Evans, P. B. (1995). *Embedded autonomy: states and industrial transformation.* Princeton University Press, USA.

FAO (2013). Prevalence of undernourishment (%) (3-year average). URL: http //faostat3.fao.org/browse/D/FS/E (accessed 13/10/2016).

FAO (2016). *Global Forest Resources Assessment 2015 - how are the world's forests changing?* Food and Agricultural Organization of the United Nations (FAO), Second edition. The figure is reproduced with permission, online source: http://www.fao.org/publications/card/en/c/770b9ce69f654a46a9af-1abbb0f71ab1/ (accessed (15/11/2016).

FAO and PROFOR (2011). Framework for assessing and monitoring forest governance. URL: http://www.fao.org/docrep/014/i2227e/i2227e00.pdf (accessed 23/02/2017).

FCPF (2015). Forest Carbon Partnership Facility REDD+ partner countries. URL: https://www.forestcarbonpartnership.org/redd-countries-1 (accessed 08/12/2015).

Ferraro, P. J., Hanauer, M. M., Miteva, D. A., Canavire-Bacarreza, G. J., Pattanayak, S. K., and Sims, K. R. (2013). More strictly protected areas are not necessarily more protective: evidence from Bolivia, Costa Rica, Indonesia, and Thailand. *Environmental Research Letters*, 8(2):025011 (7pp).

Ferreira, S. (2004). Deforestation, property rights, and international trade. *Land Economics*, 80(2):174–193.

Finan, F., Olken, B. A., and Pande, R. (2015). The personnel economics of the state. *National Bureau of Economic Research (NBER) Working Paper Series*, 2015(21825):1–49.

Fisher, M. and Shively, G. E. (2007). Agricultural subsidies and forest pressure in Malawi's "Miombo" woodlands. *Journal of Agricultural and Resource Economics*, 32(2):349–362.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F. S., Coe, M. T., Daily, G. C., Gibbs, H. K., Helkowski, J. H., Holloway, T., Howard, E. A., Kucharik, C. J., Monfreda, C., Patz, J. A., Prentice, I. C., Ramankutty, N., and Snyder, P. K. (2005). Global consequences of land use. *Science*, 309(5734):570–574.

Galinato, G. I. and Galinato, S. P. (2016). The effects of government spending on deforestation due to agricultural land expansion and CO_2 related emissions. *Ecological Economics*, 122:43–53.

Garnett, S., Sayer, J., and Du Toit, J. T. (2007). Improving the effectiveness of interventions to balance conservation and development: a conceptual framework. *Ecology and Society*, 12(1):online.

Gaveau, D. L., Pirard, R., Salim, M. A., Tonoto, P., Yaen, H., Parks, S. A., and Carmenta, R. (2016). Overlapping land claims limit the use of satellites to monitor no-deforestation commitments and no-burning compliance. *Conservation Letters*, 10(2):257–264.

Geist, H. J. and Lambin, E. F. (2002). Proximate causes and underlying driving forces of tropical deforestation. *BioScience*, 52(2):143–150.

Gennaioli, N. and Rainer, I. (2007). The modern impact of precolonial centralization in Africa. *Journal of Economic Growth*, 12(3):185–234.

Goulder, L. and Pizer, W. (2006). The economics of climate change. URL: http://www.rff.org/files/sharepoint/WorkImages/Download/RFF-DP-06-06.pdf (accessed 19/01/2017).

Guerineau, S., Motel, P. C., and Combes, J.-L. (2008). Deforestation and credit cycles in Latin American countries. *CERDI Working Paper*, 2008(200808):1–27.

Hargrave, J. and Kis-Katos, K. (2013). Economic causes of deforestation in the Brazilian Amazon: a panel data analysis for the 2000s. *Environmental and Resource Economics*, 54(4):471–494.

Hayes, T. and Persha, L. (2010). Nesting local forestry initiatives: revisiting community forest management in a REDD+ world. *Forest Policy and Economics*, 12(8):545–553.

Hecht, S. B., Kandel, S., Gomes, I., Cuellar, N., and Rosa, H. (2006). Globalization, forest resurgence, and environmental politics in El Salvador. *World Development*, 34(2):308–323.

Heisenberg, W. (1927). Über den anschaulichen Inhalt der quantentheoretischen Kinematik und Mechanik. *Zeitschrift für Physik*, 43(3):172–198.

Herbst, J. (2000). *States and power in Africa: comparative lessons in authority and control*. Princeton University Press, Princetion, USA.

Hertel, T. W. (2012). Implications of agricultural productivity for global cropland use and GHG emissions: Borlaug vs. Jevons. *Global Trade Analysis Project (GTAP) Working Paper*, 2012(69):1–36.

Hett, C., Heinimann, A., Epprecht, M., Messerli, P., and Hurni, K. (2012). Carbon pools and poverty peaks in Lao PDR: spatial data inform policy-making for REDD+ at the national level. *Mountain Research and Development*, 32(4):390–399.

Hidalgo, C. A. and Hausmann, R. (2009). The building blocks of economic complexity. *Proceedings of the National Academy of Sciences*, 106(26):10570–10575.

Holland, M. B., de Koning, F., Morales, M., Naughton-Treves, L., Robinson, B. E., and Suárez, L. (2014). Complex tenure and deforestation: implications for conservation incentives in the Ecuadorian Amazon. *World Development*, 55(2014):21–36.

Hosonuma, N., Herold, M., Sy, V. D., Fries, R. S. D., Brockhaus, M., Verchot, L., Angelsen, A., and Romijn, E. (2012). An assessment of deforestation and forest degradation drivers in developing countries. *Environmental Research Letters*, 7(4):044009.

Houghton, R. A., House, J. I., Pongratz, J., Van der Werf, G. R., DeFries, R. S., Hansen, M. C., Le Quéré, C., and Ramankutty, N. (2012). Carbon emissions from land use and land-cover change. *Biogeosciences*, 9(12):5125–5142.

INPE (2017). Projeto Prodes - monitoramento da floresta Amazônica Brasileira por satélite. URL: http://www.obt.inpe.br/prodes/index.php (accessed 23/02/2017).

Janssen, M. A., Walker, B. H., Langridge, J., and Abel, N. (2000). An adaptive agent model for analysing co-evolution of management and policies in a complex rangeland system. *Ecological Modelling*, 131(2):249–268.

Jones, D. W. and O'Neill, R. V. (1995). Development policies, urban unemployment and deforestation: the role of infrastructure and tax policy in a two-sector model. *Journal of Regional Science*, 1995(1):135–153.

Joppa, L. N. and Pfaff, A. (2009). High and far: biases in the location of protected areas. *PLOS ONE*, 4(12):e8273.

Kalkuhl, M. and Edenhofer, O. (2016). Ramsey meets Thünen: the impact of land taxes on economic development and land conservation. *International Tax and Public Finance*, 24(2):1–31.

Kanninen, M., Murdiyarso, D., Seymour, F., Angelsen, A., Wunder, S., and German, L. (2007). *Do trees grow on money? The implications of deforestation research for policies to promote REDD.* CIFOR, Bogor, Indonesia.

Karsenty, A. and Ongolo, S. (2012). Can "fragile states" decide to reduce their deforestation? The inappropriate use of the theory of incentives with respect to the REDD mechanism. *Forest Policy and Economics*, 18(2012):38–45.

Kaufmann, D. and Kraay, A. (2008). Governance indicators: where are we, where should we be going? *The World Bank Research Observer*, 23(1):1–30.

Kishor, N. and Belle, A. (2004). Does improved governance contribute to sustainable forest management? *Journal of Sustainable Forestry*, 19(1):55–79.

Kongsamut, P., Rebelo, S., and Xie, D. (1997). Beyond balanced growth. *National Bureau of Economic Research (NBER) Working Paper*, 1997(6159):1–57.

Lambin, E. F. and Meyfroidt, P. (2010). Land use transitions: socio-ecological feedback versus socio-economic change. *Land Use Policy*, 27(2):108–118.

Le Quéré, C. L., Andres, R. J., Boden, T., Conway, T., Houghton, R. A., House, J. I., Marland, G., Peters, G. P., Van der Werf, G. R., and Ahlström, A. (2013). The global carbon budget 1959 - 2011. *Earth System Science Data*, 5(1):165–185.

Lenton, T. M. (2011). Early warning of climate tipping points. *Nature Climate Change*, 1(4):201–209.

Li, Q. and Reuveny, R. (2006). Democracy and environmental degradation. *International Studies Quarterly*, 50(4):935–956.

Liscow, Z. D. (2013). Do property rights promote investment but cause deforestation? Quasi-experimental evidence from Nicaragua. *Journal of Environmental Economics and Management*, 65(2):241–261.

Lobianco, A., Delacote, P., Caurla, S., and Barkaoui, A. (2015). The importance of introducing spatial heterogeneity in bio-economic forest models: insights gleaned from FFSM++. *Ecological Modelling*, 309-310(2015):82–92.

López, R. and Galinato, G. I. (2005). Deforestation and forest-induced carbon dioxide emissions in tropical countries: how do governance and trade openness affect the forest-income relationship? *The Journal of Environment & Development*, 14(1):1–29.

Maestad, O. (2001). Timber trade restrictions and tropical deforestation: a forest mining approach. *Resource and Energy Economics*, 23(2):111–132.

Mantyka-Pringle, C. S., Visconti, P., Di Marco, M., Martin, T. G., Rondinini, C., and Rhodes, J. R. (2015). Climate change modifies risk of global biodiversity loss due to land-cover change. *Biological Conservation*, 187(2015):103–111.

Mather, A. and Needle, C. (2000). The relationships of population and forest trends. *The Geographical Journal*, 166(1):2–13.

Mather, A. S. (1992). The forest transition. *Area*, 24(4):367–379.

Mather, A. S. and Fairbairn, J. (2000). From floods to reforestation: the forest transition in Switzerland. *Environment and History*, 6(4):399–421.

Mather, A. S., Fairbairn, J., and Needle, C. L. (1999). The course and drivers of the forest transition: the case of France. *Journal of Rural Studies*, 15(1):65–90.

Mather, A. S., Needle, C. L., and Coull, J. R. (1998). From resource crisis to sustainability: the forest transition in Denmark. *International Journal of Sustainable Development & World Ecology*, 5(3):182–193.

Mendelsohn, R. (1994). Property rights and tropical deforestation. *Oxford Economic Papers*, 46(Special Issue on Environmental Economics):750–756.

Meyfroidt, P. and Lambin, E. F. (2008). Forest transition in Vietnam and its environmental impacts. *Global Change Biology*, 14(6):1319–1336.

Meyfroidt, P., Rudel, T. K., and Lambin, E. F. (2010). Forest transitions, trade, and the global displacement of land use. *Proceedings of the National Academy of Sciences*, 107(49):20917–20922.

Michalopoulos, S. and Papaioannou, E. (2013). Pre-colonial ethnic institutions and contemporary African development. *Econometrica*, 81(1):113–152.

Midlarsky, M. I. (1998). Democracy and the environment: an empirical assessment. *Journal of Peace Research*, 35(3):341–361.

Miteva, D. A., Pattanayak, S. K., and Ferraro, P. J. (2012). Evaluation of biodiversity policy instruments: what works and what doesn't? *Oxford Review of Economic Policy*, 28(1):69–92.

Murillo, R., Kilian, B., and Castro, R. (2011). *Leveraging and sustainability of PES. Ecosystem services from agriculture and agroforestry. Measurement and payment.* Earthscan, London, United Kingdom.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A. B., and Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, 403(6772):853–858.

Nabuurs, G., Masera, O., Andrasko, K., Benitez-Ponce, P., Boer, R., Dutschke, M., Elsiddig, E., Ford-Robertson, J., Frumhoff, P., Karjalainen, T., Krankina, O., Kurz, W., Matsumoto, M., Oyhantcabal, W., Ravindranath, N., Sanz Sanchez, M., and Zhang, X. (2007). Chapter 9: Forestry - AR4 WGIII. In *Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, United Kingdom.

Nash, J. (1951). Non-cooperative games. *Annals of Mathematics*, 54(2):286–295.

Neeff, T., Göhler, D., and Ascui, F. (2014). Finding a path for REDD+ between ODA and the CDM. *Climate Policy*, 14(2):149–166.

Nelson, A. and Chomitz, K. M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLOS ONE*, 6(8):e22722.

Nelson, G. C. and Hellerstein, D. (1997). Do roads cause deforestation? Using satellite images in econometric analysis of land use. *American Journal of Agricultural Economics*, 79(1):80–88.

Nunn, N. and Trefler, D. (2013). Domestic institutions as a source of comparative advantage. *National Bureau of Economic Research (NBER) Working Paper*, 2013(18851):1–66.

O'Callaghan, J. R. (1995). NELUP: an introduction. *Journal of Environmental Planning and Management*, 38(1):5–20.

Oldekop, J. A., Holmes, G., Harris, W. E., and Evans, K. (2016). A global assessment of the social and conservation outcomes of protected areas. *Conservation Biology*, 30(1):133–141.

Ollivier, H. (2012). Growth, deforestation and the efficiency of the REDD mechanism. *Journal of Environmental Economics and Management*, 64(3):312–327.

Osafo-Kwaako, P. and Robinson, J. A. (2013). Political centralization in pre-colonial Africa. *Journal of Comparative Economics*, 41(1):6–21.

Ostrom, E. (1990). *Governing the commons: the evolution of institutions for collective action*. Cambridge University Press, Cambridge, USA.

Pagiola, S. (2008). Payments for environmental services in Costa Rica. *Ecological Economics*, 65(4):712–724.

Pagiola, S., Honey-Rosés, J., and Freire-González, J. (2016). Evaluation of the permanence of land use change induced by payments for environmental services in Quindío, Colombia. *PLOS ONE*, 11(3):e0147829.

Parry, I., Shang, B., Wingender, P., Vernon, N., and Tarun, N. (2016). Climate mitigation in China: which policies are most effective? *IMF Working Paper, International Monetary Fund*, 2016(WP/16/148):1–71.

Perman, R., Ma, Y., McGilvray, J., Common, M. S., and Maddison, D. (2011). *Natural Resource and Environmental Economics*. Pearson Education Limited, Harlow, Essex, USA, 4th edition.

Perz, S. G. (2007). Grand theory and context-specificity in the study of forest dynamics: forest transition theory and other directions. *The Professional Geographer*, 59(1):105–114.

Pfaff, A., Amacher, G. S., and Sills, E. O. (2013). Realistic REDD: improving the forest impacts of domestic policies in different settings. *Review of Environmental Economics and Policy*, 7(1):114–135.

Pfaff, A., Robalino, J., Sanchez-Azofeifa, G. A., Andam, K. S., and Ferraro, P. J. (2009). Park location affects forest protection: land characteristics cause differences in park impacts across Costa Rica. *The B.E. Journal of Economic Analysis & Policy*, 9(2):1–26.

Pineda Jaimes, N. B., Bosque Sendra, J., Gómez Delgado, M., and Franco Plata, R. (2010). Exploring the driving forces behind deforestation in the state of Mexico using geographically weighted regression. *Applied Geography*, 30(4):576–591.

Pongratz, J., Reick, C. H., Houghton, R. A., and House, J. (2013). Terminology as a key uncertainty in net land use flux estimates. *Earth System Dynamics Discussions*, 4(2):677–716.

Porras, I., Barton, D. N., Miranda, M., and Chacón-Cascante, A. (2013). *Learning from 20 years of payments for ecosystem services in Costa Rica*. International Institute for Environment and Development, London, United Kingdom.

Rahman, P. M. M., Matsui, N., and Ikemoto, Y. (2013). Poverty and food security. In *Dynamics of poverty in rural Bangladesh*. Springer, Tokyo, Japan.

Rauch, J. E. and Evans, P. B. (2000). Bureaucratic structure and bureaucratic performance in less developed countries. *Journal of Public Economics*, 75(1):49–71.

Ray, D. (2008). Development economics. In Durlauf, S. N. and Blume, L. E., editors, *The New Palgrave Dictionary of Economics*, pages 468–483. Nature Publishing Group, Basingstoke, United Kingdom, 2 edition.

Reid, W. V., Chen, D., Goldfarb, L., Hackmann, H., Lee, Y. T., Mokhele, K., Ostrom, E., Raivio, K., Rockström, J., Schellnhuber, H. J., and Whyte, A. (2010). Earth system science for global sustainability: grand challenges. *Science*, 330(6006):916–917.

Robinson, B. E., Holland, M. B., and Naughton-Treves, L. (2014). Does secure land tenure save forests? A meta-analysis of the relationship between land tenure and tropical deforestation. *Global Environmental Change*, 29(2014):281–293.

Rodríguez Zúñiga, J. (2016). *Paying for forest environmental services: the Costa Rican experience*. Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Rounsevell, M. D. A., Arneth, A., Alexander, P., Brown, D. G., de Noblet-Ducoudré, N., Ellis, E., Finnigan, J., Galvin, K., Grigg, N., and Harman, I. (2014). Towards decision-based global land use models for improved understanding of the earth system. *Earth System Dynamics*, 5(1):117–137.

Rudel, T. K., Bates, D., and Machinguiashi, R. (2002). A tropical forest transition? Agricultural change, out-migration, and secondary forests in the Ecuadorian Amazon. *Annals of the Association of American Geographers*, 92(1):87–102.

Rudel, T. K., Coomes, O. T., Moran, E., Achard, F., Angelsen, A., Xu, J., and Lambin, E. (2005). Forest transitions: towards a global understanding of land use change. *Global Environmental Change*, 15(1):23–31.

Rudel, T. K., Perez-Lugo, M., and Zichal, H. (2000). When fields revert to forest: development and spontaneous reforestation in post-war Puerto Rico. *The Professional Geographer*, 52(3):386–397.

Sayer, J., Sunderland, T., Ghazoul, J., Pfund, J.-L., Sheil, D., Meijaard, E., Venter, M., Boedhihartono, A. K., Day, M., Garcia, C., Oosten, C. v., and Buck, L. E. (2013). Ten principles for a landscape approach to reconciling agriculture, conservation, and other competing land uses. *Proceedings of the National Academy of Sciences*, 110(21):8349–8356.

Schomers, S. and Matzdorf, B. (2013). Payments for ecosystem services: a review and comparison of developing and industrialized countries. *Ecosystem Services*, 6(2013):16–30.

Seymour, F. and Angelsen, A. (2012). Summary and conclusions: REDD+ without regrets. In *Analysing REDD+: challenges and choices*. Center for International Forestry Research (CIFOR), Bogor, Indonesia.

Sierra, R. and Russman, E. (2006). On the efficiency of environmental service payments: a forest conservation assessment in the Osa Peninsula, Costa Rica. *Ecological Economics*, 59(1):131–141.

Sills, E., Myers Madeira, E., Sunderlin, W. D., and Wertz-Kanounnikoff, S. (2009). The evolving landscape of REDD+ projects. In *Angelsen A., Realising REDD+: National Strategy and Policy Options*. Center for International Forestry Research CIFOR, Bogor, Indonesia.

Skutsch, M., Zahabu, E., and Karky, B. (2009). Community forest management under REDD: policy conditions for equitable governance. In *13th World Forestry Congress: Forests in development, a vital balance. Buenos Aires, Argentina*.

Smith, P., House, J. I., Bustamante, M., Sobocká, J., Harper, R., Pan, G., West, P. C., Clark, J. M., Adhya, T., Rumpel, C., Paustian, K., Kuikman, P., Cotrufo, M. F., Elliott, J. A., McDowell, R., Griffiths, R. I., Asakawa, S., Bondeau, A., Jain, A. K., Meersmans, J., and Pugh, T. A. M. (2016). Global change pressures on soils from land use and management. *Global Change Biology*, 22(3):1008–1028.

Smith, P. M., Bustamante, M., H. Ahammad, H. Clark, H. Dong, E.A. Elsiddig, H. Haberl, R. Harper, J. House, M. Jafari, O. Masera,, and C. Mbow, N.H. Ravindranath, C.W. Rice, C. Robledo Abad, A. Romanovskaya, F. Sperling, and F. Tubiello (2014). Agriculture, forestry and other land use (AFOLU). In *Climate change 2014: mitigation of climate change. Contribution of working group III to the fifth assessment report of the Intergovernmental Panel on Climate Change*. IPCC, Cambridge University Press, Cambridge, United Kingdom.

Somanathan, E., Prabhakar, R., and Mehta, B. S. (2009). Decentralization for cost-effective conservation. *Proceedings of the National Academy of Sciences*, 106(11):4143–4147.

Southgate, D. (1990). The causes of land degradation along "spontaneously" expanding agricultural frontiers in the third world. *Land Economics*, 66(1):93–101.

Spracklen, B. D., Kalamandeen, M., Galbraith, D., Gloor, E., and Spracklen, D. V. (2015). A global analysis of deforestation in moist tropical forest protected areas. *PLOS ONE*, 10(12):e0143886.

Stern, N. (2006). *Stern review: the economics of climate change*. The National Archives (HM Treasury), London, United Kingdom.

Tacconi, L. (2016). Preventing fires and haze in Southeast Asia. *Nature Climate Change*, 6(7):640–643.

Takasaki, Y. (2007). Dynamic household models of forest clearing under distinct land and labor market institutions: can agricultural policies reduce tropical deforestation? *Environment and Development Economics*, 12(3):423–443.

UN-REDD (2015). List of UN-REDD partner countries. URL: http://www.un-redd.org/ (accessed 08/12/2015).

UNEP (2012). *The emissions gap report 2012 - a UNEP synthesis report*. United Nations Environment Programme, Nairobi, Kenya.

UNEP (2015). *The emissions gap report 2015*. United Nations Environment Programme, Nairobi, Kenya.

UNFCCC (2005). Submissions from parties (Costa Rica and Papua New Guinea). Reducing emissions from deforestation in developing countries: approaches to stimulate action. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2005/Misc.1).

UNFCCC (2007). Decision 2/CP.13. Reducing emissions from deforestation in developing countries: approaches to stimulate action. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2007/6/Add.1).

UNFCCC (2009). Decision 4/CP.15. Methodological guidance for activities relating to reducing emissions from deforestation and forest degradation and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2009/11/Add.1).

UNFCCC (2010). Decision 1/CP.16. The Cancun agreements: outcome of the work of the ad hoc working group on long-term cooperative action under the convention. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2010/7/Add.1).

UNFCCC (2011). Decision 2/CP.17. Outcome of the work of the ad hoc working group on long-term cooperative action under the convention. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2011/9/Add.1).

UNFCCC (2013). Afforestation and reforestation projects under the clean development mechanism - a reference manual. URL: http://unfccc.int/resource/docs/publications/cdm_afforestation_bro_web.pdf (accessed 21/12/2016).

UNFCCC (2013a). Decision 10/CP.19. Coordination of support for the implementation of activities in relation to mitigation actions in the forest sector by developing countries, including institutional arrangements. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2013/10/Add.1).

UNFCCC (2013b). Decision 11/CP.19. Modalities for national forest monitoring systems. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2013/10/Add.1).

UNFCCC (2015). Paris agreement. URL: http://unfccc.int/paris_agreement/items/9485.php (accessed 25/03/2017).

Veldkamp, A. and Fresco, L. (1996). CLUE: a conceptual model to study the conversion of land use and its effects. *Ecological Modelling*, 85(2-3):253–270.

Villoria, N. B., Byerlee, D., and Stevenson, J. (2014). The effects of agricultural technological progress on deforestation: what do we really know? *Applied Economic Perspectives and Policy*, 36(2):211–237.

Voinov, A., Costanza, R., Wainger, L., Boumans, R., Villa, F., Maxwell, T., and Voinov, H. (1999). Patuxent landscape model: integrated ecological economic modeling of a watershed. *Environmental Modelling & Software*, 14(5):473–491.

Wade, R. (1990). *Governing the market: economic theory and the role of government in East Asian industrialization*. Princeton University Press, USA.

Walker, R. and Smith, T. E. (1993). Tropical deforestation and forest management under the system of concession logging: a decision-theoretic analysis. *Journal of Regional Science*, 33(3):387–419.

Wang, X., Biewald, A., Dietrich, J. P., Schmitz, C., Lotze-Campen, H., Humpenöder, F., Bodirsky, B. L., and Popp, A. (2016). Taking account of governance: implications for land-use dynamics, food prices, and trade patterns. *Ecological Economics*, 122(2016):12–24.

Weber, J. G., Sills, E. O., Bauch, S., and Pattanayak, S. K. (2011). Do ICDPs work? An empirical evaluation of forest-based microenterprises in the Brazilian Amazon. *Land Economics*, 87(4):661–681.

Weber, M. (1947). *The theory of social and economic organization*. Simon and Schuster, London, United Kingdom.

Wertz-Kanounnikoff, S. and Angelsen, A. (2009). Global and national REDD+ architecture. In Angelsen, A., editor, *Realising REDD*. Center for International Forestry Research, Bogor, Indonesia.

Williamson, O. E. (1975). *Markets and hierarchies, analysis and antitrust implications*. Free Press, New York, USA.

Wolfersberger, J., Delacote, P., and Garcia, S. (2015). An empirical analysis of forest transition and land-use change in developing countries. *Ecological Economics*, 119(2015):241–251.

World Bank (2013a). Agriculture, value added (% of GDP). The World Development Indicators. URL: http://data.worldbank.org/indicator/NV.AGR.TOTL.ZS (accessed (13/04/2016).

World Bank (2013b). GNI per capita. The World Development Indicators. URL: http://data.worldbank.org/indicator/NV.IND.MANF.Z (accessed 13/04/2016).

World Bank (2014). Rule of law. The Worldwide Governance Indicators. URL: http://info.worldbank.org/governance/wgi/index.aspx#home (accessed 13/04/2016).

World Bank (2015). GDP growth (annual %). The World Development Indicators. URL: http://data.worldbank.org/indicator/NY.GDP.MKTP.KD.ZG (accessed 20/12/2016).

World Bank (2016). The cost of fire an economic analysis of Indonesia's 2015 fire crisis. URL: http://pubdocs.worldbank.org/en/643781465442350600/Indonesia-forest-fire-notes.pdf (accessed 12/12/2016).

Wright, G. D., Andersson, K. P., Gibson, C. C., and Evans, T. P. (2016). Decentralization can help reduce deforestation when user groups engage with local government. *Proceedings of the National Academy of Sciences*, 113(52):14958–14963.

Part I

Political institutions in deforestation processes

Chapter 2

Governance and deforestation - a meta-analysis [1]

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Governance and deforestation - a meta-analysis

Abstract

Understanding which aspects of forest governance have the potential to effectively reduce deforestation is central to reversing trends in global deforestation and biodiversity loss. There is a multitude of empirical studies examining this relationship using various measures of governance and study designs, coming to diverse conclusions. In order to identify the source of variation across studies, this article conducts a meta-analysis of 32 empirical cross-country studies in the field of economics, containing 227 estimates of the impact of different governance measures on deforestation. Using an ordered probit model, we find that the choice of the governance measure is the main factor in explaining variations in the outcome of the studies. In particular, studies using environmental policy, ownership rights, presence of environmental NGOs, and rule of law as measures of governance, are more likely to find that better governance reduces deforestation. In contrast, studies using democracy and rights as a measure of governance are more likely to find that deforestation increases, when governance is improved. The finding that not all aspects of governance improvements are equally supportive of forest conservation suggests that more nuanced analyses of specific aspects of environmental governance are required to guide evidence-based policy making.

JEL classification: Q23, Q28, Q56, Q58

Keywords: Deforestation, governance, meta-analysis, institutions

1 Introduction

In an effort to reduce emissions from deforestation, growing attention is being paid to the role of political institutions in national forest conservation strategies: out of 70 countries planning to roll out national REDD+ (Reducing Emissions from Deforestation and Forest Degradation) programs, 54 explicitly state in their national REDD+ documents that governance issues are a concern for forest conservation, or that they want to address such issues in order to reduce deforestation (UN-REDD, 2016; FCPF, 2016).

There is a rapidly growing empirical literature examining deforestation drivers (Busch and Ferretti-Gallon, 2017; Choumert et al., 2013) and the effect of the quality of governance on deforestation (Deacon, 1994; Bhattarai and Hammig, 2001; Arvin and Lew, 2011; Ehrhardt-Martinez et al., 2002). Studies in the latter strand of the literature come to fundamentally diverging conclusions on the central question whether better governance leads to a reduction in deforestation, hereafter referred to as the governance hypothesis. While a number of studies support the hypothesis, others yield inconclusive results, or reject it.

Taking stock of the literature is hampered by substantial heterogeneity in terms of study design. Most notably, a broad spectrum of governance measures is used to operationalize the quality of governance. These various governance measures can reflect very different components of political institutions. Learning from previous studies is further complicated by significant variations in the methodology employed. Indeed, studies that use the very same governance measure in some cases still come to contradicting conclusions, which suggests that other study design choices, such as the estimation technique, also influence the results. For example, while Li and Reuveny (2006) or Buitenzorgy and Mol (2011) find that more democracy is likely to reduce deforestation, Midlarsky (1998), Marquart-Pyatt (2004), and Ehrhardt-Martinez (1998) find that it can actually increase deforestation.

In this study we conduct a meta-analysis of the literature in the field of economics to provide a systematic analysis of the relationship between deforestation and the quality of governance. The analysis is based on a sample of 227 estimates originating from 32 studies conducted between 1994 and 2016. We classify the estimates by direction and statistical significance and use an ordered probit to draw systematic comparisons across studies. The analysis provides a quantitative insight into which factors explain the variation in the multitude of study outcomes. In particular, we seek to identify which aspects (general or environmental) and levels (decision process, rules or enforcement) of governance tend to have a robust deforestation-reducing effect across different study designs. Theory guides us in hypothesizing that studies using an environmental governance measure (e.g. environmental expenditures) are more likely to yield results that are supportive of the governance hypothesis than those using a general governance measure (such as liberal democratic institutions). Furthermore, emphasis is placed on the role of control variables, model specification and estimation, as well as the spatial context and study period.

In contrast to a literature review, a meta-analysis is a systematic analysis of empirical research using objective criteria for the selection of literature and statistical tools for the identification of systematic patterns across studies that can be reproduced (Waldorf and Byun, 2005; Stanley and Doucouliagos, 2012). Existing meta-studies in the field allow conclusions to be drawn on the current scientific consensus on (i) deforestation and land tenure rights security (better land tenure rights are likely to reduce deforestation (Robinson et al., 2014)), (ii) deforestation and income (more recent publications find less evidence for the hypothesis that higher income countries are likely to experience less deforestation (Choumert et al., 2013)) or (iii) forest restoration (a meta-analysis reveals that forest restoration bears the potential to significantly enhance biodiversity (Crouzeilles et al., 2016)). Furthermore, Busch and Ferretti-Gallon (2017), Angelsen and Kaimowitz (1999), as well as Geist and Lambin (2002) provide reviews of the literature on drivers of deforestation, without identifying and quantifying systematic patterns. There is, to the best of our knowledge to date, no meta-analysis examining the relationship between governance and deforestation.

The remainder of this paper is organized as follows. We will first present the conceptual framework for the analysis in Section 2. In Section 3, we present the data selection strategy, describe the main moderator variables and provide descriptive statistics for our analysis. In Section 4 we present the meta-analytical model. In Section 5 we report the results and Section 6 concludes with a broader discussion.

2 Conceptual framework

Guided by the conceptual literature we present a simple framework in this section that decomposes two basic dimensions of governance, which lead to two more refined versions of the general governance hypothesis and indicate possible different underlying mechanisms (Ferraro and Hanauer, 2014; Meyfroidt, 2016).

As a vertical dimension, we distinguish between different levels of governance. We build upon the forest governance framework proposed by the FAO and PROFOR (2011) and differentiate between the three levels: (i) decision making processes, (ii) rules and policies, and (iii) enforcement. These three levels can be conceived as following a (vertical) ordering, because decision making processes (processes that are required to change the status quo (Tsebelis, 1995)), produce rules, (*de jure* dimension of governance (Kaufmann et al., 2007)) that are subsequently enforced (*de facto* dimension of governance (Kaufmann et al., 2007)). Of course, weaknesses in the enforcement of existing rules can also trigger new decision making processes. However, we can assume that no new rules are going to be enforced without having been subject to decision making processes beforehand. We hypothesize that the use of governance measures at different levels can partly explain the inconclusive findings on the governance hypothesis across studies. We refer to this refinement as the vertical dimensions of the governance hypothesis. If good governance at a specific level was more likely to reduce deforestation, this would also allow more focused policy advice on reforming political institutions.

As a horizontal dimension, we follow Ceddia et al. (2014) and differentiate general from specific environmental measures of governance. Ceddia et al. (2014) argue that better general governance (e.g. liberal democratic institutions) is likely to increase the demand for agricultural land and thus implicitly leads to more deforestation. Such a Jevons effect (Jevons, 1866) occurs, when efficiency increases (here induced by governance improvements) for an input factor to production (here agricultural land) allow to increase output levels of production. On the other hand, specific measures of the quality of environmental governance (e.g. environmental expenditures) are predicted to reduce demand for land as an input to agricultural production. This is referred to as a landsparing Borlaug effect. A Borlaug effect (Borlaug, 2007) occurs when efficiency improvements lead to a reduced use of natural resource based input factors to production. Along these lines, we hypothesize that studies using an environmental governance (general governance) measure are more (less) likely to yield results that are supportive of the governance hypothesis. We refer to this refinement as the horizontal dimensions of the governance hypothesis.

Figure 1 illustrates the two dimensions of governance described above that will guide the subsequent analysis. It depicts the vertical and horizontal refinement of the general governance hypothesis. For illustrative purposes, we display the six governance variables that we identify in the meta-study sample as the most frequently used governance variables (details follow in Section 2).

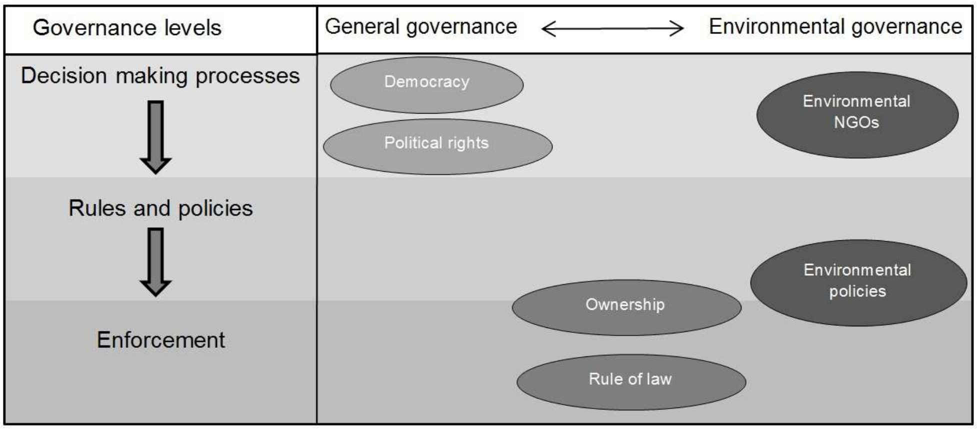

Figure 1: Two-dimensional framework of governance

3 Data

3.1 Data selection

We follow the data selection standards for meta-studies proposed by Stanley et al. (2013). We impose four study selection criteria. First, we decide to restrict the scope to peer-reviewed articles and academic working papers and exclude all other types of publications. Second, we restrict our selection to the disciplines of economics, econometrics, and finance (see Appendix A for details). Third, we only include studies that cover at least three different countries, in order to make sure that results are not simply valid for a very specific (e.g. national) context. Finally, we restrict the search to papers written in English.

To build an initial study sample, we draw upon the review paper by Angelsen and Kaimowitz (1999) and the meta-analysis by Choumert et al. (2013)[1]. 77 of the articles in Choumert et al.'s meta-analysis and 27 articles from Angelsen and Kaimowitz's review are considered relevant according to

[1]Both publications have a disciplinary background in economics. The review paper has a focus on economic models and the article is published in an economic journal.

our criteria and selected for the analysis based on their title. As a next step, we develop a keyword combination for a systematic search for publications. We test different types of boolean connectors and eventually use the keyword combination "deforestation AND (governance OR institutions) AND (regression OR empirical)" for our search. We use a simplified version of the keyword combination if the actual combination is technically not possible in a search portal (see Appendix A for the precise combination and search criteria for each platform, given its technical restrictions). We then conduct a keyword search using ScienceDirect, Wiley, JSTOR, Springer, SAGE, and Taylor and Francis for peer reviewed publications, and EconPapers (REPEC) and SSRN for working papers. We extract the results corresponding to our search criteria for EconPapers, ScienceDirect, Wiley, JSTOR, Springer, and SSRN on the 21st of February 2016 and for SAGE Publications and Taylor and Francis on the 29th of February 2016.

In total, across all different search portals, the keyword search yields 1,740 results. Adding the 104 publications from the initial study population, we have a total of 1,844 publications.

We then assess whether the 1,844 articles correspond to the purpose of our meta-analysis based on their abstracts. We exclude (i) 392 publications that use non-empirical methods, (ii) 713 with local or national scope (we are only interested in cross-country assessments), (iii) 29 duplicates, (iv) 670 studies that are concerned with a different topic, (v) 5 non-English studies, and (vi) 3 studies with contradictory results. The final sample comprises 32 studies.

These 32 studies contain a total of 277 estimations of the impact of governance on deforestation. We cannot take estimates into account that include various governance variables indicating different directions of causality (and thus a contradictory link). We therefore have to exclude 50 observations. The sample is thus further constrained to 227 observations for our analysis. While the key word and criteria driven sample selection ensures replicability, the procedure also implies that we meta-analyze only a well-defined sample of the literature and not the entire population of the literature on the topic of governance and deforestation.

Appendix A indicates how many studies are attributable to the respective search portals (in the first search and in the final sample). All coding steps are verified by three independent coders. Spot checks were undertaken by two further coders.[2] The codebook (coding instructions) and the coded information (coding form and data) are available upon request.

3.2 Descriptive statistics and exploratory analysis

3.2.1 General information on the sample

Table 1 summarizes the 32 studies included in the analysis and the respective number of estimates. We include only empirical studies that examine the effect of governance on deforestation across countries. All studies were published between 1994 and 2015. The total study population contains three working papers, corresponding to 60 estimates, and 29 articles published in academic journals, corresponding to 167 estimates. The average impact factor (SCImago Journal Rank) of the sample journal articles is 1.33 (min. 0.31;

[2] All coding steps were undertaken by Sebastian Lübbers, Annika Marxen, and Johanna Wehkamp. Spot checks were undertaken by Nicolas Koch and Sabine Fuss.

max. 8.05). The average number of citations (Google Scholar) of all sample studies amounts to almost 59 (min. 1; max. 380).

Table 1: Studies included in the analysis

Author	Year	Number of estimates
Arcand J.L., Guillaumont P., Jeanneney Guillaumont S.	(2008)	12
Arvin B.M., Lew B.	(2011)	12
Barbier E., Damania R., Léonard D.	(2005)	4
Bhattarai M., Hammig M.	(2001)	3
Bohn H, Deacon R.T.	(2000)	1
Buitenzorgy M., Mol A P.J.	(2011)	4
Culas R. J.	(2007)	3
Damette O., Delacote P.	(2012)	15
Damette O., Delacote P.	(2011)	14
Deacon R.T.	(1994)	5
Deacon R.T.	(1999)	4
Didia D.O.	(1997)	2
Ehrhardt-Martinez K.	(1998)	3
Ehrhardt-Martinez K., Crenshaw E.M., Jenkins J.C.	(2002)	4
Ferreira S.	(2004)	5
Ferreira S., Vincent J.R.	(2010)	3
Galinato G., Galinato S.	(2013)	7
Jorgenson A. K.	(2008)	3
Kishor N., Belle A.	(2004)	6
Kuusela O.P, Amacher G.S.	(2016)	3
Li Q., Reuveny R.	(2006)	9
Mainardi S.	(1998)	1
Marchand S.	(2011)	25
Marchand S., Diarra G.	(2011)	6
Marquart-Pyatt S.	(2004)	5
Nguyen V. P., Azomahou T.	(2007)	1
Novoa D. C.	(2008)	22
Rock M.T.	(1996)	2
Shandra J.M.	(2007a)	14
Shandra J.M.	(2007b)	15
Tole L.	(2004)	9
Wolfersberger J., Delacote P., Garcia S.	(2015)	5

3.2.2 Definition of the effect categories

The empirical studies in our sample use very different econometric modeling strategies and variables, making it extremely difficult to define a common, scale-free metric of estimated effect sizes for the impact of governance on deforestation. This broad variation allows us to only meta-analyze the direction and statistical significance of the effects rather than their magnitude. By using a simple classification of sign and significance (rather than pooling inconsistent actual effect sizes), we can exploit the full sample of study results and still focus on the main issue of whether a link between governance and deforestation exists. We interpret the results with respect to the widespread hypothesis (Geist and Lambin, 2002; Mendelsohn, 1994) that better governance leads to a decrease in deforestation. Following this logic, the reported results are categorized as 'positive' (supportive of the governance hypothesis) if an increase in the quality of governance leads to a statistically significant decrease in deforestation, 'inconclusive' if no statistically significant relationship can be established, and 'negative' (non-supportive of the governance hypothesis) if an increase in institutional quality leads to a statistically significant decrease in forest cover. Better governance is defined differently from one study to the other. The respective governance variables can use ordinal (e.g democracy), or cardinal (e.g. amount of environmental NGOs in the country) units. In addition, they can be scaled differently: a high score can correspond to better governance in one study, while it corresponds to worse governance in the other. The three effect categories allow to consistently classify effects across studies, in spite of study specific differences.

We use a 10% critical significance level for the categorization.[3] Out of the 227 observations, 22 are negative, finding evidence against the governance hypothesis, 82 do not find statistically significant results and 123 support the governance hypothesis (see Table 2).

3.2.3 Moderator variables

Moderator variables are used to capture different features and specifications of the estimates that influence the outcome. We group the moderator variables into five moderator variable categories: (i) governance measures, (ii) control variables, (iii) spatial, (iv) temporal, and (v) econometric specifications.[4] Including the moderator variables allows us to investigate the extent to which different governance or control variables as well as different spatial, temporal and econometric specifications, systematically influence the direction and significance of the observed effect.

Table 2 lists each moderator variable, its abbreviation, definition, mean, standard deviation, as well as minimum and maximum values.

We subsequently provide a short description and explanatory analysis of each category of moderator variable. We use a two-proportion z-test in order to understand whether the proportion of studies finding positive, negative, or inconclusive results is significantly different when a particular moderator variable is used. Table 4 displays the results of the test for each category.

Governance variables: We include the following six governance measures in the moderator variable category 'governance': (i) *environmental_policy*, (ii) *ownership*, (iii) *democracy*, (iv) environmental *NGOs*, (v) political *rights*, and (vi) *rule_of_law*. We select these six governance measures from a set of 13 governance measures initially identified in the sample with very minor forms of aggregation (listed in table 3).

More specifically, we cluster environmental policy, international environmental policy, and environmental compliance (which is a composite indicator constructed with the variables corporate ethics, burden of government regulation, and stringency of environmental regulation) into the category *environmental_policy*. The variable *environmental_policy* can be defined as measuring in how far the national policy framework internalizes environmental concerns. Due to the fact that international environmental policy measures have to be endorsed by national governments, they can be considered as being part of domestic environmental policies. Furthermore, given the way the variable environmental compliance is constructed, it indirectly also measures environmental policy.

[3]We also experiment with a 5% critical significance level. However, this lower threshold reduces the sample size (particularly, of the category 'negative') to an extent that prevents us from conducting a meaningful meta-regression analysis.

[4]We also experiment with the type of forest data that is used in the sample studies. However, this potential additional moderator does not seem to explain the variation in study outcomes.

Table 2: Definition of variables and summary measures

Dependent variable	*Effect category*	*Frequency*	*Percentage*		
	Positive	123	54.19		
	Insignificant	82	36.12		
	Negative	22	9.69		
Moderator variables	*Definition*	*Mean*	*Std. Dev.*	*Min*	*Max*
environmental_policy	Environmental policy	0.11	0.31	0	1
ownership	Ownership and land tenure rights	0.11	0.31	0	1
NGOs	Presence of environmental NGOs	0.09	0.29	0	1
democracy	Democracy	0.11	0.32	0	1
rights	Political rights	0.22	0.41	0	1
rule_of_law	Rule of law and enforcement	0.18	0.38	0	1
population	Population density	0.85	0.35	0	1
income	Income	0.70	0.46	0	1
area	Forest area	0.53	0.50	0	1
timber	Timber	0.36	0.48	0	1
agriculture	Agriculture	0.21	0.41	0	1
developing_countries	Non-high income countries	0.48	0.50	0	1
start	Start year of the analysis	1981.73	11.07	1960	2005
end	End year of the analysis	1998.87	6.00	1985	2010
panel	Panel data	0.45	0.50	0	1
dynamic	Dynamic effects	0.22	0.42	0	1
nonlinear	Nonlinear specifications (squared variables, interaction terms)	0.26	0.44	0	1
OLS	Ordinary Least Squares vs. more complex estimators	0.56	0.50	0	1
date	Publication date	2007.60	4.39	1994	2016
size	Sample size	439.39	669.51	20	3441
type	Type of publication	0.74	0.44	0	1

The different studies use data from Dietz and Kalof (1992), data from the WTO's government finance statistics (2016), the Center for International Earth Science Information Network (CIESIN, 1990), the World Resources Institute's guide to the global environment (1998), the Global Competitiveness Report (Schwab and Porter, 2009), or the World Economic Forum's (2012) indicator on corporate ethics, and on the stringency of environmental regulations.

The moderator variable *ownership* measures the probability of expropriation and the protection of property rights by a political system. The underlying studies use Banks (2008) International Country Risk Guide (2015), Fraser Institute (2016) data, or data from Acemoglu et al. (2001).

The variable *democracy* measures the extent to which a country has liberal democratic institutions (separation of powers) and processes. The respective studies use data from Marshall and Jaggers 2002 polity index (2002), Bollen's political regime index (1993), or from Banks' (2008).

The moderator variable *NGOs* measures the presence of environmental NGOs in a country. The respective data come from the yearbook of international associations as compiled by Smith and Wiest (2005), Smith (2015), the Center for International Earth Science Information Network (Shandra, 2007b), or Dietz and Kalof (1992).

The moderator variable *rights* represents political rights and civil liberties. Studies use data provided either by Freedom House (2009), or by Kaufmann et al. (1999). The variables grouped in this category have different subcomponents, but all focus on the quality of the political process. Subcomponents are for instance the electoral process, political pluralism, the right to associate, as well as the protection of individual rights.

Table 3: Different governance measures initially identified in the sample

Governance variables	Number of estimates
Environmental policy	9
Rule of law	36
Quality of the administration	5
Political rights	49
Corruption	18
Democracy	26
Enforcement	4
NGOs	21
International environmental policy	10
Ownership	25
Inequality	14
Stability	4
Environmental compliance	6

We group the categories rule of law and enforcement into the category *rule_of_ law*, because they are used interchangeably in the literature. The variable can be defined according to the World Bank (2014) definition, as capturing "*perceptions of the extent to which agents have confidence in and abide by the rules of a society*". The corresponding World Bank dataset (World Governance Indicators) has been used for the majority of estimates (22 out of 36). Other sources are the International Country Risk Guide (2015) and the Fraser Institute (2016).

There are less than 20 observations in the initially identified categories of quality of the administration, corruption, inequality, and stability (a common threshold in meta-analysis). These categories are therefore not included as distinctive moderator variables.

Table 4 provides us with a first insight into whether the proportion of findings changes significantly when different governance variables are used in the analysis.[5] Indeed, the bivariate analyses suggest that the choice of the governance variable can have a meaningful impact on study outcomes as reflected by significant variations in the proportions of positive, negative, and inconclusive results. While the effect on the study outcome of using *environmental_policy* and *ownership* as governance measures is either weak or non-existent, the use of the other governance variables seems to make a significant difference to the respective outcome proportions. In particular, studies that use *democracy* and *rights* show a significantly lower proportion of positive estimates, while the proportion of negative estimates is significantly higher. The variables *NGOs* and *rule_of_law* have the contrary effect; for both these variables we can observe a significant increase in the proportion of positive findings and a significant decrease in the proportion of negative findings.

Control variables: Five control variables are selected as additional moderator variables: (i) *population* (including population density and population growth rate), (ii) *income* (including different measures of income such as GDP, GNI, GNP etc.), (iii) *area* (including different measures of the forest area), (iv) *timber* (including different measures of timber harvest), and (v) *agriculture* (including agricultural yields, agricultural production, agricultural exports

[5] Baseline proportions: positive 54.19% (123 obs.); insignificant 36.12% (82 obs.); negative 9.69% (22 obs.); significance level |1.64|.

Table 4: Binary moderators and effect types

		N	*Proportion of estimates*		
			Negative	Inconclusive	Positive
Governance variables					
environmental_policy	yes	25	0	0.36	0.64
	no	202	0.11	0.36	0.53
	z-value		1.74*	0.01	-1.04
ownership	yes	25	0.04	0.36	0.6
	no	202	0.1	0.36	0.53
	z-value		1.02	0.01	-0.62
NGOs	yes	21	0	0.05	0.95
	no	206	0.11	0.39	0.5
	z-value		1.58	3.14*	-3.96*
democracy	yes	26	0.23	0.42	0.35
	no	201	0.08	0.35	0.57
	z-value		-2.45*	-0.7	2.13*
rights	yes	49	0.2	0.49	0.31
	no	178	0.07	0.33	0.61
	z-value		-2.86*	-2.12*	3.74*
rule_of_law	yes	40	0	0.3	0.7
	no	187	0.12	0.37	0.51
	z-value		2.28*	0.89	-2.21*
Control variables					
population	yes	194	0.1	0.36	0.54
	no	33	0.06	0.36	0.58
	z-value		-0.76	0.03	0.42
income	yes	158	0.09	0.37	0.54
	no	69	0.12	0.35	0.54
	z-value		0.64	-0.28	-0.11
area	yes	120	0.13	0.33	0.55
	no	107	0.07	0.4	0.53
	z-value		-1.51	1.2	-0.26
timber	yes	82	0.06	0.54	0.4
	no	145	0.12	0.26	0.62
	z-value		1.38	-4.14*	3.17*
agriculture	yes	47	0.02	0.32	0.66
	no	180	0.12	0.37	0.51
	z-value		1.97*	0.67	-1.82*
Spatial and econometric variables					
developing_countries	yes	108	0.12	0.4	0.48
	no	119	0.08	0.33	0.6
	z-value		-1.14	-1.1	1.74*
panel	yes	103	0.08	0.52	0.4
	no	124	0.11	0.23	0.66
	z-value		0.89	-4.66*	3.96*
dynamic	yes	51	0.04	0.59	0.37
	no	176	0.11	0.3	0.59
	z-value		1.58	-3.83*	2.76*
nonlinear	yes	58	0.12	0.4	0.48
	no	169	0.09	0.35	0.56
	z-value		-0.71	-0.65	1.05
OLS	yes	128	0.12	0.26	0.63
	no	99	0.07	0.49	0.43
	z-value		-1.17	3.69*	-2.86*
type	yes	167	0.13	0.35	0.52
	no	60	0.02	0.38	0.6
	z-value		-2.45*	0.42	1.05

etc.). The respective control variables are selected as moderator variables, because they are most frequently used in the regressions of our study population. There are intensive discussions in the literature, especially related to the variables: (i) *population* (whether more population density and growth increases (Mather and Needle, 2000; DeFries et al., 2010) or decreases (Pfaff, 1999) deforestation) and (ii) *income* (whether income growth leads to less or more deforestation (Choumert et al., 2013)). Moreover, existing reviews find that *agriculture* is the main direct driver of deforestation worldwide (Angelsen and Kaimowitz, 1999; Rudel et al., 2009).

Table 4 shows that two variables seem to significantly impact the outcome proportions. First of all, including the variable *timber* significantly reduces the proportion of positive outcomes and increases the proportion of inconclusive ones. Second, including the variable *agriculture* significantly increases the proportion of positive and reduces the proportion of negative outcomes. The other control moderator variables (*population*, *income*, and *area*) do not have a significant impact on the outcome proportions.

Spatial variable: The only spatial moderator variable that is consistently identified across studies is the variable *developing_countries*. It allows to account for whether a study includes high income countries (according to the World Bank (2016) income classification of the year of publication) or not. All studies that exclude high income countries are categorized as developing countries studies. Studies that do not list the countries included in their sample, but state that only developing countries have been used for the analysis, are also included in the category *developing_countries*. Table 4 shows that studies that exclusively rely on *developing_countries* have a statistically significant lower proportion of positive findings.

Temporal variables: To account for the potential influence of the temporal context, we include the variable *period* into the regression model, which records the average year (rather than both the start and end year) of the sample period. Given the differences in length of the study periods, we also include the square root of the sample size through the variable *size* and use this variable for robustness tests. Figure 2 shows the median of the start and end years of the time periods used for the estimations in the sample. It reveals that positive results are on average associated with later periods (1990 - 2000) than negative results (1980 - 1995). Studies with inconclusive results are on average based on longer time periods.

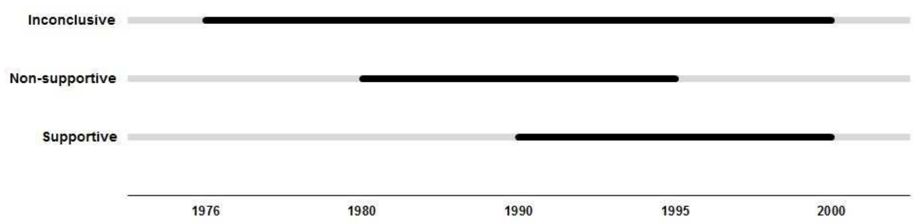

Figure 2: Median start and end years of studies in sample

Econometric variables: The first moderator variable in this category relates to data characteristics. The dummy variable *panel* distinguishes panel data settings from cross-sectional settings. Given that the fixed effects of panel data models can control for time-invariant unobservable differences between countries or regions, the research design of studies based on panel data is considered to be more robust than that of cross-sectional studies. Table 4 shows that using panel data significantly lowers the proportion of positive effects and increases the occurrence of insignificant findings. Furthermore, the dummy variables *dynamic* and *nonlinear* code studies that build on dynamic regression specifications (e.g. lags in levels and/or changes) and nonlinear specifications (e.g. quadratic and/or interaction terms) of the relationship between deforestation and governance, respectively. While the bivariate test suggests that particularly dynamic models are more likely to yield insignificant effects and less likely to yield positive effects, there are no significant differences in the distribution of the effect category when nonlinearities are accounted for. Finally, 56% of the estimates are based on OLS estimation.[6] The results for the corresponding bivariate *OLS* variable indicates that an OLS estimation is more likely to produce positive effects and less likely to produce insignificant effects, compared to more sophisticated estimation techniques. This observation is in line with expectations given the typically low standard errors of OLS.

Publication bias variable: A publication bias could be present in the analysis if there was a tendency to publish a certain type of outcome in peer-reviewed journals. Several methods are available to detect and correct for potential publication biases in settings with continuous effect size estimates (e.g. Hedges, 1992; Stanley, 2008). However, these techniques cannot be applied in our categorical model design. We therefore seek to mitigate potential biases by controlling for the type of publication. More specifically, the dummy variable *type* distinguishes published journal articles from working papers. The exploratory bivariate analysis in Table 4 suggests that peer reviewed articles may have a significantly larger proportion of negative findings. In the subsequent analyses, we also include the publication *year* to avoid any biases from time trends in the published literature.

4 Model

The preceding exploratory analysis points to a number of moderator variables that may significantly contribute to the observed variations in the findings across our study universe. However, the bivariate z-test neglects the potential simultaneous influence of various other variables and thus, may suggest relationships that do not exist. Therefore, we turn to a multivariate meta-analysis to properly identify the salient sources of variations in terms of direction and significance. Given that the three effect categories have a natural ordering by the t-statistic, an ordered probit model is appropriate for our modeling purposes. This is also the approach taken in Waldorf and Byun (2005) and Card et al. (2010).

We denote the three observed effect categories as y, taking the values 0, 1, or 2 when the estimated effect is negative, inconclusive, or positive, respectively. The observation y is assumed to be related to the latent continuous variable y^*, which denotes the exact but unobserved estimated effect size derived in each analysis. y^* is assumed to be a linear combination of some measurable moderator variables x, i.e. $y^* = x\beta + \varepsilon$, with ε being the disturbance term that has a standard normal distribution. More specifically, the link between y and y^* can be defined as follows (Greene, 2012)

[6]Other estimators used include for instance GMM, logit, probit, fixed, and random effect specifications

$$\begin{array}{ll} y = 0 & if\ y^* \leq 0 \\ y = 1 & if\ 0 < y^* \quad \leq \mu_1 \\ y = 2 & if\ \mu_2 \leq y^* \end{array} \tag{1}$$

Having established the link between y and y^*, we are also interested in estimating how a change in a moderator x translates into the probability of observing a particular effect category j. It is given by

$$P\left(y = j\right) = \Phi\left(\mu_j - x\beta\right) - \Phi\left(\mu_{j-1} - x\beta\right)\ for\ j = 0, 1, 2 \tag{2}$$

where Φ is the standard normal distribution function, and $\Phi\left(\mu_0 - x\beta\right) \equiv 0$ and

$\Phi\left(\mu_2 - x\beta\right) \equiv 1$.

The coefficients β and the thresholds μ in the ordered probit model can be estimated straightforwardly by maximum likelihood. Note that a positive β coefficient suggests a positive effect of the moderator variable x on the probability of finding a positive result (i.e. $P\left(y = 2\right)$) and a negative effect on the probability of a negative result (i.e. $P\left(y = 0\right)$). Even though the coefficient estimates can be interpreted based on their significance and signs, they do not reveal the magnitude of changes in the probability of observing each effect category in response to changes in x (precisely because the model is nonlinear). To this end, we present the corresponding marginal effects. We use the pseudo-R^2 of McFadden (1974) as a measure of fit of the model.

Two data characteristics of our sample require particular attention and we tailor our estimation techniques to these features (see Nelson and Kennedy (2009) for an excellent discussion of the often neglected specificities of the data-generating process in meta-regression analyses). First, we have multiple effect estimates from the same primary study (see Table 1), i.e. there is potential within-study correlation. To deal with this common problem in the meta-regression literature, we use robust standard errors that are clustered by study (Nelson and Kennedy, 2009). Second, the sample sizes clearly vary across the effect estimates in our sample (see Table 2). This raises the concern that there may be a 'mechanical' effect of the sample size on the distribution of t-statistics (and thereby on our dependent variable), i.e. bigger study samples are more likely to deliver significant t-statistics (Card et al., 2010). Our strategy to account for this concern is two-fold. On the one hand, we present estimation results that are based on weighted observations. More specifically, we use (the logarithm of the square root of) the sample size as a weight, which is a common method to correct for robustness differences in the literature (Waldorf and Byun, 2005; Brons et al., 2008; Nelson and Kennedy, 2009).[7] On the other hand, we conduct further specification tests to evaluate the validity of our analysis, which in fact will show that the mechanical effect of the sample size seems to be mitigated by other design factors.

5 Results

Table 5 reports the estimation results of the ordered probit model. The first numerical column shows the coefficient estimates. The last three columns

[7] Ideally, we would use the inverse of the variance as weights. However, the variances are only reported in a small sub-sample of our primary studies.

show the marginal effects for each effect category (negative, inconclusive, and positive).

In a nutshell, three main findings emerge. First, none of the different governance levels (processes, rules, and enforcement) homogenously increases the likelihood of a certain study outcome. Second, studies using specific environmental dimensions of governance are more likely to yield results that are supportive of the governance hypothesis. Third, the choice of control variables as well as the estimation technique can significantly influence the study outcome. We subsequently present and discuss these results in more detail.

Table 5: Results ordered probit model

Moderator variable	Coefficient	Marginal effects		
		Negative	*Inconclusive*	*Positive*
Governance variables				
environmental_policy	1.153*	-0.042**	-0.333**	0.375**
	(0.657)			
ownership	1.264**	-0.044**	-0.355***	0.399***
	(0.494)			
NGOs	2.320***	-0.053**	-0.468***	0.522***
	(0.517)			
democracy	-1.011**	0.149	0.226***	-0.375**
	(0.510)			
rights	-1.518***	0.246*	0.286***	-0.532***
	(0.550)			
rule_of_law	0.987*	-0.045*	-0.3**	0.345**
	(0.519)			
Control variables				
population	0.943*	-0.128	-0.227***	0.356**
	(0.490)			
income	0.0904	-0.007	-0.029	0.036
	(0.517)			
area	-1.362***	0.112*	0.386***	-0.498***
	(0.496)			
timber	-0.169	0.013	0.054	-0.067
	(0.330)			
agriculture	0.132	-0.009	-0.043	0.052
	(0.397)			
Spatial, temporal and econometric variables				
developing_countries	-0.458	0.036	0.144	-0.18
	(0.386)			
period	0.0534	-0.004	-0.017	0.021
	(0.0467)			
panel	-0.228	0.017	0.073	-0.09
	(0.878)			
dynamic	-0.491	0.047	0.146	-0.194
	(0.707)			
nonlinear	-0.436	0.04	0.132	-0.172
	(0.354)			
OLS	-0.947**	0.069	0.289***	-0.358***
	(0.390)			
date	-0.0736	0.006	0.024	-0.029
	(0.0553)			
type	0.580	-0.057	-0.171	0.228
	(0.640)			
size	0.0148	-0.001	-0.005	0.006
	(0.0210)			
N	227			
Pseudo R2	0.2522			

Standard errors (clustered by study) in parentheses.
Estimated cutpoints are not reported. * p<0.10, **p<0.05, *** p<0.01

5.1 Influence of the choice of the governance measure

The results in Table 5 show that the choice of the governance measure is the primary source of variation in the sign and significance of effect estimates. This holds true both for the significance of the coefficients and the magnitude of marginal effects. While the variables *environmental_policy*, *ownership*, *NGOs*, and *rule_of_law* increase the likelihood of positive findings, using the variables *democracy* and *rights* has the contrary effect.

5.1.1 Decision making processes

The variables *NGOs* and *democracy* measure the quality of decision making processes. The meta-analysis shows that studies are more likely to find that better governance increases deforestation, when the general governance measure *democracy* or *rights* are used. In contrast, we find that studies using the presence of environmental *NGOs* as an specific environmental governance variable are more likely to find that deforestation decreases.

More specifically, when governance is operationalized by *democracy*, the likelihood of a positive outcome is reduced by 38% (at a 5% significance level). Furthermore, the marginal effects suggest that using this variable increases the probability of inconclusive findings by 23% (at a 1% significance level). Similarly, using the variable *rights* reduces the likelihood of a supportive outcome. The strong marginal effects of the *rights* variable are particularly noteworthy. They suggest that the probability for positive findings is reduced by 53%. At the same time, using rights as a governance measure increases the likelihood of finding an insignificant or negative result by 29% or 25%, respectively. In contrast, the marginal effects for the governance measure *NGOs* indicate that using this variable increases the likelihood of positive results by 52%, while the probability of finding inconclusive (negative) results decreases by 47% (5%).

The presented results reflect both the role of the openness of the political process to civil society participation and the specific role of environmental awareness. The example of Brazil illustrates the channel through which civil society lobbying for environmental policy can turn forest conservation into a political priority: set under pressure by civil society organizations, a range of large-scale agricultural corporations agreed to the soy moratorium (Gibbs et al., 2015). This situation fostered further political action to reduce deforestation (Nepstad et al., 2014). Our analysis indicates that using more disaggregated data and conducting a more refined analysis on a specific aspect of a policy process, the role of the environmental civil society, rather than the degree of democracy in general, generates drastically different and more nuanced conclusions. This also allows more practical insights into which policies allow the systematic reduction of deforestation.

In particular, the results on the governance measure *democracy* and *rights* suggest that deliberative political processes do not necessarily lead to environmentally sustainable outcomes. If *democracy* in fact measures, whether a country has democratic decision making processes, then the results show that more opportunities for democratic participation are not necessarily used to express environmental concerns. This reflects the contradiction in the literature on the role of democracy for forest conservation; while Didia (1997), as well as Li and Reuveny (2006) find a positive impact of more democracy on forest covers, Midlarski (1998), Ehrhardt-Martinez (1998) and Marquart-Pyatt (2004) find that more democracy leads to more deforestation. Buitenzorgy and Mol (2011) reconcile the opposing views by describing an inverted U-shaped curve, representing the transition. However, in their linear model specification, an increase in democracy also leads to an increase in deforestation. Arvin and Lew (2011) reconcile the opposing views by arguing that democracy reduces deforestation only in certain regions, while it increases deforestation in other regions. Kuusela and Amacher (2016) also point to the importance of political instability and the newness of a democratic regime in explaining why deforestation might increase in phases of democratic transition.

The results show that while political processes are likely to strongly impact the deforestation process, more research on the effect of specific forms of public involvement would be useful to draw policy relevant conclusions.

5.1.2 Rules

The variables *ownership* (i.e. land rights and general ownership security), and *environmental_policy* can be understood as forming part of the second pillar 'Rules', even though both of these moderator variables also contain governance measures that could be attributable to the governance level 'Enforcement'.[8] Using the variable *ownership* in an estimate increases the likelihood to find supportive results. The findings for the specific environmental governance variable *environmental_policy* document that studies based on this particular governance measure are significantly more likely to deliver estimates that support the governance hypothesis. This highlights again the importance of differentiating between general and environmental governance variables.

It is noteworthy that this result is not in line with the simple bivariate analysis discussed above. For *environmental_policy*, the coefficient estimate is only marginally significant at the 10% level. However, the marginal effects corroborate the relevance of the positive relationship and they are statistically significant at the 5% level.

The results for the variable *rights* are comparable to the findings for the variable *democracy*. They imply that neither more democratic decision making processes, nor a better protection of political rights through the rules of a state, necessarily lead to civil society commitment to forest conservation. The findings for the variable *ownership* are in line with Robinson et al.'s (2014) meta-analysis of individual case studies (rather than cross-national studies) on land rights security. While the impact of better land rights security on deforestation has been extensively discussed in the literature (cf. Liscow (2013); Brown and Pearce (1994); Angelsen and Kaimowitz (1999); Mendelsohn (1994)), our finding confirms the hypothesis that better land tenure security tends to decrease deforestation. The result strengthens the narrative that while under certain local circumstances, land tenure rights might have led to temporary increases in deforestation rates (Liscow, 2013), forest conservation is more likely to operate effectively if a functioning land rights system is in place that allows the correct delimitation and enforcement of forest conservation areas. The findings for the variable *environmental_policy* are in line with the prediction that countries that are more committed to international environmental policy agreements and better domestic environmental policies are

[8] Given the fact that for both of these variables *de facto* and *de jure* dimensions of governance are used conjointly, no clear attribution to one or the other category can be made.

likely to be associated with less deforestation. We find that this relationship is consistent across different studies and study designs.

5.1.3 Enforcement

The third governance level 'Enforcement' is analyzed with the variable *rule_of _law*. The results show that using this governance measure in a study significantly increases the likelihood to yield results that are supportive of the governance hypothesis. In terms of magnitude, the likelihood of finding a positive result increases by 35% while the probability of inconclusive results decreases by 30% (at a 5% significance level).

The variable *rule_of_law* is a general measure of governance, but it arguably has specific implications for environmental policy making. We can therefore not clearly attribute it to being either a general, or a specific environmental governance variable. Our finding is consistent with the more recent literature, for example they are in line with Barbier and Tesfaw (2015) who find that high scores in rule of law increase the likelihood of reaching a tipping point in deforestation, whereas high scores in regulatory quality reduces the probability of reaching this point.

5.2 Influence of other study design choices

5.2.1 Control variables

Overall, we can observe that the inclusion of different control variables does not impact the outcome to the same extent as the governance variables. Still, the choice of control variables does influence the study outcomes.

In contrast with the bivariate tests, the variables *agriculture* and *timber* are not significant in the multivariate estimation model. However, and again in contrast to the bivariate analysis, the coefficients for the variables *population* and *area* are statistically significant at the 10% and at the 1% level, respectively. Including the control variable *population* increases the likelihood of finding positive results, while the inclusion of the variable *area* reduces this likelihood. The marginal effects confirm these findings. Furthermore, they show that it is 39% more likely to find insignificant results when the variable *area* is included, and 23% less likely to find inconclusive results when the variable *population* is included.

The results show that if an estimate controls for demographic changes, it is more likely to find that better political institutions reduce deforestation. This relationship could suggest that political institutions are more important when demographic pressures to convert agricultural land are high. While this interpretation is only tentative, we believe that the institutions-population-deforestation nexus deserves further investigation.

Furthermore, the results on the variable *area* underline the importance of examining deforestation drivers in relation to existing forests stocks. This finding is in line with the strand of literature arguing that the existing forest stock influences deforestation trends. In this context the forest scarcity path (Wolfersberger et al., 2015; Rudel et al., 2005) is discussed most prominently. It argues that a country only starts to control deforestation politically, if the relative share of forest land has become small and forest products are thus scarce.

5.2.2 Spatial, temporal, and econometric variables

The spatial context of the sample studies evaluating the governance hypothesis does not seem to explain the variation between the estimated effects. More specifically, the dummy variable for the non-high income country group (*developing_countries*) remains statistically insignificant, i.e. studies that only focus on developing countries do not find different results.

Turning to the temporal context, the exploratory analysis suggests that more recent sample periods are more likely to yield positive results. In contrast, the multivariate results unambiguously suggest that the study period (here taken into account through the variable *period*) has no effect on the probability of the three effect categories. Similarly, the (square root of the) sample size (*size*) is not a significant moderator variable. The insignificant marginal effects are in line with sampling theory, showing that a larger sample size tends to decrease the probability of insignificant findings. We return to the sample size effect in the evaluation of the model below.

Examining the moderator variables that control for econometric issues, we find that the estimated coefficients associated with data characteristics – measured by the variable *panel* – and model specifications – measured by the variables *dynamics* and *nonlinear* – remain insignificant in the baseline model, suggesting that these technical choices are not a relevant source of variation between the estimated effects.
However, the key finding concerning the econometric issues is that the choice of the estimator is non-trivial. More specifically, the coefficient estimate and marginal effects show that using *OLS* estimation rather than more sophisticated estimators reduces the probability of obtaining positive results and significantly increases the probability of obtaining inconclusive results. This result contrasts with the exploratory analysis, where the proportion of positive findings is significantly larger when OLS models are taken into account. The finding could indicate a potential endogeneity bias in estimates of the effects of governance on deforestation. Measurement error in the governance variable seems a particular likely source of endogeneity bias. Indeed, if the independent variables are subject to measurement error, then the OLS coefficient estimates will be biased downwards (Wooldridge, 2002), which could explain our findings.[9]

Finally, and in contrast to the binary test, there is no clear indication of a publication bias in the results. When looking at the variable *date*, there is no support for a publication trend or turn-around in publication bias as the effect of the publication year remains both statistically and substantively trivial. There is also no clear indication of a publication bias in the sample when looking at the variable *type* (of publication). The estimated coefficient for published journal articles suggests that these studies are neither more nor less likely to show significant effects of governance on deforestation than working papers. We also recall that the sample size is not a significant moderator variable, but it should again be noted that these tests for publication biases are only indicative. Future research should apply more sophisticated methods to detect and correct for potential publication bias, but this would go beyond the scope of our objectives.

[9] Similarly, measurement error in the dependent deforestation variable would increase the variance of the error term, i.e. lead to inefficient coefficient estimates.

5.3 Robustness checks

We perform several additional analyses to assess the robustness of our results and methodology. First, to account for concerns that there may be a 'mechanical' effect of the sample size on the distribution of t-statistics and our dependent variable, we reestimate our ordered probit model based on weighted observations, using the square root of the sample size (Appendix table B, column 1) and the logarithm of the square root of the sample size as weights (Appendix table B, column 2)(similar to Waldorf and Byun (2005)). The weighted regression results are generally consistent with the unweighted estimates reported in Table 5.[10]
Second, to further investigate the role of sample size differences, we implement a simple specification test proposed in Card et al. (2010). We estimate separate probit models for the likelihood of significantly positive (Appendix table B, column 3) and significantly negative (Appendix table B, column 4) effect estimates that include the square root of the sample size. If there was a mechanical effect, a larger sample size should lead to larger positive t-statistics *and* larger negative t-statistics, i.e. we should find in both probit models a significantly positive coefficient for the square root of the sample size (in contrast, in our ordered probit regression the two effects might offset each other). Both estimated coefficients of interest are relatively small and clearly statistically insignificant, suggesting that there is no relationship between the sample size and the probability of either a significantly positive or significantly negative t-statistic. This finding suggests that the mechanical effect of sample size is mitigated by other design factors. Under a valid ordered probit model specification we would also expect the coefficients of the probit model for a significantly positive effect to be quite close to the ordered probit coefficients in table 5, while the coefficients from the probit model for a significantly negative effect should be close in magnitude but opposite in sign. For the concerning data, the estimated coefficients are not in perfect alignment with this prediction, but the predicted pattern seems qualitatively correct, particularly for the moderators that turned out to be highly significant.[11]

Finally, we check whether our results are influenced by the choice of the number of effect categories used. More specifically, we use four rather than three effect categories (Appendix table B, column 5) by splitting the insignificant findings into insignificant positive and insignificant negative effect estimates. The inferences concerning the effects of the moderator variables are highly consistent with the main estimation, with the only notable difference being that the variable *population* turns out to be insignificant.

6 Discussion and conclusion

Existing empirical evidence of the effect of governance on deforestation is based on a variety of different governance measures, as well as diverse study designs.

[10] Given that the square root of the sample size varies widely (min. 4.5; max. 58.7), using this variable for weighting may over-correct for robustness differences. We have, however, also experimented using it as weights. The results are materially the same and available upon request.

[11] Note that we cannot include the full set of moderator variables in the probit model for significantly negative estimates because some moderators predict the outcome perfectly.

As a result, the existing literature yields supportive, inconclusive, and non-supportive results for the hypothesis that better governance reduces deforestation (governance hypothesis).

In this paper, we have used a meta-regression analysis to provide a systematic synthesis of the literature in the field of economics on the effect of governance on deforestation. In particular, we have examined, whether (i) the level of governance (decision making processes, rules, or enforcement), (ii) the specificity of the deployed governance measure (general vs. specific environmental governance variables), or (iii) other methodological study design elements, significantly influence the results.

We have found no evidence that a particular level of governance systematically favors a supportive or non-supportive study outcome. However, our findings have corroborated the hypothesis that studies using specific environmental dimensions of governance, namely *environmental_policy* and the presence of environmental *NGOs*, are more likely to yield results that are supportive of the governance hypothesis. Furthermore, our meta-analysis has highlighted the role of other study design choices. While using the control variable *population* significantly increases the likelihood of a study to find a supportive result, using the control variable *area* significantly reduces it. Moreover, we have shown that the estimation technique significantly influences the likelihood of a study to yield results that are supportive of the governance hypothesis. In particular, using an ordinary least squares estimation technique significantly lowers the probability to find supportive results.

While the insights of this meta-analysis offer a first systematic evaluation of different studies that analyze the effect of governance on deforestation, the following limitations need to be taken into account, when interpreting the results.

First, the heterogeneity in scales (ordinal and cardinal) and units of the different governance and forest measures in our sample allows us to only meta-analyze the direction and statistical significance of effects rather than their magnitude. Thus, our meta-analysis does not reveal relevant information regarding the size of deforestation-reducing effects of different governance measures. As the literature is still growing, future meta-studies may eventually build on a more homogenous study sample and close this research gap.

Second, in contrast to a literature review, a meta-analysis examines a very specific phenomenon and thus has a more confined range (Waldorf and Byun, 2005). In the context of our study, the literature sample is restricted to the field of economics and economics related publications. Our choice reflects the trade-off in meta-analysis between the benefits of having a broad sample universe of studies from the literature and the significant costs involved in the coding and verification of the sample studies, following the standards in meta-analytical research laid out by Stanley et al. (2013). Notwithstanding, research expanding the scope to studies from other fields, such as environmental sciences, political sciences, or sociology is required to corroborate the here presented conclusions.

Finally, as discussions in the literature on the effect of governance on economic growth reveal, the effects of general governance measures, such as democracy, civil rights, and rule of law have to be interpreted cautiously due to potential measurement errors. In particular, Kaufmann and Kraay (2008) draw attention to measurement errors in governance variables and conclude that one should think of governance indicators as *"all providing imperfect signals of fundamentally unobservable concepts of governance"*. Devarajan (2008) argues that aggregated national governance indicators, do not account for local differences in governance, which can imply further fundamental measurement errors. Kurtz and Schrank (2007) furthermore argue that such general governance measures suffer from perceptual bias and adverse selection problems in the sampling process. Furthermore, according to Kishor and Belle (2004) general governance measures can capture a variety of phenomena and thus do not allow to measure the specific effects of forest sector governance on deforestation.

In spite of these limitations, our analysis provides avenues and insights for future research on the effect of governance on deforestation. We would like to use the remainder of this concluding section to discuss these insights.

First of all, while we find that a differentiation of the three governance levels does not conclusively explain the variation in study outcomes in our sample, more explicit analysis of whether these different governance levels are likely to have different effects on deforestation patterns seems valuable.

Second, our analysis indicates that contrary to the widespread assumption that all aspects of governance improvements are equally supportive to forest conservation, it is worthwhile to examine the effects of specific aspects of governance in the forest conservation context and to understand the underlying mechanisms. Based on the above described criticism of general governance measures, an alternative explanation to the positive effect of general democracy and civil rights improvements on deforestation could also be that the highly aggregated and ordinal governance measures may not constitute adequate proxies for governance. Yet, more research is required to draw robust conclusions.

Third, a better understanding of environmental governance seems necessary. The aggregation undertaken in the moderator variable environmental policy still proposes a very high level of aggregation of different dimensions of environmental policy. The study sample does not yet allow such a refinement, because it contains too few observations on individual dimensions of environmental policy. Future research may focus on providing a more nuanced analysis of environmental and forest sector governance and aim at collecting more specific data on environmental and forest sector governance. Currently, a central obstacle to the refinement of such analysis remains the availability of data. The State of the World's Forests Report in 2005 already highlighted that the lack of data on forest governance is a problem for evidence-based policy making in the field.

Most of the currently available data on forest governance is subject to various limitations: the available data focusing on specific elements of forest governance are reported in non-discrete units (mostly nominal or ordinal data, such as the presence or absence of a forest code (e.g. World Resources Institute (2014), Global Witness for Peru (2012)). Existing attempts to provide data on specific aspects of forest governance also use heterogeneous reporting standards for different countries. Consequently, the results are not comparable in a quantitative analyses. In addition, most initiatives to date only provide data for a selected number of countries (UN-REDD PGA for instance, for Indonesia

(UN-REDD, 2012), Vietnam (UN-REDD, 2013), Nigeria and Ecuador). Finally, most data are only available for short time periods. The absence of panel data prevents the use of more sophisticated estimation techniques, which may yield more robust results.

Forest governance data that have been added to the global Forest Resources Assessments conducted by the Food and Agriculture Organization of the United Nations (FAO) in 2005, 2010, and 2015 represent a notable exception. The global Forest Resources Assessments contains data on forest ownership and management rights, the designated functions of forests, forest management and legal status, employment in the forest sector, forest policy and legal framework, human resources within public forest institutions, forest education and research, forest revenue and public expenditure on forestry, and the status of ratification of international conventions and agreements. While this could become a promising data source for a more nuanced analysis of the effect of specific aspects of forest governance, to date only a few studies have used these sources (e.g. Galinato and Galinato (2016); Barbier and Tesfaw (2015); Whiteman et al. (2015)).

In order to enable policy advice to be tailored to country specific contexts, future research should focus on providing and using such more specific and more precise data on different aspects of forest governance. Kishor and Rosenbaum (2003), as well as Tegegne et al. (2014) discuss useful indicators for an improved measurement of forest governance in the European Forest Law Enforcement, Governance and Trade policy context. Kishor an Rosenbaum (2003) suggest, for instance, to measure the percentage of concession awards involving bribery, the records of concessions voided after discovery of illegality, or a record for concession holders to disclose familial connections to the government, as alternative and more precise measures for forest sector governance. Tegegne et al. (2014), suggest measuring the occurrence of illegal activities (illegal deforestation), the effective adjunction in forest related offenses, or the amount of resolved public disputes. Cowling et al. (2006) and Kishor and Rosenbaum (2012) provide guidelines for the collection of forest governance data. In view of the global increase of freedom of information laws (Ackerman and Sandoval-Ballesteros, 2006), freedom of information requests could be a promising method to collect such data for future research.

Furthermore, there are many deforestation drivers that lay outside of the forest sector that might also be related to weak governance. Future research could also evaluate in how far governance in other sectors, does affect deforestation rates. Ideally, research could provide guidance that would allow policy efforts for governance improvement, for instance in the REDD+ context, to be concentrated towards areas of governance that are most likely to be effective levers for forest conservation and furthermore exhibit the highest potential to yield positive spill-overs for other policy goals.

References

Acemoglu, D., Johnson, S., and Robinson, J. A. (2001). The colonial origins of comparative development: an empirical investigation. *American Economic Review*, 91(5):1369–1401.

Ackerman, J. M. and Sandoval-Ballesteros, I. E. (2006). The global explosion of freedom of information laws. *Administrative Law Review*, 58(1):85–130.

Angelsen, A. and Kaimowitz, D. (1999). Rethinking the causes of deforestation: lessons from economic models. *The World Bank Research Observer*, 14(1):73–98.

Arcand, J.-L., Guillaumont, P., and Jeanneney, S. G. (2008). Deforestation and the real exchange rate. *Journal of Development Economics*, 86(2):242–262.

Arvin, B. M. and Lew, B. (2011). Does democracy affect environmental quality in developing countries? *Applied Economics*, 43(9):1151–1160.

Banks, A. (2008). Cross-national time series data archives. URL: http://www.cntsdata.com/ (accessed 16/06/2016).

Barbier, E. B., Damania, R., and Léonard, D. (2005). Corruption, trade and resource conversion. *Journal of Environmental Economics and Management*, 50(2):276–299.

Barbier, E. B. and Tesfaw, A. (2015). Explaining forest transitions: the role of governance. *Ecological Economics*, 119(2015):252 – 261.

Bhattarai, M. and Hammig, M. (2001). Institutions and the environmental Kuznets curve for deforestation: a crosscountry analysis for Latin America, Africa and Asia. *World Development*, 29(6):995–1010.

Bohn, H. and Deacon, R. T. (2000). Ownership risk, investment, and the use of natural resources. *American Economic Review*, 90(3):526–549.

Bollen, K. (1993). Liberal democracy: validity and method factors in cross-national measures. *American Journal of Political Science*, 37(4):1207–1230.

Borlaug, N. (2007). Feeding a hungry world. *Science*, 318(5849):359–359.

Brons, M., Nijkamp, P., Pels, E., and Rietveld, P. (2008). A meta-analysis of the price elasticity of gasoline demand. A SUR approach. *Energy Economics*, 30(5):2105–2122.

Brown, K. and Pearce, D. (1994). *The causes of tropical deforestation.* University College London Press Ltd.: London, United Kingdom.

Buitenzorgy, M. and Mol, A. P. (2011). Does democracy lead to a better environment? Deforestation and the democratic transition peak. *Environmental and Resource Economics*, 48(1):59–70.

Busch, J. and Ferretti-Gallon, K. (2017). What drives deforestation and what stops it? A meta-analysis. *Review of Environmental Economics and Policy*, 11(1):3–23.

Card, D., Kluve, J., and Weber, A. (2010). Active labour market policy evaluations: a meta-analysis. *The Economic Journal*, 120(548):F452–F477.

Ceddia, M. G., Bardsley, N. O., Gomez-y Paloma, S., and Sedlacek, S. (2014). Governance, agricultural intensification, and land sparing in tropical South America. *Proceedings of the National Academy of Sciences*, 111(20):7242–7247.

Choumert, J., Combes Motel, P., and Dakpo, H. K. (2013). Is the environmental Kuznets curve for deforestation a threatened theory? A meta-analysis of the literature. *Ecological Economics*, 90(2013):19–28.

CIESIN (1990). Membership to environmental intergovernmental organizations - Center for International Earth Science Information Network. http://sedac.ciesin.columbia.edu/data/sets/browse?facets=theme:governance (accessed 13/04/2016).

Corderí Novoa, D. (2008). Deforestation and property rights: a comparison between former British and Spanish colonies. *Economic Analysis Working Papers*, 7(2008):1–14.

Cowling, P., DeValue, K., and Rosenbaum, K. (2006). *Assessing forest governance - a practical guide to data collection, analysis, and use*. PROFOR and Food and Agriculture Organization of the United Nations, Rome, Italy.

Crouzeilles, R., Curran, M., Ferreira, M. S., Lindenmayer, D. B., Grelle, C. E., and Benayas, J. M. R. (2016). A global meta-analysis on the ecological drivers of forest restoration success. *Nature Communications*, 7(11666):1–8.

Culas, R. J. (2007). Deforestation and the environmental Kuznets Curve: an institutional perspective. *Ecological Economics*, 61(2):429–437.

Damette, O. and Delacote, P. (2011). Unsustainable timber harvesting, deforestation and the role of certification. *Ecological Economics*, 70(6):1211–1219.

Damette, O. and Delacote, P. (2012). On the economic factors of deforestation: what can we learn from quantile analysis? *Economic Modelling*, 29(6):2427–2434.

Deacon, R. T. (1994). Deforestation and the rule of law in a cross-section of countries. *Land Economics*, 70(4):414–430.

Deacon, R. T. (1999). Deforestation and ownership: evidence from historical accounts and contemporary data. *Land Economics*, 75(3):341–359.

DeFries, R. S., Rudel, T., Uriarte, M., and Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience*, 3(3):178–181.

Devarajan, S. (2008). Two comments on governance indicators: where are we, where should we be going? By Daniel Kaufmann and Aart Kraay. *The World Bank Research Observer*, 23(1):31–36.

Diarra, G. and Marchand, S. (2011). Environmental compliance, corruption and governance: theory and evidence on forest stock in developing countries. *HAL archives-ouvertes*, 2011(00557677):1–21.

Didia, D. O. (1997). Democracy, political instability and tropical deforestation. *Global Environmental Change*, 7(1):63–76.

Dietz, T. and Kalof, L. (1992). Environmentalism among nation-states. *Social Indicators Research*, 26(4):353–366.

Ehrhardt-Martinez, K. (1998). Social determinants of deforestation in developing countries: a cross-national study. *Social Forces*, 77(2):567.

Ehrhardt-Martinez, K., Crenshaw, E. M., and Jenkins, J. C. (2002). Deforestation and the environmental Kuznets curve: a cross-national investigation of intervening mechanisms. *Social Science Quarterly*, 83(1):226–243.

FAO and PROFOR (2011). Framework for assessing and monitoring forest governance. URL: http://www.fao.org/docrep/014/i2227e/i2227e00.pdf (accessed 13/04/2016).

FCPF (2016). FCPF REDD+ countries. URL: https://www.forestcarbonpartnership.org/redd-countries-1 (accessed 13/05/2016).

Ferraro, P. J. and Hanauer, M. M. (2014). Quantifying causal mechanisms to determine how protected areas affect poverty through changes in ecosystem services and infrastructure. *Proceedings of the National Academy of Sciences*, 111(11):4332–4337.

Ferreira, S. (2004). Deforestation, property rights, and international trade. *Land Economics*, 80(2):174–193.

Ferreira, S. and Vincent, J. R. (2010). Governance and timber harvests. *Environmental and Resource Economics*, 47(2):241–260.

Fraser Institute (2016). Legal index. URL: http://www.freetheworld.com/ (accessed 13/05/2016).

Freedom House (2009). Freedom in the world comparative and historical data. URL: https://freedomhouse.org/report-types/freedom-world (accessed 13/05/2016).

Galinato, G. and Galinato, S. (2013). The role of government spending on deforestation and carbon dioxide emissions from land use change. *EconPapers Working Papers*, 2013(14):1–22.

Galinato, G. I. and Galinato, S. P. (2016). The effects of government spending on deforestation due to agricultural land expansion and CO_2 related emissions. *Ecological Economics*, 122(2016):43–53.

Geist, H. J. and Lambin, E. F. (2002). Proximate causes and underlying driving forces of tropical deforestation. *BioScience*, 52(2):143–150.

Gibbs, H., Rausch, L., Munger, J., Schelly, I., Morton, D., Noojipady, P., Soares-Filho, B., Barreto, P., Micol, L., and Walker, N. (2015). Brazil's soy moratorium. *Science*, 347(6220):377–378.

Greene, W. H. (2012). *Econometric analysis*, volume 7. Pearson Education Limited, Essex, United Kingdom.

Hedges, L. V. (1992). Meta-analysis. *Journal of Educational and Behavioral Statistics*, 17(4):279–296.

ICRG (2015). International Country Risk Guide. URL: http://epub.prsgroup.com/products/icrg-historical-data (accessed 13/05/2016).

Jevons, W. S. (1866). *The coal question.* Macmillan and Co, London, United Kingdom.

Jorgenson, A. K. (2008). Structural integration and the trees: an analysis of deforestation in less-developed countries, 1990–2005. *The Sociological Quarterly*, 49(3):503–527.

Kaufmann, D. and Kraay, A. (2008). Governance indicators: where are we, where should we be going? *The World Bank Research Observer*, 23(1):1–30.

Kaufmann, D., Kraay, A., and Mastruzzi, M. (2007). Growth and governance: a reply. *Journal of Politics*, 69(2):555–562.

Kaufmann, D., Kraay, A., and Zoido, P. (1999). Governance matters. *World Bank Policy Research Working Paper*, 1999(2196):1–64.

Kishor, N. and Belle, A. (2004). Does improved governance contribute to sustainable forest management? *Journal of Sustainable Forestry*, 19(1-3):55–79.

Kishor, N. and Rosenbaum, K. (2012). Assessing and monitoring forest governance: a user's guide to a diagnostic tool. URL: http://www.profor.info/node/1998 (accessed 17/06/2016).

Kishor, N. M. and Rosenbaum, K. L. (2003). Indicators to monitor progress of forest law enforcement and governance initiatives to control illegal practices in the forest sector. *International Forestry Review*, 5(3):211–218.

Kurtz, M. J. and Schrank, A. (2007). Growth and governance: models, measures, and mechanisms. *Journal of Politics*, 69(2):538–554.

Kuusela, O.-P. and Amacher, G. S. (2016). Changing political regimes and tropical deforestation. *Environmental and Resource Economics*, 64(3):445–463.

Li, Q. and Reuveny, R. (2006). Democracy and environmental degradation. *International Studies Quarterly*, 50(4):935–956.

Liscow, Z. D. (2013). Do property rights promote investment but cause deforestation? Quasi-experimental evidence from Nicaragua. *Journal of Environmental Economics and Management*, 65(2):241–261.

Mainardi, S. (1998). An econometric analysis of factors affecting tropical and subtropical deforestation. *Agrekon*, 37(1):23–65.

Marchand, S. (2011). Historical and comparative institutional analysis: evidences from deforestation. *HAL archives-ouvertes*, 2010.16(00552243):1–42.

Marquart-Pyatt, S. (2004). A cross-national investigation of deforestation, debt, state fiscal capacity, and the environmental Kuznets curve. *International Journal of Sociology*, 34(1):33–51.

Marshall, M. G. and Jaggers, K. (2002). *Polity IV project: political regime characteristics and transitions, 1800-2002*, volume 1. Integrated Network for Social Conflict Research, Washington DC, USA.

Mather, A. and Needle, C. (2000). The relationships of population and forest trends. *The Geographical Journal*, 166(1):2–13.

McFadden, D. (1974). The measurement of urban travel demand. *Journal of Public Economics*, 3(4):303–328.

Mendelsohn, R. (1994). Property rights and tropical deforestation. *Oxford Economic Papers*, 46(Special Issue on Environmental Economics):750–756.

Meyfroidt, P. (2016). Approaches and terminology for causal analysis in land systems science. *Journal of Land Use Science*, 11(5):501–522.

Midlarsky, M. I. (1998). Democracy and the environment: an empirical assessment. *Journal of Peace Research*, 35(3):341–361.

Nelson, J. P. and Kennedy, P. E. (2009). The use (and abuse) of meta-analysis in environmental and natural resource economics: an assessment. *Environmental and Resource Economics*, 42(3):345–377.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J., da Motta, R. S., et al. (2014). Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, 344(6188):1118–1123.

Pfaff, A. S. (1999). What drives deforestation in the Brazilian Amazon? Evidence from satellite and socioeconomic data. *Journal of Environmental Economics and Management*, 37(1):26–43.

Robinson, B. E., Holland, M. B., and Naughton-Treves, L. (2014). Does secure land tenure save forests? A meta-analysis of the relationship between land tenure and tropical deforestation. *Global Environmental Change*, 29(2015):281–293.

Rock, M. T. (1996). The stork, the plow, rural social structure and tropical deforestation in poor countries? *Ecological Economics*, 18(2):113–131.

Rudel, T. K., Coomes, O. T., Moran, E., Achard, F., Angelsen, A., Xu, J., and Lambin, E. (2005). Forest transitions: towards a global understanding of land use change. *Global Environmental Change*, 15(1):23–31.

Rudel, T. K., Defries, R., Asner, G. P., and Laurance, W. F. (2009). Changing drivers of deforestation and new opportunities for conservation. *Conservation Biology*, 23(6):1396–1405.

Schwab, K. and Porter, M. (2009). *The global competitiveness report 2008–2009*, volume 1. World Economic Forum, Cologny, Switzerland.

Shandra, J. M. (2007a). International nongovernmental organizations and deforestation: good, bad, or irrelevant? *Social Science Quarterly*, 88(3):665–689.

Shandra, J. M. (2007b). The world polity and deforestation a quantitative, cross-national analysis. *International Journal of Comparative Sociology*, 48(1):5–27.

Smith, J. (2015). Exploring connections between global integration and political mobilization. *Journal of World-Systems Research*, 10(1):255–285.

Smith, J. and Wiest, D. (2005). The uneven geography of global civil society: national and global influences on transnational association. *Social Forces*, 84(2):621–652.

Stanley, T., Doucouliagos, H., Giles, M., Heckemeyer, J. H., Johnston, R. J., Laroche, P., Nelson, J. P., Paldam, M., Poot, J., Pugh, G., et al. (2013). Meta-analysis of economics research reporting guidelines. *Journal of Economic Surveys*, 27(2):390–394.

Stanley, T. D. (2008). Meta-regression methods for detecting and estimating empirical effects in the presence of publication selection. *Oxford Bulletin of Economics and Statistics*, 70(1):103–127.

Stanley, T. D. and Doucouliagos, H. (2012). *Meta-regression analysis in economics and business*, volume 5. Routledge, London, United Kingdom.

Tegegne, Y. T., Van Brusselen, J., Tuomasjukka, D., and Lindner, M. (2014). Proposing an indicator framework for FLEGT voluntary partnership agreements impact monitoring. *Ecological Indicators*, 46(2014):487–494.

Tole, L. (2004). A quantitative investigation of the population-land inequality-land clearance nexus. *Population and Environment*, 26(2):75–106.

Transparency International (2012). *Making the forest transparent - annual transparency report.* Transparency International, Washington DC, USA.

Tsebelis, G. (1995). Decision making in political systems: veto players in presidentialism, parliamentarism, multicameralism and multipartyism. *British Journal of Political Science*, 25(03):289–325.

UN-REDD (2012). Participatory governance assessment: the 2012 Indonesia forest, land, and REDD+ governance index. URL: http://www.undp.org/content/dam/indonesia/docs/envi/PGA(accessed 15/03/2016).

UN-REDD (2013). *Analysis of indicators and small scale data collection for Participatory Governance Assessment (PGA) for the REDD+ in Viet Nam.* United Nations REDD+ Programme, Hanoi, Vietnam.

UN-REDD (2016). UN-REDD countries. URL: http://www.unredd.net/regions-and-countries/regions-and-countries-overview.html (accessed 13/05/2016).

Van, P. N. and Azomahou, T. (2007). Nonlinearities and heterogeneity in environmental quality: an empirical analysis of deforestation. *Journal of Development Economics*, 84(1):291–309.

Waldorf, B. and Byun, P. (2005). Meta-analysis of the impact of age structure on fertility. *Journal of Population Economics*, 18(1):15–40.

WEF (2012). 2.01 Stringency of environmental regulation. URL: http://www3.weforum.org/docs/TTCR/2013/TTCR_DataTables2_2013.pdf (accessed 01/12/2016).

Whiteman, A., Wickramasinghe, A., and Piña, L. (2015). Global trends in forest ownership, public income and expenditure on forestry and forestry employment. *Forest Ecology and Management*, 352(2015):99–108.

Wolfersberger, J., Delacote, P., and Garcia, S. (2015). An empirical analysis of forest transition and land-use change in developing countries. *Ecological Economics*, 119(2015):241–251.

Wooldridge, M. J. (2002). Econometric analysis of cross section and panel data. *Massachusetts Institute of Technology.*

World Bank (2014). The Worldwide Governance Indicators. URL: http://info.worldbank.org/governance/wgi/index.aspx#home (accessed 13/04/2016).

World Bank (2016). New country classifications. URL: https://datahelpdesk.worldbank.org/knowledgebase/articles/378834-how-does-the-world-bank-classify-countries (accessed 20/07/2016).

WRI (1998). *World resources 1998-99 : a guide to the global environment, environmental change and human health.* World Resources Institute, Washington DC, USA.

WRI (2014). *Rights to Resources: the status of local natural resource rights in national laws in Sub-Saharan Africa.* World Resources Institute, Washington DC, USA.

WTO (2016). Government finance statistics. URL: http://www.imf.org/external/data.htm (accessed 13/05/2016).

Appendix

A.Details on the identification of the study population

Table 6: Details on the identification of the study population

Search databases	Keywords (as of the 16.2.2016)	Search specification	Date of search	Exportable entries	Final relevant entries
Econ Papers	deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”)	no further specification	21.02.2016	29	3
ScienceDirect (Economics, Econometrics and Finance)	deforestation AND (“governance”OR “institutions”) AND (“regression” OR “empirical”)	- Search criteria: Deforestation AND (“governance” OR “institutions”) AND (“regression” or “empirical”) - Economics, Econometrics and Finance - Type: Article - Time selection: All years	21.02.2016	692	7
Wiley (simple search)	deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”)	- Specification: article	21.02.2016	29	0
JSTOR (journals)	deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”)	- Search criteria: Deforestation AND (“governance” OR “institutions”) AND (“regression” or “empirical”) - Item type: Articles - Discipline and or journal: Economics - All years	21.02.2016	330	4
Springer	deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”)	- Deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”) - Discipline: Economics - Language: English	21.02.2016	427	4
SSRN	deforestation governance institutions	- Deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”) - Did not yield any results. Consequently, we use “deforestation governance institutions”.	21.02.2016	3	0
SAGE Journals	deforestation AND (“governance” OR “institutions”) AND (“regression” OR “empirical”)	- Specification: Economics and Development	29.02.2016	74	0
Taylor and Francis	deforestation AND (“governance”OR “institutions”) AND (“regression” OR “empirical”)	- Specification: Journal - Specification: Areas Economics, Finance - Business & Industry - Environment and Sustainability	29.02.2016	156	3
Choumert et al. 2013	-		29.02.2016	77	9
Angelsen and Kaimowitz, 1999.	-		29.02.2016	27	2

B. Robustness tests

	(1) Weigthed [sq(N)]	(2) Weigthed [log(sq(N))]	(3) Probit for significantly positive	(4) Probit for signficantly negative	(5) Four categories
Governance moderator variables					
environmental_policy	0.873	1.042	0.811	0	1.120*
	(0.645)	(0.639)	(0.696)	(.)	(0.678)
ownership	0.593	1.058**	1.656***	-0.0145	1.291***
	(0.569)	(0.492)	(0.447)	(0.844)	(0.494)
democracy	-1.110**	-1.013**	-1.121**	0.750	-0.939*
	(0.533)	(0.499)	(0.551)	(0.585)	(0.491)
NGOs	2.379***	2.328***	1.922***	0	2.450***
	(0.621)	(0.534)	(0.549)	(.)	(0.504)
rights	-1.396***	-1.462***	-1.335**	2.563***	-1.563***
	(0.529)	(0.532)	(0.550)	(0.704)	(0.536)
rule_of_law	0.741	0.912*	0.899*	0	0.984**
	(0.462)	(0.496)	(0.503)	(.)	(0.485)
Control moderator variables					
population	1.113**	1.029**	1.326**	-0.801	0.736
	(0.554)	(0.508)	(0.532)	(0.805)	(0.461)
income	0.611	0.286	0.0221	0.669	0.268
	(0.505)	(0.512)	(0.544)	(0.672)	(0.512)
area	-1.047**	-1.293**	-1.127**	1.271*	-1.328***
	(0.531)	(0.505)	(0.477)	(0.738)	(0.482)
timber	-0.545	-0.275	-0.692**	-1.475**	-0.134
	(0.350)	(0.331)	(0.348)	(0.581)	(0.317)
agriculture	0.337	0.209	-0.0639	-0.881	0.0851
	(0.393)	(0.391)	(0.487)	(0.674)	(0.366)
Spatial, temporal and econometric moderator variables					
developing_countries	-0.739*	-0.528	-0.540	0.893	-0.577
	(0.409)	(0.371)	(0.432)	(0.587)	(0.366)
average	0.0171	0.0415	0.0714*	-0.0180	0.0427
	(0.0470)	(0.0462)	(0.0426)	(0.0575)	(0.0458)
panel	-1.007	-0.467	-0.104	-0.125	0.0631
	(0.884)	(0.860)	(1.015)	(1.196)	(0.840)
dynamic	-0.367	-0.413	-1.114		-0.581
	(0.715)	(0.704)	(0.700)		(0.690)
nonlinear	-0.758**	-0.560	-0.168	1.137*	-0.428
	(0.352)	(0.342)	(0.375)	(0.598)	(0.363)
OLS	-1.076***	-0.935***	-1.028*	1.163	-0.793**
	(0.284)	(0.334)	(0.597)	(0.738)	(0.397)
date	-0.0644	-0.0676	-0.0999*	0.0169	-0.0824
	(0.0567)	(0.0544)	(0.0604)	(0.0708)	(0.0528)
type	-0.267	0.300	0.304	-1.618*	0.454
	(0.681)	(0.645)	(0.635)	(0.927)	(0.638)
size	0.0296	0.0185	0.0110	-0.00840	0.00702
	(0.0219)	(0.0207)	(0.0234)	(0.0225)	(0.0198)
N	227	227	227	141	227
Pseudo R2	0.2525	0.2447	0.3111	0.3597	0.2234

Standard errors (clustered by study) in parentheses.
Estimated cutpoints are not reported. * $p<0.10$, ** $p<0.05$, *** $p<0.01$

Chapter 3

Accounting for institutional capacity in global forest modeling[1]

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Accounting for institutional capacity in global forest modeling

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Abstract

The current state of the art in modeling forest cover change is to combine a detailed representation of biophysical processes with economic decision-making principles. Yet, there is an increasing consensus that the quality of political institutions is another relevant component in determining forest cover change patterns.

In this paper, the Global Forest Model is used to analyze whether including an index, measuring the capacity of political institutions to guarantee sustainable natural resource management, allows to improve the precision of the modelled forest cover trend. The analysis shows that incorporating the index indeed allows reducing the gap between the estimated and observed forest cover trends for the 2000 to 2010 calibration period.

Keywords: Global Forest Model, forest cover change modeling, institutional capacity for natural resource management

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1 Introduction

Deforestation accounts for 12% of anthropogenic CO_2 emissions (Smith et al., 2014), causes biodiversity loss (Gibson et al., 2011), soil erosion (Smith et al., 2016), ground water stress, and changes in local rainfall patterns (Garcia-Carreras and Parker 2011). The literature has widely acknowledged the conversion of forest land to agricultural land as the main driver of deforestation (Busch and Ferretti-Gallon 2017; Gibbs et al., 2010; Mayaux et al., 2013). This logic is typically reflected in land use change models. In the Global Forest Model (G4M global v.4.0), a representative land owner makes a return-maximizing land use decision, based on a comparison of net present values of agricultural and forest land. The biophysical properties and the agricultural suitability of the land are taken into account for the decision (Kindermann et al., 2006; 2008).

After Brazil's historic success in curbing deforestation by more than 70% (Tollefson, 2015) through enhanced enforcement and fining of illegal deforestation (Cisneros et al., 2015; Hargrave and Kis-Katos, 2012; Nepstad et al., 2014), more recent literature on deforestation suggests that, next to economic and biophysical factors, the quality of political institutions is a key parameter influencing land use change decisions (Bhattarai and Hammig, 2001; 2004; Buitenzorgy and Mol, 2011; Galinato and Galinato, 2012; Koyuncu and Yilmaz, 2013) [1]. Furthermore, examples such as Costa Rica and Colombia, with similar development of gross national income (World Bank, 2015c), but diverging forest cover trends from 1990 to 2015. Costa Rica experienced 7.5% forest cover gain and Colombia 9% forest cover loss (FAO, 2015). This situation suggest that the quality of political institutions can influence deforestation regardless of the level of economic deforestation pressure.

By providing intertemporal contracts, institutions help generate regularity in social behavior and can prevent the overuse of common goods (Aoki, 2001). In this article we refer to this capacity with the concept of 'environmental institutional quality'. It measures the extent to which existing political institutions lead to a sustainable use of common environmental resources. In order to measure environmental institutional quality, this paper builds on the FAO's forest governance framework (2011) that suggests to measure the quality of political institutions in the forest sector by three components: (i) the regulatory frameworks, (ii) planning and decision making processes, and (iii) the implementation and enforcement of policies.

Despite the increasing attention that is paid to the quality of political institutions in deforestation processes in the empirical literature, it is still uncommon to take differences in the quality of institutions into account when modeling forest cover change trends (Benítez et al. (2007) and Wang et al. (2016) represent notable exceptions). Magliocca et al. (2015) make a strong case for using synthesis knowledge to improve process-based land change models. This paper aims at taking this new trend in the empricial literature into account for the forest cover change simulations of the Global Forest Model. In the Global Forest Model, all factors causing deviations of purely economically motivated land use change decisions are captured by the residual calibration

[1] In spite of this success, deforestation rates in the Brazilian Amazon have recently increased again by 29% between 2015 and 2016 (INPE, 2017).

factor. This factor is multiplied by the estimated net present value of forestry, to yield an adjusted net present value of forest land use.

This paper tests the hypothesis whether the residual calibration factor can be reduced by including an additional parameter into the model, which measures the quality of political institutions that are relevant for the sustainable management of environmental resources like forests. Reducing the residual calibration factor would reduce unexplained factors influencing the forest cover change decisions and thus improve the representation of deforestation processes in the model. The hypothesis is first tested through a regression analysis using the residual calibration factor for the 2000 to 2010 period as a dependent variable and environmental institutional quality indicators as independent variables. In a second step, the index is applied to the model, to evaluate, in a third step, whether this can substantially improve the simulation. Finally, a test of the significance of the findings is conducted using data on forest cover change for 2015. The test indicates that for the countries selected, a better match between the model forecast and observed forest cover change trends can be achieved by accounting for environmental institutional quality.

The remainder of the paper is organized as follows: in section 2 the data and the construction of the environmental institutional quality indicator are presented. Section 3 presents the methodological steps taken to integrate the indicator into the model. Section 4 presents the results, and section 5 discusses the relevance of the findings for other forest cover change models and ecological process models in general.

2 Data

2.1 Environmental Institutional Quality Index

An in-depth review of existing sources reveals that comprehensive cross country datasets, measuring differences in the quality of institutions affecting the management of natural resources, are unavailable. At the same time, Kishor and Belle (2004) highlight that general governance indicators are unsuitable to measure the performance of institutions charged with the management of natural resources, because general and specific environmental governance trends can be fundamentally different. In the absence of specific measures on the quality of forest sector institutions, an indicator is constructed relying upon the FAO's forest governance framework (2011). The index aims at representing the three pillars of the forest governance framework, which are (i) the quality of regulatory frameworks, (ii) the effectiveness of decision making processes, and (iii) the enforcement of existing policies. Existing indicators on the quality of institutions are used as a proxy for these components and aggregated to form the composite environmental institutional quality index.[2]

More precisely, component 1 is represented by the indicator "*Environmental Policy*" developed by Hartmann and Reimann (2010). This indicator is defined as measuring the extent to which "*the externalization of costs or inadequate time horizons are avoided or restrained by environmental regulation*"

[2]In the absence of suitable specific measures on the quality of forest sector institutions, we revert to more general measures on the quality environmental institutions in order to approximate the quality of forest institutions.

Table 1: Data sources for the composition of the environmental institutional quality index

Indicator	Source	Institution	Availability
Environmental policy	Hartmann and Reimann (2010)	Bertelsmann Foundation	2006-2014
Number of days to start a business	Porter et al. (2008)	World Economic Formum	2003, 2006-2014
	The World Bank Group (2015a)	World Bank Group	2004-2014
Structural constraints	Hartmann and Reimann (2010)	Bertelsmann Foundation	2006-2014

(Hartmann and Reimann, 2010). Indicator values are based on expert assessments and provided in an ordinal scale ranging from low (1) to high (10).

Component 2 is represented by the indicator "*number of days to start a business*" by Porter et al. (2008), which is complemented with the same indicator produced by the World Bank (2015a) taking data for Niger, the Democratic Republic of the Congo, Belize, Tajikistan, Laos, Myanmar, Dominican Republic, Brunei, Sierra Leone, Togo, Uzbekistan, Liberia, Eritrea, Republic of Congo, Gabon, Sudan, Central African Republic, Belarus Guyana, Bhutan, Suriname, and Gambia. The indicator measures the time required to start a business by the number of "*calendar days needed to complete the procedures to legally operate a business*" (World Bank, 2015a; Porter, 2008). It refers to a standard business that is 100% domestically owned. The indicator is used as a proxy for component 2 of the FAO forest governance framework (2011), reflecting how long it takes for an administration to make a decision and thus how effective decision making processes are. Data is provided in cardinal units, counting the number of days.

Component 3 is represented by the indicator "*structural constraints*" devised by Hartmann and Reimann (2010). This indicator is defined as measuring "*structural difficulties [that] constrain the political leadership's governance capacity*" (Hartmann and Reimann, 2010). Structural difficulties include "*a lack of educated labor force*" and "*severe infrastructural deficiencies*" (Hartmann and Reimann, 2010). It is therefore used as a proxy for the government's ability to implement existing environmental and forest conservation policies. The indicator is based on an expert assessment on an ordinal scale ranging from 0 to 10 points. A score of 10 corresponds to a low level of structural constraints. Table 1 presents details on data sources for each of the components. The complete dataset is available upon request.

In order to construct the index, the different indicator components have to be normalized. Values for the variables "*number of days to start a business*" and "*structural constraints*" are rescaled, such that for all components of the composite index a high value represents high environmental institutional quality. In a second step, all values are normalized to values between 0 and 1 in order to ensure that different components of the index are weighted equally. All index components are then totaled and the resulting value is normalized again to make sure that upper and lower bounds for index values are defined. Average index values are calculated for two periods, corresponding to the two simulation periods of the Global Forest Model, 2000 to 2010 (for the available years) and 2010 to 2015. The index is available for a total sample of

115 countries for the period 2000 to 2010 and for 116 countries for the period 2010 to 2015. Indicator values range from 0 to 1. Data is only available for economies in transition (Hartmann and Reimann, 2010). Consequently, developed countries (countries in North America and Western Europa, as well as Japan, Australia and New Zealand) are excluded. This limitation means that countries that score relatively well do so in comparison to all other economies in transition (and not compared to developed countries). Furthermore, the index does not provide results for very small countries (especially small islands), non-independent territories, and countries that were in a phase of civil war or unclear statehood during the calibration period. This is also due to limited data availability for these countries. A detailed list of indicator values is available in the appendix.

2.2 Control variables

In order to test how far the residual calibration factor can be explained by the quality index for environmental institutions, account needs to be taken of alternative explanations. The two main alternatives are geographic factors and differences in levels of economic development (DeFries et al., 2010; Mather and Needle, 1998 ;2000).

More precisely, data on (i) land area in 1000 ha (FAO, 2010), (ii) forest cover in 1000 ha (FAO, 2010), (iii) gross domestic product purchasing power parity in current US$ (World Bank, 2015b), (iv) population density measured by the amount of people per sq. km of land area (World Bank, 2015d), and (v) a dummy variable indicating, whether there is tropical, or subtropical vegetation in a country, based on the FAO's Ecological Zones (FAO, 2000), are used to control for possible alternative explanations in the model. Averages for all control variables are calculated for the calibration period of the global forestry model (2000-2010) in order to test the regression model.

Gross domestic product is controlled for, despite the fact that it is already included in the Global Forest Model, because in the current model version, gross domestic product only influences the forestry net present value indirectly through the price of wood. However, some economies have become less dependent on agricultural expansion for economic growth (Acemoglu, 2008). As a result, they could experience less deforestation pressure. This effect can be controlled for by including an additional measure of gross domestic product into the model.

3 Methods

The Global Forest Model compares net present agricultural (A_i) and forestry (F_i) values for a grid cell (resolution 0.5° x 0.5°) to simulate the land use change decision of a representative land owner, who may be a state or individual, depending on the ownership regime (Gusti and Kinderman, 2011; Kindermann et al., 2006; 2008). The subscript i indicates that the values vary for each country. A net present value calculation allows to optimize investment decisions, based on future expectable costs and revenues. In the case of a forest manager, it allows to optimize the length of the rotational period, by choosing the age at which the net present value of the timber will be maximized (Perman, 2013) It thereby allows evaluating competing long-term investment options. A deforestation decision is made when $A_i > F_i$, an afforestation decision is made when $A_i < F_i$ and no land use change decision is made when $F_i = A_i$ (Gusti, 2015; Rokityanskiy et al., 2007).

A_i is estimated using a Cobb-Douglas production function (Gusti et al., 2008). The parameter is influenced by agricultural suitability, population density, the amount of forest area in neighboring grid cells, past deforestation rates, road density, and the gross domestic product. A_i is estimated following Kindermann (2006) and multiplied by the relative change in agricultural land prices provided by the Global Biosphere Management Model GLOBIOM (Havlík et al., 2011), which is a global, bottom-up, partial equilibrium land use change model. The linkage between G4M and GLOBIOM is described in Gusti and Kindermann (2011).

F_i is calculated assuming that the representative forest manager optimizes the wood value, which is influenced by harvesting costs, transportation costs, available forest biomass, wood prices, and the representative forest manager's discount rate. As with the A_i, these parameters are estimated following Kindermann et al. (2006) and are adjusted at cell level to relative prices derived from an agricultural land use model (cf. Gusti and Kindermann, 2011), the Global Biosphere Management Model (Havlík et al., 2011). Figure 2 provides a schematic illustration of the Global Forest Model. The model cannot account for spatial heterogeneity. Furthermore, the model does not include non-agricultural or non-forestry land use forms. The model also excludes forest areas under protected area status, based on the World Database on Protected Areas (IUCN and UNEP-WCMC, 2006).

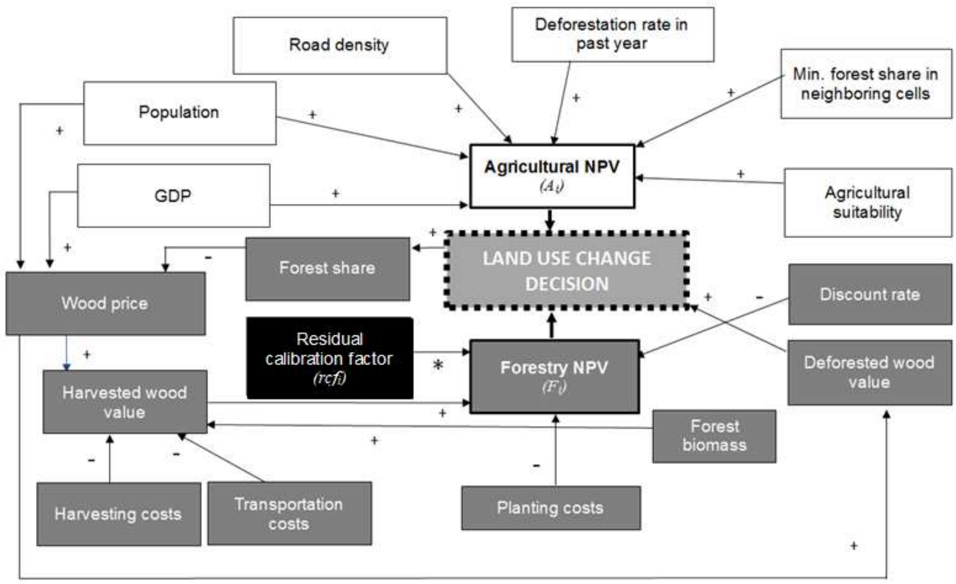

Figure 1: Global Forest Model schematic illustration: the land use change decision is based on a comparison between agricultural net present values (A_i) (influenced by parameters in white) and forestry net present values (F_i) (influenced by parameters in dark grey) and makes a land use change decision, based on this comparison. A '+' sign indicates that a parameter influences the respective net present value positively, a '-' sign indicates a negative influence and an asterix is used to illustrate that the residual calibration factor is multiplied with F_i. Source: Gusti (2015).

The Global Forest Model is calibrated to match country level deforestation and afforestation data based on the FAO's Forest Resources Assessment for the period from 2000 to 2010 (FAO FRA 2015). Within a country, values are assigned to cells according to Hansen (2013).

In the model the residual calibration factor (rcf_i) captures all factors causing deviations of purely economically motivated land use change decisions. These calibration values again vary from one country to the other, hence we use the subscript i in our notation. It is multiplied by the estimated net present value of forestry, which yields an adjusted net present value of forestry, as described by equation (1).

$$F_{i,adjusted} = rcf_i \cdot F_{i,estimated} \tag{1}$$

In order to estimate whether the residual calibration factor is likely to be explained by the environmental institutional quality index, an ordinary least squares regression model is used

$$ln(rcf_i) = \beta_0 + \beta_1 EIQ_i + \gamma_j CV_{i,j} + \epsilon_i, \tag{2}$$

where EIQ_i is the environmental institutional quality index. β_0, β_1 are coefficients that indicate the effect of the EIQ_i on $ln(rcf_i)$, holding all other factors fixed. $CV_{i,j}$ is a vector composed of the five control variables identified above (CV_1), GDP purchasing power parity (CV_2), land size (CV_3), population density (CV_4), and tropical and subtropical vegetation (CV_5). γ is a coefficient that indicates the effect of each control variable (j) on the outcome variable, holding all other factors fixed. ϵ_i is the error term.

A semi-logarithmic specification is used, based on the assumption that a one unit increase in environmental institutional quality leads to a percentage increase in the residual calibration factor (and thus in the multiplier of the net present value of forestry). Different versions of the model in equation (2) are estimated: (i) without control variables, (ii) with each single control variable, and (iii) with all control variables. Furthermore, as a robustness test, the regressions are also estimated for a restricted range of residual calibration factor values (values between 0.05 and 15).

The index is incorporated into the model using the following procedure: the adjusted forestry net present value ($F_{i,adjusted}$) is calculated using the estimated forestry net present value multiplied by the "*old*" residual calibration factor, as used in previous model applications ($rcf_{i,old}$):

$$F_{i,adjusted} = rcf_{i,old} \cdot F_{i,estimated}. \tag{3}$$

The old residual calibration factor matches model simulations with observed deforestation trends and thereby accounts for all deviations from purely economic and biophysical land use change decision making parameters. Values of the $rcf_{i,old}$ are multiplied with the composite index to achieve a residual calibration factor to be used in future model applications ($rcf_{i,new}$). The values are exponentially distributed. Therefore the logarithm of the residual calibration factor multiplied by the composite index, is equal to the natural logarithm of $rcf_{i,new}$:

$$ln(rcf_{i,old}) \cdot EIQ_i = ln(rcf_{i,new}). \tag{4}$$

From equation (4) it follows that

$$ln(rcf_{i,old}) = ln(rcf_{i,new}^{\frac{1}{EIQ_i}}). \tag{5}$$

The application of the index to the model provides a measure of the reduction in the residual calibration factor and thus model quality. The percent reduction of the residual calibration factor at the country level is then calculated as

$$\%reduction = 100 \cdot (1 - \frac{1 - rcf_{i,new}}{1 - rcf_{i,old}}). \tag{6}$$

In order to test the significance of the results found for the calibration period, the percentage reduction of the residual calibration factor, as given in equation (6), was again calculated using (i) data on environmental institutional quality for the period 2010 to 2015 to (ii) construct the environmental institutional quality index for the period 2010 to 2015 as derived from the regression results for the calibration period 2000 to 2010.

4 Results

4.1 Regression analysis

Table 2: Ordinary least squares regression using the logarithm of the residual calibration factor and the index. The significance level is indicated by the stars next to the coefficient values. The bracketed value below the coefficient value corresponds to the standard errors.

Variable name	(1) $logRCF$	(2) $logRCF$	(3) $logRCF$	(4) $logRCF$	(5) $logRCF$	(6) $logRCF$	(7) $logRCF$
EIQ	4.396***	4.836***	5.077***	4.582***	5.108***	5.359***	4.228***
	(3.22)	(4.69)	(4.85)	(3.92)	(4.97)	(5.07)	(3.76)
CV_1	-0.00000295		0.00000325				
	(-0.56)		(1.53)				
CV_2	0.0000174			0.00000559			
	(1.03)			(0.44)			
CV_3	0.00000293				0.00000199*		
	(1.15)				(1.96)		
CV_4	-0.000616*					-0.000620*	
	(-1.77)					(-1.88)	
CV_5	-0.638						-0.865
	(-1.04)						(-1.46)
ϵ	-2.581**	-3.089***	-3.301***	-3.031***	-3.393***	-3.260***	-2.044**
	(-2.58)	(-5.46)	(-5.66)	(-5.19)	(-5.85)	(-5.75)	(-2.26)
N	105	112	111	111	112	112	107
t statistics	in brackets						
* p<0.10	**p<0.05	*** p<0.01					

Table 2 shows that the regression yields statistically significant results at the 0.1% level (p-value = 0.001), when the model is tested with all control variables (column 1), no control variables (column 2), and each single control variable (column 3-7). Furthermore, table 3 reports results of a robustness test with a restricted range (0.05-15) of residual calibration factor values and shows that these tests also yield statistically significant results.

4.2 Reduction of the residual calibration factor

When using the environmental institutional quality index in the Global Forest Model, the residual calibration factor can, on average, be reduced by 43% for the 2000 to 2010 calibration period. Figure 3 shows the percentage reduction

Table 3: Ordinary least squares regression using the logarithm of the residual calibration factor and the index for a restricted range of residual calibration factor values (0.05 to 15).The significance level is indicated by the stars next to the coefficient values. The bracketed value below the coefficient value corresponds to the standard errors.

Variable name	(1) $logRCF$	(2) $logRCF$	(3) $logRCF$	(4) $logRCF$	(5) $logRCF$	(6) $logRCF$	(7) $logRCF$
EIQ	1.652*	1.379**	1.545**	1.193*	1.620**	1.740***	1.261*
	(1.99)	(2.17)	(2.39)	(1.69)	(2.57)	(2.67)	(1.78)
CV_1	0.000		0.00000193				
	(-0.81)		(1.64)				
CV_2	0.00000927			0.00000411			
	(0.99)			(0.58)			
CV_3	0.00000227				0.00000124**		
	(1.58)				(2.23)		
CV_4	-0.000392**					-0.000362**	
	(-2.06)					(-2.01)	
CV_5	0.0470						-0.160
	(0.13)						(-0.45)
ϵ	-1.191*	-0.792**	-0.946**	-0.754**	-1.037***	-0.927**	-0.618
	(-1.89)	(-2.15)	(-2.49)	(-2.00)	(-2.75)	(-2.51)	(-1.07)
N	82	88	87	87	88	88	84
t statistics	in brackets						
* p<0.10, **	p<0.05,	*** p<0.01					

in the residual calibration factor per region (regions are clustered according to the ISO3166 list of countries (ISO, 2016), with within continent differences for Africa and South and Latin America, a relatively uniform reduction pattern for Asia, and the lowest influence of the environmental institutional quality index in Eastern European countries.

A more detailed look at individual countries indicates differences within regions: the reduction in the residual calibration factor is of 58% for Indonesia, while it is only of 22% for Malaysia. Specifically, that means that for Indonesia, the use of the environmental institutional quality index helps model forest cover change, but for Malaysia, the index is not useful in reducing the residual calibration factor. This is a strong indication that there must be explanations for the residual calibration factor in Malaysia, other than the quality of Malaysia's environmental institutions. These are not currently captured by the model.

4.3 Testing the index

Testing the index was complicated by two factors: (i) Missing values still present a challenge in interpreting the data provided by the 2015 Forest Resources Assessment (FAO, 2015a) at the global level. In order to provide data for countries or territories that did not provide an assessment at all the FAO has used desk studies. Such desk study estimates correspond to 1.2% of the forest area (MacDicken et al. 2015). In addition, a range of countries have used projections in their national reports, instead of actual inventories (cf. FAO, 2015b). These projections are based on the 2010 data-set. (ii) Although index values for the two calibration periods (2000-2010 and 2010-2015) are different for almost all countries, no significant difference in the mean environmental institutional quality index could be observed (0.516 and 0.499 for 2000 to 2010 and 2010 to 2015 respectively; Mann and Whitney test: p=0.508).

Consequently, the overall average percentage reduction in the residual calibration factor does not change significantly, being 43% and 46% for 2000 to 2010 and 2010 to 2015, respectively (Mann and Whitney test p=0.868). For individual countries, however, results exhibit an improvement of the model

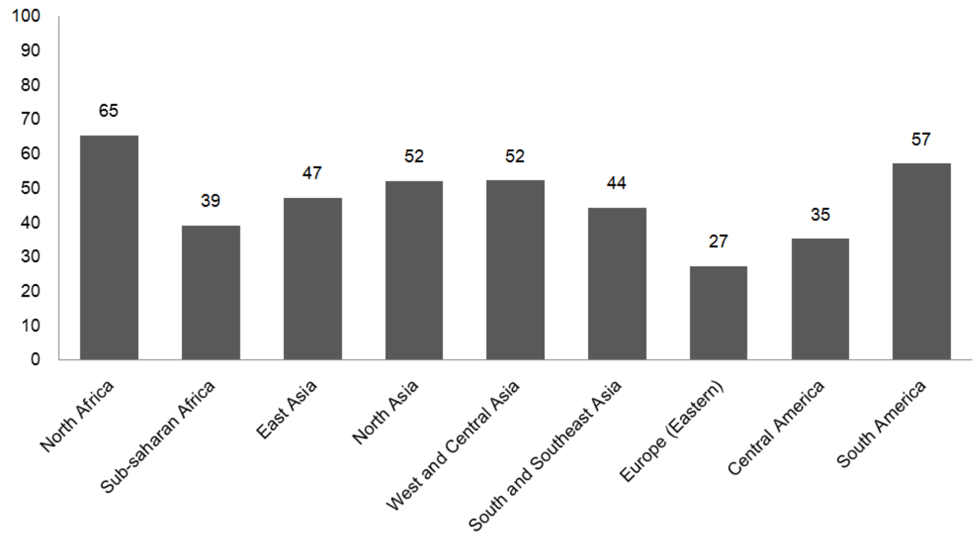

Figure 2: The bars indicate the average reduction of the residual calibration factor per region in percent. The regional clustering is undertaken according to the ISO3166 norm (ISO, 2016).

estimates. For instance, there is an increase in environmental institutional quality of about 26% in Brazil (EIQ index of 0.306 and 0.387 for 2000 to 2010 and 2010 to 2015, respectively), resulting in a decrease in estimated mean annual forest cover loss of 43% (-2282 vs. -984 km^2 per year for 2000 to 2010 and 2010 to 2015, respectively), in line with the observed reduction in deforestation. For Paraguay on the other hand, a decrease in environmental institutional quality by 13% (index value of 0.452 and 0.293 for 2000 to 2010 and 2010 to 2015, respectively) resulted in an estimated increase annual net forest cover loss of 35% (242 and 325 km^2 per year for 2000 to 2010 and 2010 to 2015, respectively).

5 Discussion

The above analysis shows that accounting for differences in environmental institutional quality through the EIQ_i in the Global Forest Model, improves the representation of past deforestation trends. While a wider application of this technique could be interesting for other ecological models, data constraints currently considerably limit this potential. This section discusses the findings of our analysis on the context of the broader land use change and environmental modeling literature, as well as the implications of data constraints for such analysis in the future.

5.1 Implications for global forest modeling

Agrawal et al. (2002) and Turner et al. (2016) highlight the importance of including a better representation of human decision making systems in process-based forest models. Most innovations in forest modeling focus on the representation of ecological processes. Studies aim for example at better representing the effect of reduced precipitation under climate change on forests stocks (Fischer et al., 2014), the effect of temperature rise on tree species (Liang and Zhou, 2010), or carbon storage dynamics (Liu and Han, 2009). Others discuss the benefits of more complex physiological models over hybrid empirical-physiological models of forest ecosystems (Jin et al., 2016). However, most model innovations stay within the realm of purely ecological processes and do not take the impact of human decision making systems into account in their analysis.

Of the few models taking this aspect more explicitly into account, most focus on behavioral aspects in the form of agent based models (An, 2012). Alternatively, they only analyze the effects of local political and institutional settings (Janssen et al., 2000; Lobianco et al., 2015) rather than taking the influence of entire political systems into account. In some specific cases, data on national institutional quality have been incorporated into land use change models, mainly through the channel of the discount rate. During an earlier application of the Global Forest Model, Benítez et al. (2007) included country risk rating data in order to adjust discount rates to simulate afforestation investment. A recent contribution by Wang et al. (2016), using an agro-economical model, also takes the quality of institutions into account by applying the discount rate. In their study, weak governance leads to a higher discount rate, which disincentivizes investment into better technologies and consequently leads to the expansion of croplands (which translates into deforestation). This approach, however, is not based on an explicit biophysical and economic model of change in forest cover and only deals with one specific channel of influence (technological change) with deforestation conceived as an externality of cropland expansion. With respect to the current state of the literature on forest cover change modeling, it thus becomes clear that institutional quality is currently not systematically included, even though there is agreement that it makes a difference in terms of deforestation. This study contributes to the existing literature by analyzing the explanatory power of an index on environmental institutional quality and assessing whether contributes to reducing the residual calibration factor of the Global Forest Model (cf. figure 2).

While the difference in reduction of the residual calibration factor for Northern and Sub-Saharan Africa (65 vs. 39% for 2000 to 2010, respectively) may be due to the difference in forest covered area (4 vs. 40% on average per country), the difference between Central and South America may be accounted for by the difference in environmental institutional quality (mean EIQ index for 2000 to 2010 of 0.571 and 0.487, for Central and South America, respectively). The implementation of the index into the Global Forest Model represents a further refinement in assessing the process of deforestation. Preliminary testing of the extended model (cf. section 4.3) with yet partially uncomplete data (MacDicken, 2015) already indicates the relevance of the inclusion of the environmental institutional quality index into the model. With more data available over time and further changes in environmental institutional quality at the country, regional, and global level, the significance of model improvement may be further demonstrated.

5.2 Data availability

A growing number of datasets on general institutional quality has been developed over the last two decades (for instance, Gastil, 1990; Gwartney et al., 1996; Johnson and Sheehy, 1995; Kaufmann et al., 2010; Messick, 1996;

World Bank, 2014). However, the time series involved are still short and older datasets have been particularly criticized for endogeneity and ordinality issues (Aron, 2000; Locke, 2013).

There are to date only a few datasets measuring in how far political institutions are capable to guarantee the sustainable management of environmental goods (Sauter, 2012; Brunel and Levinson, 2016). While datasets measuring environmental outcomes, like the concentration of NO_2, or access to drinking water, are increasingly available (such as Yale's Environmental Performance Index (Emerson et al., 2012)), these indicators do not assess whether the environmental outcome is attributable to the quality of the political institutions.

The few existing datasets measuring, for instance, the stringency of environmental policies still faces limitations. Either, because they only provide a single cross-section (cf. the indicator on environmental stringency by Dasgupta et al. (2001)), or because they are not based on discrete units, but use surveys and provide data in ordinal scales, such as the Environmental Policy Index (Hartmann and Reimann, 2010), used in this report as a subcomponent, or the World Economic Forums "Environmental Policy Stringency" indicator (Browne et al., 2012). Furthermore, to the best knowledge of the authors, there is no available index which currently combines environmental policy, the effectiveness of decision making on such policies, and the enforcement of such policies. Even fewer data are available for the quality of forest sector policies and institutions.

Existing initiatives to improve this situation mainly provide nominal data (Global Witness, 2009; WRI, 2015), which is then not comparable in a cross-section analysis, due to nationally determined indicators and measurement standards for each country (cf. (Kishor and Rosenbaum, 2012) or UN-REDD Participatory Governance Assessment, for instance Situmorang (2012)). A first attempt to provide cross-country and time-series data on the quality of forest institutions in discrete units is represented by the FAO's datasets on the amount of people employed in the forest sector, human resources within public forest institutions and forest revenue and public expenditure on forest (FAO, 2010). While these datasets currently contain many missing entries, they are likely to become more reliable and broadly available in the future. Departing from this situation, the environmental institutional quality index constructed in this study builds on existing data on environmental policy and combines them with measures on the effectiveness of decision making and enforcement. The index thus presents an innovation in this regard. However, in the absence of specific data on the quality of forest sector institutions, results have to be treated with caution.

A second limitation of this analysis, concerns the quality of forest cover data provided by the FAO Forest Resources Assessment that have been used for the calibration of the Global Forest Model. Especially for earlier data different authors report heterogeneity in reporting standards, definitions and data collection methods (Angelsen and Kaimowitz, 1999; Bhattarai and Hammig, 2004; Grainger, 1996; Mather and Needle, 1998). Hansen et al (2013) provide deforestation data based on Landsat satellite images and thus presents an alternative. However, it does not distinguish between plantations and natural forests (Tropek et al., 2014) and counts all forms of forest cover change as deforestation. FAO data are therefore still widely used in forest cover change studies.

In summary, the Global Forest Model in the extended version (including the environmental institutional quality index ((G4M global v.4.1)) shows that the forest management strategy for a representative forest planner looks different for countries at different stages of institutional development. This is taken into account by adjusting the value of forestry accordingly through the environmental institutional quality index. Capturing these non-linear effects provides an improved reflection of the complex interaction of environmental institutions, economic pressures on forests, and deforestation. Provided that more and better data become available in the future, the model could simulate the forest conserving effect of environmental institutional quality, independent of economic deforestation drivers. In addition, models addressing processes other than forests (e.g. hydrological models) could benefit from applying the same methodology developed here in order to improve their match between model estimate and observation. Hereby going beyond purely ecological process models even if they follow an economic decision-making rationale - and moving towards more comprehensive socio-ecological process modeling, can further enhance the understanding of environmental degradation processes.

6 Conclusion

Incorporating an indicator on environmental institutional quality into the Global Forest Model significantly reduces the residual calibration factor of the model for the period 2000 to 2010. First tests for the subsequent calibration period (2010-2015) using data from the FAO 2015 Forest Resources Assessment (2015a), show that including the index can contribute to a better forecast of deforestation trends. Future research should concentrate on providing better data on the quality of institutions that are necessary to manage natural resources sustainably. In particular more data would be necessary on the quality of specific institutions in the forest sector. Provided that in future more such data will be made available, the here proposed approach could also be useful for other types of ecological models, for instance hydrological, soil, or fisheries models. Improved data availability would also enable scenarios on different evolutions of the quality of environmental institutions to be constructed to determine their respective impact on the ecological system.

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References

Acemoglu, D. (2008). *Introduction to modern economic growth.* Princeton University Press, Princeton, USA.

Agarwal, C., Green, G. M., Grove, J. M., Evans, T. P., and Schweik, C. M. (2002). *A review and assessment of land-use change models: dynamics of space, time, and human choice.* US Forest Service, Washington D.C., USA.

An, L. (2012). Modeling human decisions in coupled human and natural systems: review of agent-based models. *Ecological Modelling*, 229(2012):25–36.

Angelsen, A. and Kaimowitz, D. (1999). Rethinking the causes of deforestation: lessons from economic models. *The World Bank Research Observer*, 14(1):73–98.

Aoki, M. (2001). *Toward a comparative institutional analysis.* MIT University Press Group Ltd, Cambridge, USA.

Aron, J. (2000). Growth and institutions: a review of the evidence. *The World Bank Research Observer*, 15(1):99–135.

Benítez, P. C., McCallum, I., Obersteiner, M., and Yamagata, Y. (2007). Global potential for carbon sequestration: geographical distribution, country risk and policy implications. *Ecological Economics*, 60(3):572–583.

Bhattarai, M., H. M. (2004). Governance, economic policy, and the environmental Kuznets curve for natural tropical forests. *Environment and Development Economics*, 9(3):367–382.

Bhattarai, M. and Hammig, M. (2001). Institutions and the environmental Kuznets curve for deforestation: a crosscountry analysis for Latin America, Africa and Asia. *World Development*, 29(6):995–1010.

Browne, C.and Geiger, T. and Gutknecht, T. (2012). *The executive opinion survey: the voice of the business community - The Global Competitiveness Report 2012–2013.* World Economic Forum, Geneva, Switzerland.

Brunel, C. and Levinson, A. (2016). Measuring the stringency of environmental regulations. *Review of Environmental Economics and Policy*, 10(1):1–21.

Buitenzorgy, M. and Mol, A. P. (2011). Does democracy lead to a better environment? Deforestation and the democratic transition peak. *Environmental and Resource Economics*, 48(1):59–70.

Busch, J. and Ferretti-Gallon, K. (2017). What drives deforestation and what stops it? A meta-analysis. *Review of Environmental Economics and Policy*, 11(1):3–23.

Cisneros, E., Zhou, S. L., and Börner, J. (2015). Naming and shaming for conservation: evidence from the Brazilian Amazon. *PLOS ONE*, 10(9):e0136402.

Dasgupta, S., Mody, A., Roy, S., and Wheeler, D. (2001). Environmental regulation and development: a cross-country empirical analysis. *Oxford Development Studies*, 29(2):173–187.

DeFries, R. S., Rudel, T., Uriarte, M., and Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience*, 3(3):178–181.

Emerson, J. W., Hsu, A., Levy, M. A., de Sherbinin, A., Mara, V., Esty, D. C., and Jaiteh, M. (2012). *Environmental performance index and pilot trend environmental performance index.* Yale Center for Environmental Law and Policy, New Haven, USA.

FAO (2000). Ecological zones. FAO global forest resources assessment 2000. URL: http://www.fao.org/forestry/country/19971/en/ago/ (accessed 14/11/2015).

FAO (2010). *Global forest resources assessment 2010.* Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

FAO (2011). *Framework for assessing and monitoring forest governance.* Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

FAO (2015a). *Global forest resources assessment 2015.* Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

FAO (2015b). *Global forest resources assessments country reports 2015.* Food and Agriculture Organization of the United Nations (FAO), Rome, Italy.

Fischer, R., Armstrong, A., Shugart, H. H., and Huth, A. (2014). Simulating the impacts of reduced rainfall on carbon stocks and net ecosystem exchange in a tropical forest. *Environmental Modelling & Software*, 52(2014):200–206.

Galinato, G. I. and Galinato, S. P. (2012). The effects of corruption control, political stability and economic growth on deforestation-induced carbon dioxide emissions. *Environment and Development Economics*, 17(1):67–90.

Garcia-Carreras, L. and Parker, D. (2011). How does local tropical deforestation affect rainfall? *Geophysical Research Letters*, 38(L19802):1–12.

Gastil, R. D. (1990). The comparative survey of freedom: experiences and suggestions. *Studies in Comparative International Development*, 25(1):25–50.

Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., and Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proceedings of the National Academy of Sciences*, 107(38):16732–16737.

Gibson, L., Lee, T. M., Koh, L. P., Brook, B. W., Gardner, T. A., Barlow, J., Peres, C. A., Bradshaw, C. J., Laurance, W. F., Lovejoy, T. E., et al. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, 478(7369):378–381.

Global Witness (2009). *Annual transparency report.* Global Witness, Washington D.C., USA.

Grainger, A. (1996). An evaluation of the FAO tropical forest resource assessment, 1990. *Geographical Journal*, 162(1):73–79.

Gusti, M. (2015). *Global Forestry Model (G4M) overview – global version.* Ecosystem Services and Management, International Institute for Applied Systems Analysis, Laxenburg, Austria.

Gusti, M., Havlík, P., and Obersteiner, M. (2008). *Technical description of the IIASA model cluster.* International Institute for Applied Systems Analysis (IIASA), Laxenburg, Austria.

Gusti, M. and Kindermann, G. (2011). An approach to modeling landuse change and forest management on a global scale. In *Proceedings, 1st International Conference on Simulation and Modeling Methodologies, Technologies and Applications (SIMULTECH)*, volume 7–2011, pages 180–185.

Gwartney, J. D., Lawson, R., and Block, W. (1996). *Economic freedom of the world, 1975-1995.* Fraser Institute, Washington D.C., USA.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S., Tyukavina, A., Thau, D., Stehman, S., Goetz, S., Loveland, T., et al. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, 342(6160):850–853.

Hargrave, J. and Kis-Katos, K. (2013). Economic causes of deforestation in the Brazilian Amazon: a panel data analysis for the 2000s. *Environmental and Resource Economics*, 54(4):471–494.

Hartmann, H. and Reimann, S. (2010). *Bertelsmann Transformation Index 2010 - political management in international comparison.* Bertelsmann Foundation, Gütersloh, Germany.

Havlík, P., Schneider, U. A., Schmid, E., Böttcher, H., Fritz, S., Skalský, R., Aoki, K., De Cara, S., Kindermann, G., Kraxner, F., et al. (2011). Global land-use implications of first and second generation biofuel targets. *Energy Policy*, 39(10):5690–5702.

INPE (2017). Projeto Prodes - monitoramento da floresta Amazônica Brasileira por satélite. URL: http://www.obt.inpe.br/prodes/index.php (accessed 23/02/2017).

ISO (2016). Country codes - ISO 3166. International Organization for Standardization. URL: http://www.iso.org (accessed 17/5/2016).

IUCN and UNEP-WCMC (2006). WDPA consortium world database on protected areas. UNEP-world conservation monitoring centre (UNEP-WCMC). URL: www.protectedplanet.net (accessed 16/04/2016).

Janssen, M. A., Walker, B. H., Langridge, J., and Abel, N. (2000). An adaptive agent model for analysing co-evolution of management and policies in a complex rangeland system. *Ecological Modelling*, 131(2):249–268.

Jin, W., He, H. S., and Thompson, F. R. (2016). Are more complex physiological models of forest ecosystems better choices for plot and regional predictions? *Environmental Modelling & Software*, 75(2016):1–14.

Johnson, B. and Sheehy, T. (1995). The index of economic freedom. Heritage Foundation. URL: www.heritage.org (accessed 16/05/2016).

Kaufmann, D., Kraay, A., and Mastruzzi, M. (2011). The worldwide governance indicators: methodology and analytical issues. *Hague Journal on the Rule of Law*, 3(2):220–246.

Kindermann, G., McCallum, I., Fritz, S., and Obersteiner, M. (2008). A global forest growing stock, biomass and carbon map based on FAO statistics. *Silva Fennica*, 42(3):387–396.

Kindermann, G. E., Obersteiner, M., Rametsteiner, E., and McCallum, I. (2006). Predicting the deforestation-trend under different carbon-prices. *Carbon Balance and Management*, 1(15):1–17.

Kishor, N. and Belle, A. (2004). Does improved governance contribute to sustainable forest management? *Journal of Sustainable Forestry*, 19(1-3):55–79.

Kishor, N. and Rosenbaum, K. (2012). *Assessing and monitoring forest governance: a user's guide to a diagnostic tool.* PROFOR, Washington D.C., USA.

Koyuncu, C. and Yilmaz, R. (2013). Deforestation, corruption, and private ownership in the forest sector. *Quality & Quantity*, 47(1):227–236.

Liang, J. and Zhou, M. (2010). A geospatial model of forest dynamics with controlled trend surface. *Ecological Modelling*, 221(19):2339–2352.

Liu, G. and Han, S. (2009). Long-term forest management and timely transfer of carbon into wood products help reduce atmospheric carbon. *Ecological Modelling*, 220(13):1719–1723.

Lobianco, A., Delacote, P., Caurla, S., and Barkaoui, A. (2015). The importance of introducing spatial heterogeneity in bio-economic forest models: insights gleaned from FFSM++. *Ecological Modelling*, 309(2015):82–92.

Locke, A. (2013). *Property rights and development briefing: property rights and economic growth.* Oversee Development Institute, London, United Kingdom.

MacDicken, K. G. (2015). Global forest resources assessment 2015: what, why and how? *Forest Ecology and Management*, 352(2015):3–8.

Magliocca, N. R., Van Vliet, J., Brown, C., Evans, T. P., Houet, T., Messerli, P., Messina, J. P., Nicholas, K. A., Ornetsmüller, C., Sagebiel, J., et al. (2015). From meta-studies to modeling: using synthesis knowledge to build broadly applicable process-based land change models. *Environmental Modelling & Software*, 72(2015):10–20.

Mather, A. and Needle, C. (2000). The relationships of population and forest trends. *The Geographical Journal*, 166(1):2–13.

Mather, A. S. and Needle, C. (1998). The forest transition: a theoretical basis. *Area*, 30(2):117–124.

Mayaux, P., Pekel, J.-F., Desclée, B., Donnay, F., Lupi, A., Achard, F., Clerici, M., Bodart, C., Brink, A., Nasi, R., et al. (2013). State and evolution of the African rainforests between 1990 and 2010. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, 368(20120300):1–10.

Messick, R. (1996). *World survey of economic freedom 1995-1996: a freedom house study*. NJ Transaction Publishers, New Brunswick, USA.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J., da Motta, R. S., et al. (2014). Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, 344(6188):1118–1123.

Perman, R., Ma, Y., McGilvray, J., and Common, M. (2003). *Natural resource and environmental economics*, volume 3. Pearson Education, London, United Kingdom.

Porter, M. E., Delgado, M., Ketels, C., and Stern, S. (2008). *Moving to a new global competitiveness index - The Global Competitiveness Report*, volume 2009. World Economic Forum, Geneva, Switzerland.

Rokityanskiy, D., Benítez, P. C., Kraxner, F., McCallum, I., Obersteiner, M., Rametsteiner, E., and Yamagata, Y. (2007). Geographically explicit global modeling of land-use change, carbon sequestration, and biomass supply. *Technological Forecasting and Social Change*, 74(7):1057–1082.

Sauter, C. (2014). How should we measure environmental policy stringency? URL: www.unine.ch/files/content/sites/irene/files/shared/documents/WP_2014 /wp1401_1.pdf (accessed 2016/06/15).

Situmorang, A., Kartodihardjo, H., Nababan, A., Khatarina, J., Santosa, M., Safitr, M., Soeprihanto, P., Effendi, S., and Solvberg, T. (2012). Participatory governance assessment: the 2012 Indonesia forest, land, and REDD+ governance index. URL: www.undp.org/content/dam/indonesia/docs/envi/PGA%20Report %20English%20Final.pdf (accessed 16/04/2015).

Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E., Haberl, H., Harper, R., House, J., Jafari, M., Masera, O., Mbow, C., Ravindranath, N., Rice, C., Robledo Abad, C., Romanovskaya, A., Sperling, F., and Tubiello, F. (2014). *Agriculture, Forestry and Other Land Use (AFOLU). In: Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, USA.

Smith, P., House, J. I., Bustamante, M., Sobocká, J., Harper, R., Pan, G., West, P. C., Clark, J. M., Adhya, T., Rumpel, C., et al. (2016). Global change pressures on soils from land use and management. *Global Change Biology*, 22(3):1008–1028.

Tollefson, J. (2015). Stopping deforestation: battle for the Amazon. *Nature*, 520(2015):20–23.

Tropek, R., Sedláček, O., Beck, J., Keil, P., Musilová, Z., Šímová, I., and Storch, D. (2014). Comment on high-resolution global maps of 21st-century forest cover change. *Science*, 344(6187):981–981.

Turner, K. G., Anderson, S., Gonzales-Chang, M., Costanza, R., Courville, S., Dalgaard, T., Dominati, E., Kubiszewski, I., Ogilvy, S., Porfirio, L., et al. (2016). A review of methods, data, and models to assess changes in the value of ecosystem services from land degradation and restoration. *Ecological Modelling*, 319(2016):190–207.

Wang, X., Biewald, A., Dietrich, J. P., Schmitz, C., Lotze-Campen, H., Humpenöder, F., Bodirsky, B. L., and Popp, A. (2016). Taking account of governance: implications for land-use dynamics, food prices, and trade patterns. *Ecological Economics*, 122(2016):12–24.

World Bank (2014). Country policy and institutional assessment (No. 88991). URL: http://data.worldbank.org/data-catalog/CPIA (accessed 29/03/2017).

World Bank (2015a). Doing business - measuring business regulations. URL: http://data.worldbank.org/data-catalog/doing-business-database (accessed 15/06/2016).

World Bank (2015b). GDP, purchasing power parity (PPP) (current international US dollar). URL: http://data.worldbank.org/indicator/NY.GDP.MKTP.PP.CD (accessed 15/06/2016).

World Bank (2015c). GNI per capita, purchasing power parity (PPP) (current international). URL: http://data.worldbank.org/indicator/NY.GNP.PCAP.PP.CD (accessed 15/06/2016).

World Bank (2015d). Population density (people per sq. km of land area). URL: http://data.worldbank.org/indicator/EN.POP.DNST (accessed 15/06/2016).

World Resources Institute (2015). Rights to resources map. URL: http://www.wri.org/resources/maps/rights-to-resources (accessed 15/06/2016).

Appendix

Table 4: Index values for the two calibration periods

Country	EIQ index (2000-2010)	EIQ index (2010-2015)
Mauritania	0.3171394	0.3673393
Papua New Guinea	0.397242	0.3200316
Pakistan	0.3721446	0.3878617
Democratic Republic of the Congo	0	0.0730059
Nigeria	0.3254328	0.2786485
Rwanda	0.4775212	0.4751759
Liberia	0.2439197	0.3282564
Ethiopia	0.3739209	0.3852964
Nepal	0.3939269	0.361145
Botswana	0.4354828	0.5439387
Togo	0.2553422	0.2957524
Benin	0.3671045	0.4067261
Burundi	0.3016624	0.348381
El Salvador	0.5285966	0.4975285
Kenya	0.3973252	0.3742531
Lesotho	0.3564456	0.3211012
Burkina Faso	0.3892538	0.3381198
Namibia	0.455623	0.4776162
Republic of the Congo	0.2344352	0.2141114
Senegal	0.4362165	0.5074613
Eritrea	0.3093842	0.1046131
Ghana	0.5171459	0.5322384
Sierra Leone	0.3404256	0.2940403
Cambodia	0.2094748	0.0903175
Cameroon	0.4537116	0.4468637
Sri Lanka	0.5155814	0.5065228
Mali	0.3855268	0.3956202
Chad	0.210749	0.1335523
Zimbabwe	0.2122357	0.1504458
Zambia	0.4673449	0.381339
Singapore	1	0.9567183
Thailand	0.6599068	0.586845
Haiti		0
Nicaragua	0.50863	0.4593461
Honduras	0.4804755	0.4292507
Venezuela	0.26084	0.1356171
Malaysia	0.6873467	0.700047
Myanmar		0.1650243
Laos	0.2643547	0.3594023
Algeria	0.4719794	0.4494916
Indonesia	0.3375796	0.3797591
Tajikistan	0.2592033	0.2863131
Mexico	0.6058961	0.6014391
Bolivia	0.4534022	0.376662
Uganda	0.5104258	0.4256843
Dominican Republic	0.5174142	0.5356953
Jamaica	0.7170074	0.6456125
Bahrain		0.6617238
Czech Republic	0.938825	0.9591271
Slovenia	0.9056307	1
Slovakia	0.8934652	0.8577351
Guatemala	0.4862107	0.3601595
Madagascar	0.5631235	0.4000313
Lebanon	0.509325	0.3970593
Latvia	0.8653687	0.8754419

Table 5: Index values for the two calibration periods (continued)

Country	EIQ index (2000-2010)	EIQ index (2010-2015)
Latvia	0.8653687	0.8754419
Moldova	0.4540012	0.5235101
Belarus	0.5499613	0.6054799
Poland	0.8415589	0.8256844
Panama	0.6270987	0.6235257
Kuwait	0.531165	0.5110277
Bangladesh	0.4167516	0.5327546
Lithuania	0.8814737	0.9223057
Mauritius	0.7789161	0.7735648
Croatia	0.8207038	0.8955733
Azerbaijan	0.4634438	0.6014704
Ecuador	0.4232783	0.4867511
Iran	0.4513364	0.4559049
Armenia	0.6212281	0.6058814
Angola	0.2535988	0.3708744
South Africa	0.66122	0.5967152
Côte d'Ivoire	0.3283491	0.3345847
Turkey	0.6914983	0.6706398
Morocco	0.4739097	0.4572501
Yemen		0.2734554
Hungary	0.959854	0.9515407
Costa Rica	0.6696005	0.6983576
Guinea		0.3402159
Sudan	0.2735806	0.200871
Qatar	0.7174708	0.7755827
Romania	0.7935344	0.7902237
South Korea	0.9171007	0.9463006
Jordan	0.5447595	0.4587204
Bulgaria	0.737747	0.7702487
Ukraine	0.5610575	0.5509776
Malawi	0.3186362	0.253496
Central African Republic	0.2935574	0.2317163
Estonia	0.8754485	0.8977788
United Arab Emirates	0.626616	0.6542469
Colombia	0.4310604	0.4336618
Russia	0.4972755	0.4449241
Kazakhstan	0.6145857	0.5717191
Chile	0.7488309	0.7992648
Argentina	0.6018605	0.5891444
Mozambique	0.3887711	0.408697
Bhutan	0.584792	0.5989641
China	0.5049224	0.4652588
Peru	0.4705113	0.6214923
Uruguay	0.7216804	0.8977788
Paraguay	0.4518384	0.3933052
Brazil	0.3061037	0.3870014
Georgia	0.5749223	0.5707805
Vietnam	0.4604315	0.5547631
Philippines	0.4962905	0.5462224
Oman	0.6586711	0.6646958
Mongolia	0.4352315	0.4145785
Albania	0.5147323	0.5567183
Egypt	0.4931621	0.4721101
Tanzania	0.3990634	0.3361489
Serbia	0.6952302	0.7343814
Saudi Arabia	0.5708622	0.5743626
Bosnia and Herzegovina	0.5386382	0.5842953
India	0.5081084	0.5111216
Macedonia	0.6102987	
Libya	0.6171345	0.5105584
Kyrgyzstan	0.475841	0.417613
Tunisia	0.6642132	0.5517285
Uzbekistan	0.4350021	0.3954238
Montenegro	0.6295896	
Niger	0.3107071	
Taiwan	0.8139259	

Part II

Forest conservation in institutionally weak countries

Chapter 4

Analyzing the perception of deforestation drivers by African policy makers in light of possible REDD+ policy responses[1]

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Analyzing the perception of deforestation drivers by African policy makers in light of possible REDD+ policy responses

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Abstract

As REDD+ countries are moving towards the implementation phase of their national REDD+ programs, it becomes crucial for them to improve their understanding of deforestation drivers in order to determine possible policy responses. As states are the official owners of forests in most African countries, analyzing the perspective of policy makers on deforestation in this context, helps revealing deforestation drivers that are harder to quantify, define, and measure with usual proxies. It also potentially allows identifying politically and institutionally feasible deforestation reduction measures. In this paper content analysis is used to assess how African policy makers perceive deforestation drivers. We find that they strongly emphasize the role of institutional and policy drivers. Furthermore, we find that some of the complex issues related to forest governance in general, can be narrowed down to very specific problems by policy makers. In the light of these findings, we will argue that mechanisms and standards have to be found to allow institutional and policy drivers of deforestation to be addressed in the result-based payments phase of REDD+.

Keywords:Drivers of deforestation, REDD+, institutions, scale of REDD+, content analysis, results-based payments phase

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1 Introduction

With deforestation accounting for around $3.0 \pm 1.1Gt$ CO_2 of global greenhouse gas emissions between 2000 and 2005 (Harris et al., 2012), REDD+ (reducing emissions from deforestation and forest degradation) is a critical policy instrument for climate change mitigation and has been discussed at the international negotiations on climate change since 2005 (UNFCCC, 2005). REDD+ is planned to be rolled out in three phases: (i) the readiness phase, when national technical and institutional capacity is developed, (ii) the implementation phase, when institutions are established, and (iii) the results-based payments phase, when payments for reduced emissions from deforestation are made to tropical countries (Meridian Institute, 2009).

Countries are moving progressively towards the implementation phase of REDD+, with an ongoing reflection on strategies to reduce deforestation based on analyses of its drivers (Aquino and Guay, 2013). Deforestation drivers describe the causes of the removal of trees and the conversion of land to other uses (Van Kooten, G.C. and Bulte, E.H., 2000). In order to be able to identify feasible policy responses to deforestation, it is important to put the analysis of drivers into the perspective of possible policy responses. However, determining the relative importance of different drivers of deforestation is challenging for two reasons in particular.

First of all, measurements of deforestation are still unreliable, especially for many African countries, despite the improvement of access to and quality of satellite images over recent years (Grainger, 2008; Lewis et al., 2009; Rudel, 2013; Williams et al., 2007). The availability of data on explanatory variables is an even greater problem. There are many drivers that remain hard to identify, define, and measure as such, and so finding adequate proxies remains a challenge. Unlike in the energy sector, for instance, it is very hard for most countries to attribute percentage shares of emissions from deforestation to specific drivers. This situation complicates both policy focus and response.

Secondly, because decision-making impacting deforestation occurs on many levels, causal chains can be very long and various different drivers may coexist. The causes of deforestation can therefore take different shapes depending on the chosen perspective.

However, when trying to put the analysis of deforestation drivers into the perspective of a possible policy response, it becomes important to consider the perspective of actors who are in key positions to structurally reduce deforestation. In most African countries, forests are officially owned by the state (Agrawal et al., 2013). While many actors might indirectly drive deforestation, states are at least theoretically in a key position to establish and enforce rules for sustainable resource use (Wehkamp et al., 2013). Taking a closer look at their perception of the problem is a first step towards a better understanding of drivers that are hard to define, quantify, and measure.

In this paper we use content analysis to analyze the perception of deforestation drivers as described by African policy makers in national REDD+ Readiness documents.[1] Section 2 will provide an overview of the existing literature on

[1]Next to many bilateral initiatives (most notably supported by the Norwegian government), two multi-donor programs, the Forest Carbon Partnership Facility managed by the World Bank, and the UN-REDD Programme, provide support to over 54 countries in the REDD+ readiness phase.

deforestation drivers and explain how this study adds to it. Section 3 describes the methodology of the analysis, and section 4 presents the results showing that institutional and policy drivers of deforestation are perceived as key by African policy makers.

2 Literature review

Academic research on deforestation drivers, especially in the African context, has been shaped by pronounced controversies in the past. For instance, while international development banks have supported logging projects, arguing that they allow countries to sustainably harvest tropical forests (Bowles et al., 1998; Wilkie, 1992), others argue that logging is a major cause of deforestation (Laporte et al., 2007; Rice et al., 1997). The role of population growth in deforestation has been similarly disputed, with a number of authors arguing that it is an important predictor of deforestation (DeFries et al., 2010; Jha and Bawa, 2006; Myers, 1993; Pahari and Murai, 1999; Rudel, 1989), and others arguing that population density only drives deforestation, if combined with certain socio-economic factors (Westoby, 1979). Furthermore, while some argue that land tenure rights decrease deforestation (Jaramillo and Kelly, 1997; Mendelsohn, 1994), others find that securing land tenure rights can actually increase deforestation, if the general policy frameworks sets deforestation-increasing incentives (Angelsen, 2007).
More recently and in reaction to an analysis conducted by DeFries et al. (2010), Fisher (2010) highlights that population growth and urbanization alone, do not explain deforestation in the African context. Instead, he identifies subsistence farming and the extraction of fuelwood as major drivers there. Other authors support this view (Boucher et al., 2011; Brink and Eva, 2009; Sanford et al., 2011).
In a meta-analysis Busch and Ferretti-Gallon (2017) find that while the number of peer-reviewed, spatially-explicit publications on deforestation has drastically increased in the last 20 years - from 2 to up to 20 new articles published per year - most of them still reach contradicting conclusions.
This becomes even more pronounced when taking qualitative analysis and case study evidence into account as well, as in a meta-study completed by Geist and Lambin (2001). The authors distinguish between direct and indirect drivers of deforestation and find that next to the above mentioned drivers, infrastructure extension (also supported by (ITTA and FAO, 1999; Laurance, 2009; Reid and Bowles, 1998)), agriculture and wood extraction (Buys et al., 2007; Gibbs et al., 2010; Hosonuma et al., 2012; Mitchard and Flintrop, 2013), soil quality and other biophysical drivers, as well as social trigger events, economic factors (Chidumayo, 1989; Rademaekers et al., 2010; Schueler et al., 2011; Swenson et al., 2011; Von Amsberg, 1998), technological, policy, and institutional factors (Angelsen and Kaimowitz, 1999; Bhattarai and Hammig, 2001, 2004; Buitenzorgy and P J Mol, 2011; Culas, 2007; Galinato and Galinato, 2012; Kissinger et al., 2011; Koyuncu and Yilmaz, 2013; Rametsteiner, 2009) and cultural factors drive deforestation. An overview of their framework is provided in Figure 1.

This reveals that finding a definite and generally acceptable explanatory model of the causes of deforestation is a complex task. There are two notable dimensions of complexity that deserve further attention.

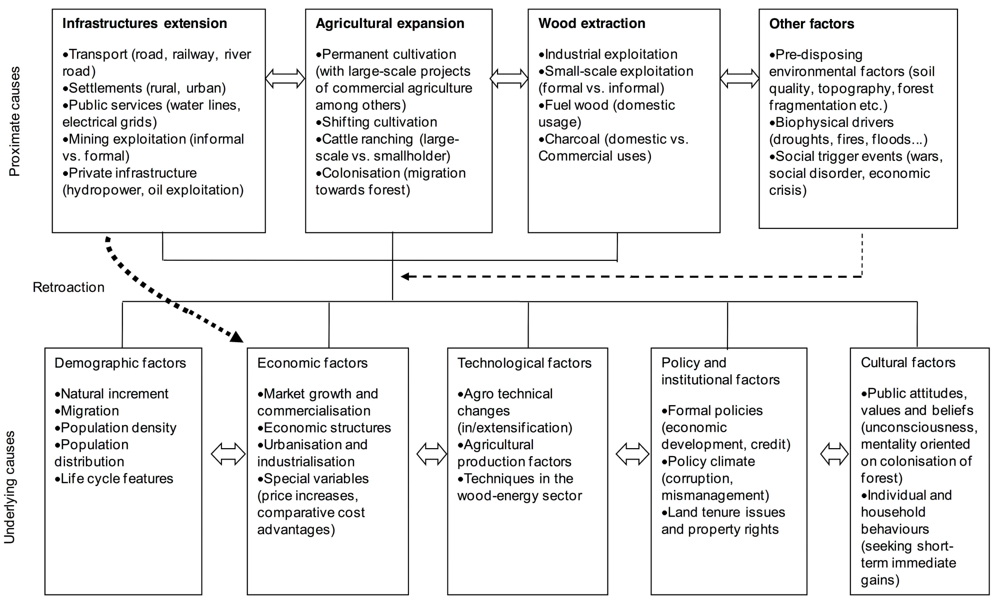

Figure 1: Framework for the analysis of drivers of deforestation provided by Geist and Lambin, 2001 (as cited in the R-PP of the Democratic Republic of the Congo).

First, the availability of verified historical data on deforestation is a problem for many countries (Angelsen and Kaimowitz, 1999; Grainger, 1999; Lewis et al., 2009; Williams et al., 2007).[2] While this situation has generally improved, especially since satellite images have been made freely available (Mayaux et al., 2013; Hansen et al., 2013; Rudel, 2013), remaining problems for many countries are the lack of cloud-free high resolution data and the capacity for verification, which are especially important to understand small scale drivers of deforestation (Malhi et al., 2013) and in order to differentiate plantations from natural forest (Tropek et al., 2014). However, the quality of satellite images of deforestation is constantly improving and therefore data availability should become less of a problem in the future (Casey et al., 2015).
The more important second dimension is however not related to deforestation data as such, but to the measurement of explanatory variables. There are many drivers of deforestation that are poorly understood and beyond qualitative descriptions on a case-study basis very hard to identify, define, and measure. Finding quantifiable proxies is complicated, especially for drivers related to the institutional frameworks, such as the quality of forest laws, or the power to enforce forest conservation. Almost no data exists on these aspects on a cross-country and time series basis.[3] Consequently, these drivers are very hard to integrate into spatially explicit models, which would provide a broader

[2]Some authors also go as far as saying that all studies on deforestation prior to 2000 use ‘unreliable measures of forest loss’ (Grainger, 2008; Rudel, 2013).

[3]Some efforts have been undertaken recently to collect data on forest governance, most notably by the World Resource Institute (2013), Transparency International and Global Witness (2009) and Chatam House (2015). However, their initiatives do not cover all tropical countries, only provide nominal data and are not available for more than 1 year. Furthermore, the FAO has added governance indicators to the Global Forest Resources Assessment in 2010, contain data on forest ownership and management rights, forest management and legal status, the designated functions of forests, employment in the forest sector, human resources within public forest institutions, forest policy and legal framework, forest education and

picture of their relative importance.
Secondly, deforestation is a multidimensional and context-specific phenomenon, which is often reduced to two dimensions (direct and indirect drivers). However, the causal chains resulting in deforestation can be very long. A simple example may illustrate this point: when looking at a farmer cutting a tree in a central African country, the intuitive observation could be that the farmer's need for agricultural land directly drives deforestation. However, when zooming out a little further, we may find that the municipality has problems enforcing the borders of the protected areas, or the national government has decided to construct a road leading into the protected area without providing any support against illegal deforestation to the local government, or that an international commodity trading company has speculated on food prices and therefore impacted the farmer's decision to cut down trees, etc. Thus there could be many explanatory models. In fact, all of them could interact simultaneously, making it difficult to isolate any one of these factors. Therefore, reducing the complexity of deforestation drivers to general concepts, such as agriculture or logging, does not seem to be very useful when trying to put the analysis of the problem into the perspective of a possible policy response.
Our research takes this situation as a starting point and adds to the existing literature by analyzing the perspective of policy makers, revealing the mechanisms that they consider deficient and leading to deforestation in their countries.

3 Method

Content analysis can be used for the quantification of the content of communication (Berelson, 1952). The methodology is particularly useful for the analysis of text sources and captures tendencies or atmospheres in texts. In the absence of comparable quantitative sources of data, content analysis extracts quantified information (in form of frequencies) from qualitative sources. For this paper a content analysis methodology specified by Früh (2007) is used. Seven major steps constitute this analysis: (i) construction of the hypothesis; (ii) selection of sampling material; (iii) development of a category system; (iv) definition of operational units, (v) coding; and, finally, (vi) reliability and validity tests.

3.1 Construction of the hypothesis

In this paper we use content analysis to assess the role of institutional and policy drivers of deforestation as described in national REDD+ documents of African REDD-countries. It assumes that due to state ownership of forest lands, institutional and policy drivers tend to be perceived as playing a preeminent role in African countries. The null hypothesis is that institutional and policy drivers are not mentioned more extensively than other drivers of deforestation. This hypothesis will be tested on a representative sample of national documents.

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research, forest revenue and public expenditure on forestry, and the status of ratification of international conventions and agreements.

3.2 Selection of the sampling material

There are many documents on national REDD+ strategies generated by African countries that could be included in the analysis (country progress sheets, R-PP formulation grant agreements, R-PIN and a variety of additional documentation, etc.(FCPF website, 2013)). However, in order to guarantee that the descriptions of drivers of deforestation are comparable, only two types of documents are selected for the analysis: the national REDD-Readiness Preparation Proposal (R-PPs)[4] and for one country (Zambia) the UN-REDD strategy document. These documents are available for 18 African countries.[5] They are comparable, because they are all national funding requests to the World Bank's Forest Carbon Partnership Facility and the UN-REDD programme. They all date back to the same period (2010 to 2013)[6]and have very similar structures, dedicating one entire section, of comparable length, to the description of drivers of deforestation ((FCPF and UN-REDD, 2012), for more details on the documents see appendix A.1).

The documents are subjective estimates of deforestation drivers by REDD+ policy makers and national policy makers who are directly or indirectly involved into the forest sector. It has to be taken into account that often times national or international consultants are involved into the drafting process of these documents (gathering of information and coordination of the process). There is therefore a risk of the consultants view influencing the outcomes. However, since all documents are the results of consultative processes, and have all been endorsed by national policy-makers, the R-PPs can still be considered as a robust choice for the analysis.

Secondly, policy makers' views on deforestation drivers are likely to be subjective, potentially shaped by personal experiences and impressions. Important drivers of deforestation might therefore be unintentionally or intentionally omitted. Nevertheless, they have a priori been developed in a participatory process involving national experts and are frequently endorsed by different political institutions and a variety of stakeholders. Therefore they still form an adequate basis for our study.[7]

In the absence of a cross-country database on structural drivers of deforestation, these national REDD+ documents are, despite their limitations, an important starting point to better understand these drivers.

3.3 Development of the category system

A category system allows classifying the content of communication and revealing the relative importance of certain aspects. In this context, categories represent a broader type of deforestation driver specified in intermediate and subcategories. The category system relies on a combination of deductive and

[4](Burkina Faso, 2012; Cameroon, 2013; Central African Republic, 2013; DRC, 2010; Ethiopia, 2011; Ghana, 2010; Ivory Coast, 2013; Kenya, 2010; Liberia, 2012; Madagascar, 2013; Mozambique, 2013; Nigeria, 2013; Republic of the Congo, 2010; Sudan, 2013; Tanzania, 2010; Togo, 2013; Uganda, 2011; Zambia, 2010).

[5]status 2013

[6]For all countries using the R-PP format, the context unit is section 2a., which describes drivers of deforestation, for Zambia it is the section on "problem analysis for REDD+ in Zambia".

[7]See section 1.b. of each R-PP and in Zambia's national UN-REDD Programme document, p.27

inductive methodologies as suggested by Früh (2007, p. 153). The broad selection of categories, intermediate and subcategories is based on the framework provided by Geist and Lambin (2001) as discussed in section 2 above. They propose categories as well as more detailed intermediate and subcategories in their framework. Their theoretical framework is then in a second step verified through a small and representative sample (Früh, 2007, p. 157).[8] Two main differences to Geist and Lambin's framework have to be retained: first, they propose a two-dimensional framework (direct and indirect drivers of deforestation). However, these two dimensions cannot always be disentangled. While countries generally all refer to this distinction, they do not refer to it in a comparable and consistent manner. Therefore, this distinction will be ignored in this analysis. Secondly, some subcategories have to be slightly adapted to the context of analysis, with the only significant change being that natural resource extraction (an absent driver in Geist and Lambin's framework) had to be added.

3.4 Definition of operational units

There are four relevant types of operation units in content analysis: the sampling unit, the unit of analysis, the coding unit and context unit (Früh, 2007, p. 164).[9] The sampling units are those elements that are included in an analysis for a specific reason, which are here the 18 national REDD documents. The units of analysis and recording units are often defined identically, as units indicating the absence or presence of the phenomena (Krippendorff, 2004, p. 99). However, Früh (2007, p. 95) proposes a useful distinction between unit of analysis and recording unit, according to which in our context the unit of analysis would be the national strategy document, while the recording units would be the quasi-sentences indicating drivers of deforestation (because we are interested in finding out how many times a country mentions a subcategory). Finally, the context unit includes all the text that has to be taken into account in order to make a specific coding decision on a coding unit. In this context, it is again the whole document.
Having established the definitions of the most relevant units of analysis, quantification can be defined. This will be based on subcategories. As no hierarchical order of drivers of deforestation exists, quantification of all subcategories take a binary form and have nominal values of '1' when identified in a docu-

[8]For this purpose, two coders read five national documents independently and identify all the categories and subcategories (here drivers of deforestation) that can possibly be identified in the documents (list available in appendix A.2). Subsequently, the two lists are compared and a Holsti's reliability coefficient of 0.93, (1969) allows the establishment of the category and subcategory framework based on the results. Krippendorff's alpha (which will be used for the rest of the analysis) could not be used here, since it assesses co-variation, which does not exist in the identification of recording units. If both coders note '0' for its absence then it will simply not be mentioned. Consequently, there is no co-variation and Krippendorff's alpha would give a misleading result. This can be illustrated, when another indicator is used. Holsti's formula for instance, yields a reliability level of 0.94, whereas Krippendorff's alpha (2004, p. 237) only 0.65. A list of the two assessments is available in appendix A.2.

[9]Distinguishing between these different units is relevant, because the sources of the documents often do not correspond to discrete units (like interviews with an individual) and content analysis often tries to assess a phenomena, rather than the attitude of different document types towards a phenomena (Krippendorff, 2004, p. 98).

ment and '0' when considered to be absent.[10] The addition of all subcategories by category and text eventually allows us to assess how frequently different categories of drivers are mentioned in each of the texts, as well as on average.

3.5 Coding

The proposed full list of possible categories and subcategories is tested several times: first with another small selection of documents and later on the entire sample. In a test coding, it is verified, whether there are other categories or subcategories that might have been overseen in the rest of the documents. It is also noted whether subcategories have to be dropped, because they are already included in existing categories or subcategories. Furthermore, a codebook is prepared, describing each type of category and subcategory that allows to clearly distinguish subcategories following Züll's (2010) methodology. If test coders encounter ambiguities, the codebook is revised and adjusted. Coding is carried out by three coders, including two new coders that have not participated in the test coding. In the coding phase, each of them codes half of the documents independently, while the first coder codes all the documents again. Coding the entire sample twice for reliability, allows us to achieving a high degree of certainty for all categories and subcategories (Krippendorff, 2004, p. 239).

3.6 Intercoder reliability and validity

The two independent assessments of the text can now be compared on the basis of Krippendorff's reliability coefficient α (Hayes and Krippendorff, 2007). The coefficient allows the assessment of levels of agreement between the different coding results, where $\alpha = 1$, indicates perfect reliability and $\alpha = 0$ the absence of reliability. Out of the 79 subcategories, 10 subcategories are dropped.[11] Acceptable intercoder reliability can be achieved for 69 subcategories (appendix A.3 and A.4).[12] Krippendorff's alpha ranges between 0.64 and 1 (for detailed results see appendix A.4). Finally, validity is a last robustness criterion, which requires that the findings could at least be theoretically verified. This requirement is fullfilled by this sample, because findings could theoretically be confirmed with interviews.

[10] As explicitly expressed in the majority of all REDD+ strategy documents, none of the analyzed countries has based its assessment on up-to-date data on drivers of deforestation. Therefore, even if a hierarchy of drivers of deforestation was expressed in the document, it has not been weighted in the analysis, assuming that in view of the lack of data it was rather established ad hoc. Consequently, in the analysis, all present drivers of deforestation were considered to have an equal value of '1'.

[11] These were (1) honey production, (2) rural insecurity, (3) limited legal and institutional mechanisms for conflict management in rural areas multiplicity of customary institutions, (4) undervaluation of land, (5) economic crisis, (6) insufficient forest practice, (7) conversion to plantations (energy goods), (8) low competitiveness of standing forests or reforestation, (9) inefficient (wood waste generating) land use techniques, (10) domestic land grabbing/elite monopoly, reasons were: (1) contextual insignificance, (2) α reliability levels below 0.3.

[12] Only 8 subcategories had reliability levels in the range of 0.65 to 0.69, but are considered to be still in the accepted range of reliability, due to the difficulty of applying Krippendorff's alpha to a binary data set. In none of the cases, there were more than two disagreements, but Krippendorff's alpha punishes the absence of co-variation, meaning that if everything is coded '1' by both coders, apart from one variable, Krippendorff's alpha shows low reliability, in spite of actual high agreement (Krippendorff, 2004, p. 237).

4 Results

There are three major findings. First of all, when looking at the results on the level of the categories only, it can be observed that institutional and policy drivers of deforestation are mentioned more frequently than any other type of driver. Secondly, it can be seen that there is a strong agreement across African REDD+ policy makers that institutional and policy issues are perceived to be relevant drivers. Third, within the intermediate of institutional and policy drivers of deforestation the strongest agreement could be observed on the subcategory land rights issues. Finally, when zooming into the subcategory 'institutional and policy drivers of deforestation', it is noticeable that countries identify similar issues as being important to them.

4.1 Institutional and policy issues are mentioned frequently

The process of deductive and inductive construction of categories and subcategories has generated nine categories. Specific drivers of deforestation are attributed to each of these as subcategories. For instance, a subcategory of the category 'infrastructure' is 'private infrastructure projects'. It was possible to identify 24 subcategories for the category 'policy and institutional drivers of deforestation', while for the other eight categories only an average of 7.6 subcategories was identified. This process thus shows a strong presence of institutional and policy drivers of deforestation in the documents.

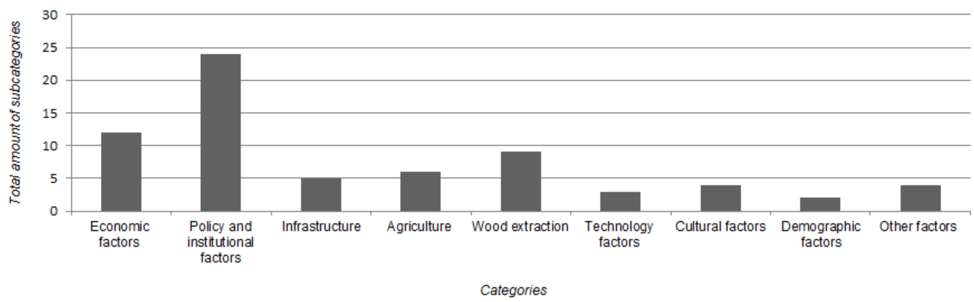

Figure 2: Amount of identified subcategories of drivers of deforestation per categories

4.2 Agreement across African countries on the role of institutional and policy drivers of deforestation

The apparent frequency of institutional and policy drivers of deforestation in the documents could possibly also be linked to the complexity of such drivers, i.e. a larger number of subcategories is needed to describe the problem set. Alternatively, each country might have its own special institutional problems, which could be entirely different from one country to another. This would mean that this type of driver is only described very much in detail, but not that institutional and policy drivers occur with particular frequency. However, on average, countries identified 19.61 out of 24 institutional and policy drivers of deforestation.[13] This shows that there was not only a detailed description of, and frequent reference to such drivers, but also that there is a strong agreement across countries that the majority of institutional and policy drivers of

[13]With a range of 17 to 23 and one out layer of 9.

deforestation seem to apply in the various different African contexts.
For instance, all 18 countries considered that a lack of land-use planning, weak land tenure security, a lack of forest management and illegal logging, drive deforestation in their country. The strength of this agreement is only equivalent to drivers that have been extensively treated and prominently featured in the academic literature, such as agricultural expansion and population growth. This finding is also in agreement with the findings of Williams (2013).
Even when examining the drivers upon which only 16 or 17 out of 18 countries agreed, most of them were still found to be in the institutional and policy category. 17 out of 18 countries agreed that the agricultural and land tenure legal framework, weak governance, weak communities and community rights, as well as weak governance, and a lack of resources and personnel for the implementation of policies, drive deforestation. This quite strong agreement only existed four times in other categories (again in agricultural expansion and wood and natural resource extraction).
Finally, a relatively strong agreement (16 out of 18 countries) could be found on five further policy and institutional subcategories and one agricultural category. The results are summarized in Table 1.

As a next step, it is interesting to investigate the intermediate categories of institutional and policy drivers of deforestation (formal national policy framework, political climate, informal policies and implementation difficulties, and land tenure rights issues). Taking the relative agreement for each intermediate category illustrates that there is no one single issue that monopolizes the agreements, but that there is a rather strong relative agreement for each issue. Furthermore, it shows that the strongest agreement on institutional and policy drivers can be found on land tenure and rights issues. These problems are as much associated with the legal framework itself, as with its enforcement.

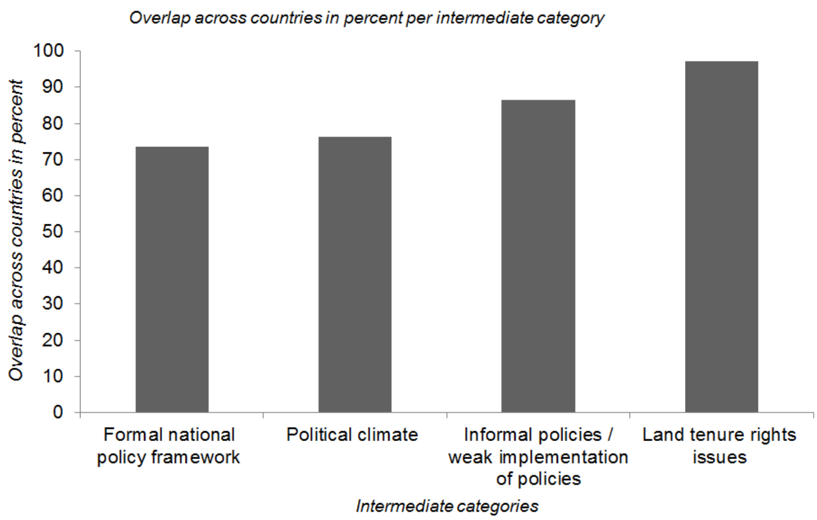

Figure 3: Relative agreement on different intermediate categories from the category policy and institutional drivers of deforestation.

This can be illustrated in more detail, when taking a look at the absolute number of subcategories for each intermediate category identified by country, shown in Figure 4. Apart from Mozambique, where legal and land tenure issues (especially community rights issues) were recently addressed (at least

Table 1: Agreement across countries on drivers of deforestation

Amount of agreements	Subcategory of deforestation driver	Category of deforestation driver
18/18	- Lack of land-use planning	2. Policy and institutional drivers of deforestatio
	- Weak land tenure security - Lack of forest management - Illegal logging	
	- Small scale agriculture	4. Agricultural expansion
	- Small scale logging	5. Wood and natural resource extraction
	- Fuelwood energy products	
	- Population growth	8. Demographic factors
17/18	- Agricultural policy framework driving deforestation	2. Policy and institutional drivers of deforestation
	- Legal framework for land tenure driving deforestation - Weak communities and community rights - Weak governance - Lack of resources and personnel for the implementation of policies	
	- Slash and burn or shifting cultivation	4. Agricultural expansion
	- Large scale / industrial agriculture	
	- Commercial logging	5. Wood and natural resource extraction
	- Charcoal energy products	
16/18	- Forest policy framework driving deforestation	2. Policy and institutional drivers of deforestation
	- Inconsistencies in policy frameworks - Weak law enforcement - Lack of knowledge and capacity in the forestry sector - Insufficient monitoring	
	- Small holder cattle ranching	4. Agricultural expansion

according to the document), all countries have identified all four subcategories related to land tenure issues.

Examining the absolute number of subcategories per category allows identifying tendencies in terms of were the bulk of the problem lies. While some countries (especially Tanzania, Madagascar, and Togo) identified many issues and reform-needs related their legal framework in their R-PP, in the R-PP of the Republic of the Congo only four drivers connected to the policy and legal framework are identified. The R-PP document of Zambia pays the least attention to institutional and policy drivers of deforestation. It is also the only document where issues related to the formal national policy framework and the political climate are not mentioned as drivers.[14]

[14] Again, this does not necessarily reflect the reality of the country, but it does reflect the reality reported in the R-PP documents.

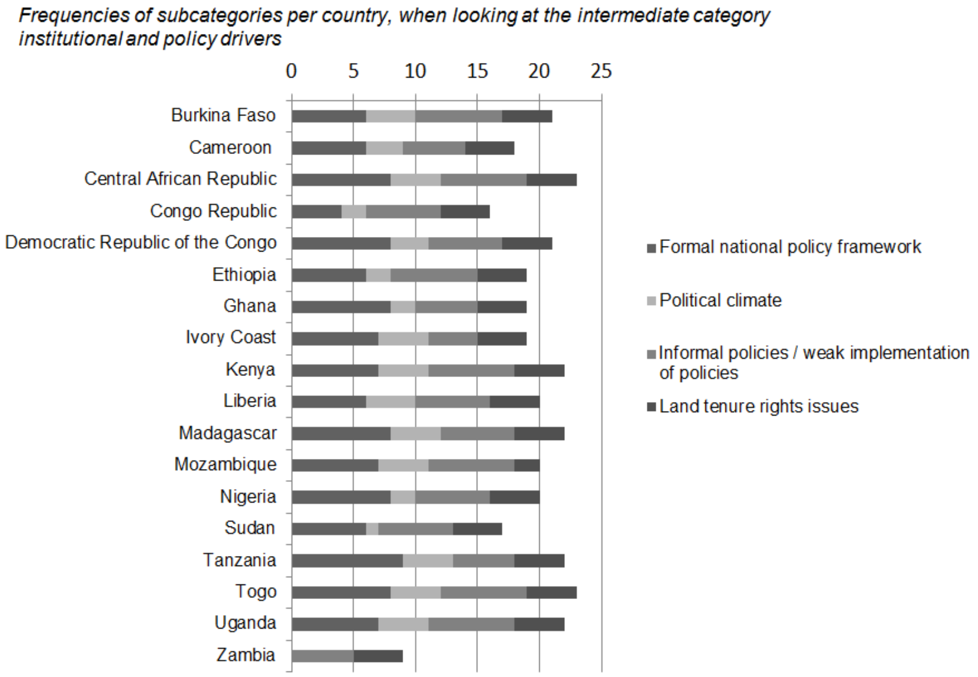

Figure 4: Mapping country emphasis of different aspects of institutional and policy drivers of deforestation (on the ordinate, amount of subcategories identified per country)

4.3 Policy and institutional drivers of deforestation can be very concrete

Through a detailed look at the intermediate and subcategories in Figure 5 some further observations on drivers of deforestation can be made, closing the loop with the initial discussion on how to place the analysis of drivers of deforestation into the perspective of possible REDD+ policy responses. Especially column 3 reveals a list of relatively concrete problems translating into broader governance weaknesses.

Category 2: Policy and institutional drivers of deforestation			
Formal national policy framework	*Political climate*	*Informal policies and implementation*	*Land tenure and rights issues*
- Forest policy framework - Agriculture policy framework - Incentives structures - Industrialization policies (driving industrial agri-business/ deforestation) - Inconsistency in policies or framework - Taxation or subsidies - Lack of benefit sharing from forest resources - Lack of coordination among sectors and policy frameworks - No harmonized decentralization process	- Weak governance - Weak law enforcement - Corruption, illicit tax collection - Lack of accountability of forest sector officials, lack of moral, clientelism	- Lack of knowledge and capacity in forest sector - Lack of resources/personnel for implementation of policies - Lack of scientific information and science-policy link - Lack of forest management - Lack of distribution of information on legal framework to local level - Illegal logging - Insufficient monitoring	- Land tenure legal framework driving deforestation (positive law, e.g. *mise en valeur*) - Weak security of land tenure rights (overlapping legal systems) - Lack of land use planning (rural and urban) - Weak communities and community rights

Figure 5: All subcategories for policy and institutional drivers of deforestation

5 Discussion and conclusion

Figure 5 constitutes a tentative list of problems perceived as driving deforestation that could have the potential to structurally reduce emissions from deforestation systematically. The feasibility of addressing these institutional and policy drivers in the REDD+ context can only be assessed, when taking the respective national and international institutional environments into account.

A fundamental open question, which we would like to discuss in this last part, is therefore whether such issues can and should addressed in the context of REDD+. Some scholars underline the important role of jurisdictions, because they are in a position to address structural drivers of deforestation (Corbera and Schroeder, 2011; Den Besten et al., 2014; Gregersen et al., 2010; Karsenty et al., 2012; Megevand et al., 2013; Umemiya et al., 2010). Umemiya et al. (2010), for instance, underline the link between weak governance and deforestation. Gregersen et al. (2010) stress the importance of well-designed governance schemes as a crucial tool for addressing deforestation. Karsenty et al. (2012) argue that direct incentive payments to land owners alone will be insufficient to address the causes of deforestation and that better regulations are needed. Corbera and Schroeder (2011) emphasize the importance of addressing governance issues at country level in the context of REDD+. Megevand et al. (2013) focus on the Congo Basin and argue that inter-alia, forest-sensitive land-use planning and land tenure rights as well as inter-sectorial policy coordination will be indispensable requirements to curb deforestation in the long term.

Others argue that many institutional and policy drivers of deforestation are too complex to be addressed by a single policy tool such as REDD+ anyways and consequently prefer project-oriented approaches (Hall, 2013; Neeff et al., 2014; Chagas et al., 2011). It is argued that many countries do not yet have the necessary institutional capacities at a national level to effectively reduce deforestation especially in the context of the discussion on the scale of REDD+ (Chagas et al., 2011). Furthermore, it has been argued that focusing too much on such issues in the context of REDD+ might come at the cost of not achieving any real emission reductions (Neeff et al., 2014).

This last part aims to contribute to this discussion by putting the findings into the perspective of (i) the policy context of African countries, (ii) the theory of collective action, and (iii) the challenges that including institutional investments into the international REDD+ policy portfolio in an possible future implementation phase.

An important differentiation can be made concerning the intermediate categories in figure 5: while efforts to protect forests will clearly benefit from progress in all the intermediate categories, some of them might be too complex to be addressed with a single policy tool such as REDD+.

Political climate

The intermediate category 'political climate' (column 2) includes subcategories describing general drawbacks that are likely to hinder effective policy making in almost every area concerning deforestation. The majority of African countries have relatively low scores in assessments of their political climate (Ber-

telsmann, 2014; Bräutigam and Knack, 2004; Ndulu and O'Connell, 1999). Problems such as corruption and weak governance are not specific to the context of forest policy and are unlikely to be addressed through a single policy tool, such as REDD+.
Some also argue that nepotism and corruption distort the effects of policies to such a large extent (Bayart, 2006) that it would seem more logical to aim at influencing legal practice, rather than supporting abstract legal frameworks. Karsenty and Ongolo (2012) question the idea that decision-makers in countries with limited statehood act solely according to welfare maximization principles and consequently criticize the application of the theory of incentives to such contexts. In contrast, a number of policy issues in the first intermediate category 'formal national policy frameworks' (column 1) such as the forest, agricultural, and industrial policy framework, as well as taxation and subsidies, weak coordination and collaboration between different ministries on policy frameworks at a national as well as at a decentralized level, and in the last intermediate category 'land tenure and rights issues' (column 4) can offer policy entry points to reducing deforestation.

Policy planning and policy making - money can't buy reforms

More generally it has been questioned whether externally induced conditionality based policy changes can work at all (Collier, 2000; Collier and Dollar, 2002). In fact there does not seem to be convincing evidence that conditionality has had any impact on actual disbursement (Svensson, 2003). Angelsen (2013) puts this analysis into the REDD+ context and argues that there is a considerable principal-agent problem - between the recipient and the donor countries and that REDD+ lacks an enforcement mechanism. There even seems to be an incentive for the money to be disbursed on the donor country side. Burnside and Dollar (2004) highlight that a major challenge to addressing institutional and policy issues on a conditional payment basis is that the domestic processes to implement reforms are often poorly understood and their complexity underestimated.

Improving coordination across ministries

Improving the coordination and collaboration across ministries and existing policy frameworks is, in many cases, likely to be obstructed by parties that have a vital interest in maintaining the status-quo. For example, in the Democratic Republic of the Congo, different ministries have the right to allocate concessions for different types of land-use (e.g. mining, logging, agriculture) which then often overlap. A common regime for land rights allocation and a resolution mechanism for overlapping land-right claims, bears the risk of significantly reducing the power and the sources of income of individual ministries (Adams et al., 1999; Forsyth, 2009; Lange, 2008; Toulmin, 2009). In Cameroon, Ongolo (2015) observes an interest of forest administration officials in maintaining an incoherent and uncoordinated status-quo situation with respect to the power of different ministries and agreements with international partners. This preserves scope for their personal agendas.

Addressing land tenure rights issues

Land tenure problems are not merely a problem of formal laws, but rather an issue created through legal practice. In many African countries, where forests

are theoretically owned by states, in practice they are managed by communities, or individuals, who have no formalized rights to their land (Sikor and Lund, 2009; Wily, 2008, 2002; Wily et al., 2000). Customary institutions to regulate access to property coexist with official institutions (Faye, 2014). This is most notably the case for community rights (Chomba et al., 2014; Ostrom, 2009). In many African countries formal laws are inherited from colonial times and do not constitute a historically grown rights structure (Hatcher, 2011; Hatcher et al., 2009; Karsenty, 1998). Consequently, the state often settles on practices allocating concessions for different land-use types, rather than exclusive property rights and in general encounters many difficulties in formalizing customary rights (Comby, 1991; Lund, 2002; Ouedraogo, 2011).
A good example is the loose formal recognition of community rights in the Democratic Republic of the Congo (Loi Nr. 011/, 2002) and its weak implementation (Doherty and Schroeder, 2011).[15] Even with clear intentions and initial implementation success, weak enforcement capacity often hinders authorities from following through when faced with unforeseen adverse effects that question the overall legitimacy of the reform project (Damnyag et al., 2012).

While there might be many more practical challenges to addressing institutional and policy drivers of deforestation, it is perhaps necessary to put the question the other way round: can deforestation be sustainably reduced without addressing some of the most burning institutional and policy drivers of deforestation?
The overwhelming attention paid to these issues in national REDD+ strategy documents, as well as the cross-cutting nature of many of these issues, suggests that it will be difficult. While it might be challenging to estimate the exact mitigation potential of addressing institutional and policy drivers of deforestation at the current stage, theory on collective action and governance of common goods underlines their fundamental importance.
In her work on the governance of common-pool resources, Ostrom (1990) argues for instance, that individuals can be assumed to use common-pool resources sustainably, if there are institutions in place allowing them to do so. In the context of deforestation it seems very unlikely that emissions can be sustainably reduced without embedding this idea into an institutional framework that transcends the lifespans of individuals and projects. This is particularly the case, because long-term cooperation for forest conservation requires institutions that provide at least the theoretical opportunity to guarantee it.
No matter how stable a country is and how strong its capacities to enforce laws may be, laws are the fundamental norms (*Grundnorm*) that condition the emergence of other subordinated rules (enforcing decrees, application, and policies) (Kelsen, 1934; Merle, 2007). While many attempts to improve institutional frameworks with external support might have failed in the past, it is important to learn from these experiences rather than accepting that they cannot be changed. Yet, the question still remains, how and to which extent can institutional and policy drivers of deforestation be addressed in the con-

[15] Article 22 of the 2002 forest code foresaw the possibility for a formal recognition of community rights, yet the necessary decree on the modalities of implementation was only issued in 2014.

text of REDD+.

In light of the results on the role of institutional and policy drivers of deforestation, it appears worthwhile to conclude this section with a discussion of the challenges that institutional weaknesses represent to the international policy framework for REDD+.

Concrete opportunities with weak barriers and low costs
A feasibility perspective can be taken in order to identify opportunities for institutional and policy interventions in the framework of REDD+ with weak barriers and low implementation costs. While policies addressing drivers related to the political climate, the general legal framework and also the more specific legal framework on land tenure are likely to face strong resistance to change, REDD+ policy makers have also identified a variety of more concrete issues related to weak implementation of policies and a lack of capacity. As these are part of existing, but poorly functioning institutional frameworks and policies, they are likely to face less opposition than more fundamental reforms (see table 2, column 3) and might have favorable co-benefits. Such issues range from the lack of knowledge and capacity in the forest sector, the lack of resources and personnel for the implementation of policies, the lack of scientific information and a sciences-policy link, missing or poor forest management and distribution of information on the legal framework to the local level, to the limited control of illegal logging and insufficient monitoring.[16] In other words: starting from what was agreed, but is not implemented, could be a promising avenue. Beyond these likely low resistance aspects, it is also imaginable that some more fundamental political changes (such as reforms in the agricultural subsidies or the allocation of land concessions) could become politically feasible if accompanied by investments stimulating the emergence of economic alternatives to deforestation. If REDD+ money could be used to provide capital for these investments, a structural change in the sectors currently still dependent on deforestation could possibly be triggered.

Pricing the costs of investments into political institutions
Once countries identify such more concrete and politically feasible policy options amongst policy and institutional drivers of deforestation, clear strategies can be established. Angelsen (2013) highlights the importance of logical frameworks and long-term impact logic, with measurable indicators for success, when deciding on activities. Once short-, mid- and long-term activities to address such drivers are established, it becomes possible to estimate the associated cost. Experiences with such cost estimates exist, for instance, in the land tenure sector: the Millennium Challenge Corporation (2006) has estimated the costs of securing rural land tenure rights (through participatory cartography) in Benin. Putting a price tag on such institutional and policy drivers of deforestation would allow their integration into the result-based payments logic that constituted much of the initial attractiveness of REDD+. Performance would not necessarily be immediately measured in terms of carbon emission reductions, because many of the activities related to the political framework

[16] Oyono (2004) observed (referring to the decentralization process of forest management in Cameroon) that in many parts of the country, institutions that could guarantee sustainable management are simply non existent.

would only result in carbon emission reductions in the mid-term.

Sovereignty, national ownership, and national coalitions

In contrast to many aid programs, in the REDD+ context tropical countries provide a service to the international community: the reduction of emissions from deforestation. At the same time, institutional and policy drivers of deforestation are often sensitive national issues, and therefore country autonomy is crucial. National sovereignty in the REDD+ strategy development process has been repeatedly underlined in the UNFCCC framework (2013). Furthermore, national ownership of REDD+ processes could be an important way to reduce the principal-agent problem between donor and recipient countries (Svensson, 2003). A good example is Ethiopia, where some experimentation with result-based payments on a jurisdictional level has already taken place. In the state of Oromia in Ethiopia, REDD+ funding is used to promote structural forest conservation though landscape management plans. The program uses a two-phase results-based finance model; in the first phase international donors bear the risk of non-compliance, while in the second phase payments are disbursed based only on emission reductions achieved by the jurisdiction. Furthermore, given that domestic political processes and path dependencies should not be underestimated (Burnside and Dollar, 1997), it might be a promising strategy to try to identify the possible losers to a change in status-quo, and how the benefits of more forest-friendly institutions and policies could also become beneficial to them (Fay et al., 2013). Such an approach might allow path-dependencies to be overcome and create ‘coalitions for change’ (Brockhaus et al., 2014).

Funding opportunities and new performance indicators

Reliable funding opportunities can empower coalitions for change and create incentives for countries to address the institutional and policy drivers of deforestation. International funding could be made available to REDD+ countries, in a first phase, against performance indicators (milestones) of the development of a ‘policy environment’ more favorable to forest conservation. An example might be the reduction of subsidies to commercial agriculture activities in forest areas, accompanied by international financial support for non-deforestation activities (such as sustainable intensification of agriculture, in order to avoid a welfare loss). This type of ‘performance payment’ will not interfere with a country’s sovereignty, as countries will decide themselves which measures to take. However, these performance indicators are certainly more complex than simply measuring changes in carbon emission levels, and require a broadening of the notion of performance in the discussions on REDD+. A crucial element of REDD+, and part of its uniqueness, is that it goes beyond traditional official development assistance (ODA) and requires accountability for agreed-upon results. At the same time methodologies need to be flexible enough to accommodate diverse country circumstances (Neeff et al., 2014). A challenging question is where to set the cursor between sovereignty and conditionality. While showing short-term results in terms of carbon emission reductions can be difficult, other short-term proxy indicators for success in reducing deforestation can be chosen when addressing institutional and policy drivers. This allows progressive payments against results with an evaluation of their effectiveness in terms of emission reductions in the mid-term.

Possibility of financing a spectrum of activities in the international REDD+ framework

In order to maintain the uniqueness of REDD+, the idea of results-based payments has to be flexible enough to embrace a broad spectrum of potential activities, including a mix of indicators evaluating the effective and sustained implementation of policies (Karsenty et al., 2012). Guaranteeing national sovereignty in REDD+ strategy making would also mean allowing tropical countries to choose activities to reduce emissions from deforestation and forest degradation from a spectrum of possible activities. A crucial challenge to the future of REDD+ will be to improve the assessment and understanding of the mitigation potential and costs of different feasible REDD+ policy options at the national level in the absence of any historical experience from which to learn (Fosci, 2013; Mbatu, 2015). Future research should focus on country-specific analysis of the costs and relative mitigation potential of addressing institutional and policy drivers of deforestation, as well as of other possible REDD+ activities.

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References

Adams, M., Sibanda, S., and Turner, S. D. (1999). *Land tenure reform and rural livelihoods in Southern Africa*. Overseas Development Institute, London, United Kingdom.

Agrawal, A., Cashore, B., Hardin, R., Shepherd, G., Benson, C., and Miller, D. (2013). Economic contributions of forests. In *Background paper prepared for the United Nations Forum on Forests*. Cambridge, USA.

Angelsen, A. (2007). Forest cover change in space and time: combining the von Thünen and forest transition theories. *World Bank Policy Research Working Paper*, 2007(4117):1–43.

Angelsen, A. et al. (2013). REDD+ as performance-based aid: general lessons and bilateral agreements of Norway. *UNU-WIDER Working Paper*, 2013(135):1–26.

Angelsen, A. and Kaimowitz, D. (1999). Rethinking the causes of deforestation: lessons from economic models. *The World Bank Research Observer*, 14(1):73–98.

Aquino, A. and Guay, B. (2013). Implementing REDD+ in the Democratic Republic of Congo: an analysis of the emerging national REDD+ governance structure. *Forest Policy and Economics*, 36(2013):71–79.

Bayart, J.-F. (2006). *L'Etat en Afrique: la politique du ventre*. Fayard, Paris, France.

Berelson, B. (1952). *Content analysis in communication research*. The Free Press, New York, USA.

Bertelsmann Stiftung (2014). Bertelsmann Transformation Index (BTI). URL: http://www.bti-project.org/index/(accessed 28/04/2015).

Bhattarai, M. and Hammig, M. (2001). Institutions and the environmental Kuznets curve for deforestation: a crosscountry analysis for Latin America, Africa and Asia. *World Development*, 29(6):995–1010.

Bhattarai, M. and Hammig, M. (2004). Governance, economic policy, and the environmental Kuznets curve for natural tropical forests. *Environment and Development Economics*, 9(3):367–382.

Boucher, D., Elias, P., Lininger, K., May-Tobin, C., Roquemore, S., and Saxon, E. (2011). *The root of the problem: what's driving tropical deforestation today?* Union of Concerned Scientists, Cambridge, USA.

Bowles, I. A., Rice, R. E., Mittermeier, R., and Da Fonseca, G. A. (1998). Logging and tropical forest conservation. *Science*, 280(5371):1899–1900.

Bräutigam, D. A. and Knack, S. (2004). Foreign aid, institutions, and governance in sub-Saharan Africa. *Economic Development and Cultural Change*, 52(2):255–285.

Brink, A. B. and Eva, H. D. (2009). Monitoring 25 years of land cover change dynamics in Africa: a sample based remote sensing approach. *Applied Geography*, 29(4):501–512.

Brockhaus, M., Di Gregorio, M., and Mardiah, S. (2014). Governing the design of national REDD+: an analysis of the power of agency. *Forest Policy and Economics*, 49(2014):23–33.

Buitenzorgy, M. and Mol, A. P. (2011). Does democracy lead to a better environment? Deforestation and the democratic transition peak. *Environmental and Resource Economics*, 48(1):59–70.

Burkina Faso (2012). Readiness preparation plan for REDD+ (R-PP Burkina Faso). URL: https://www.forestcarbonpartnership.org/burkina-faso (accessed 23/03/2015).

Burnside, A. C. and Dollar, D. (2004). Aid, policies, and growth: revisiting the evidence. *World Bank Policy Research Working Paper*, 2004(3251):1–36.

Busch, J. and Ferretti-Gallon, K. (2017). What drives deforestation and what stops it? A meta-analysis. *Review of Environmental Economics and Policy*, 11(1):3–23.

Buys, P. (2007). *At loggerheads? Agricultural expansion, poverty reduction, and environment in the tropical forests*. World Bank Publications, Washington D.C., USA.

Cameroon (2013). Readiness preparation proposal (R-PP) country: Cameroon. https://www.forestcarbonpartnership.org/cameroon (accessed 23/03/2015).

Casey, M. (2015). To save tropical forests, scientists are turning to satellites in space. URL: http://www.cbsnews.com/news/scientist-have-come-up-with-a-way-to-accurately-measure-tropical-forests-from-space/ (accessed 23/03/2015).

Central African Republic (2013). Readiness preparation proposal for the Central African Republic. URL: https://www.forestcarbonpartnership.org/central-african-republic, (accessed 23/03/2015).

Chagas, T., Streck, C., O'Sullivan, R., Olander, J., and Seifert-Granzin, J. (2011). *Nested Approaches to REDD+: an overview of issues and options*. Forest Trends and Climate Focus, Washington D.C., USA.

Chatam House (2015). Illegal logging portal. URL: http://www.illegal-logging.info/ (accessed 23/03/2015).

Chidumayo, E. (1989). Land use, deforestation and reforestation in the Zambian Copperbelt. *Land Degradation & Development*, 1(3):209–216.

Chomba, S., Treue, T., and Sinclair, F. (2015). The political economy of forest entitlements: can community based forest management reduce vulnerability at the forest margin? *Forest Policy and Economics*, 58(2015):37–46.

Collier, P. and Dollar, D. (2002). Aid allocation and poverty reduction. *European Economic Review*, 46(8):1475–1500.

Comby, J. (1990). *L'impossible propriété*. ADEF, Paris, France.

Corbera, E. and Schroeder, H. (2011). Governing and implementing REDD+. *Environmental Science & Policy*, 14(2):89–99.

Culas, R. J. (2007). Deforestation and the environmental Kuznets curve: an institutional perspective. *Ecological Economics*, 61(2):429–437.

Damnyag, L., Saastamoinen, O., Appiah, M., and Pappinen, A. (2012). Role of tenure insecurity in deforestation in Ghana's high forest zone. *Forest Policy and Economics*, 14(1):90–98.

Deacon, R. T. (1994). Deforestation and the rule of law in a cross-section of countries. *Land Economics*, 70(4):414–430.

DeFries, R. S., Rudel, T., Uriarte, M., and Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nature Geoscience*, 3(3):178–181.

Den Besten, J. W., Arts, B., and Verkooijen, P. (2014). The evolution of REDD+: an analysis of discursive-institutional dynamics. *Environmental Science & Policy*, 35(2014):40–48.

Doherty, E. and Schroeder, H. (2011). Forest tenure and multi-level governance in avoiding deforestation under REDD+. *Global Environmental Politics*, 11(4):66–88.

DRC (2002). Loi nr 011/2002 du 29 août 2002, portant code forestier en République Démocratique du Congo. URL: http://leganet.cd (accessed 11/03/2014).

DRC (2010). Readiness plan for REDD 2010 to 2012, R-PP final version Democratic Republic of Congo v.3.1.

Ethiopia (2011). Readiness preparation proposal (R-PP) country submitting the proposal: Federal Democratic Republic of Ethiopia date resubmitted (formal): 25 May 2011. URL: https://www.forestcarbonpartnership.org/ethiopia (accessed 11/03/2014).

Fay, M., Hallegatte, S., and Vogt-Schilb, A. (2013). Green industrial policies: when and how. *World Bank Policy Research Working Paper*, 2013(6677):1–26.

Faye, P. (2015). Choice and power: resistance to technical domination in Senegal's forest decentralization. *Forest Policy and Economics*, 2015(60):19–26.

Fisher, B. (2010). African exception to drivers of deforestation. *Nature Geoscience*, 3(6):375–376.

Forsyth, T. (2009). *Multilevel, multiactor governance in REDD+: participation, integration and coordination*. Center for International Forestry Research (CIFOR), Bogor, Indonesia.

Fosci, M. (2013). Balance sheet in the REDD+: are global estimates measuring the wrong costs? *Ecological Economics*, 89(2013):196–200.

Früh, W. (2011). *Inhaltsanalyse: Theorie und Praxis.* UTB Verlag, Germany.

Galinato, G. I. and Galinato, S. P. (2012). The effects of corruption control, political stability and economic growth on deforestation-induced carbon dioxide emissions. *Environment and Development Economics*, 17(1):67–90.

Geist, H. J. and Lambin, E. F. (2001). What drives tropical deforestation. *LUCC Report series*, 2001(4):1–136.

Ghana (2010). Readiness preparation proposal Ghana. URL: https://www.forestcarbonpartnership.org/ghana (accessed 11/03/2014).

Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., and Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proceedings of the National Academy of Sciences*, 107(38):16732–16737.

Global Witness (2010). *Making the forest sector transparent: annual transparency report 2009.* Global Witness, Washington D.C., USA.

Grainger, A. (1999). Constraints on modelling the deforestation and degradation of tropical open woodlands. *Global Ecology and Biogeography*, 8(3-4):179–190.

Grainger, A. (2008). Difficulties in tracking the long-term global trend in tropical forest area. *Proceedings of the National Academy of Sciences*, 105(2):818–823.

Gregersen, H., El Lakany, H., Karsenty, A., and White, A. (2010). *Does the opportunity cost approach indicate the real cost of REDD+.* Rights and Resources Initiative, Washington D.C., USA.

Hall, R. (2013). *REDD+ and the underlying causes of deforestation and forest degradation.* Global Forest Coalition, Asunción, Paraguay.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S., Tyukavina, A., Thau, D., Stehman, S., Goetz, S., Loveland, T., et al. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, 342(6160):850–853.

Harris, N. L., Brown, S., Hagen, S. C., Saatchi, S. S., Petrova, S., Salas, W., Hansen, M. C., Potapov, P. V., and Lotsch, A. (2012). Baseline map of carbon emissions from deforestation in tropical regions. *Science*, 336(6088):1573–1576.

Hatcher, J. (2009). *Securing tenure rights and reducing emissions from deforestation and degradation (REDD).* Rights and Resources Initiative, Washington D.C., USA.

Hatcher, J., Liddle, M., et al. (2008). *From exclusion to ownership? Challenges and opportunities in advancing forest tenure reform.* Washington D.C., USA.

Hayes, A. F. and Krippendorff, K. (2007). Answering the call for a standard reliability measure for coding data. *Communication Methods and Measures*, 1(1):77–89.

Holsti, O. R. (1969). *Content analysis for the social sciences and humanities.* Addison-Wesley Reading, Boston, USA.

Hosonuma, N., Herold, M., De Sy, V., De Fries, R. S., Brockhaus, M., Verchot, L., Angelsen, A., and Romijn, E. (2012). An assessment of deforestation and forest degradation drivers in developing countries. *Environmental Research Letters*, 7(044009):1–13.

ITTO (1999). *Road Infrastructures in Tropical Rainforests: road to Development or Road to Destruction?* Food and Agriculture Organization of the United Nations, Rome, Italy.

Ivory Coast (2013). Proposition de mesures pour l'état de préparation (R-PP), pays : République de Côte d'Ivoire R-PP version finale. URL: https://www.forestcarbonpartnership.org (accessed 11/03/2014).

Jaramillo, C. F. and Kelly, T. (1997). *Deforestation and property rights in Latin America.* Inter-American Development Bank, Washington D.C., USA.

Jha, S. and Bawa, K. S. (2006). Population growth, human development, and deforestation in biodiversity hotspots. *Conservation Biology*, 20(3):906–912.

Karsenty, A. and Ongolo, S. (2012). Can "fragile states" decide to reduce their deforestation? The inappropriate use of the theory of incentives with respect to the REDD mechanism. *Forest Policy and Economics*, 18(2012):38–45.

Karsenty, A., Tulyasuwan, N., and Ezzine de Blas, D. (2012). *Financing options to support REDD+ activities: based on a review of the literature.* Centre de coopération internationale en recherche agronomique pour le développement, Montpellier, France.

Kelsen, H. (1934). *Reine Rechtslehre Einleitung in Die Rechtswissenschaftliche Problematik.* Number 1. Mohr Siebeck, Tübingen, Germany.

Kenya (2010). Revised REDD readiness preparation proposal for Kenya submitted to the Forest Carbon Partnership Facility. URL: https://www.forestcarbonpartnership.org/kenya-0 (accessed 11/03/2014).

Kissinger, G. and Herold, M. (2012). *Drivers of deforestation and forest degradation - A synthesis report for REDD+ Policymakers.* Oslo, Norway.

Koyuncu, C. and Yilmaz, R. (2013). Deforestation, corruption, and private ownership in the forest sector. *Quality & Quantity*, 47(1):227–236.

Krippendorff, K. (2004). *Content analysis: an introduction to its methodology.* Sage Publications, New York, USA.

Lange, S. (2008). *Land tenure and mining in Tanzania.* Chr. Michelsen Institute, Bergen, Norway.

Laporte, N. T., Stabach, J. A., Grosch, R., Lin, T. S., and Goetz, S. J. (2007). Expansion of industrial logging in central Africa. *Science*, 316(5830):1451–1451.

Laurance, W. (2009). Roads to rainforest ruin. *New Scientist*, 203(2723):24–25.

Lavigne Delville, P. (1998). *Quelles politiques foncières pour l'Afrique rurale? Réconcilier pratiques, légitimité et légalité*. Karthala, Coopération Française, Paris, France.

Lewis, S. L., Lopez-Gonzalez, G., Sonké, B., Affum-Baffoe, K., Baker, T. R., Ojo, L. O., Phillips, O. L., Reitsma, J. M., White, L., Comiskey, J. A., et al. (2009). Increasing carbon storage in intact African tropical forests. *Nature*, 457(7232):1003–1006.

Liberia (2012). Readiness preparation proposal (R-PP) for country: Republic of Liberia approved: Resolution PC/9/2011/2 date: March 19, 2012. URL: https://www.forestcarbonpartnership.org/liberia (accessed 11/03/2014).

Lund, C. and Benjaminsen, T. A. (2003). *Securing land rights in Africa*, volume 14. Taylor & Francis, New York, USA.

Madagascar (2013). Readiness preparation proposal (R-PP) Madagascar v12 08/10/10. URL: https://www.forestcarbonpartnership.org/madagascar (accessed 11/03/2014).

Malhi, Y., Adu-Bredu, S., Asare, R. A., Lewis, S. L., and Mayaux, P. (2013). African rainforests: past, present and future. *Phil. Trans. R. Soc. B*, 368(20120312):1–10.

Mayaux, P., Pekel, J.-F., Desclée, B., Donnay, F., Lupi, A., Achard, F., Clerici, M., Bodart, C., Brink, A., Nasi, R., et al. (2013). State and evolution of the African rainforests between 1990 and 2010. *Philosophical Transactions of the Royal Society of London, Biological Sciences*, 368(20120300):1–10.

Mbatu, R. S. (2015). Domestic and international forest regime nexus in Cameroon: an assessment of the effectiveness of REDD+ policy design strategy in the context of the climate change regime. *Forest Policy and Economics*, 52(2015):46–56.

MCC (2006). Millennium challenge compact between the United States of America acting through the Millennium Challenge Corporation and the government of the Republic of Benin. URL: https://assets.mcc.gov/agreements/081606benincompact.pdf (accessed 23/03/2015).

Megevand, C., Mosnier, A., Hourticq, J., Sanders, K., Doetinchem, N., and Streck, C. (2013). *Deforestation trends in the Congo Basin: reconciling economic growth and forest protection*. World Bank Group, Washington D.C., USA.

Mendelsohn, R. (1994). Property rights and tropical deforestation. *Oxford Economic Papers*, 46(Special Issue on Environmental Economics):750–756.

Meridian Institute (2009). *REDD+ institutional options assessment: developing an efficient, effective, and equitable institutional framework for REDD+ under the UNFCCC.* Meridian Institute, Washington D.C., USA.

Merle, J.-C. (2007). La conception du droit de Hermann Cohen et de Hans Kelsen. *Revue Germanique Internationale*, 2007(6):123–134.

Mitchard, E. T. and Flintrop, C. M. (2013). Woody encroachment and forest degradation in sub-Saharan Africa's woodlands and savannas 1982–2006. *Phil. Trans. R. Soc. B*, 368(20120406):1–7.

Mozambique (2013). Readiness preparation proposal (R-PP) for country: Mozambique submission format: Working draft version 5 (revised). URL: https://www.forestcarbonpartnership.org/mozambique (accessed 11/03/2014).

Myers, N. (1993). Tropical forests: the main deforestation fronts. *Environmental Conservation*, 20(1):9–16.

Ndulu, B. J. and O'Connell, S. A. (1999). Governance and growth in sub-Saharan Africa. *The Journal of Economic Perspectives*, 13(3):41–66.

Neeff, T., Göhler, D., and Ascui, F. (2014). Finding a path for REDD+ between ODA and the CDM. *Climate Policy*, 14(2):149–166.

Nigeria (2013). REDD+ readiness preparation proposal (R-PP) Federal Republic of Nigeria. URL: https://www.forestcarbonpartnership.org/nigeria (accessed 11/03/2014).

Ongolo, S. (2015). On the banality of forest governance fragmentation: exploring "gecko politics" as a bureaucratic behaviour in limited statehood. *Forest Policy and Economics*, 53(2015):12–20.

Ostrom, E. (1990). *Governing the commons: the evolution of institutions for collective action.* Cambridge University Press, Cambridge, USA.

Ostrom, E. (2009). Design principles of robust property-rights institutions: what have we learned? In *Property Rights and Land Policies, K. Gregory Ingram, Yu-Hung Hong, eds.* Lincoln Institute of Land Policy, Cambridge, USA.

Ouedraogo, H. M. (2011). De la connaissance à la reconnaissance des droits fonciers africains endogènes. *Études Rurales*, (1):79–93.

Oyono, P. R. (2004). *Institutional deficit, representation, and decentralized forest mangement in Cameroon: elements of natural resource sociology for social theory and public policy.* World Resources Institute, Washington D.C., USA.

Pahari, K. and Murai, S. (1999). Modelling for prediction of global deforestation based on the growth of human population. *Journal of Photogrammetry and Remote Sensing*, 54(5):317–324.

Rademaekers, K., Eichler, L., Berg, J., Obersteiner, M., and Havlik, P. (2010). *Study on the evolution of some deforestation drivers and their potential impacts on the costs of an avoiding deforestation scheme*. European Commission, Brussels, Belgium.

Rametsteiner, E. (2009). Governance concepts and their application in forest policy initiatives from global to local levels. *Small-scale Forestry*, 8(2):143–158.

Reid, J. W. and Bowles, I. A. (1997). Reducing the impacts of roads on tropical forests. *Environment: Science and Policy for Sustainable Development*, 39(8):10–35.

Republic of the Congo (2010). REDD+ readiness preparation proposal (R-PP) Republic of the Congo. URL: https://www.forestcarbonpartnership.org/republic-congo (accessed 23/03/2015).

Rice, R. E., Gullison, R. E., and Reid, J. W. (1997). Can sustainable management save tropical forests? *Scientific American*, 276(4):44–49.

Rudel, T. K. (1989). Population, development, and tropical deforestation: a cross-national study. *Rural Sociology*, 54(3):327–328.

Rudel, T. K. (2013). The national determinants of deforestation in sub-Saharan Africa. *Phil. Trans. R. Soc. B*, 368(20120405):1–7.

Sanford, T., Boucher, D., Elias, P., Lininger, K., May-Tobin, C., Roquemore, S., Saxon, E., Martin, J., and Mulik, K. (2011). The drivers of tropical deforestation: a comprehensive review. In *Proceeding of the AGU Fall Meeting Abstracts*, volume 1, page 1087. American Geophysical Union, Washington D.C., USA.

Schueler, V., Kuemmerle, T., and Schröder, H. (2011). Impacts of surface gold mining on land use systems in western Ghana. *Ambio*, 40(5):528–539.

Sikor, T. and Lund, C. (2009). Access and property: a question of power and authority. *Development and Change*, 40(1):1–22.

Sudan (2013). Readiness preparation proposal (R-PP) for country: Sudan. URL: https://www.forestcarbonpartnership.org/sudan (accessed 11/03/2014).

Svensson, J. (2003). Why conditional aid does not work and what can be done about it? *Journal of Development Economics*, 70(2):381–402.

Swenson, J. J., Carter, C. E., Domec, J.-C., and Delgado, C. I. (2011). Gold mining in the Peruvian Amazon: global prices, deforestation, and mercury imports. *PloS ONE*, 6(4):e18875.

Tanzania (2013). United Republic of Tanzania final submission to the Forest Carbon Partnership Facility (FCPF): Readiness preparation proposal (R-PP). URL: https://www.forestcarbonpartnership.org/tanzania (accessed 11/03/2014).

Togo (2013). Proposition de mesures pour l'état de préparation R-PP pays : Togo. URL: https://www.forestcarbonpartnership.org/togo (accessed 11/03/2014).

Toulmin, C. (2009). Securing land and property rights in sub-Saharan Africa: the role of local institutions. *Land Use Policy*, 26(1):10–19.

Tropek, R., Sedláček, O., Beck, J., Keil, P., Musilová, Z., Šímová, I., and Storch, D. (2014). Comment on "high-resolution global maps of 21st-century forest cover change". *Science*, 344(6187):981–981.

Uganda (2011). REDD readiness preparation proposal for Uganda. https://www.forestcarbonpartnership.org/uganda, (accessed 11/03/2014).

Umemiya, C., Rametsteiner, E., and Kraxner, F. (2010). Quantifying the impacts of the quality of governance on deforestation. *Environmental Science & Policy*, 13(8):695–701.

UNFCCC (2005). FCCC/CP/2005/online.1 Reducing emissions from deforestation in developing countries: approaches to stimulate action. URL: http://unfccc.int/resource/docs/2005/cop11/eng/online01.pdf (accessed 23/03/2015).

UNFCCC (2013). FCCC/CP/2013/5 Work programme on results based finance. URL: http://unfccc.int/resource/docs/2013/cop19/eng/05.pdf (accessed 23/03/2015).

Van Kooten, G. C. and Bulte, E. H. (2001). *The economics of nature: managing biological assets.* Blackwell, Oxford, United Kingdom.

Von Amsberg, J. (1998). Economic parameters of deforestation. *The World Bank Economic Review*, 12(1):133–153.

Wehkamp, J., Devlin, S., and Williams, C. (2013). *Ownership models for sustainable natural resource governance.* New Economics Foundation Working Paper, London, United Kingdom.

Westoby, J. C. (1979). Forest industries for socio-economic development. *The Commonwealth Forestry Review*, 58(2):107–116.

Wilkie, D. S., Sidle, J. G., and Boundzanga, G. C. (1992). Mechanized logging, market hunting, and a bank loan in Congo. *Conservation Biology*, 6(4):570–580.

Williams, C., Hanan, N., Neff, J., Scholes, R., Berry, J., Denning, A., and Baker, D. (2007). Africa and the global carbon cycle. *Carbon Balance Management*, 2(3):1–13.

Williams, L. G. (2013). Putting the pieces together for good governance of REDD+: an analysis of 32 REDD+ country readiness proposals. *World Resources Institute (WRI), Washington D.C., USA.*

Wily, L. A. (2002). Participatory forest management in Africa: an overview of progress and issues. In *Second international workshop on participatory forestry in Africa. Defining the way forward.*, pages 18–22. Food and Agricultural Organization (FAO), Rome, Italy.

Wily, L. A. (2008). Custom and commonage in Africa rethinking the orthodoxies. *Land Use Policy*, 25(1):43–52.

Wily, L. A. et al. (2000). Land tenure reform and the balance of power in eastern and southern Africa. *Overseas Development Institute and Department for International Development*, Natural Resource Perspectives(58):1–4.

WRI (2013). Rights to resources interactive map. http://www.wri.org/resources/maps/rights-to-resources (accessed 23/03/2015).

Zambia (2010). UNREDD programme document Zambia quick start initiative. URL: http://www.un-redd.org/UNREDDProgramme/CountryActions/zambia/tabid/1029/language/en-US/Default.aspx (accessed 11/03/2014).

Züll, C., Scholz, E., and Schmitt, H. (2010). *Kategorienschema für die Vercodung von Assoziationen mit den politischen Richtungsbegriffen" links" und" rechts"*. GESIS, Leibniz Institute for the Social Sciences, Cologne, Germany.

Appendix

A.1 Information on text sources

See online appendix of publication in Forest Policy and Economics (http://www.sciencedirect.com/science/article/pii/S1389934115300010).

A.2 Level of agreement for identification of variables

See online appendix of publication in Forest Policy and Economics (http://www.sciencedirect.com/science/article/pii/S1389934115300010).

A.3 Results coding (results coder 1, which have been taken as a reference)

See online appendix of publication in Forest Policy and Economics (http://www.sciencedirect.com/science/article/pii/S1389934115300010).

A.4 Results intercoder reliability, calculated with Freelon's software ReCal2.

See online appendix of publication in Forest Policy and Economics (http://www.sciencedirect.com/science/article/pii/S1389934115300010).

Chapter 5

Export tariffs combined with public investments as a forest conservation policy instrument[1]

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[1]Submitted to the journal *Agricultural Economics*.
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Export tariffs combined with public investments as a forest conservation policy instrument*

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Abstract

The forest conservation policy instrument REDD+ (Reducing Emissions from Deforestation and Forest Degradation) is designed to compensate governments of tropical countries for their efforts to conserve forests. Food insecure countries that are economically specialized in the agricultural sector and have weak institutions, are likely to face difficulties to enforce forest conservation. This article explores in how far export tariffs on agricultural goods combined with public investments, could be a forest conservation policy mix in such contexts. We first show empirically that structural constraints to forest conservation policies are particularly pronounced in one third of countries where REDD+ programs are planned to be rolled out. We then develop a two sector competing land use model with a domestic food producing and an exporting agricultural sector. We show that it is possible to combine export tariffs with public investments such that deforestation decreases, while agricultural production levels and food prices remain constant.

JEL classification: O24, Q17, Q23, Q24, Q56

Keywords: Deforestation, REDD+, export tariffs, public investments, two sector competing land use model

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1 Introduction

The international forest conservation program REDD+ (Reducing Emissions from Deforestation and Forest Degradation) aims at compensating governments or jurisdictions of tropical countries for their efforts to preserve tropical forests. In this context, a variety of policy approaches are discussed, ranging from direct cash or non-cash transfers, technical assistance, stricter monitoring and enforcement of conservation, or sustainable agricultural practice programs (Fishbein and Lee, 2015) to fiscal policies, like introducing land taxes (Kalkuhl and Edenhofer, 2017).

Countries like Brazil (Cisneros et al., 2015; Nepstad et al., 2014), or Costa Rica (Andam et al., 2008) have been able to enforce forest conservation, while experiencing economic growth (FAO, 2015; World Bank, 2013). However, in many tropical countries the implementation of forest conservation policies is challenged by the absence of economic alternatives to deforestation-driving agricultural practices (Barbier, 2004) and weak institutions (Deacon, 1994; Barbier et al., 2005). This situation is further complicated by food insecurity (Ericksen et al., 2011). For this particular type of countries, it seems necessary to identify policies that allow to (i) reduce deforestation, while (ii) at least maintaining the pre-policy level of agricultural output, and (iii) keeping prices of food products stable.

In this article we propose to combine export tariffs on agricultural goods with agricultural productivity increasing public investments as a forest conservation policy mix for low and lower income countries[1] that are specialized in the agricultural sector, food insecure, and equipped with weak political institutions. In a stylized facts section we first explain why these factors are likely to challenge currently discussed policy approaches for REDD+ and show that in one third of the countries in which REDD+ programs are planned to be rolled out (UN-REDD, 2015; FCPF, 2015)[2] these structural constraints are particularly pronounced. Second, we develop an analytical model, which allows to examine the effects of the proposed policy mix on (i) land demand (deforestation), (ii) agricultural output levels, and (iii) price levels of agricultural commodities.

The hypothesis that the proposed combination of export tariffs with public investments could allow to achieve these multiple policy objectives follows from two main insights from economic literature.

First, economic theory suggests that when environmental resources like forests are not adequately conserved, opening to trade increases the market for the exploitation of the resource and thus environmental degradation (Copeland and Taylor, 2004). Empirical evidence shows that trade liberalization has indeed increased deforestation rates in the past (Barbier, 2000; Pacheco, 2006; Shandra et al., 2009). Theoretical (Bernhofen, 1997; Rodrik, 1989) and empirical (Solberg et al., 2010; Goodland and Daly, 1996) economic literature finds that export tariffs on unprocessed commodities, can stimulate the structural transformation of an economy. They can also represent a source of public revenue (Bouët and Laborde, 2010). Furthermore, Skinner et al. (1991) and

[1] According to the World Bank (2014a) income classification that we will refer to throughout this study, low and lower middle income countries are defined as countries with a GNI per capita below 4,125 US$.

[2] Countries that form part of the UN-REDD or FCPF program as of December 2015

Younger et al. (1999), show that implementing export tariffs is feasible in countries with weak political institutions, which has also been demonstrated by the increase in export tariffs in least developed countries during the 2007 food price crisis (Kim, 2010). Besides, export tariffs are one of the few explicitly tolerated trade policy instruments under WTO rules.[3]

Second, public investments can contribute to increase agricultural productivity (Craig et al., 1997). Such investments can be financed through tax-revenue recycling. In line with this logic, Jones and O'Neill (1994) have proposed to use tax revenues for deforestation-reducing and economically stimulating public investments. In the context of this study public investments are conceived as publicly provided services that lead to productivity gains and thus intensification in the agricultural sector. Examples for such productivity gains are electrification (Assunção et al., 2015), or the allocation of land tenure rights (Mendelsohn, 1994; Robinson et al., 2014). In Sub-Saharan African countries for example, the national electrification rate is very low with just 35% according to (IEA, 2016). Investments into electrification could reduce the need for land when electricity allows using electric pumps for irrigation, thus making more intensive land use possible. Alternatively, governments could also provide effective property rights on land. Abdulai et al. (2011) have for example shown that improved land rights increase the efficiency of land use through better investment incentives. Public investment induced agricultural intensification can have two simultaneous effects on land demand and thus deforestation (Villoria et al., 2014). On the one hand, productivity improvements can entail an increase in natural resource demand - an effect that is commonly referred to as the Jevons-effect (1866). Byerlee et al. (2014) find that this effect particularly dominates in cases of market-driven intensification. On the other hand, agricultural intensification can also imply a more efficient (and thus land-sparing) use of the natural resources, which is commonly referred to as a Borlaug effect (2007). In our model both effects can occur and in line with Hertel (2012) we show that the dominating effect depends on the elasticity of demand for the respective agricultural product.

We use a two sector model, in order to capture both, the effect of export tariffs, as well as the two simultaneous effects of public investments on land demand, output levels and price levels. The first sector is a domestically operating food producing sector, which satisfies an inelastic demand of the domestic population. The second sector is an international exporting agricultural sector. It produces crops like palm oil, or soybeans for the international market and is faced with an elastic international market demand. The model allows illustrating the interaction of these two types of agricultural producers.

The model shows that using the proposed policy mix, allows to limit deforestation without reducing output levels both in the exporting and domestic sector, and without increases in domestic food prices. In addition, we find that the policy package can be partly self-funding through the tariff revenues. The level of international REDD+ payments, that would be necessary for the policy to work, thus depends on the amount of export tariff-revenues and the corresponding availability of resources for public investments in the respective country.

We start by presenting stylzed facts. We then introduce the model and show that different assumptions on the elasticity of demand, corresponding

[3]See GATT rule article 2, 11.1 and 11.2 (WTO, 1947)

to different sectors, lead to different effects of public investments. Next we analyze the effect of export tariffs and public investments simultaneously. In section 5 we show that export tariffs and investments in agricultural productivity can be combined in a way that reduces deforestation and keeps agricultural prices as well as agricultural production stable. We conduct a numerical estimation of the costs of the policy for the international REDD+ donor in section 6. Finally, we discuss our findings and put them into a policy perspective in section 7. We end with a short conclusion.

2 Stylized facts

In this section we present stylized facts, showing that (i) an economic specialization in the agricultural sector, (ii) food insecurity, and (iii) weak political institutions, are likely to function as structural constraints to forest conservation. We further show that all three structural constraints are particularly pronounced in one third of the countries where REDD+ programs are planned to be rolled out.

2.1 Structural constraints to forest conservation policies

First of all, figure 1 illustrates how in countries with the lowest GNI per capita (black line), the contribution of the agricultural sector to GDP (black dashed line) is largest. Furthermore, it shows that the average contribution of the manufacturing sector to GDP is rather low, in most low and lower middle income countries (dark grey in figure 1).

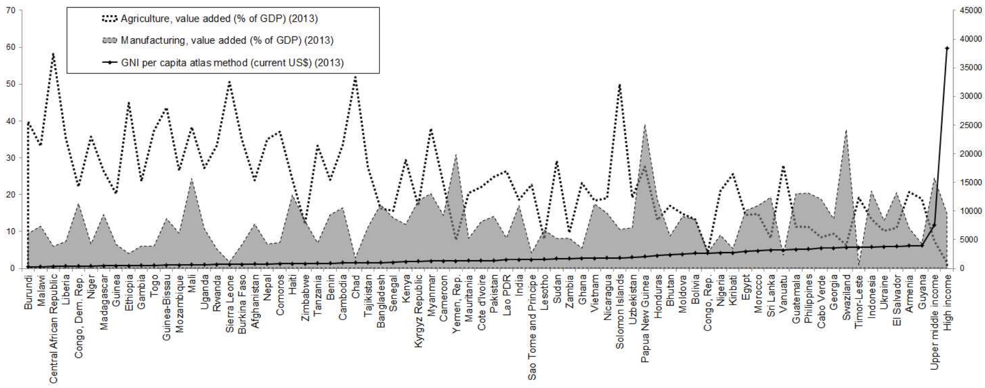

Figure 1: Agriculture and manufacturing value added to GDP and GNI per capita for low and lower middle income countries in 2013. Left axis: Relative value added of agriculture in percent (black dashed line, main source WDI, completed with CIA and AEO data) (World Bank, 2013; African Development Bank, 2013; Central Intelligence Agency, 2013) and manufacturing (dark grey, main source WDI, completed with CIA and AEO data)) in 2014 in percent (based on ISIC divisions 15-37 if available and else on ISIC divisions 10-45 (World Bank, 2013; African Development Bank, 2013; Central Intelligence Agency, 2013)). Right axis: GNI per capita (black) in 2013 (World Bank, 2013). The income groups according to the World Bank classification (World Bank, 2014a) are provided for comparison.

Kongsamut et al (2001) refer to these facts as 'Kuznets facts' and explain them with a model in which the fraction of income that a household spends on agricultural goods declines, once a certain minimum subsistence income level is reached, which triggers the diversification of the economy. From that observation it would follow that it is more likely that an economy begins to diversify after the subsistence level is reached. For countries with more diversified economies and technologically more advanced agricultural sectors it is easier to enforce forest conservation without foregoing economic growth opportunities, because the relative dependence on land as an input factor to production decreases. For example, in spite of the stringent forest conservation enforcement phase and thus policy-induced agricultural land scarcity from 2004-2008 in Brazil, GDP growth rates were high, ranging from 3.14% to 6% (World Bank, 2013) and no decline in agricultural output levels could be observed (Macedo et al., 2012).

Second, figure 2 shows that particularly in low and lower middle income countries the agricultural sector represents a large contribution to exports. On average agricultural raw materials constitute 11% of merchandised exports in low and 5% in lower middle income countries, ranging up to 49.13% for Benin, 48.49% for the Solomon Islands and 46.04% for the Central African Republic (World Bank, 2013). At the same time, figure 2 also shows that the percentage prevalence of undernourishment is higher in low (24%) and lower middle (13%) income countries (FAO, 2013). In contrast, upper middle and high income countries have both a relatively small contribution of agricultural raw materials to their exports and have a very low prevalence of undernourishment.[4]

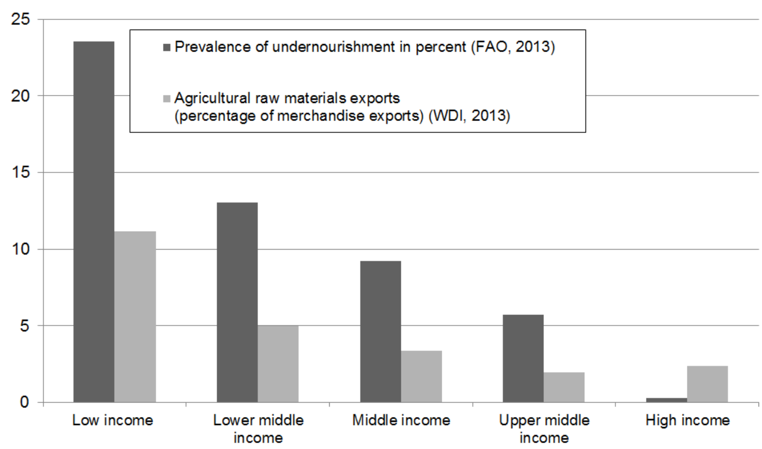

Figure 2: Relative contribution of agricultural raw materials to merchandised exports and relative prevalence of undernourishment in percent. Left axis: Prevalence of undernourishment in dark grey (FAO, 2013) and agricultural raw materials as a contribution to exports in light grey (World Bank, 2014b), respectively for low, lower middle middle, upper middle and high income countries (World Bank, 2014a).

Henson et al (2000) explain this phenomenon by a specialization in export-oriented agricultural supply chains, reliant on high-value markets in developed countries and simultaneously uncoupled and undersupplied local markets. The parallelity of the two types of markets for agricultural commodities creates a high dependence on the land demanding agricultural sector, and vulnerability to changes in local food supplies for the domestic population. This situation thus represents an obstacle to forest conservation, because increasing protected areas - especially in regions with high population density - translates into land

[4]FAO does not provide data for all countries for which the percentage of prevalence of food insecurity is below 5 %(FAO, 2013), therefore we set the value for all of these countries equal to 0 as a default value.

scarcity for agricultural goods and consequently public resistance to conservation (Brockington et al., 2006; Pullin et al., 2013; Oldekop et al., 2016).

Finally, low income countries have also the weakest scores in the quality of political institutions. They have an average score of -0.83 in rule of law and of -0.84 in control of corruption on a scale from -2.5 to 2.5 (World Bank, 2014b). In contrast, high income countries have a positive average score of 1.01 in control of corruption and 1.06 in rule of law. Acemoglu (2005), finds that the weak quality of political institutions is a central reason for diverging patterns in long run economic growth in itself. Weak institutions cause countries to be economically locked into sectors that do not require complex contracting institutions (Nunn and Trefler, 2013), such as typically agriculture. Therefore weak institutions also present a structural impediment to economic diversification towards a less land-demanding economy.

Furthermore, a growing body of empirical literature shows that the quality of political institutions is also directly a central structural parameter to forest conservation. Key elements regarding the quality of political institutions, such as the strength of rule of law (Corderí Novoa, 2008), reliable land tenure rights (Arcand et al., 2008; Bohn and Deacon, 2000), or the absence of corruption (Koyuncu and Yilmaz, 2009) significantly impact, whether a country is likely to be able to conserve its forests or not. It has also been argued that the quality of political institutions can fundamentally undermine the functioning of results-based payments schemes like REDD+. Karsenty and Ongolo (2012) argue that incentive payment-based forest conservation programs are especially likely to confront difficulties in fragile states. Angelsen (2013) points out that there is a high risk of embezzlement of funds in recipient countries with weak institutions, but at the same time little incentive for donating countries to control and potentially sanction the mismanagement of REDD+ funds. This situation can thus undermine the effectiveness of policy instruments in the REDD+ context.

To summarize, effective forest conservation is complicated in economies with a strong specialization in the agricultural sector, because there are less economic alternatives to deforestation driving agricultural practices. It is also challenged by food insecurity, because the population is very vulnerable to conservation induced land scarcity. Finally, it is difficult in countries that lack the political institutions to enforce forest conservation.

2.2 Structural constraints coincide in potential REDD+ countries

We run a Spearman's correlation to assess the relationship between the percentage contribution of agricultural products to GDP (200 observations), rule of law (181 observation) and the prevalence of undernourishment (155 observations). There is a negative and significant correlation between the agricultural share of GDP and rule of law of $\rho = -0.63$, a positive and significant correlation between the agricultural share of GDP and prevalence of undernourishment of ρ=0.62, and a negative and significant correlation between rule of law and prevalence of undernourishment of ρ=-0.6.

It thus shows that a country with a weak rule of law score in our sample, tends to have a larger share of agricultural products as a contribution to GDP and that the percentage share of undernourrished people in this context tends to be larger.

Furthermore, using a three-dimensional scatterplot (figure 3) shows that (i) a relatively large contribution of the agricultural sector to GDP (displayed in percent on the x-axis (World Bank, 2013)), (ii) weak rule of law scores (displayed on the y-axis using the original scale from -2.5 to 2.5 (World Bank, 2014b)), and (iii) the prevalence of undernourishment (displayed in percent on the z-axis (FAO, 2013)[5]), are likely to coincide in countries in which REDD+ programs (highlighted by light grey points) are planned to be rolled out.

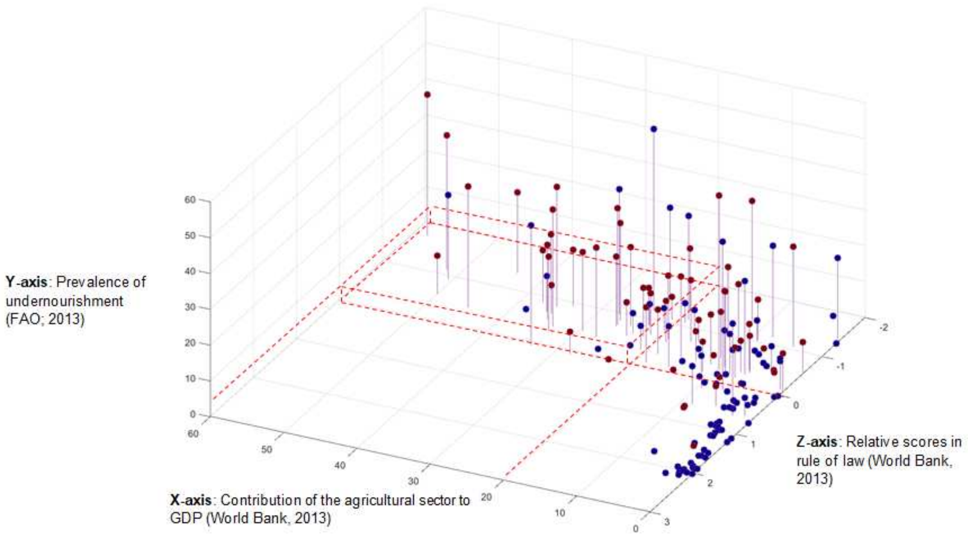

Figure 3: X-axis: contribution of the agricultural sector to GDP (World Bank, 2013), y-axis: prevalence of undernourishment (FAO, 2013), and z-axis: relative scores in rule of law on a scale from -2.5 to 2.5 (World Bank, 2014b). The red dots indicate countries in which REDD+ programs are planned to be rolled out. Note that insufficient data is available for South Sudan and the Solomon Islands and thus they are not plotted in the graph.

Figure 3 shows that REDD+ countries tend to be located in the upper left part of the coordinate. REDD+ countries have an average share of 20% of agricultural products as a contribution to their GDP, prevalence of food insecurity among 15% of their populations, and an average rule of law score of -0.55.

Finally, it is useful to understand in which countries all of the three problems are particularly pronounced to understand what it means for forest conservation. When looking particularly at low and lower middle income countries in which the agricultural sector is larger than the manufacturing sector, the prevalence of undernourishment is above 5% and the rule of law score below -0.54 (average value of lower middle income countries), we identify 36 countries. These countries host 474.7 Mha of forest, which corresponds to 12% of the world's forests, with 168 Mha of primary forest, corresponding to 13% of the world's remaining primary forests (FAO, 2015) (see figure 4 for a list of these countries).

At the same time, as figure 4 shows most of the countries experience deforestation, with an average forest cover loss of 3% over the 2010 to 2015

[5]The FAO does not provide data for countries with scores below 5%, but indicates for which countries this is the case. We set values for those countries equal to zero.

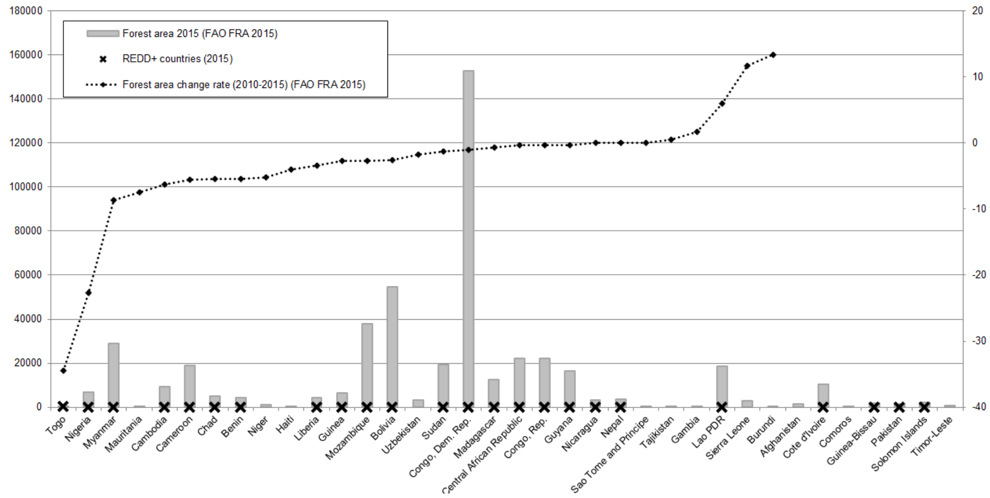

Figure 4: Forest shares for agrarian low income countries with rule of law scores below -0.5 (corresponding to the average value of lower middle income countries). Left axis: Forest area (gray) in ha (FAO, 2015). Right axis: Forest area change (dashed line) in 2015 as a percentage of the forest area in 2010, for Afghanistan, Cote d'Ivoire, Comoros, Guinea-Bissau, Pakistan, Solomon Islands and Timor-Leste no forest area change data is available (FAO, 2015). The black crosses indicate REDD+ countries.

period (FAO, 2015). Togo and Nigeria experienced particularly high deforestation rates with 34% and 22% respectively over the 2010 to 2015 period (FAO, 2015). Figure 4 illustrates the extent of forest cover, as well as the forest area change rates between 2010 and 2015 for the countries in which all of the three problems are particularly pronounced. Only 5 out of 36 of these countries have experienced forest cover gain (Tajikistan (0.5%), Gambia (2%), Laos (6%), Sierra Leone (11%) and Burundi (13%). Out of the 36 countries in which the problems are particularly pronounced, there are 24 countries in which REDD+ programs are planned to be rolled out. In total REDD+ programs are planned to be rolled out in 70 countries, meaning that countries in which the structural constraints to forest conservation are particularly articulated represent one third of all REDD+ countries.

3 The model

We use a competing land use model in order to simulate the decision of a representative farmer. Total land A contains natural forest land W and agricultural land L. Agriculture is one of the most important drivers of deforestation (Gibbs et al., 2010; Busch and Ferretti-Gallon, 2017). We assume that higher demand for agricultural land causes an expansion of agricultural land and simultaneous deforestation. As a consequence, the amount of agricultural land is a function of the price of agricultural land r_L. We model this with a land supply function,

$$L = r_L^{\varepsilon} \,, \tag{1}$$

where $\varepsilon \geq 0$ is the supply elasticity of land. An expansion of agricultural land implies a reduction of forest land, $A - L = W$. Hence, the growth rate of forest land g_W is the negative of the growth rate of agricultural land,

$$g_W = -g_L = -\varepsilon g_{r_L} \; . \tag{2}$$

The agricultural sector uses government infrastructure and institutions G, capital K and land L as inputs for production,

$$F(G, K, L) = G^{\alpha} K^{\beta} L^{\gamma}. \tag{3}$$

We assume $\beta + \gamma = 1$, $\alpha > 0$, $\beta > 0$ and $\gamma > 0$. Capital investment and land use are chosen by the representative land owner. Capital can be rented at an exogenous interest rate r_K. We consider this a realistic assumption since Karlan (2014) finds that farmers in developing countries are typically not liquidity constrained (and lack access to insurance instead). The price for land r_L is endogenous. The level of public investment G is decided by the government. Note that agricultural producers modeled this way never make negative profits, since they have constant returns to scale and land prices are endogenous.

This production function follows a typical approach (Barro, 1990; Barro and Sala-i Martin, 1992; Turnovsky, 2000) when representing returns to scale. There are increasing returns to scale, when all production factors are taken together. This means that for the resource constrained domestic government there is an optimal amount of public investment (G), above which it is not optimal to decrease private investment further through taxation. However, there is no absolute optimum. Additional investments in G, financed by outside sources, would increase production.

The government collects a tariff on the export of agricultural products τ. The representative farmer chooses K and L to maximize profits and takes the government's tariff τ and the level of price p as given,

$$\max_{K,L} (1-\tau) p (G^{\alpha} K^{\beta} L^{\gamma}) - r_K K - r_L L \; . \tag{4}$$

The representative farmer's maximization problem (4) results in the following first order conditions,

$$\begin{aligned} \frac{\partial \mathcal{L}}{\partial K} &= (1-\tau) p \beta (G^{\alpha} K^{\beta-1} L^{\gamma}) - r_K = 0 \; , && (5) \\ \frac{\partial \mathcal{L}}{\partial L} &= (1-\tau) p \gamma (G^{\alpha} K^{\beta} L^{\gamma-1}) - r_L = 0 \; . && (6) \end{aligned}$$

We represent demand for agricultural products with a constant elasticity of demand function,

$$Y = p^{-\theta} \; , \tag{7}$$

where $\theta \geq 0$ is the demand elasticity.

In section 4 we consider the effect of agricultural productivity on deforestation for variations of the elasticity of demand, θ. In section 5 we let a domestic food sector with low θ compete for land with an export sector with high θ. This framework will allow us to study the effect of agricultural productivity on several key agricultural variables like deforestation, food prices, and total export sales.

4 The role of the elasticity of demand

In our model, productivity-enhancing public investment can have both, a deforestation increasing a decreasing effect. We capture these two effects through different assumptions on the elasticity of demand in the domestic sector (inelastic demand) and the internationally exporting sector (elastic demand).

In order to understand the interaction of government investments and the demand elasticity for agricultural products, we first analyze a general case.

Proposition 1 *The amount of land demanded for agriculture is given by*

$$L = \left(\gamma^{\beta(-\frac{1}{\theta}+1)-1} (1-\tau)^{\frac{1}{\theta}-1} G^{\alpha(\frac{1}{\theta}-1)} \left(\frac{r_K}{\beta}\right)^{\beta(-\frac{1}{\theta}+1)} \right)^{\frac{\varepsilon}{-\beta(\frac{1}{\theta}-1)-1-\varepsilon\frac{1}{\theta}}} . \tag{8}$$

An increase in public investments G increases the amount of deforestation if and only if $\theta > 1$.

Proof

We solve food demand (7) for p and land supply (1) for r_L and insert the expressions into the first order conditions (5) and (6). We then solve equation (5) for K and use it to substitute for K in equation (6) and solve for L. □

This proposition shows that an increase in agricultural productivity through public investments can increase the amount of agricultural land at the expense of forests or increase it. There are two effects at work: First of all, for a given amount of production, less inputs, including land, are required. At the same time, the output can be produced at a lower price, so that production can be scaled up. Scaling up production requires to purchase more inputs. When demand is inelastic ($\theta < 1$), the first effect dominates. When demand is elastic ($\theta > 1$), the second effect dominates.

In addition to the case with general demand elasticity, we consider two special cases. In the first special case, we assume that demand is perfectly inelastic $\theta_1 = 0$. It represents a staple food producing sector that satisfies a limited local demand. An example for this could be staple foods consumed by the local population.

Corollary 1 *When demand is perfectly inelastic, public investments decrease deforestation.*

Proof

Using

$$\lim_{\theta \to 0} \frac{\varepsilon((\frac{1}{\theta}-1))}{-\beta\frac{1}{\theta}+\beta-1-\varepsilon\frac{1}{\theta}} = \frac{\varepsilon}{-\beta-\varepsilon} \tag{9}$$

we obtain

$$L_1 = \lim_{\theta \to 0} L = \left(\gamma^{-\beta} (1-\tau) G^{\alpha} \left(\frac{r_K}{\beta}\right)^{-\beta} \right)^{\frac{\varepsilon}{-\beta-\varepsilon}} . \tag{10}$$

From $\frac{\varepsilon}{-\beta-\varepsilon} < 0$ we obtain $\frac{dL_1}{dG} < 0$. □

The food sector with inelastic demand, thus reflects the Borlaug hypothesis (Borlaug, 2007; Meyfroidt and Lambin, 2008; Stevenson et al., 2013; Cohn et al., 2014), which postulates that increased agricultural productivity reduces deforestation. The Borlaug effect is caused by higher productivity (through more public investments in our case) that allows farmers to produce the same amount of food with less land.

In the second special case, we assume that demand is perfectly elastic, $\theta_2 = \infty$. This assumption applies to the sector that exports agricultural products like palm oil, soybeans, coffee, or cotton to the international market. We assume that production is taking place in a small open economy, such that changes in domestic price levels do not influence international market prices.

Corollary 2 *When demand is perfectly elastic, public investments increase deforestation.*

Proof

Using $\lim_{\theta\to\infty} \frac{1}{\theta} = 0$ we obtain

$$L_2 = \lim_{\theta\to\infty} L = \left(\gamma^{1-\beta}(1-\tau)G^{\alpha}\left(\frac{r_K}{\beta}\right)^{-\beta}\right)^{\frac{\varepsilon}{1-\beta}} . \tag{11}$$

Form $\frac{\varepsilon}{1-\beta} > 0$ we obtain $\frac{dL_2}{dG} > 0$. □

The export sector, where demand is elastic, reflects the Jevons paradox. Additional public investments makes it more attractive to use the complementary inputs capital and land. The higher use of land accelerates deforestation.

This section shows that both the Borlaug hypothesis and the Jevons paradox can be reproduced with the model. When agricultural land is used to produce products with different elasticities of demand, it is thus not straightforward to predict the aggregate effect of investments into productivity on deforestation. In the next section we analyze a simple case of such an interaction in order to show how environmental and economic objectives can be reconciled.

5 The case of a competitive land market

In the model the two sectors compete for land. Total agricultural land is thus the sum of the land used in the two sectors,

$$L = L_1 + L_2 . \tag{12}$$

In the food producing sector L_1 we assume that $\theta = 0$. Inserting this into the food demand equation (7) we obtain that total food production is fixed,

$$Y_1 = 1 . \tag{13}$$

In the export sector we have $\theta = \infty$. Again, inserting this into the food demand equation shows that the prices in the exporting sector are given by the international market $\bar{p}_2$,

$$p_2 = \bar{p}_2 . \tag{14}$$

5.1 Tariff policy and public investments

We consider the effect of government policy interventions in the form of tariffs and public investments individually. We can describe the equilibrium by equations (5), (6), and (13) for sector 1, equations (5), (6) and (14) for sector 2, as well as the land market given by (1) and (12).

Proposition 2 *Higher tariffs reduce deforestation, but also production in the export sector. They also lead to a reduction in food prices. Higher public* *investments lead to an increase in deforestation and an increase in production in the export sector. The effect of higher public investments on food prices depends on the relative size of the output elasticity of public investments in the two sectors.*

Proof

From (5) and (6) we obtain $\frac{K_1}{L_1} = \frac{\beta_1}{\gamma_1}\frac{r_L}{r_K}$. Using (13) and the production function we have that $1 = G^{\alpha_1} K_1^{\beta_1} L_1^{\gamma_1}$. Combining these two expressions and solving for r_L we obtain

$$G^{-\frac{\alpha_1}{\beta_1}} L_1^{-\frac{1}{\beta_1}} \frac{\gamma_1}{\beta_1} r_K = r_L \tag{15}$$

Solving (5) for K_2, inserting into (6) and using (14) we have

$$\gamma_2 \left(\left(\frac{\beta_2}{r_K} \right)^{\beta_2} (1-\tau)\bar{p}_2 G^{\alpha_2} \right)^{\frac{1}{1-\beta_2}} = r_L. \tag{16}$$

Combining (15) and (16) and solving for L_1 we obtain

$$L_1 = \left(\frac{\gamma_1}{\gamma_2 \beta_1} \right)^{\beta_1} G^{-\frac{\alpha_1\gamma_2 + \alpha_2\beta_1}{\gamma_2}} r_K^{\frac{\beta_1}{\gamma_2}} \beta_2^{-\frac{\beta_1\beta_2}{\gamma_2}} ((1-\tau)\bar{p}_2)^{-\frac{\beta_1}{\gamma_2}} . \tag{17}$$

From this expression we obtain $\frac{dL_1}{d\tau} > 0$ and $\frac{dL_1}{dG} < 0$. Using (16) we obtain $\frac{dr_L}{d\tau} < 0$ and $\frac{dr_L}{dG} > 0$. Combining (1) and (12) we have $\frac{dL_2}{d\tau} = \varepsilon r_L^{\varepsilon-1} \frac{dr_L}{d\tau} - \frac{dL_1}{d\tau} < 0$ and $\frac{dL_2}{dG} > 0$. Furthermore, solving (5) for K_2 and plugging L_2 into the equation, we obtain $\frac{dK_2}{d\tau} < 0$ and $\frac{dK_2}{dG} > 0$. Using the production function we have $\frac{dY_2}{d\tau} < 0$ and $\frac{dY_2}{dG} > 0$. Using (2) we observe that deforestation decreases with a tariff increase and increases with an increase in public investment.

Using the capital-labor ratio in sector 1 we have $K_1 = \frac{\beta_1}{\gamma_1}\frac{r_L}{r_K} L_1$. Since we assumed that firms produce with constant returns to scale, they make zero profits. Therefore, $p_1 = (r_K K_1 + r_L L_1) = (r_K \frac{\beta_1}{\gamma_1}\frac{r_L}{r_K} L_1 + r_L L_1) = \frac{r_K}{\beta_1} G^{-\frac{\alpha_1}{\beta_1}} L_1^{-\frac{\gamma_1}{\beta_1}} = r_K^{\frac{\gamma_2-\gamma_1}{\gamma_2}} \beta_1^{\beta_1} G^{\frac{-\alpha_1\gamma_2+\alpha_2\gamma_1}{\gamma_2}} \left(\frac{\gamma_2}{\gamma_1}\right)^{\gamma_1} \beta_2^{\frac{\beta_2\gamma_1}{\gamma_2}} ((1-\tau)\bar{p}_2)^{\frac{\gamma_1}{\gamma_2}}$. With this we have $\frac{dp_1}{d\tau} < 0$ and $\frac{dp_1}{dG} < 0 \Leftrightarrow \frac{\alpha_2}{\gamma_2} < \frac{\alpha_1}{\gamma_1}$. □

The intuition for the tariff increase is straightforward. Export tariffs reduce the net price received by the producer. The sector thus uses less inputs and produces less. The lower demand for land reduces the land price and thus deforestation.

Concerning the effect of the tariff increase on the food price, there are two opposing effects. The food sector uses more land, but it pays less per unit of land. The price drop is however stronger, meaning that the food sector pays less for the total amount of land after the tariff increase even though it uses more land.

Public investments by contrast benefit both sectors. They bid up the price for land and therefore cause more deforestation. Only the export sector can expand production. The food sector thus uses the land more intensively by substituting land with capital (as well as public infrastructure and institutions). The export sector takes over some land from the food sector. Concerning the food price, there are two effects. The public investments benefit the food sector directly, but indirectly they harm the food sector by making land more expensive. The net effect on food prices thus depends on which sector benefits more from the direct effect of the investment.

5.2 Revenue recycling

In Corollary 2 we have seen that public investments can produce a Jevons paradox: the export sector benefits and expands production, but at the same time more forest is lost to agricultural production. Proposition 2 shows that export tariffs can be used to reduce deforestation, but since the export tariff also reduces production of the export sector it remains unclear what the net effect the two policy tools in combination will be.

In order to identify a beneficial policy we first identify the interests of the different stakeholders. An international REDD+ donor is willing to make a payment to the country government, if the government introduces a policy, which reduces deforestation. We assume that the government is already undertaking maximal forest conservation efforts, given its multiple constraints. The government is thus only willing to undertake additional efforts, if they represent zero net additional cost to the government budget. We assume that the export sector is willing to accept its country's participation in the REDD+ program only, if it is not forced to downsize its operation. We assume that the size of the sector reflects the interests of the business owners through factor income, since technically profits are zero. Finally, we assume that the country's population is willing to participate in the REDD+ program only if the participation does not increase food prices.

The challenge is thus to design the REDD+ policy mix such that all stakeholders are willing to participate. We find that indeed, it is possible to design the policy in a Pareto improving manner, such that no stakeholder loses:

Proposition 3 *For any amount of public investment it is possible to raise export tariffs such that (i) production in the export sector remains constant, (ii) deforestation reduces, and (iii) food prices decline.*

Proof

Let $\bar{Y}_2$ be the level of production in the export sector before the REDD policy mix is implemented. Then condition (i) can be expressed as

$$\bar{Y}_2 = G^{\alpha_2} K_2^{\beta_2} L_2^{\gamma_2} \ . \tag{18}$$

Solving (5) for L_2 and plugging it into (12) we have

$$K_2 = \left(\frac{r_K}{(1-\tau)\bar{p}_2\beta_2} \right)^{-\frac{1}{\gamma_2}} G^{\frac{\alpha_2}{\gamma_2}} (L - L_1) \ . \tag{19}$$

Inserting (19) into (18) we can express $1 - \tau$ as a function of $L - L_1$,

$$1 - \tau = \bar{Y}_2^{\frac{\gamma_2}{\beta_2}} G^{-\frac{\alpha_2}{\beta_2}} \frac{r_K}{\bar{p}_2\beta_2} (L - L_1)^{-\frac{\gamma_2}{\beta_2}} \ . \tag{20}$$

This is the level of tariffs the government would need to set in order to induce the export sector (sector 2) to produce $\bar{Y}_2$, the same amount they produced before the policy intervention.

We can now insert this choice of tariffs by the government into the market equilibrium value for L_1 given in equation (17):

$$L_1 = \left(\frac{\gamma_1}{\gamma_2\beta_1}\right)^{\beta_1} G^{\frac{-\alpha_1\beta_2+\alpha_2\beta_1}{\beta_2}} \beta_2^{\beta_1} \bar{Y}_2^{-\frac{\beta_1}{\beta_2}} (L - L_1)^{\frac{\beta_1}{\beta_2}} \ . \tag{21}$$

Using (1) and (15) we have

$$L = r_L^{\varepsilon} = G^{-\frac{\alpha_1\varepsilon}{\beta_1}} L_1^{-\frac{\varepsilon}{\beta_1}} \left(\frac{\gamma_1}{\beta_1} r_K\right)^{\varepsilon} \ . \tag{22}$$

Inserting this into (21) we have

$$L_1 = \left(\frac{\gamma_1}{\gamma_2\beta_1}\right)^{\beta_1} G^{\frac{-\alpha_1\beta_2+\alpha_2\beta_1}{\beta_2}} \beta_2^{\beta_1} \bar{Y}_2^{-\frac{\beta_1}{\beta_2}} \left(G^{-\frac{\alpha_1\varepsilon}{\beta_1}} L_1^{-\frac{\varepsilon}{\beta_1}} \left(\frac{\gamma_1}{\beta_1} r_K\right)^{\varepsilon} - L_1\right)^{\frac{\beta_1}{\beta_2}} \ . \tag{23}$$

Using this we define

$$H = \left(\frac{\gamma_1}{\gamma_2\beta_1}\right)^{\beta_1} G^{\frac{-\alpha_1\beta_2+\alpha_2\beta_1}{\beta_2}} \beta_2^{\beta_1} \bar{Y}_2^{-\frac{\beta_1}{\beta_2}} \left(G^{-\frac{\alpha_1\varepsilon}{\beta_1}} L_1^{-\frac{\varepsilon}{\beta_1}} \left(\frac{\gamma_1}{\beta_1} r_K\right)^{\varepsilon} - L_1\right)^{\frac{\beta_1}{\beta_2}} - L_1 \ . \tag{24}$$

Employing (23) we can write $\frac{\partial H}{\partial G} = \frac{-\alpha_1\beta_2+\alpha_2\beta_1}{\beta_2} \frac{L_1}{G} - \frac{\alpha_1\varepsilon}{\beta_2} \frac{L_1}{L-L_1} \frac{L}{G}$ and $\frac{\partial H}{\partial L_1} = \frac{\beta_1}{\beta_2} \frac{L_1}{L-L_1} \left(-\frac{\varepsilon}{\beta_1} \frac{L}{L_1} - 1\right) - 1$.

Using (15) the effect of an increase in government spending is thus given by

$$\frac{dr_L}{dG} = r_L \left(-\frac{\alpha_1}{\beta_1}\frac{1}{G} - \frac{1}{\beta_1}\frac{1}{L_1}\frac{dL_1}{dG}\right) \ , \tag{25}$$

where $\frac{dL_1}{dG} = -\frac{\frac{\partial H}{\partial G}}{\frac{\partial H}{\partial L_1}}$. Inserting $\frac{dL_1}{dG}$ into (25) it can be shown that $\frac{dr_L}{dG} < 0$. Using (1) we have $\frac{dL}{dG} = \varepsilon r_L^{\varepsilon-1} \frac{dr_L}{dG} < 0$.

In the proof of Proposition 2 we have seen that $p_1 = \frac{r_K}{\beta_1} G^{-\frac{\alpha_1}{\beta_1}} L_1^{-\frac{\gamma_1}{\beta_1}}$, so that

$$\frac{dp_1}{dG} = p_1 \left(-\frac{\alpha_1}{\beta_1}\frac{1}{G} - \frac{\gamma_1}{\beta_1}\frac{1}{L_1}\frac{dL_1}{dG}\right) \ . \tag{26}$$

Again resorting to $\frac{dL_1}{dG} = -\frac{\frac{\partial H}{\partial G}}{\frac{\partial H}{\partial L_1}}$ we obtain $\frac{dp_1}{dG} < 0$. □

Thus with the policy mix of simultaneous export tariff increases and public good investments the production of the export sector remains constant. This implies that the same amount of agricultural products is exported after implementing the policy as before. The policy thus does not cause negative effects, in terms of higher food prices for example, to the trade partners of the implementing country.

The proposition shows that a combination of tariffs and government spending, which keeps production in the export sector constant, would reduce deforestation and lower food prices. However, the amount spent by the government on public investments might exceed the additional revenue from the export tariffs. In order to get the government on board, the international REDD+ donor could fill the funding gap and pay the missing amount to the government.

The international REDD+ donor has a limited budget and will not be willing to spend any amount on the outlined policy. It would thus need to trade off the amount of forest saved through the policy, with the amount needed to maintain a zero net effect on the government budget. The government revenue with the policy package is given by $\tau\bar{Y}_2$, where τ is given by equation (20). The expenditure of the government would be given by the amount of additional public investment of the policy package multiplied with the cost per unit of government investment. If the government revenue from the policy exceeds or equals the costs for the corresponding public investment levels, it would be self-financing. Otherwise REDD+-money would be required.

6 Numerical estimation

Proposition 3 suggests a combination of export tariffs and government investments, which would be acceptable to the export industry and domestic food consumers. It also implicitly contains a supply function for forest conservation: the more government investments there are, the more forest will be conserved. In this section we make this trade-off explicit and estimate it empirically.

6.1 Cost benefit analysis of forest conservation

For the numerical estimate we assume $\alpha_1 = \alpha_2, \beta_1 = \beta_2$ and $\gamma_1 = \gamma_2$, which will allow us to write equation (23) as

$$L_1 = \left(\left(\bar{Y}_2 + 1\right) G^{\frac{\alpha\varepsilon}{\beta}} \left(\frac{\gamma}{\beta} r_K\right)^{-\varepsilon} \right)^{-\frac{\beta}{\varepsilon+\beta}} . \tag{27}$$

By equation (22) we have

$$L = G^{-\frac{\alpha\varepsilon}{\beta}} L_1^{-\frac{\varepsilon}{\beta}} \left(\frac{\gamma}{\beta} r_K\right)^{\varepsilon} . \tag{28}$$

Equation (20) gives the level of τ which keeps total production of the export sector at $\bar{Y}_2$:

$$\tau = 1 - \bar{Y}_2^{\frac{\gamma}{\beta}} G^{-\frac{\alpha}{\beta}} \frac{r_K}{\bar{p}_2 \beta} (L - L_1)^{-\frac{\gamma}{\beta}} . \tag{29}$$

Inserting equations (28) and (27) into equation (29) we obtain τ as a function of exogenous parameters. Let $\bar{G}$ be the "initial" amount of government investment. It is the amount which allows firms in the export sector to produce the amount $\bar{Y}_2$, when export tariffs are equal to zero. $\bar{G}$ can be obtained by solving (29) with $\tau = 0$ for G.

By introducing agricultural export tariffs the government generates a revenue of $\tau\bar{Y}_2$. At the same time, it has expenses for increasing government investments from $\bar{G}$ to a higher level. Labeling the price for one unit of government investment as p_G, the expenses amount to $p_G(G-\bar{G})$. The government is not willing to accept a net loss due to the forest conservation policy. However, the international REDD+ donor can support the program by an amount (E) of

$$E = p_G(G - \bar{G}) - \tau\bar{Y}_2 . \tag{30}$$

Finally, we can calculate the amount of conserved forest area, which corresponds to a given level of government investment. Using again equation (22) we have

$$W = A - L = A - G^{-\frac{\alpha\varepsilon}{\beta}} L_1^{-\frac{\varepsilon}{\beta}} \left(\frac{\gamma}{\beta} r_K\right)^{\varepsilon} . \tag{31}$$

Together with equation (27) this yields the amount of conserved forest areas a function of G.

6.2 An example

It is not possible to solve equation (29) for G analytically. Therefore we cannot combine equations (30) and (31) to obtain the amount of forest as a function of the expenditure for the international REDD+ donor. However, we can produce a graph of the relationship of costs and the expectable conserved forest area numerically.

We choose the model parameters according to empirical estimates from the literature. Warner (2014) conducts a review of the empirical literature on the effect of public investments on output for lower income countries and concludes that a value of 0.15 is a realistic assumption for low income countries, so that we set $\alpha = 0.15$. The factor share of non-reproducible capital (which is mainly land) is between 20% and 30% in developing countries in Table II of Caselli and Feyrer (2007), so that we set $\gamma = 0.25$. We require the production function to have constant returns to scale, so that we obtain $\beta = 1 - \gamma = 0.75$. Udry and Anagol (2006) describe that the interest rate faced by farmers in developing countries varies strongly and can attain very high rates, exceeding 100% in some cases. For "well-established food crop cultivation" they give a range of 30% to 50%, so that we work with $r_K = 0.3$. For the supply elasticity of land, Gouel et al. (2006) give a range of 0.37 for Asia and Latin American countries and 0.75 for African countries. We thus choose $\varepsilon = 0.37$ for the reference case. We normalize the values for land area A, output in the exporting sector $\bar{Y}_2$, and the costs of public investments p_G to 1.

For these value, we have a resulting $\bar{G}$ value of 0.045. Increasing G above $\bar{G}$ generates costs given by $p_G(G - \bar{G})$, see the dotted line in Figure 5, left panel. At the same time, the export tariff increases, since τ is chosen such that agricultural exports stay constant. The revenues are given by $\tau\bar{Y}_2$ and are plotted as a solid line. Eventually, the costs exceed the revenue such that the government would make a loss. In order to support the policy and to avoid that the government would have to bear the costs, bridging this gap can be made possible through international financial support. This would be the net expense by the international REDD+ donor. The net expense is plotted in Figure 5 in the right panel on the vertical axis. As the G continues to increase agricultural producers intensify their land use without expanding production. This allows them to save on land and the amount of forest spared increases. The amount of forest spared is plotted on the horizontal axis.

While the cost per unit of government investment increase linearly, revenue is a concave function. A country with a low initial stock of government investment will become more productive when more is invested. The export tariff needed to keep production constant is thus increasing steeply. As the economy gets saturated with government investments, additional investments will increase productivity only to a small degree, thus producing only small increases in additional tariff revenue.

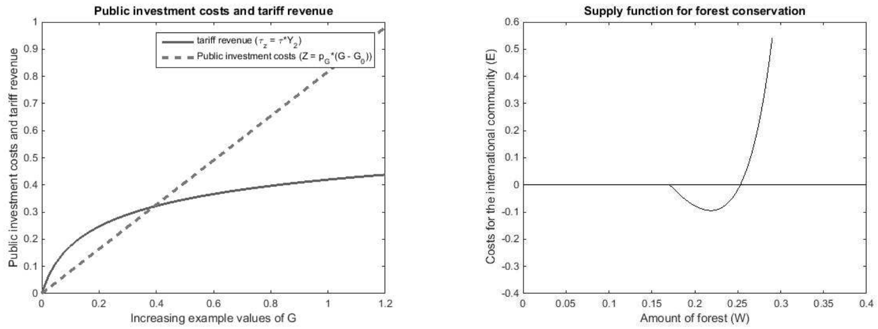

Figure 5: Cost and revenue of forest conservation (left) and net cost compared to conserved forest (right).

The numerical example is chosen in order to allow to show the progression of the curve. Initially, 0.171 units of land are covered with forests. The example is optimistic, because it is probably rather unlikely - even though possible - that the revenue of the policy initially exceeds the cost. 0.075 units of land can be reserved for conservation at zero net cost. Eventually, however, this trend reverses. Initially there would be a win-win situation: the policy generates additional government revenue and conserves more forest land. For higher levels of G, the costs increase until the total cost of the policy exceeds the revenue. This is the point, where the international REDD+ donor would need to support the public investments of country financially. At some point, the marginal value for additional government benefits decreases and hardly any additional forest can be conserved. In a less optimistic scenario, the costs could exceed the revenue from the beginning, thus creating a trade-off between higher net costs and more forest conservation from the start.

The cost curve in the right panel of Figure 5 can be read as a “forest supply curve” faced by the international REDD+ donor, where a higher transfer to the country would increase the amount of forest conservation offered. The curve shows that, as expected, the marginal cost for conserving additional units of forest increases. Defining a utility function on expenditure and forest conservation for the international REDD+ donor would allow to obtain the optimal combination of cost and conserved forest area.

6.3 Parameter sensitivity

In order to analyze, how different parameter assumptions influence the effect of the policy mix, we next vary the parameter values of p_G, A, Y_2, α, ε, and r_K. This sensitivity analysis also provides an intuition for how well the proposed policy mix could work in different countries.

An increase in the cost of public investments, p_G, obviously decreases the amount of forest that can be conserved with the policy mix. If we increase the land area of the avarage low income economy described in Section 6.2 A, the initial forest area, W, will increase. The costs of conserving forest remain the same.

Further parameter variations require a more detailed analysis. Consider a change in the output of the export sector to $\bar{Y}_2 = 2$. This case reflects an economy with a higher level of public investments such that $\bar{G} = 0.073$. As the export sector produces more, it also uses more forest land, such that initially

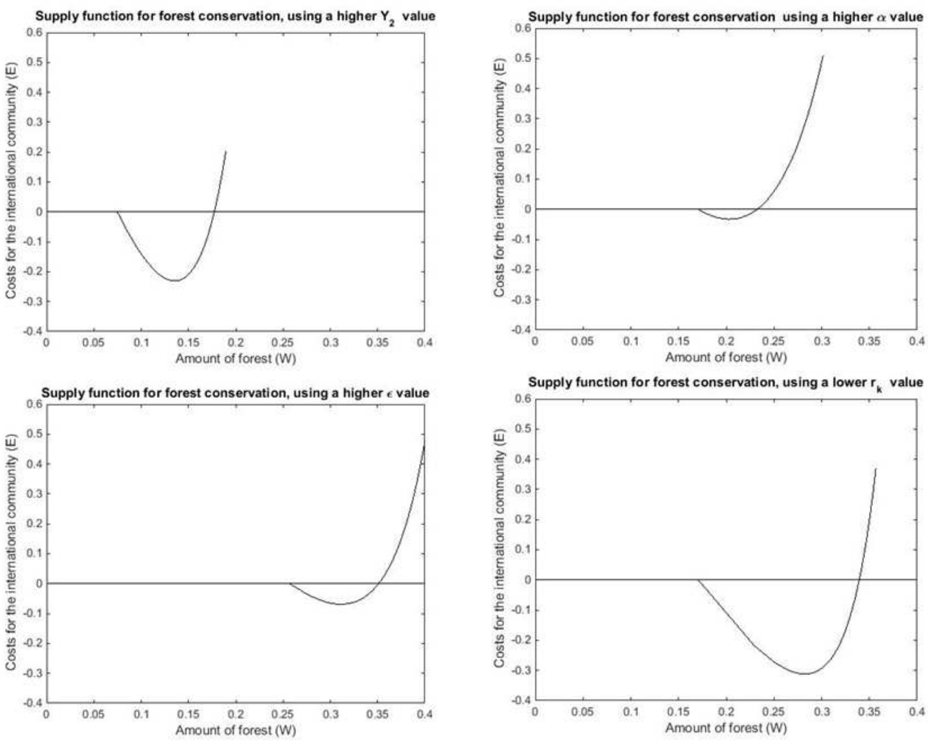

Figure 6: The four graphics above show the effect of a variation of different parameters Y_2(upper left panel), α (upper right panel), ϵ (lower left panel), r_K (lower right panel)
.

only 0.075 units of land are left for conservation. When the policy mix is introduced, it initially generates higher profits since the amount of output in the exporting sector, based on which tariffs are collected, is higher. Due to this higher revenue, 0.1 units of land can be afforested with zero net cost. The upper left panel of Figure 6 illustrates the effect.

When the output elasticity of public investments is higher, $\alpha = 0.3$, more public investments are used initially, $\bar{G} = 0.211$. The initial forest area is almost unchanged compared to the reference scenario at 0.171 . As much more public investments are used initially, less land, (0.06 units) can be afforested due to decreasing returns to further investments. The upper right panel of Figure 6 illustrates this case.

Next we consider the case, where the supply elasticity of land is at the upper end of the interval given by Gouel et al. (2006) and use a value of $\varepsilon = 0.75$. The initial level of public investments is $\bar{G} = 0.053$. 0.091 units of land can be afforested with zero net cost in this scenario. The lower left panel of Figure 6 illustrates this case.

Finally, we also consider a scenario with a lower interest rate, $r_K = 0.2$. The reduced price for capital means that more capital is used while a lower initial level of public investments is used, $\bar{G} = 0.006$. The amount of land used is not affected much, such that there is an initial forest cover area of $W = 0.17$ units of land. Given that a lot of capital is available initially, complementary public investments are taken up productively, so more land, 0.167 units, can be afforested with zero net cost. The lower right panel of Figure 6 illustrates this point.

Summarizing, we find that forest conservation is more cost effective, when public investments are more affordable (p_G), when more is exported already ($\bar{Y}_2$), when the supply elasticity of land is higher (ε), and when the interest rate is lower (r_K). An economy with a higher output elasticity of public investments (α) can be expected to have a relatively high level of public investments already, so that further additions increase productivity less and thus make the policy mix less cost effective.

7 Discussion

A total volume of 18 billion US$ has been comitted for REDD+ payments (FAO, 2016). At the same time, this article shows that political realities in one third of the countries in which REDD+ programs are planned to be rolled out, are shaped by three structural constraints to forest conservation: a strong specialization in the agricultural sector, food insecurity, and weak political institutions. It is thus important to develop forest conservation policies, which allow to reduce deforestation effectively in spite of these structural constraints.

In this article we have analyzed, in how far export tariffs in combination with public investments can represent such a policy mix. We find that the policy mix can (i) reduce deforestation, while (ii) maintaining agricultural output levels, (iii) without increasing the prices of agricultural products. By satisfying these central requirements, the policy mix provides incentives for all relevant stakeholders, namely the international donor, the exporting sector, the government, and the domestic population to accept its implementation.

When examining the political feasibility of the policy mix, four aspects require further discussion: (i) possible secondary economic impacts, (ii) the implementability of export tariffs, (iii) the type of public investments, and (iv) the political economy challenges for the policy mix to becoming effective.

First of all, the potential secondary effects of the introduction of export tariffs have been critically discussed in the literature. Warr (2001) looks at the case of Thailand's rice exports. He points out that while export tariffs reduce consumer prices, they also reduce the producer prices and thus the real wages of unskilled labor. A similar observation is made by Dennis and Iscan (2011), who find that distortionary trade policies on agricultural products reduce the real wage and bring it closer to the subsistence level, whereby the structural transformation of the economy is delayed. In our model consumer prices also decline as a consequence of the proposed policy mix. However, given the public investments, production costs also decline. Therefore production levels and thus the required labor input can be assumed to remain constant, even if the model does not explicitly analyze labor market effects. The crucial difference here is the reinvestment of tariff revenues into productivity enhancing public infrastructures or institutions. The policy mix provides disincentives to deforest and incentives to substitute the input factor deforested land with public infrastructures and institutions. The representative farmer switches from a land intensive to a public infrastructure and institutions intensive production. In the long run, higher economic growth rates can be expected, since the policy induces a more quality oriented model of development. Long run growth is driven by technology and institutional quality, both of which can be boosted with the suggested policy mix. REDD+ funds can be used to support the government budget for the required level of public investments.

Second, past experiences with export tariffs show that they have been used more frequently during the last decade (OECD, 2014), in contrast to other restrictive trade policy instruments that are less and less deployed, due to WTO rules. The use of export tariffs had been significantly reduced during the Washington Consensus period, where Bretton Woods institutions advised developing countries to liberalize trade policies (Williamson, 1993), see appendix 2 for examplary data.[6] The OECD explains the new rise in export tariffs by the volatility on the international commodity price markets and in particular, by the food price crisis in 2007. Such a trade policy reaction can for instance be observed in Bangladesh, Brazil, Cambodia, Egypt, China, Madagascar, India, Nepal, Thailand, and Vietnam on Rice and India, Argentina, Kazakhstan, Nepal, and Pakistan on wheat (Bouët and Laborde, 2010). Nevertheless, export tariffs are currently still at rather low levels compared to historic values.

Third, not all types of public investments are likely to have the same positive effect on forest cover. We therefore conceive public investments as types of investment that lead to an increase in productivity in the agricultural sector, without having simultaneously a harmful impact on forest cover. Electrification and land right allocations are likely to have such net positive effects. Assuncao et al. (2015) have empirically investigated the impact of electrification on productivity and on forest cover in Brazil. They find that electrification leads to investments into irrigation systems, mechanization, fertilizers, and pesticides and thus increases land productivity. Furthermore, Robinson et al. (2014) have conducted a meta-analysis of empirical studies on the relationship between land tenure rights and tropical deforestation. Based on evidence from 118 cases, they find that land tenure security is associated with less deforestation. In contrast, the construction of roads through forest areas (as another form of public investments), entails productivity gains in the agricultural sector, due to a reduction in transport costs, but empirical and theoretical evidence show that it also bears a high risk for additional deforestation (Busch and Ferretti-Gallon, 2017; Angelsen, 2007). This illustrates that the policy design needs to consider the type of public investment to effectively increase productivity without increasing deforestation. Only a country specific analysis could provide such a level of detail.

Finally, most of the usually discussed REDD+ policy approaches, such as compensation payments to farmers require complex implementing institutions on all levels and are thus challenging to implement in institutionally weak countries. In contrast, a combination of export tariffs with public investments is more implementable, because it is sufficient to collect export tariffs at the export hubs (such as ports or airports, railways or highways out of the country), which can build on existing custom offices and thus limits the administrative cost of the tariff collection. Furthermore, the policy mix does not require rural institutions for forest conservation to work. Nevertheless, the risk of corruption may undermine the effectiveness of the proposed policy mix. However, combining export tariff revenue with REDD+ funding has two potential sources

[6] In Malaysia, the Philippines, Sri Lanka, Ethiopia, Tanzania, Chile, Columbia, Costa Rica, and Mexico they revenues from export tariffs contributed to up to 5% to public income (FAO, 1994). In Madagascar, income from export tariffs constituted 30% of the government revenue in 1983 (Anderson and Masters, 2009).

of scrutiny: First domestic public scrutiny, which is likely to be higher due to the domestic tax collection (Ross, 2004) and second international monitoring and control of the disbursement of REDD+ funds.

8 Conclusion

The conventional approach to REDD+ takes the form of conditional payments for conservation projects. In countries with a low degree of institutional development there are substantial concerns that they can be implemented successfully. In this paper we propose an alternative approach to REDD+ policy, which is designed to have minimal requirements on institutional quality. We identify 36 countries, with a combination of a strong economic specialization in the agricultural sector, food insecurity and weak political institutions.

For these countries we suggest the imposition of export tariffs on agricultural products. The intention of this tariff increase is to reduce the incentive for unsustainable deforestation for the export market. We show that this does not need to have a negative effect on the exporting industry if the government simultaneously invests in agricultural productivity, in particular by improving land tenure security and by supplying electricity. The increased productivity would intensify agriculture, so that the same output can be generated with less land.

The proposed policy mix has costs (for the government investments) and revenues (from the export tariffs) and is thus partially self-financing. As costs are expected to exceed the revenue at least initially, the policy would need to be supported with REDD+ funds. However, the suggested forms of government investments are in line with existing policy objectives by the international community, so that there would be strong synergy effects with development objectives.

We acknowledge that investing in agricultural productivity is a challenging and long-term project and also vulnerable to the problems arising from weak governance. At the same time it offers the prospect of resolving the trade-off between development and forest conservation in a sustainable manner.

References

Abdulai, A., Owusu, V., and Goetz, R. (2011). Land tenure differences and investment in land improvement measures: theoretical and empirical analyses. *Journal of Development Economics*, 96(1):66–78.

Acemoglu, D., Johnson, S., and Robinson, J. A. (2005). Institutions as a fundamental cause of long-run growth. *Handbook of Economic Growth*, 1(2005):385–472.

African Development Bank (2013). The African Economic Outlook. URL: http://www.africaneconomicoutlook.org/en/ (accessed 13/04/2016).

Andam, K. S., Ferraro, P. J., Pfaff, A., Sanchez-Azofeifa, G. A., and Robalino, J. A. (2008). Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences*, 105(42):16089–16094.

Anderson, K. and Masters, W. A. (2009). *Distortions to agricultural incentives in Africa*. The World Bank, Washington D.C., USA.

Angelsen, A. (2007). Forest cover change in space and time: combining the von Thünen and forest transition theories. *Policy Research Working Paper*, 1(WPS4117):1–43.

Angelsen, A. (2013). REDD+ as performance-based aid: general lessons and bilateral agreements of Norway. *WIDER Working Paper*, 2013(135):1–24.

Arcand, J.-L., Guillaumont, P., and Jeanneney, S. G. (2008). Deforestation and the real exchange rate. *Journal of Development Economics*, 86(2):242–262.

Assunção, J., Lipscomb, M., and Mushfiq Mobarak, A. (2015). Infrastructure development can benefit the environment: electrification, agricultural productivity and deforestation in Brazil. *LACEA Working Paper*, 2015:1–47.

Barbier, E. B. (2000). Links between economic liberalization and rural resource degradation in the developing regions. *Agricultural Economics*, 23(3):299–310.

Barbier, E. B. (2004). Explaining agricultural land expansion and deforestation in developing countries. *American Journal of Agricultural Economics*, 86(5):1347–1353.

Barbier, E. B., Damania, R., and Léonard, D. (2005). Corruption, trade and resource conversion. *Journal of Environmental Economics and Management*, 50(2):276–299.

Barro, R. J. (1990). Government spending in a simple model of endogeneous growth. *Journal of Political Economy*, 98(5):S103–S125.

Barro, R. J. and Sala-i Martin, X. (1992). Public finance in models of economic growth. *The Review of Economic Studies*, 59(4):645–661.

Bernhofen, D. M. (1997). Strategic trade policy in a vertically related industry. *Review of International Economics*, 5(3):429–433.

Bohn, H. and Deacon, R. T. (2000). Ownership risk, investment, and the use of natural resources. *American Economic Review*, 90(3):526–549.

Borlaug, N. (2007). Feeding a hungry world. *Science*, 318(5849):359–359.

Bouët, A. and Laborde, D. (2010). Economics of export taxation in a context of food crisis. *International Food Policy Research Institute (IFPRI) Discussion Paper*, 2010(00994):1–34.

Brockington, D., Igoe, J., et al. (2006). Eviction for conservation: a global overview. *Conservation and Society*, 4(3):1–47.

Busch, J. and Ferretti-Gallon, K. (2017). What drives deforestation and what stops it? a meta-analysis. *Review of Environmental Economics and Policy*, 11(1):3–23.

Byerlee, D., Stevenson, J., and Villoria, N. (2014). Does intensification slow crop land expansion or encourage deforestation? *Global Food Security*, 3(2):92–98.

Caselli, F. and Feyrer, J. (2007). The marginal product of capital. *The Quarterly Journal of Economics*, 122(2):535–568.

Central Intelligence Agency (2013). The World Factbook 2014. URL: https://www.cia.gov/library/publications/download/ (accessed 13/04/2016).

Cisneros, E., Zhou, S. L., and Börner, J. (2015). Naming and shaming for conservation: evidence from the Brazilian Amazon. *PloS ONE*, 10(9):e0136402.

Cohn, A. S., Mosnier, A., Havlík, P., Valin, H., Herrero, M., Schmid, E., O'Hare, M., and Obersteiner, M. (2014). Cattle ranching intensification in Brazil can reduce global greenhouse gas emissions by sparing land from deforestation. *Proceedings of the National Academy of Sciences*, 111(20):7236–7241.

Copeland, B. and Taylor, M. (2004). Trade, growth, and the environment. *Journal of Economic Literature*, 42(1):7–71.

Corderí Novoa, D. (2008). Deforestation and property rights: a comparison between former British and Spanish colonies. *Economic Analysis Working Papers*, 7(2):1–14.

Craig, B. J., Pardey, P. G., and Roseboom, J. (1997). International productivity patterns: accounting for input quality, infrastructure, and research. *American Journal of Agricultural Economics*, 79(4):1064–1076.

Deacon, R. T. (1994). Deforestation and the rule of law in a cross-section of countries. *Land Economics*, 70(4):414–430.

Dennis, B. N. and Iscan, T. B. (2011). Agricultural distortions, structural change, and economic growth: a cross-country analysis. *American Journal of Agricultural Economics*, 93(3):885–905.

Ericksen, P., Thornton, P., Notenbaert, A., Cramer, L., Jones, P., and Herrero, M. (2011). Mapping hotspots of climate change and food insecurity in the global tropics. *CCAFS Report*, 2011(5):1–56.

FAO (1994). *La fiscalité agricole dans les pays en développement*. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO (2013). Prevalence of undernourishment (%) (3-year average). URL: http //faostat3.fao.org/browse/D/FS/E (accessed 13/10/2016).

FAO (2015). Global forest resources assessment 2015. URL: http://www.fao.org/forest-resources-assessment/current-assessment/en/(accessed 13/04/2015).

FAO (2016). Voluntary REDD+ database. URL:http://www.fao.org/forestry/vrd/ download/en/ (accessed 08/12/2015).

FCPF (2015). FCPF REDD+ countries. URL: https://www.forestcarbonpartnership.org/redd-countries-1 (accessed 08/12/2015).

Fishbein, G. and Lee, D. (2015). *Early lessons from jurisdictional REDD+ and low emissions development programs*. The Nature Conservancy Policy Report, Washington D.C., USA.

Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., and Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proceedings of the National Academy of Sciences*, 107(38):16732–16737.

Goodland, R. and Daly, H. (1996). Environmental sustainability: universal and non-negotiable. *Ecological Applications*, 6(4):1002–1017.

Gouel, C., Hertel, T., et al. (2006). Introducing forest access cost functions into a general equilibrium model. *GTAP Working Research Memorandum*, 2006(8):1–10.

Henson, S., Brouder, A.-M., and Mitullah, W. (2000). Food safety requirements and food exports from developing countries: the case of fish exports from Kenya to the European Union. *American Journal of Agricultural Economics*, 82(5):1159–1169.

Hertel, T. W. (2012). Implications of agricultural productivity for global cropland use and GHG emissions: Borlaug vs. Jevons. *GTAP Working Paper*, 2012(69):1–36.

IEA (2016). World Energy Outlook - International Energy Agency. URL: http://www.worldenergyoutlook.org (accessed 07/02/2016).

Jevons, W. S. (1866). *The coal question*. Macmillan and Co, London, United Kingdom.

Jones, D. W. and O'Neill, R. V. (1994). Development policies, rural land use, and tropical deforestation. *Regional Science and Urban Economics*, 24(6):753–771.

Kalkuhl, M. and Edenhofer, O. (2017). Ramsey meets Thünen: the impact of land taxes on economic development and land conservation. *International Tax and Public Finance*, 24(2):350–380.

Karlan, D., Osei, R., Osei-Akoto, I., and Udry, C. (2014). Agricultural decisions after relaxing credit and risk constraints. *The Quarterly Journal of Economics*, 129(2):597–652.

Karsenty, A. and Ongolo, S. (2012). Can "fragile states" decide to reduce their deforestation? The inappropriate use of the theory of incentives with respect to the REDD mechanism. *Forest Policy and Economics*, 18:38–45.

Kim, J. (2010). Recent trends in export restrictions. *OECD Trade Policy Papers*, 2010(101):1–47.

Kongsamut, P., Rebelo, S., and Xie, D. (2001). Beyond balanced growth. *The Review of Economic Studies*, 68(4):869–882.

Koyuncu, C. and Yilmaz, R. (2009). The impact of corruption on deforestation: a cross-country evidence. *The Journal of Developing Areas*, 42(2):213–222.

Macedo, M. N., DeFries, R. S., Morton, D. C., Stickler, C. M., Galford, G. L., and Shimabukuro, Y. E. (2012). Decoupling of deforestation and soy production in the southern Amazon during the late 2000s. *Proceedings of the National Academy of Sciences*, 109(4):1341–1346.

Mendelsohn, R. (1994). Property rights and tropical deforestation. *Oxford Economic Papers*, 46(Special Issue on Environmental Economics):750–756.

Meyfroidt, P. and Lambin, E. F. (2008). The causes of the reforestation in Vietnam. *Land Use Policy*, 25(2):182–197.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J., Motta, R. S. d., Armijo, E., Castello, L., Brando, P., Hansen, M. C., McGrath-Horn, M., Carvalho, O., and Hess, L. (2014). Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, 344(6188):1118–1123.

Nunn, N. and Trefler, D. (2013). Incomplete contracts and the boundaries of the multinational firm. *Journal of Economic Behavior & Organization*, 94(2013):330–344.

OECD (2014). Export restrictions in raw materials trade: facts, fallacies and better practices. URL: http://www.oecd.org/tad/benefitlib/export-restrictions-raw-materials.htm (accessed 13/04/2016).

Oldekop, J., Holmes, G., Harris, W., and Evans, K. (2016). A global assessment of the social and conservation outcomes of protected areas. *Conservation Biology*, 30(1):133–141.

Pacheco, P. (2006). Agricultural expansion and deforestation in lowland Bolivia: the import substitution versus the structural adjustment model. *Land Use Policy*, 23(3):205–225.

Pullin, A. S., Bangpan, M., Dalrymple, S., Dickson, K., Haddaway, N. R., Healey, J. R., Hauari, H., Hockley, N., Jones, J. P., Knight, T., et al. (2013). Human well-being impacts of terrestrial protected areas. *Environmental Evidence*, 2(1):1–41.

Robinson, B. E., Holland, M. B., and Naughton-Treves, L. (2014). Does secure land tenure save forests? A meta-analysis of the relationship between land tenure and tropical deforestation. *Global Environmental Change*, 29(2014):281–293.

Rodrik, D. (1989). Optimal trade taxes for a large country with non-atomistic firms. *Journal of International Economics*, 26(1-2):157–167.

Ross, M. L. (2004). Does taxation lead to representation? *British Journal of Political Science*, 34(2):229–249.

Shandra, J. M., Leckband, C., and London, B. (2009). Ecologically unequal exchange and deforestation: a cross-national analysis of forestry export flows. *Organization & Environment*, 22(3):293–310.

Skinner, J. (1991). If agricultural land taxation is so efficient, why is it so rarely used? *The World Bank Economic Review*, 5(1):113–133.

Solberg, B., Moiseyev, A., Kallio, A. M. I., and Toppinen, A. (2010). Forest sector market impacts of changed roundwood export tariffs and investment climate in Russia. *Forest Policy and Economics*, 12(1):17–23.

Stevenson, J. R., Villoria, N., Byerlee, D., Kelley, T., and Maredia, M. (2013). Green revolution research saved an estimated 18 to 27 million hectares from being brought into agricultural production. *Proceedings of the National Academy of Sciences*, 110(21):8363–8368.

Turnovsky, S. J. (2000). Fiscal policy, elastic labor supply, and endogenous growth. *Journal of Monetary Economics*, 45(1):185–210.

Udry, C. and Anagol, S. (2006). The return to capital in Ghana. *American Economic Review*, 96(2):388–393.

UN-REDD (2015). List of UN-REDD countries. URL: http://www.unredd.net/regions-and-countries/regions-and-countries-overview.html (accessed 08/12/2015).

Villoria, N. B., Byerlee, D., and Stevenson, J. (2014). The effects of agricultural technological progress on deforestation: what do we really know? *Applied Economic Perspectives and Policy*, 36(2):211–237.

Warner, A. (2014). Public investment as an engine of growth. *IMF Working Papers*, 2014(WP/14/148):1–76.

Warr, P. G. (2001). Welfare effects of an export tax: Thailand's rice premium. *American Journal of Agricultural Economics*, 83(4):903–920.

Williamson, J. (1990). What Washington means by policy reform. In Williamson, J., editor, *Latin American adjustment: how much has happened?* Institute for International Economics, Washington D.C., USA.

Williamson, J. (1993). Democracy and the "Washington Consensus". *World Development*, 21(8):1329–1336.

World Bank (2013). The world development indicators. URL: http://data.worldbank.org/indicator/NV.IND.MANF.Z (accessed 13/04/2016).

World Bank (2014a). New country classifications. URL: http://data.worldbank.org/news/new-country-classifications-2015 (accessed 13/04/2016).

World Bank (2014b). The worldwide governance indicators. URL: http://info.worldbank.org/governance/wgi/index.aspx#home (accessed 13/04/2016).

WTO (1947). The general agreement on tariffs and trade (1947). URL: https://www.wto.org/english/docs_e/legal_e/gatt47_01_e.htm (accessed 13/04/2016).

WTO (2004). Trade policy review Benin, Burkina Faso, Mali. URL: https://www.wto.org/english/tratop_e/tpr_e/s236_sum_e.pdf (accessed 13/04/2016).

Younger, S. D., Sahn, D. E., Haggblade, S., and Dorosh, P. A. (1999). Tax incidence in Madagascar: an analysis using household data. *The World Bank Economic Review*, 13(2):303–331.

9 Appendix

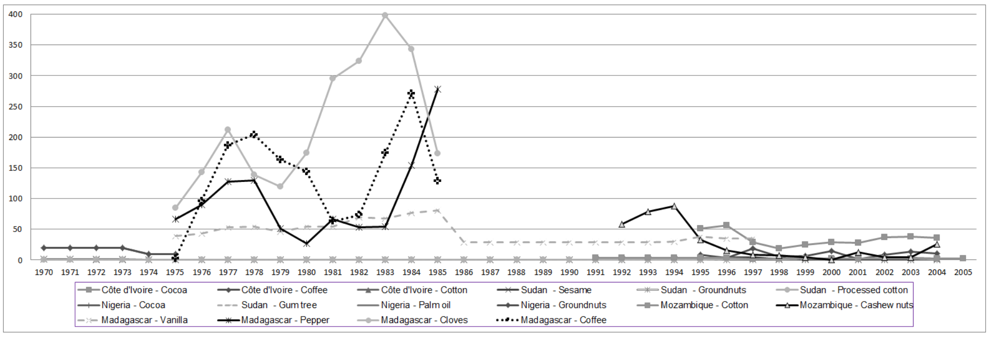

Figure 7: The evolution of the use of export tariffs for some exemplary crops in Cote d'Ivoire, Sudan, Nigeria, Mozambique and Madagascar in percent of the export value (Anderson and Masters, 2009).

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As figure 7 illustrates in an exemplary manner (time series are unavailable for most countries), the Washington Consensus (Williamson, 1990) marked a period, when Bretton Woods institutions advised developing countries to liberalize their trade policies, which consequently significantly reduced most of existing export tariffs (Williamson, 1993). In Benin, for instance, most export tariffs were suppressed in 1993 (WTO, 2004).

Chapter 6

Synthesis and outlook

The first part of this dissertation consists of a meta-analysis of the empirical economics literature on deforestation processes and uses an index of institutional quality based to improve the precision of a forest cover models. Central research questions for the first part of this dissertation have been: which role do political institutions play in deforestation processes according to the existing economics literature? What can be learnt from the existing literature on the effect of improved political institutions on deforestation? How could the role of political institutions be better reflected in forest cover models?

Chapter 2 presents a meta-regression analysis of the empirical literature on the effect of improved governance on deforestation. Two types of observations on the existing literature motivate the analysis. First, there is a multitude of studies examining the hypothesis that improved governance leads to a reduction in deforestation. However, empirical studies come to diverging conclusions: while there are a number of studies with supportive results, other studies find inconclusive results, or even reject the governance hypothesis. A closer look at the respective studies reveals that they use a multitude of different governance measures to examine the effect of governance on deforestation. This suggests that, contrary to the hypothesis that all forms of governance improvement similarly reduce deforestation, either not all aspects of forest governance have the same directionality of effect, or some measures used are not suited to operationalize the effect of governance on deforestation. Second, studies that use the same governance measure in their analysis (for example democracy) still come to diverging results. This finding suggests that other factors of study design than the choice of the governance measure alone are likely to affect the outcome.

These two observations are the main motivation to conduct a systematic analysis of the existing literature on the effect of governance improvements on deforestation. An ordered probit model is used in order to examine the sources of variation across studies to conduct the meta-analysis. The sample is based on 32 studies that contain 227 estimates of the effect of governance on deforestation. The effect on study outcomes of the choice of governance measures and of other elements of study design (such as the choice of control variables, the type of data deployed in the study, the estimation technique etc.) is analyzed.

The analysis reveals that the choice of the governance variable is the main source of variation. In particular, it can be shown that specific environmental governance variables such as environmental policy and the presence of NGOs significantly increase the likelihood of a study to find results that support the governance hypothesis. Furthermore, the variables security of ownership and rule of law have a statistically significant deforestation decreasing effect across studies. The results also show that studies using the variables democracy and political rights in order to measure governance are more likely to find that better governance increases deforestation.

Moreover, the results show that elements of study design are likely to influence the outcome. In particular, including the control variables forest area and control variables on the size or density of the population has a statistically significant effect on the results. The results also show that using an ordinary least squares estimator, rather than other more complex estimation techniques, significantly reduces the likelihood for a study to find results that are supportive of the governance hypothesis.

The study's discussion highlights the need to work on the understanding of the causal effects of specific environmental and forest sector governance elements on deforestation and reflects on the challenges and opportunities for this research agenda.

Chapter 3 contributes to the literature on political institutions in forest cover change modelling. The literature review on the state-of-the-art forest cover change models in the introduction revealed that such models currently take the role of political institutions only to a very limited extent into account. Chapter 3 explores whether including political institutions in the Global Forest Model (G4M) could improve its representation of deforestation processes within the model.

The Global Forest Model integrates biophysical and economic model components to represent deforestation processes. The model assumes that there is a return maximizing land owner (who can be a public or a private entity) per grid cell. The representative land owner compares the net present values of forestry with the net present value of agricultural land use. Based on this net present value comparison, she decides whether to deforest or maintain a forested area. The net present value estimation is informed by the biophysical properties of the respective grid cell and corresponding prices.

In the calibration process a residual calibration factor is used to match the estimated with observed deforestation trends. It allows accounting for parameters that influence the forest cover change decision that are not represented by the biophysical and economic components of the model. The residual calibration factor is multiplied with the net present value of forest land. The analysis tests the hypothesis that the residual calibration factor can be significantly reduced, if differences in the quality of political institutions in charge of the management of environmental goods are taken into account in the modeling process.

In order to test the hypothesis an indicator on environmental institutional quality is constructed. The indicator is based on the FAO's and PROFOR's (2011) framework on forest governance and has three components, which are (i) policy, legal, institutional, and regulatory frameworks, (ii) decision-making and planning processes, and (iii) the implementation, and enforcement of, and compliance with such regulatory frameworks.

The indicator on “environmental quality” (Hartmann and Reimann, 2010) is used to represent component (i), an indicator on the effectiveness of administrative decision making (Porter et al., 2008; World Bank, 2015) is used in order to represent component (ii), and data on structural macroeconomic constraints provided by Hartmann and Reimann (2010), in order to represent component (iii). Indicator values are per-country rankings that use ordinal scales. In order to construct the composite indicator for the 2000 to 2010 calibration period individual indicator values are averaged for the entire period, normalized to values between 0 and 1, summated and then again normalized in order to make the composite indicator values comparable.

Subsequently, an ordinary least squares regression model is used to test, whether the composite index allows explaining an important share of the residual calibration factor. In the regression model, the logarithm of the residual calibration factor is used as a dependent variable and the composite index on environmental institutional quality as an independent variable. Residual calibration factor values are used in the logarithmic form, because it is assumed that political institutions have a non-linear effect on forest cover change. Furthermore, a range of control variables (land area, forest cover, gross domestic product, population density, and tropical or subtropical vegetation) are included. It is then tested whether the indicator can explain an important part in the variation of the residual calibration factor with different model specifications - using all control variables, each single control variable, and no control variables at all.

The results are significant for all model specifications and show that the indicator explains the residual calibration factor in all different model specifications. It is concluded that including the indicator in the Global Forest Model can reduce the unexplained factors in the model. First tests using the calibration period 2010 to 2015 confirm this result.

The second part of this dissertation analyzes policy options for forest conservation, with a focus on countries with weak political institutions. Central research questions are: how do Sub-Saharan African policy makers perceive deforestation drivers? Which role do institutional and policy issues play in their perception and which role could they play in policy responses to deforestation? Which fiscal policy instruments could be used to reduce deforestation, while maintaining economic output levels?

Chapter 4 concentrates on Sub-Saharan African countries and analyzes policy makers’ perceptions of deforestation drivers using content analysis. The analysis is motived by the idea that the implementation phase of REDD+ requires a more nuanced understanding of deforestation drivers including structural drivers in order to formulate targeted policy responses. Given that collecting consistent data on deforestation drivers is still a challenge in most Sub-Saharan African countries and that most forest areas are officially owned by the state in the majority of African countries, studying the perception of deforestation drivers by policy makers is a promising source of relevant insights.

While a range of broad analyses of deforestation drivers for African countries discusses for instance the role of agriculture, population growth, fuelwood collection, urbanization, economic factors, or political institutions in deforestation processes, most do not provide nuanced conclusions indicating specific entry points for policy making. Analyzing the perception of deforestation drivers of African REDD+ policy makers as expressed in REDD+ policy documents allows identifying such concrete entry points for policy making and determining deforestation drivers that are harder to identify, define, and measure with standard proxies in empirical studies of deforestation.

Content analysis is used to quantify the occurrence of different deforestation drivers in texts and to draw comparative conclusions. Data is collected in a nominal form. A dummy variable distinguishes documents in which a respective deforestation driver subcategory is mentioned from documents that do not mention the driver. Three independent coders undertake the coding of the documents. The entire body of documents is coded twice by each coder, in order to ensure a higher degree of reliability. Subsequently, a reliability coefficient is used in order to calculate the level of agreement between different coders. Based on the reliability coefficient, it can be concluded, whether the level of agreement is sufficiently high for the results to be considered valid.

The analysis reveals that institutional and policy drivers of deforestation are mentioned much more frequently in the sample than other types of deforestation drivers. While the frequency of occurrence of institutional and policy drivers of deforestation in the text could also indicate that policy makers use different terms to refer to the same problem, it can be shown that there is a high level of agreement across African countries on the specific type of institutional and policy drivers of deforestation. The highest level of agreement could be observed for the category of land rights issues.

The chapter shows that beyond more general governance problems, such as corruption, policy makers are able to identify a range of concrete forest sector specific institutional problems. Such problems range from reform needs in formal national policy frameworks, such as the agricultural or forest policy framework, to barriers to the implementation of such policies. Barriers to the implementation of policies are, for example, a lack of personnel and capital equipment (e.g. vehicles) in the forest sector administration, or the absence of scientific policy advice to forest policy making. The analysis thus provides more concrete entry points to forest conservation policy making than the existing literature.

Chapter 5 uses an analytical model in order to assess potential effects of a forest conservation policy that combines export tariffs on unprocessed agricultural commodities with public investments. The objective of the study is to understand, whether the policy mix can be used to reduce deforestation, while maintaining agricultural output levels - a challenge faced by many developing countries.

REDD+ policy making theoretically offers room to support national and fiscal policy instruments that help address deforestation drivers structurally. However, large areas of the remaining intact tropical forests are located in low and lower middle income countries. Forest conservation policy making is complicated by three structural constraints in many of these countries.

First, a specialization in the agricultural sector leads to a high dependence upon land as an input to production. Low income countries tend to have a stronger specialization in agricultural activities and less developed manufacturing sectors. Such non-diversified economies do have few economic alternatives to land-demanding agricultural activities resulting in pressure on forests.

Second, systematic forest conservation efforts are challenged, if countries have two parallel agricultural sectors - a phenomenon that tends to occur in low and lower middle income countries. An internationally exporting sector produces crops that can be sold at comparably high prices on international markets (e.g. cocoa, coffee, soy, or palm oil). Simultaneously, there is a domestic agricultural sector that produces staple food products. Thus, in spite of the strong economic specialization in the agricultural sector, low income economies tend to experience food shortages for the domestic population. Consequently, in particular in countries with high population density an increase in the scarcity of the input factor land - caused by forest conservation measures - is likely to be met by public resistance.

Thirdly, forest conservation can be systematically constrained by weak political institutions. Weak political institutions are a direct barrier to forest conservation because they prevent the enforcement of sustainable forest management practices or protected areas. They are also an indirect impediment to forest conservation, because countries with weak institutions tend to be locked into sectors of economic activity that do not rely on complex contracting institutions. These are mostly agricultural and extractive industries. Low income countries tend to be locked into sectors of economic activity that are usually land-demanding and thus more difficult to reconcile with forest conservation.

Since a lot of forest rich countries fit into the described category it appears necessary to develop REDD+ policies that are realistically implementable in these contexts. Such policies need to reduce deforestation, while at least maintaining constant agricultural output levels, such that the country does at least not experience any negative secondary economic effects of forest conservation and ideally can benefit from positive economic externalities.

The chapter examines under which circumstances export tariffs in combination with public investments can fulfill these requirements. Export tariffs are one of the few tolerated trade policies under the rules of the World Trade Organization. Empirical and theoretical economic analyses show that export tariffs can stimulate the structural transformation of an economy, if they are levied on unprocessed commodities, such as agricultural products. These are also the main source of deforestation in many low and lower middle income countries.

In the context of the analysis, public investments are conceptualized as publically provided goods or services that allow farmers to attain productivity increases in the agricultural sector. Examples for such publically provided goods and services could be electricity infrastructure and land tenure rights.

A static mode of competing land uses is used in order to analyze the effect of the introduction of the policy mix. The model assumes two different agricultural sectors: a domestic sector that produces domestic food products and an internationally exporting sector. It is assumed that the two sectors are shaped by different elasticities of demand. The domestic food producing sector faces a perfectly inelastic demand, while the internationally exporting sector is a taker of international market prices and thus faces a perfectly elastic demand. Both sectors use land, capital, and public institutions and infrastructure as inputs to production and maximize output levels with their choice of the input factors capital and land.

The equilibrium analysis reveals that the level of export tariffs can be combined with public investments in a way that deforestation is reduced, food prices in the domestic sector decline, and the output level in the exporting sector remains constant. A numerical example shows that under general parameter value assumptions, the policy can initially be self-funded (through the export tariff revenue), but that an international donor would have to support the public investments in order to make the policy mix scalable. Using an inverse supply function, the numerical example allows depicting the cost function for the international REDD+ donor with a convex shape, implying that the costs for forest conservation are increasing exponentially when more land is reserved for conservation. Chapter 5 concludes by discussing the feasibility of introducing such a policy mix, as well as potential associated risks.

6.1 Common insights and broader significance

The four chapters of this dissertation discuss the role of political institutions in deforestation processes. The chapters show that the quality of political institutions is a central determinant of whether forest conservation is likely to succeed or not. It becomes evident that the current literature does not provide sufficient data and analysis of the role of specific environmental and forest sector institutions. It therefore does not take the quality of political institutions sufficiently into account when discussing policy options for forest conservation - especially for institutionally weak countries. The dissertation helps to fill this gap in the literature by providing a systematic overview of the status quo of the discussion of the topic in the empirical literature. It also analyzes in how far global forest cover change models such as the G4M model could take the influence of environmental sector specific political institutions more explicitly into account.

Furthermore, the dissertation shows that beyond the proxies that are frequently used in order to analyze governance drivers of deforestation, a useful entry point for more refined analysis and data collection can be the perception of political and institutional deforestation drivers by policy makers. Finally, it is examined in how far the particular challenges to forest conservation in institutionally weak low-income countries can be addressed with a specific policy mix that combines export tariffs with public investments. All chapters enrich the existing literature with a more nuanced understanding of the influence of specific elements of political institutions on deforestation processes, as well as a more in-depth discussion of forest conservation policy options for institutionally weak countries.

It has been acknowledged in the policy discussion on REDD+ that it is challenging to guarantee forest conservation without addressing central institutional problems. This dissertation suggests that the analysis of institutional problems can become more concise and oriented towards policy applications in order to provide scientific policy advice.

6.2 Policy implications

A central implication of the research conducted in the context of this dissertation is that addressing the more structural causes of deforestation, through for example institutional or fiscal reforms, bears great potential for an effective reduction of deforestation drivers. Bringing this finding back to the policy sphere, a central remaining question is in how far such activities, aiming at addressing the more systematic causes of deforestation, are compatible with REDD+. It is thus of interest to examine whether the current international REDD+ policy framework enables countries to undertake such activities supported by REDD+ funds and where barriers to such more structural approaches to forest conservation can be found. Decision 1/CP.16 (2010) requests developing countries to develop national strategies or action plans to address "*the drivers of deforestation and forest degradation, land tenure issues, forest governance issues*". However, this decision does not imply how such activities are going to be financed in the future.

While basic agreements on the financial architecture of REDD+ have been reached in the past years, specific agreements on where the bulk of the funding is going to come from, which volumes of funding can be realistically calculated with, and under which conditions it is going to be transferred to the implementing countries are still missing.

Decision 1/CP.16 (UNFCCC, 2010) highlights that results-based funding could come from a variety of sources, including bilateral, multilateral, public, private, and alternative sources (decision 2/CP.17., paragraph 65 UNFCCC, 2011b). This implies non-market approaches (decision 2/CP.17. paragraph 66 UNFCCC, 2011b), such as joint mitigation and adaptation approaches (decision 9/CP.19 UNFCCC, 2013b), as well as market-based approaches (decision 2/CP.17 paragraph 66 UNFCCC, 2011b).

Non-market approaches or public sources have funded the bulk of REDD+ activities so far. Readiness funding has been provided bilaterally or by multilateral agencies, such as UN-REDD and the World Bank's Forest Carbon Partnership Facility (FCPF). While technically, these sources of funding could be used for governance improvements and all sorts of reforms, the grants have in most cases been too small to trigger such transformational changes (the REDD+ Readiness grants provided by the FCPF are usually worth 3.6 million US$ per country for example). Furthermore, there are multilateral and bilateral initiatives that have funded REDD+ programs targeting institutional reforms in the form of improved forest monitoring or land rights mapping (Wibowo and Giessen, 2015). The governments of Norway, the UK, and Germany, for instance, have articulated their commitment to tropical forest conservation and backed it financially with bilaterat support (Barrett and Goldstein, 2015). The decision 2/CP.17 (UNFCCC, 2011b) highlights that joint mitigation and adaptation measures could be used to strengthen governance.

While less straightforward, market based approaches could technically also represent a source of funding for institutional and fiscal reforms for forest conservation. Inasmuch as one could imagine that market-based carbon offset REDD+ payments are transferred to forest conservation project managers, they could also be transferred to jurisdictions or national governments as a compensation for the emission reducing policies they implement. Such a scenario seems plausible given that the only obligation for the receiving party of REDD+ payments is to guarantee that emissions from deforestation or forest degradation are reduced, while the implementing country can by itself sovereignly define the means to achieve such emission reductions. The World Bank's Carbon Fund (2014) has for instance piloted results-based payments contracts with jurisdictional entities such as the state of Oromia in Ethiopia.

Existing emission trading schemes around the world have considered including forest conservation based emission reduction certificates. The EU-ETS or the Californian ETS have so far excluded such emission reduction certificates. It is however possible that in the future a regional or global emission trading scheme accepts forest conservation based emission reduction certificates.

The fact that none of the existing emission trading schemes accepts forest carbon offsets implies that REDD+ countries are continuing with the implementation of their domestic REDD+ policies, without knowing under which conditions emission trading schemes are likely to accept forest carbon offset credits in the future. This situation bears the risk of creating institutional stranded assets.

If the governing jurisdictions of carbon markets are going to accept all types of implementing parties for emission reductions and all types of activities, the path-dependency risk for REDD+ implementing countries is likely very low.

However, in light of the experiences with the Clean Development Mechanism (CDM) (Lederer, 2011), a more plausible scenario appears to be that the hosting legislations of the emission trading schemes will define their own standards to accept emission reduction permits based on forest conservation.

In order to avoid a situation, in which such requirements turn current institutional REDD+ readiness efforts turn into stranded institutional assets, decisions of the UNFCCC on three REDD+ finance questions appear pivotal. These points are (i) the definition of what countries with tropical forests are supposed to be paid for, (ii) the type of eligible contracting entity, and (iii) the type of activities that are going to be acceptable.

Two international policy options on these three points appear feasible, either a UNFCCC decision that guarantees that all diverse approaches to REDD+ must be accepted in future emission trading schemes, or decisions that clarify, as soon as possible, which types of contracting entities or activities can be systematically excluded by authorities that host emission trading schemes that consider to include REDD+.

The following three subsections will discuss why these three points are of fundamental importance for future prospects to address the more structural causes of deforestation through REDD+.

6.2.1 What does the UNFCCC mean by positive incentives?

REDD+ is supposed to provide payments to developing countries as a compensation for reduced deforestation. The idea behind this concept is that deforestation and resulting CO_2 emissions represent a negative externality in the form of climate change to the international community. A typical collective action problem presents itself in the context of the REDD+ mechanism: how much do the two parties (international community and forest-rich countries) have to contribute respectively in order to solve the problem? Where to draw the lines between domestic and international responsibilities?

The Yasuní initiative in Ecuador, for instance, took a relatively clear stance on the question. In order to conserve the Yasuní national park in Ecuador and to refrain from oil exploitations in the region, the Ecuadorian government requested industrialized countries for financial compensation for at least half of the foregone revenue, based on their historical responsibility on climate change (climate debt) (Larrea and Warnars, 2009). However, the international community was unwilling to accept the deal and refused to make the payments. This case highlights that in this emerging field of international environmental policy the responsibilities are not at all clearly defined.

The UNFCCC decisions do not yet provide an answer to these questions. Different concepts have been used to refer to the idea of compensation for forest conservation throughout the negotiations that could have fundamentally different implications: in their initial proposal, Papua New Guinea and Costa Rica referred to the term "*financial compensation*" for developing countries (UNFCCC, 2005). COP13 uses the term of "*positive incentives*" (decision 2/CP.13 UNFCCC, 2007). The appendix to (decision 1/CP.16, e. - i. UNFCCC, 2010) furthermore notes that REDD+ activities should be undertaken in "*accordance with national development priorities*" and "*national sustainable development needs and goals*".

According to Chipman and Moore (2008) "*the compensation principle holds that one of two possible states constitutes an improvement over the other if the gainer could compensate the losers for their losses and still be at least as well off as in the original state.*" According to this definition, the term compensation payment in the UNFCCC context could imply that the marginal return to conservation only has to be equal to the marginal return to alternative deforestation-intensive land uses, such as agriculture.

The term positive incentives however, could refer to an entirely different idea. Varian (2010) argues that for incentive payment contracts to be Pareto improving, two conditions need to be fulfilled. First of all, a *participation constraint* (Laffont and Martimort, 2001) needs to be met. The participation constraint implies that the utility of the REDD+ contract for all participating parties must be at least equal to the utility of both parties without it. This first condition corresponds to the concept of compensation. The second condition however, the *incentive compatibility constraint* (Hammond, 1987), requires that the incentives to participate must be larger than other options. This implies that the marginal return to forest conservation must be larger - and not only equal to - the alternative revenue from conversion to agriculture.

The concept of positive incentives therefore is more demanding and provides forest rich developing countries with reasons for higher expectations for financial compensation than the formulation of compensation payments. Varian (2010) argues that three parameters are crucial to determine whether these two conditions (participation and incentive comparability constraint) can be fullfilled and whether hence the agreement is Pareto improving.

First of all, it depends in this context on each party's willingness to pay for forest conservation. This requires for parties to define, which reserve price they would be individually willing to pay/or accept for forest conservation. This is a challenging endeavor. While some argue that REDD+ could be a relatively low cost option for emission reductions (e.g. Kindermann et al. (2008) 5 US$/ton CO_2), others argue that the costs for REDD+ are generally underestimated and should be expected to be much higher (Gregersen et al., 2010).

Based on which criteria should a developing forest rich country set its REDD+ reserve price? This is where the third formulation of REDD+ payments becomes relevant (decision 1/CP.16, e. - i. UNFCCC, 2010)): if REDD+ payments should be in "*accordance with national development goals*", it needs to be understood in how far REDD+ could potentially be in misalignment with national development goals and how high the associated additional compensation payments would have to be. More than one third of REDD+ countries are economically specialized in the agricultural sector and thus rely to a certain extent on deforested land as an input to economy activity (for details see chapter 5). Consequently, forest conservation without reduced agricultural output levels (and thus compatibility with development goals) is challenging in the absence of economic diversification.

On the one hand, it could be argued that the forest rich country would not depart from its deforestation intensive development trajectory in the absence of the REDD+ payments. As in the Yasuní case, it is argued that industrial countries have caused the problem of climate change and that developing countries have an equal right to development and thus implicitly deforestation and that any deviation from this development trajectory should be financially compensated at least to a important extent by the international community. Along these lines and assuming that a forest rich country undertakes no forest conservation based on its own initiative, Ollivier (2012) for instance argues that there is a risk of a REDD+ Laffer curve: while payments are increasing agricultural productivity and welfare to a certain extent, beyond a critical threshold (meaning very high REDD+ payments) they do not offset the foregone revenue from agriculture production.

Along similar lines, a developing forest rich country could set a very high reserve price for REDD+ possibly involving for example agricultural technology transfers, because it could be argued that agricultural development is necessary for its economy to undergo structural change (Kongsamut et al., 1997) and that international forest conservation payments fail to provide the economy with the skills it would acquire through a usual development process (Hidalgo and Hausmann, 2009).

On the other hand, arguing that the forest rich country bears the damage of the policy through foregone economic development is simplified way to frame the problem. A reserve price could also take into account that the implementing country actually also experiences domestic benefits such as less soil erosion, or less climate change damages from forest conservation. So the tropical country could also acknowledge domestic benefits of forest conservation and chose a lower reserve price.

Similar problems arise in the willingness to pay calculation of the international donor. Based on which criteria should an industrial country set its reserve price for forest conservation? To what extent can it request shared efforts? For instance, to which extent can it request that governments or jurisdictions of forest rich countries only receive REDD+ payments, if they agree to abolish domestic subsidies that actually drive deforestation? Which level of domestic effort of a forest rich country can it realistically request?

A second parameter that influences the Pareto optimality of the REDD+ contract (Varian, 2010) is the initial distribution of wealth between the two involved countries. What does the marginal cost of forest conservation mean for each contracting party, given its total budget?

Finally, Varian (2010) highlights that two aspects related to the method of incentive payment contract making can undermine the Pareto efficiency of the contract. These two elements are (i) that the formulated reserve prices do not correspond to the true willingness to pay of each party and (ii) that there is information asymmetry between parties. An efficient level of forest conservation would occur, if the sum of the respective marginal willingnesses to pay for the common good, would equal the marginal cost of an extra unit of forest conservation.

In light of the different possible interpretations of what is meant by REDD+ payments and corresponding reserve prices, including possible distortions, it is unclear what parties to the UNFCCC mean by decisions that state that funding should be adequate, predictable (decision 1/CP.16), and additional (decision 2/CP.17 UNFCCC, 2011b).

To summarize the central question is, whether REDD+ payments are only meant to provide compensation payments and thus fulfill the participation constraint of forest rich developing countries, or whether they should provide positive incentives and thus represent incentives that are larger than the economic alternatives to forest conservation. If the latter was the case, then a more in-depth understanding of likely development trajectories, as well as the domestic costs and benefits of forest conservation needs to be developed. Applying the concept of positive incentives thus is more likely to offer opportunities to address the more structural drivers of deforestation in forest rich developing countries.

6.2.2 Results-based payments for whom?

According to the UNFCCC decision 10/CP.19 (2013a), paragraph 2, the national focal points nominate the entities that receive results-based payments. While such a national entity could nominate only private forest carbon project managers, it could also nominate public entities, such as local or national governments. The forest carbon credit supplying entity is supposed to provide a service (forest conservation or restoration) to the demanding entity (international donor).

The literature mostly refers to domestic definitions of entities entitled to results-based payments with the term carbon rights (Palmer, 2011; Lederer, 2011; Karsenty et al., 2014). Defining forest carbon rights is challenging, because forest carbon is for the majority of jurisdictions a new type of legal asset.

It is a new type of asset because its legal ownership must not necessarily be bound to the ownership of the tree (Burtraw and Sekar, 2014). Different concession types in the land tenure law of the Democratic Republic of the Congo, for instance, allocate concession rights for different purposes. A *concession d'usufruit*, allows to consume the fruits of the land (article. 132 (DRC, 2002)), but does not grant exclusive ownership rights to the concession holder. Consequently, the owner also has no right to sell the concession. Only a *concession perpétuelle* (art. 96.(DRC, 2002)) grants absolute rights - including the right to sell (*aliénation*) the concession - to the beneficiary.

So the definition of carbon rights could, but does not need to be bound to land rights definitions (Loft et al., 2015). Consequently, a variety of ownership concepts appear feasible. The domestic definition of carbon rights hence needs to correspond to the predominant political philosophy of ownership in the respective country. Of course there is usually in any country a balance of competing principles and in no country the development of new laws follows one principle alone. However, extending existing laws requires building on the principles that appear to be dominanating in existing laws.

Different legal traditions are likely to offer different scopes for more systematic national REDD+ activities. Four concepts are useful to illustrate this point. First, in countries in which ownership rights are defined according to a libertarian philosophical tradition, forest carbon rights are likely to be defined corresponding to the principle of full liberal ownership (Honoré, 1961). Following this philosophical tradition, the rights to land and all corresponding natural resources constitute a natural or historical right of the land owners, which could be an individual or a group (e.g. indigenous community). Such a legal tradition offers no scope for more systematic activities for deforestation reduction, because no legal entity higher than the individual land owner would be entitled to decide over the resource use.

Second, in countries with a left-libertarian political philosophy of land ownership (Stephan and Muller-Furstenberger, 1998), forest carbon rights are likely to be defined individually, but some scope for a jurisdictional entity to regulate the access to forests can remain.

Third, in countries where an egalitarian welfare view is predominant (Beitz, 1979), it might appear justified that there is no absolute right to the land and that for instance a jurisdiction or a national government gets the rights to the forest carbon credits, even though the trees are located on private lands.

Finally, in countries where a utilitarian view on property is dominating, it may appear justified and necessary for a jurisdiction or national government to undertake structural activities for emission reduction from deforestation and be rewarded with the benefits from such activities, even if forests are located on private lands. This would be the case only of course, if forest conservation in the respective country is perceived as welfare enhancing by decision makers.

On an international level, REDD+ is so vaguely defined that it could correspond to all of these normative positions. However, on a country level very different normative traditions can dominate. Nominating these entities and thereby legally defining the ownership of carbon domestically, bears the risk of creating domestic legislations that are incompatible with possible future international carbon market rules.

6.2.3 Results-based payments for which activities?

Different UNFCCC decisions underline the principle of sovereignty for domestic REDD+ strategy making (e.g. (decision 1/CP.16 UNFCCC, 2010); (decision 12/CP.17 UNFCCC, 2011a); (decision 10/ CP.19 UNFCCC, 2013a)), i.e. processes should be country-driven (decision 12/CP.17 UNFCCC, 2011a). Consequently, countries are in theory free to decide domestically how they want to reduce emissions and in theory only have to demonstrate that such emission reductions have actually occurred as a consequence of their activities. Different types of carbon rights definitions are likely to entail different types of emission reduction activities.

If carbon rights are defined exclusively, according to a libertarian normative tradition, the only form of emission reduction activity needs to be undertaken by those exclusive right-holders. Consequently, a national forest conservation strategy can only rely on the atomized efforts of these individual right holders and carbon payments would be transferred directly to these right holders (or right holding communities).

There are a range of cost calculations in the REDD+ context that implicitly assume individual and exclusive form of land ownership (Smith and Scherr, 2003). Accordingly many studies use an individual (or community-based) opportunity cost approach in order to estimate the cost resulting from foregone revenues to individual farmers from avoided deforestation. Concretely, these scenarios calculate the net present value of forest and non-forest land use and estimate, based on this calculation, how high international payments would have to be to pay out what land owners would have earned alternatively from deforestation. The Stern review (2006) for instance applies this logic and finds that emission reductions from avoided deforestation could be of less than 5 US\$/tCO_2. The fifth assessment report of the IPCC (Smith et al., 2014) summarizes different global mitigation potential scenarios using different carbon prices. A carbon price of less than 20 US\$/tCO_2 would have a global mitigation potential of almost 6 GtCO_2 eq/yr (Kindermann et al., 2008), a carbon price of up to 20 US\$/tCO_2 would have a mitigation potential of more than 9 GtCO_2 eq/yr (Sohngen and Sedjo, 2006; Kindermann et al., 2008), a carbon price of up to 100 US\$/tCO_2 would have a mitigation potential of more than 12 GtCO_2 eq/yr in the forestry sector (Sohngen and Sedjo, 2006; Kindermann et al., 2008). Compared to other mitigation options, avoiding deforestation is thus considered to be a low cost mitigation option. For instance, the estimated cost of changing global agricultural tilling practices to reduce one GtCO_2/year is of 27 US\$/tCO_2e (Stern, 2006).

However, scholars that are coming from a rather left-libertarian or egalitarian philosophical tradition criticize that an approach that merely focuses on paying out the revenue form deforestation to the land owner, underestimates the costs for REDD+. Fosci (2013) for instance argues that the implementation costs of the implementing countries are not considered.

Furthermore, Karsenty et al. (2014) point at the role of the transaction costs to fund the administrative side of the projects, because it neglects the transaction costs that would allow reducing emissions. Pfaff et al. (2013) argue: "*Yet claims about the opportunities for forest conservation have likely been overstated, as the sometimes low opportunity cost of land in production is just a lower bound on REDD's costs. There are also costs to clarify land tenure, distribute payments, establish, manage, and monitor protected areas, and reform agriculture and infrastructure policies that affect forests.*"

A utilitarian welfare perspective would build on an entirely different logic. Rather than asking what the individual has to be paid in order to accept not to deforest, it would ask how much a jurisdiction would have to be paid in order to implement and enforce policies that allow guaranteeing forest conservation. This logic has been applied in an analysis by de Souza Cunha et al. (2016) of the expenditure of the Brazilian government for its efforts in forest conservation. The authors conclude that mitigation costs for avoided deforestation for the federal Brazilian government and the municipal and state level costs were between US$ 1.09 and 3.25/tCO_2.

While this approach appears very cost-effective, it has to be taken into account that such costs could potentially be higher in countries with less stable institutions than Brazil. More recent discussions - in particular those that recognize the importance of structural drivers of deforestation stimulate more policy debates in this direction. The UN-REDD program (2015) for instance, recently published a policy brief discussing fiscal incentives for agricultural commodity production and the incompatibility of many fiscal incentives with REDD+ opening a discussion on how fiscal incentives that drive deforestation could be dealt with in the REDD+ context. A policy brief by the Overseas Development Institute (2015) discusses a similar matter and shows that the amount of REDD+ to conserve forests is small compared to the amount of subsidies for agricultural expansion that drive deforestation.

Irawan et al. (2013) discuss the opportunity costs of oil palm plantations in Indonesia and highlights in this context that estimates that focus on the opportunity costs alone do not take the governments revenue derived from such agricultural activities into account. They argue that if such revenue flows are not taken into account (if no equal benefit for the government can be derived from REDD+ funding, as from agricultural activities), public authorities are very unlikely to support REDD+ effectively in its implementation.

Wertz-Kanounnikoff (2008) argues that different cost estimation techniques are suitable for different purposes. While global cost estimates appear useful to frame the discussions, more disaggregated regional or country level estimates appear necessary to correctly estimate the costs for avoided deforestation, also in light of different local forms of implementation.

In the absence of clarity on where the bulk of future funding for REDD+ is going to come from and whether the potentially largest source (markets) will be restricted to a certain type of activity, deciding on future activities for emission reductions from deforestation remains challenging.

6.3 Future research

The research conducted in this dissertation discusses answers to some of the most pressing issues in forest conservation, but also proposes avenues for future research.

The literature on governance and deforestation lacks data and studies that examine the effects of specific elements of forest governance on deforestation. Only if such analyses become more explicit and can more succinctly point to the most relevant governance channels through which forest conservation is guaranteed, policy action can become more targeted. Such more refined analysis could potentially allow making multilaterally supported forest conservation policies that aim at addressing deforestation related governance issues more effective. Studies on the effect of governance on deforestation would furthermore benefit from using data in cardinal units, rather than ordinal units, in order to avoid biased results.

Second, global forest cover change models would benefit from taking the role of political institutions more explicitly into account. As it is also noted in the IPCC's Fifth Assessment Report (Smith et al., 2014), institutions are an important barrier to forest-based mitigation measures. Consequently, forest conservation mitigation cost estimates are likely to underestimate the real costs of forest conservation, because global forest cover models do not take implementation costs related to weak institutions, such as the absence of land tenure rights, or the costs of enforcement into account. Based on a refined understanding and better data on the effect of different levels of institutional quality on deforestation models could take institutional parameters more explicitly into account for future estimates of mitigation potentials.

Third, the dissertation shows that more research needs to be conducted on policy options for forest conservation for institutionally weak countries. In general, more research on opportunities for fiscal and institutional reforms would be useful in order to analyze the implications of different policy pathways. Building on a better understanding of the effect of political institutions on deforestation, future discussions of policy options for forest conservation could compare the effectiveness of different forest conservation strategies in institutionally weak countries.

Fourth, beyond institutional reforms, fiscal reforms could bear the potential to reduce emissions from deforestation more structurally than project-based forest conservation measures as well. Fiscal policy systems could potentially be reformed such that deforestation is reduced and more revenue for the government than through existing fiscal policies could be collected. This revenue could also be redistributed.

In particular, future research should examine in how far agricultural subsidies that drive deforestation could be reformed. Understanding political economy barriers to such reforms and identifying further strategies for fiscal reforms that maintain or improve the situation of all stakeholders, would also be interesting issues for future research.

While the discussion part of this dissertation offers some reflections on how far REDD+ could be used to finance such institutional and fiscal reforms in forest rich countries, future research could explore in how far such reforms could be financed more systematically by REDD+. Furthermore, it could examine under which conditions the governments of forest rich countries would accept to change existing incentive structures. Measuring the costs of REDD+ in this context would involve not only the direct foregone revenue, but also possible forgone future opportunities for economic development. Future research could also examine the concept of positive incentives more explicitly and asses how REDD+ payments can be designed such that they truly represent positive incentives for implementing countries.

The discussion part of this dissertation furthermore highlights that central issues of future bargaining between forest rich countries and industrialized countries on REDD+ emission reduction contracts will be how the costs for emission reductions should be estimated and which types of activities are going to be eligible. While global estimates offer very rough first estimates, regional and local cost estimates, would be useful to have a more realistic and applied discussion of the costs of REDD+.

If such analysis would be undertaken for a range of countries, an understanding could be developed of which type of activities are likely to yield the largest and most sustainable impact on deforestation reduction at the lowest costs. Such marginal abatement cost curves for REDD+ could allow indicating whether there is a risk that future funding for REDD+ is likely to be skewed towards a certain type of activity or region, while underfunding other regions and activities. It would be interesting to analyze whether such an effect exists particularly considering that existing emission trading schemes may accept REDD+ carbon offsets in the future.

If such research indicated the risk that funding is likely to be directed to a particular type of activity or region, it would be interesting to complement the marginal abatement cost framework with further dimensions, such as could be biodiversity priorities, equity, or economic development concerns. Economic development concerns could be, for instance, whether lower levels of economic development in the implementing country lead to larger implementation costs for REDD+ across different implementation strategies. If this was the case one would need to analyze which mechanisms could be found to provide solutions for countries with low levels of economic development as well.

The discussion also highlighted that the question of domestic and international responsibilities for forest conservation could be an interesting subject of future research. While the discussions on REDD+ basically depart from the idea that forest rich countries have no responsibility to reduce deforestation, it also appears difficult to justify that the international community finances forest conservation, while fiscal policies (at much higher orders of magnitude) fuel deforestation domestically in forest rich countries.

Beyond thinking about how incentive structures leading to deforestation can be reformed within a country, it is also a promising avenue to think about demand-side measures that governments of countries that consume deforestation-intensive products could implement, while bearing in mind the effects on sustainable development in forest-rich exporting countries.

References

Barrett, K. and Goldstein, A. (2015). Norway, Germany, United Kingdom pledge $ 5 billion to combat tropical deforestation. URL: http://www.ecosystemmarketplace.com/articles/norway-germany-uk-pledge-5-billion-to-combat-tropical-deforestation/ (accessed 15/03/2017).

Beitz, C. R. (1979). *Political theory and international relations.* Princeton University Press, Princeton, USA.

BioCarbon Fund (2014). Oromia forested landscape program. URL: https://www.biocarbonfund-isfl.org/ethiopia-program (accessed 14/03/2017).

Burtraw, D. and Sekar, S. (2014). Two world views on carbon revenues. *Journal of Environmental Studies and Sciences*, 4(1):110–120.

Chipman, J. S. and Moore, J. C. (2008). Compensation principle. *The New Palgrave. A Dictionary of Economics*, 1(2):online. Eds. Steven N. Durlauf and Lawrence E. Blume. Palgrave Macmillan.

De Souza Cunha, F. A. F., Börner, J., Wunder, S., Cosenza, C. A. N., and Lucena, A. F. (2016). The implementation costs of forest conservation policies in Brazil. *Ecological Economics*, 130(2016):209–220.

DRC (2002). Loi numéro 011/2002 du 29 août 2002, portant code forestier en République Démocratique du Congo. URL: http://leganet.cd (accessed 11/03/2014).

FAO and PROFOR (2011). Framework for assessing and monitoring forest governance. URL: http://www.fao.org/docrep/014/i2227e/i2227e00.pdf (accessed 23/02/2017).

Fosci, M. (2013). Balance sheet in the REDD+: are global estimates measuring the wrong costs? *Ecological Economics*, 89:196–200.

Gregersen, H., El Lakany, H., Karsenty, A., and White, A. (2010). *Does the opportunity cost approach indicate the real cost of REDD+? Rights and realities of paying for REDD+.* Rights and Resources Initiative, Washington DC, USA.

Hammond, P. J. (1987). Markets as constraints: multilateral incentive compatibility in continuum economies. *The Review of Economic Studies*, 54(3):399–412.

Hartmann, H. and Reimann, S. (2010). Bertelsmann Transformation Index 2010 - political management in international comparison. URL: http://www.bti2010.bertelsmann-transformation-index.de/bti/ (accessed 25/01/2016).

Hidalgo, C. A. and Hausmann, R. (2009). The building blocks of economic complexity. *Proceedings of the National Academy of Sciences*, 106(26):10570–10575.

Honoré, A. M. (1961). Ownership. In Gordon Guest, A., editor, *Oxford Essays in Jurisprudence*, volume 107, pages 107–147. Oxford University Press Oxford, New York, USA.

Irawan, S., Tacconi, L., and Ring, I. (2013). Stakeholders' incentives for land-use change and REDD+: the case of Indonesia. *Ecological Economics*, 87(2013):75–83.

Karsenty, A., Vogel, A., and Castell, F. (2014). "Carbon rights", REDD+ and payments for environmental services. *Environmental Science & Policy*, 35(2014):20–29.

Kindermann, G., Obersteiner, M., Sohngen, B., Sathaye, J., Andrasko, K., Rametsteiner, E., Schlamadinger, B., Wunder, S., and Beach, R. (2008). Global cost estimates of reducing carbon emissions through avoided deforestation. *Proceedings of the National Academy of Sciences*, 105(30):10302–10307.

Kissinger, G. (2015). Fiscal incentives for agricultural commodity production: options to forge compatibility with REDD+. *UN-REDD Programme Policy Brief*, 7(2015):online.

Kongsamut, P., Rebelo, S., and Xie, D. (1997). Beyond balanced growth. *National Bureau of Economic Research (NBER) Working Paper*, 1997(6159):1–57.

Laffont, J.-J. and Martimort, D. (2001). *The theory of incentives: the principal-agent model.* Princeton University Press, Princeton, USA.

Larrea, C. and Warnars, L. (2009). Ecuador's Yasuni-ITT initiative: avoiding emissions by keeping petroleum underground. *Energy for Sustainable Development*, 13(3):219–223.

Lederer, M. (2011). From CDM to REDD+ what do we know for setting up effective and legitimate carbon governance? *Ecological Economics*, 70(11):1900–1907.

Loft, L., Ravikumar, A., Gebara, M. F., Pham, T. T., Resosudarmo, I. A. P., Assembe, S., Tovar, J. G., Mwangi, E., and Andersson, K. (2015). Taking stock of carbon rights in REDD+ candidate countries: concept meets reality. *Forests*, 6(4):1031–1060.

McFarland, W., Whitley, S., and Kissinger, G. (2015). *Subsidies to key commodities driving forest loss: implications for private climate finance*, volume (ODI). Overseas Development Institute, London, United Kingdom.

Ollivier, H. (2012). Growth, deforestation and the efficiency of the REDD mechanism. *Journal of Environmental Economics and Management*, 64(3):312–327.

Palmer, C. (2011). Property rights and liability for deforestation under REDD+: implications for 'permanence' in policy design. *Ecological Economics*, 70(4):571–576.

Pfaff, A., Amacher, G. S., and Sills, E. O. (2013). Realistic REDD: improving the forest impacts of domestic policies in different settings. *Review of Environmental Economics and Policy*, 7(1):114–135.

Porter, M. E., Delgado, M., Ketels, C., and Stern, S. (2008). *Moving to a new Global Competitiveness Index. The Global Competitiveness Report.* Number 2008. World Economic Forum, Geneva, Switzerland.

Smith, J. and Scherr, S. J. (2003). Capturing the value of forest carbon for local livelihoods. *World Development*, 31(12):2143–2160.

Smith, P. M., Bustamante, M., H. Ahammad, H. Clark, H. Dong, E.A. Elsiddig, H. Haberl, R. Harper, J. House, M. Jafari, O. Masera,, and C. Mbow, N.H. Ravindranath, C.W. Rice, C. Robledo Abad, A. Romanovskaya, F. Sperling, and F. Tubiello (2014). Agriculture, forestry and other land use (AFOLU). In *Climate change 2014: mitigation of climate change. Contribution of working group III to the fifth assessment report of the Intergovernmental Panel on Climate Change.* IPCC, Cambridge University Press, Cambridge, United Kingdom.

Sohngen, B. and Sedjo, R. (2006). Carbon sequestration in global forests under different carbon price regimes. *The Energy Journal*, 27(Special Issue: Multi-Greenhouse Gas Mitigation and Climate Policy):109–126.

Stephan, G. and Muller-Furstenberger, G. (1998). The double dividend of carbon rights. *International Journal of Environment and Pollution*, 10(3-4):485–494.

Stern, N. (2006). *Stern review: the economics of climate change.* The National Archives (HM Treasury), London, United Kingdom.

UNFCCC (2005). Submissions from parties (Costa Rica and Papua New Guinea). Reducing emissions from deforestation in developing countries: approaches to stimulate action. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2005/Misc.1).

UNFCCC (2007). Decision 2/CP.13. Reducing emissions from deforestation in developing countries: approaches to stimulate action. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2007/6/Add.1).

UNFCCC (2010). Decision 1/CP.16. The Cancun agreements: outcome of the work of the ad hoc working group on long-term cooperative action under the convention. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2010/7/Add.1).

UNFCCC (2011a). Decision 12/CP.17. Report of the conference of the parties on its seventeenth session, held in Durban from 28 november to 11 december 2011 addendum part two: action taken by the conference of the parties at its seventeenth session. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2011/9/Add.2).

UNFCCC (2011b). Decision 2/CP.17. Outcome of the work of the ad hoc working group on long-term cooperative action under the convention. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2011/9/Add.1).

UNFCCC (2013a). Decision 10/CP.19. Coordination of support for the implementation of activities in relation to mitigation actions in the forest sector by developing countries, including institutional arrangements. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2013/10/Add.1).

UNFCCC (2013b). Decision 9/CP.19. Coordination of support for the implementation of activities in relation to mitigation actions in the forest sector by developing countries, including institutional arrangements. *United Nations Framework Convention on Climate Change*, (FCCC/CP/2013/10/Add.1).

Varian, H. R. (2010). *Intermediate microeconomics: a modern approach*, volume 8. W.W. Norton & Company, New York, USA.

Wertz-Kanounnikoff, S. (2008). Estimating the costs of reducing forest emissions: a review of methods. *Center for International Forestry Research (CIFOR), Bogor, Indonesia Working Paper*, 2008(42):1–22.

Wibowo, A. and Giessen, L. (2015). Absolute and relative power gains among state agencies in forest-related land use politics: the ministry of forestry and its competitors in the REDD+ programme and the one map policy in Indonesia. *Land Use Policy*, 49(2015):131–141.

World Bank (2015). Doing business - measuring business regulations. URL: http://data.worldbank.org/datacatalog/doingbusinessdatabase (accessed 19/11/2016).

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Tools and resources

All chapters of this thesis were written with LaTeX using Miktex (Version 2.9) and Texmaker (Version 4.5). All used data sources are indicated in the respective chapters.

Introduction:

- Figure 1 was published in FAO (2016) and has been reproduced with permission.
- Figure 2 has been published in Smith et al (2014) and has been reproduced with permission.
- Figure 3 was produced by the author using the MathWorks Software MATLAB (Version R2016a).

Chapter 2:

- The analysis in chapter 1 was undertaken using Stata (Version 14).
- Figure 1 was produced by the authors using Microsoft Office (Word) (Version 2013).
- Figure 2 was produced by the authors using Microsoft Office (Excel) (Version 2013).

Chapter 3:

- The analysis in chapter 2 was undertaken using Stata (Version 14).
- The Global Forest Model (G4M) is written in C++.
- Figure 1 was produced by the authors using Microsoft Office (Word) (Version 2013).
- Figure 2,3,4, and 5 were produced by the authors using Microsoft Office (Excel) (Version 2013).

Chapter 4:

- Figure 1 is adapted from a figure published in Geist and Lambin (2002).
- The data was generated and evaluated using Microsoft Office (Word) (Version 2013).
- Intercoder reliability was calculated using ReCal2.

Chapter 5:

- The numerical analyses in chapter 4 were undertaken using Microsoft Office (Excel), MathWorks Software MATLAB (Version R2016a).
- Figure 1, 2, and 4 were produced by the authors using Microsoft Office (Excel).
- Figure 3, 5, and 6 were produced by the authors using the MathWorks Software MATLAB (Version R2016a).

References

FAO (2016). *Global Forest Resources Assessment 2015 - how are the world's forests changing?* Food and Agricultural Organization of the United Nations (FAO), Second edition. The figure is reproduced with permission, online source: http://www.fao.org/publications/card/en/c/770b9ce69f654a46a9af-1abbb0f71ab1/ (accessed (15/11/2016).

Geist, H. J. and Lambin, E. F. (2002). Proximate causes and underlying driving forces of tropical deforestation. *BioScience*, 52(2):143–150.

Smith, P. M., Bustamante, M., H. Ahammad, H. Clark, H. Dong, E.A. Elsiddig, H. Haberl, R. Harper, J. House, M. Jafari, O. Masera,, and C. Mbow, N.H. Ravindranath, C.W. Rice, C. Robledo Abad, A. Romanovskaya, F. Sperling, and F. Tubiello (2014). Agriculture, forestry and other land use (AFOLU). In *Climate change 2014: mitigation of climate change. Contribution of working group III to the fifth assessment report of the Intergovernmental Panel on Climate Change.* IPCC, Cambridge University Press, Cambridge, United Kingdom.

List of publications

All articles forming part of this dissertation are in the publication process or published with the below indicated journals.

Part I: Political institutions in deforestation processes

- Chapter 2: Wehkamp J., Koch N., Luebbers S., Fuss S. (2018). Governance and deforestation - a meta-analysis in economics. *Ecological Economics*, 144(2018):214-227. DOI: https://doi.org/10.1016/j.ecolecon.2017.07.030
- Chapter 3: Wehkamp J., Pietsch S. A., Fuss S., Gusti M., Reuter W. H., Koch N., Kindermann G., Kraxner F. (under review). Accounting for institutional capacity in global forest modeling. Under review in the journal Environmental Modelling and Software. Reproduction of the article as part of a dissertation is permitted according to ELSEVIER's policies: https://www.elsevier.com/about/our-business/policies/copyright/personal-use

Part II: Forest conservation in institutionally weak countries

- Chapter 4: Wehkamp J., Aquino A., Fuss S., Reed W. E., (2015). Analyzing the perception of deforestation drivers by African policy makers in light of possible REDD+ policy responses. *Forest Policy and Economics*, 59(2015):7-18. DOI: https://doi.org/10.1016/j.forpol.2015.05.005
- Chapter 5: Schwerhoff G., Wehkamp J. (under review). Export tariffs combined with public investments as a forest conservation policy instrument. Under review in the journal *Agricultural Economics*. The editor Prof. Abdulai did not object the online publication forming part of the dissertation project.