Land use spillovers of bioeconomy-driven trade shocks under imperfect environmental law enforcement

J. Börner[1]; M. Bruckner[2]; R. Flach[3]; B. Soares-Filho[4]; S. Wunder[5]

1: University of Bonn, Institute for Food and Resource Economics, Germany, 2: Vienna University of Economics and Business, , Austria, 3: University of Hamburg, , Germany, 4: Federal University of Minas Gerais, , Brazil, 5: Center for International F

Corresponding author email: jborner@uni-bonn.de

Abstract:

Green growth strategies and bioeconomic technological innovation affect global demand and supply of agricultural and forestry-based commodities. What trade-mediated impacts has this fledging transformation on land-use change at ecologically sensitive tropical forest margins? Standard global trade models only provide impact assessments at aggregate regional scales, implicitly assuming either perfect or zero environmental enforcement. However, emerging empirical impact evaluations suggest that conservation policies only partially constrain illegal land conversion with highly variable effectiveness in space. We present a spatially explicit cropland allocation tool simulating imperfectly functioning conservation policies. We shock cropland allocation with a land demand scenario derived from a multi-regional input-output model to assess land-use spillovers under two common policy scenarios of imperfect environmental enforcement under spatial heterogeneity: (1) protection of specific flagship biomes through protected area networks or (2) cost-efficient enforcement in accessible zones immediately threatened by illegal agricultural expansion. Both scenarios result in land use spillover effects, but combining the two strategies does not generally perform better than flagship biome protection alone. Outcomes depend on country-specific spatial distributions of returns to cropland expansion, law enforcement costs, and environmental service provision. In closing, we discuss the implications of our findings for land-use governance in a globalized bioeconomy.

Acknowledgement: This work was supported by the Robert Bosch Foundation, the German Federal Ministry of Education and Research, the German Federal Ministry for Economic Cooperation and Development, and is part of CIFOR's Global Comparative Study on REDD+

JEL Codes: Q57, Q51

#1397

IAAE

Land use spillovers of bioeconomy-driven trade shocks under imperfect environmental law enforcement

Abstract

Green growth strategies and bioeconomic technological innovation affect global demand and supply of agricultural and forestry-based commodities. What trade-mediated impacts has this fledging transformation on land-use change at ecologically sensitive tropical forest margins? Standard global trade models only provide impact assessments at aggregate regional scales, implicitly assuming either perfect or zero environmental enforcement. However, emerging empirical impact evaluations suggest that conservation policies only partially constrain illegal land conversion with highly variable effectiveness in space. We present a spatially explicit cropland allocation tool simulating imperfectly functioning conservation policies. We shock cropland allocation with a land demand scenario derived from a multi-regional input-output model to assess land-use spillovers under two common policy scenarios of imperfect environmental enforcement under spatial heterogeneity: (1) protection of specific flagship biomes through protected area networks or (2) cost-efficient enforcement in accessible zones immediately threatened by illegal agricultural expansion. Both scenarios result in land use spillover effects, but combining the two strategies does not generally perform better than flagship biome protection alone. Outcomes depend on country-specific spatial distributions of returns to cropland expansion, law enforcement costs, and environmental service provision. In closing, we discuss the implications of our findings for land-use governance in a globalized bioeconomy.

1 Introduction

Amid global per capita income and population growth, industrialized and developing countries around the world increasingly adopt strategies to reduce fossil fuel dependency by expanding renewable resource use (Staffas et al. 2013; Meyer 2017). Biomass from agriculture, forestry, and fisheries will thus play an ever more important role as a source of food, feed, fiber, and fuel in the future global bioeconomy (Slade et al. 2011; Deininger 2013). As in the 20th century, innovation-induced boosts in primary sector productivity and biomass use efficiency may alleviate though not entirely neutralize the corresponding additional pressure on land and land-based ecosystem services (Tilman et al. 2011; Popp et al. 2017). The efficient use of the world's productive land resources, balancing human needs across all types of terrestrial ecosystem services as well as national boundaries, thus becomes a major global governance challenge.

The challenge is not new, however. Already today, the costs of land degradation due to land use and land cover change (LUCC) are estimated at USD 231 billion annually, half of which accrue as externalities, such as carbon emissions and biodiversity loss (Nkonya et al. 2016). Between 2003 and 2012, above ground biomass carbon loss was largest in tropical forests, i.e. 230 million tons per year or roughly 2.4% of global fossil fuel based carbon emissions in 2012 (Liu et al. 2015).

An unknown, but large share of the degradation and deforestation of the world's tropical forests and savannahs violates existing national environmental legislation (Benatti, da Cunha Fischer 2017; Graham et al. 2017; Wehkamp et al. 2015). Important reasons for the poor enforcement of environmental law at the world's tropical forest margins, such as lack of financial and human resources, insufficient local institutional capacity and contrary development interests, policy incoherence or corruption have been discussed in a growing theoretical and empirical literature (Ceddia et al. 2014; Robinson et al. 2010; Ceddia et al. 2013).

Standard economic theory dictates that each hectare of productive land should be put to its best use considering a global social welfare function. Clearly, such allocation would require an unrealistic degree of alignment of national policies and development strategies. Improving forest and environmental law enforcement strategies in developing and emerging economies thus comes to be a promising and potentially powerful complementary strategy to avoid uncontrolled illegal deforestation and the associated social and environmental costs.

In this paper we study the potential role of land use spillovers from spatially heterogeneous policy enforcement. We explore conditions under which imperfect forest and environmental law enforcement in biomass exporting tropical countries can aggravate rather than alleviate the impact of deforestation on the provision of globally valued ecosystem services. This happens basically because enforcement diminishes the more we move towards market-remote lands, which pushes illegal land conversion and land grabbing towards the agricultural frontier, where in turn the environmental values (e.g. of relatively dense tropical forests) are highest. We proceed in two steps:

First, we extend the basic von Thünen land rent model to account for enforcement that diminishes moving towards the margins, due to the rise in field-based enforcement costs – a pattern that has been shown to dominate in tropical forest settings. The model helps to explain one key underlying mechanism of environmental policy spillovers or leakage and motivates our hypothesis that such spillovers can mean that growingly imperfect enforcement at frontiers results in less environmental service provision than both 'no enforcement' and 'full enforcement' strategies: environmental benefits go up and enforcement efforts go down simultaneously as we move toward the frontier thus reinforcing a suboptimal land allocation pattern with too much occupation of environmentally sensitive frontier lands.

In a second step, we develop and implement a simple global spatially explicit land allocation model to simulate the effect of alternative stylized enforcement scenarios on global ecosystem service provision given a positive shock in EU oilseed demand. In principle, this land allocation model is well suited for coupling with standard economy-wide and multi-regional trade models and could thus contribute to enhancing the reliability of global impact assessments by relaxing the common assumption of spatially homogeneous environmental policy incentives at national scale. Model results support the hypothesis that poor law enforcement can be worse than no enforcement in terms of key globally valued ecosystem service outcomes. However, both direction and size of the effect are highly sensitive to policy goals and associated enforcement strategies as well as country-specific spatial patterns of infrastructure, cropland, and primary forest cover.

The remainder of the paper is structured as follows. Section 2 conceptually demonstrates how imperfect environmental law enforcement may lead to land use spillover effects. In section 3, we document data and modelling approaches. Results are presented in section 4 and discussed in section 5. Section 6 concludes with implications for policy.

2 Imperfect environmental law enforcement in the *Von Thünen* land rent model

Von Thünen's land rent framework remains a popular conceptual approach to explain deforestation at and around agricultural frontiers in the tropics (Angelsen 2010; Kalkuhl, Edenhofer 2017). According to the framework, land rents, defined as revenues minus costs, reduce as distance to markets increases (Figure 1, upper panel) up to a point where agricultural production and associated primary forest conversion ceases, i.e. the agricultural frontier.

Formally, this model suggests that all land up to the agricultural frontier must be cleared given positive rents. Yet, in practice we usually observe a gradient from intensive to extensive land uses resulting in a transition from open, to fragmented, to uniform (forest cover) landscapes at increasing distance to markets and urban centers (Chomitz 2006). Forest cover change is typically low close to the markets and urban centers and at the forest frontier, but for different reasons. Around commercial centers most available forest land has already been converted for high-value residential and agricultural or other commercial purposes, whereas deforestation is just not profitable enough around the agricultural frontier. Instead deforestation tends to be highest in the intermediate range of market distance as depicted in gray in Figure 1.

Clearly, a number of factors not captured by the standard land rent framework ultimately determine the actual location and scale of deforestation under specific local settings (Walker 2004). The stylized distribution of deforestation in Figure 1, nonetheless, represents a theoretically consistent landscape scale manifestation of the forest transition curve, where deforestation is highest at intermediate levels of development (Angelsen 2007).

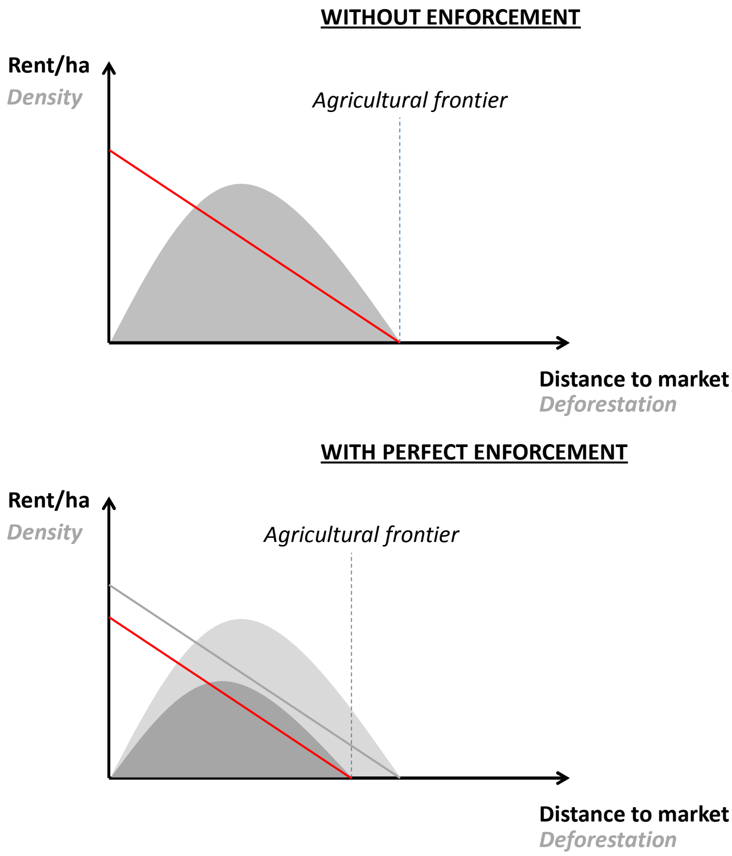

Figure 1: Land rent (red line) and distribution of annual change in forest cover (grey shaded areas) without (upper panel) and with (lower panel) perfect environmental law enforcement

Most policies to combat deforestation are intended to tip land users' rent calculus in favor of less or zero forest conversion. For example, disincentive-based instruments, such as protected areas or use restrictions on private land are typically enforced through fines or asset confiscation. If enforced perfectly across space, such policies will shift per hectare rents downwards and the agricultural frontier to the left as in the lower panel of Figure 1. Correspondingly, we would expect less forest loss and a relocation of deforestation hotspots to zones that are somewhat closer to the market. Incentive-based policies, such as payments for environmental services would have an equivalent effect on land rents by increasing the opportunity costs of forest conversion, which are assumed to be zero in our framework under the no enforcement baseline scenario (Figure 1 upper panel).

In rural areas of developing and emerging countries, however, such spatially homogeneous delivery of policy incentives is not the norm. Robinson et al. (2010) distinguish between full and incomplete enforcement to motivate the notion that reducing illegal activity to zero is seldom an economically optimal enforcement strategy. Since enforcement is costly and tends to exhibit diminishing returns to effort, it is often neither viable nor desirable to punish each and every law offense.

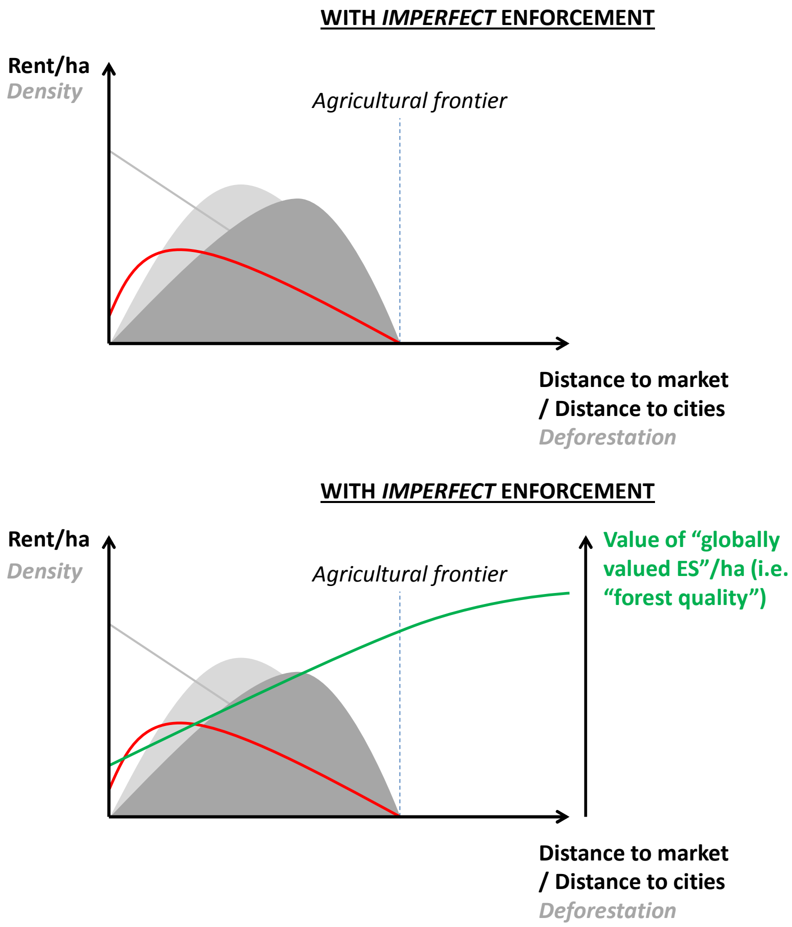

Figure 2: Land rent (red line) and distribution of annual change in forest cover (grey shaded areas) with imperfect environmental law enforcement (upper panel) and implications for ecosystem service provision (green line, lower panel).

The costs of enforcing environmental land use regulations are inherently linked to spatial characteristics. For example, Albers (2010) shows formally that optimal enforcement of protected areas in developing country settings involves balancing rural welfare with budgetary tradeoffs between protected area size, shape, and enforcement costs. Environmental law enforcement in remote inaccessible areas (such as agricultural frontiers) is generally costlier than in infrastructure dense regions (i.e. close to urban settlements), leading to spatially varying levels of enforcement effort and thus probability of punishment (Börner et al. 2014).

Following Albers (2010) and Börner et al. (2015a), this notion is depicted in Figure 2, where we assume constant levels of nominal fines per hectare of deforested land and adjust land rents by a penalty that diminish as enforcement costs increase, and correspondingly, probability of detection and punishment reduces with distance to urban centers. Depending on assumptions about the functional forms of the land rent and enforcement costs, the resulting adjusted land rent curve may look like the red line in Figure 2, i.e. relocating peak rents away from zero distance to markets.

We expect that this imperfect (i.e. incomplete) enforcement scenario will affect the spatial distribution of deforestation by shifting the mode towards the agricultural frontier (Figure 2). The size and relevance of this effect depends on the functional forms of the land rent and enforcement cost curves.

The potential spatial relocation of deforestation in response to the imperfect enforcement shock constitutes a spatial spillover effect. Whether or not spatial spillovers are relevant depends on their overall impacts on ecosystem service provision. Figure 2 (lower panel) illustrates a common spatial configuration of tropical forest frontiers where forest quality, and thus carbon storage and biodiversity related option, existence, and amenity values increase with distance to markets and correspondingly lower levels of anthropogenic impacts. Deforestation including imperfect enforcement spillovers may here, all else equal, produce an overall higher loss of ecosystem services than in the baseline scenario without enforcement. Clearly, the opposite would happen in contexts where environmental quality reduces with distance to markets.

Considering equilibrium effects and farm-level response to environmental policy incentives, the size of the spillover effect will depend on: (1) the elasticity of output prices with stronger spillovers when prices increase in response lower levels of production due to environmental policy enforcement, and (2) the elasticity of input demand with stronger spillovers when land use intensity is unresponsive to increases in the costs of primary forest conversion.

In short, the theoretical discussion above shows that imperfectly enforced environmental land use regulations may produce negative environmental net-effects compared to no law enforcement under some combinations of the factors captured by our land rent model. In the remainder of this paper we explore how these factors may play out under simulations of real world conditions.

3 Data & Methods

Our stylized modelling approach relies on two separate tools that are documented below. First, we rely on a multi-regional input output (MRIO) model to identify current land demand for the production of biofuels and vegetable oils, i.e. important globally traded resources for the bioeconomy. We then calculate country-level cropland expansion based on projected EU demand for these bio-based resources in 2020. We are aware of limitations of this approach, as it ignores potential demand and supply response via global commodity markets. However, here we are primarily interested in country level (not aggregate global) responses: the MRIO allows us to cover a large number of countries with different landscape configurations, i.e. the key outcome determinants of land use spillover processes.

Second, increases in cropland demand are allocated within countries based on a simple algorithm that identifies the most profitable units of available land in the proximity of existing cropland. Profitability is expressed in terms of conversion probability and imperfect enforcement is simulated by adjusting conversion probabilities as indicated in Figure 2 above. We then determine changes in spatially explicit indicators of ecosystem service provision assuming conversion of native vegetation for cropland in these land units.

Bioeconomy trade shocks spurring land demand

In order to track land use from crop production via several trade and processing steps to the non-food use of vegetable oils and bioethanol, we rely on a global multi-regional physical input–output table (mrPIOT) covering production and use of crops, vegetable oils and ethanol.

The mrPIOT documents the flows of crops and derived products circulating through the global economy, as proposed by Bruckner et al. (2015). The system merges the production, trade, and utilization data for agricultural products provided by the Food and Agriculture Organization Corporate Statistical Database (FAOSTAT 2017) into a physical input–output table covering 175 countries.

Figure 3 shows the hypothetical supply chains of vegetable oils for non-food purposes in two countries (A and B), thereby illustrating how the FAO data were used to track global biomass flows. In this example, country A produces rapeseed while country B produces soybeans which are partly exported to country A. All oilseeds are used for the production of vegetable oils. The conversion efficiency of oilseeds into oils is based on the FAO's country-specific technical conversion factors (FAO 2003). Besides being produced domestically, soybean oil is also imported into country A from country B. Finally, a certain share of the domestic supply of each vegetable oil is reported in the FAO's commodity balance sheets to be used for non-food purposes (FAO 2001).

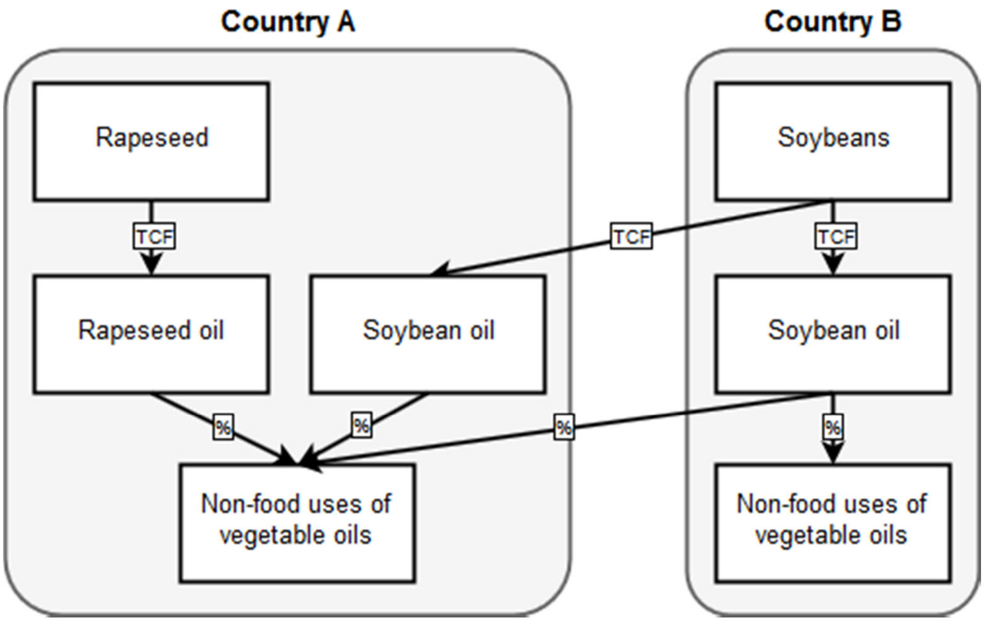

Figure 3: Schematic illustration of hypothetical supply chains of vegetable oils used for non-food purposes in two countries. TCF = technical conversion factor. % = non-food share.

The resulting international commodity tree structure for 175 countries and 35 commodities was implemented in matrix format, thus constituting an mrPIOT. Input–output relations for all processing steps were defined based on technical conversion factors. A Leontief inverse (Leontief 1936) derived from this mrPIOT was then multiplied with a land use extensions, documenting the hectares of physical land used per ton of crop produced, thereby calculating the land footprint of non-food uses of vegetable oils and bioethanol in each country.

Input–output models have fixed technical coefficients and cannot address dynamic aspects of the economy such as price changes and resulting structural changes and their effects on trade volumes. The assumption of the Leontief model of a linear marginal relationship between a sector's output and its intermediate inputs (i.e. a change in a sector's output requires proportional changes in its consumption of intermediate inputs), while appropriate for the ex-post assessment of biomass flows, is critical when conducting an ex-ante analysis, as it implies the absence of scale effects (Ferng 2009). However, as (Ferng 2009) acknowledges, the application of input–output coefficients to scenario analysis would be appropriate when the production techniques can be assumed to remain unchanged and the input flows are measured in physical terms, which is the case for the model we intend to set up.

Despite of the existing shortcomings, environmental input–output analysis was used before for scenario analyses, for example, for the assessment of the environmental impacts of different diet scenarios for the EU (Tukker et al. 2011) considering changes in final demand, and for the analysis of the long term environmental pressures resulting from technology changes by changing both industry input requirements and environmental intensities of production (Wilting et al. 2008). Besides, population, affluence and technology have been described as the most fundamental drivers of environmental pressures (Ehrlich, Holdren 1971). These variables are represented in input–output models and can be easily modified in order to implement a set of scenario assumptions. Therefore, it has been argued that IO models provide a useful framework to investigate future scenarios (Barrett, Scott 2012; Guan et al. 2008).

In IO models, population and affluence and resulting consumption levels and patterns are represented by the final demand vectors. By changing the magnitude and composition of household and government expenditures for different products, the impact of assumed future consumption profiles can be analysed. This can be operationalised by introducing changes into the final demand matrix of a country. While some studies assume technology to remain constant (Kronenberg 2009; Tukker et al. 2011; Washizu, Nakano 2010) others also adapt technology coefficients (Hubacek, Sun 2001; Wilting et al. 2008), or even introduce new products by adding rows and columns to the variables of the IO model (Malik et al. 2014).

Finally, the Leontief model assumes static supply shares, meaning that an additional demand resulting from a set of scenario assumptions will increase current supplies proportionally, disregarding for example competitiveness and production capacities of the suppliers. This is particularly challenging for the analysis of future land use and biomass flows, as production and supply are constrained and will therefore differ substantially from current patterns in order to adapt to future demands. Different modelling approaches have been employed, including the mixed IO model in combination with the so called RAS technique (Hubacek, Sun 2001; Kerschner, Hubacek 2009), dynamic input–output models (Cruz et al. 2009), and fuzzy optimization (Aviso et al. 2011; Tan et al. 2012).

For this study, we modelled land use change as the result of an increase in EU final demand for non-food uses of vegetable oils and bioethanol by 9.33 % and 46.23 % respectively for the EU. These growth shares refer to the period from 2011 to 2020 and were obtained from global projections (OECD-FAO Agricultural Outlook 2016-2025 2016).

Land use change allocation and impact assessment

To allocate additional cropland in a given country we assume that spatial units of land are chosen based on expected return and accessibility vis-à-vis existing cropland. Expected returns (π) to agriculture are measured in USD per hectare and accessibility in terms of the Euclidean distance (*d*) to the nearest existing cropland unit. Conversion probability is then defined by the joint probability density function of expected returns and accessibility:

$$f_{\Pi,D}(\pi,d) = f_{\Pi|D}(\pi,d) f_D(d) \tag{1}$$

To simulate imperfect enforcement we penalize conversion probabilities by stylized functions of travel time that represent enforcement probabilities (Börner et al. 2014). We consider two types of enforcement strategies that mimic enforcement practice in the real world (Figure 4). "least cost" enforcement minimizes enforcement costs subject to a budget constraint and thereby implicitly prioritizes accessible (low costs of access for field-based enforcement action) over inaccessible (high access costs) areas. This results in spatial enforcement probability distributions similar to the gray and black curves in Figure 3, which have been observed by Börner et al. (2015) in Brazil. "Flagship biome" enforcement maximizes deterrence in formally protected areas, which are often declared in and around environmentally valuable landscapes (Pouzols et al. 2014). The green line in Figure 3 illustrates an extreme form of a flagship biome protection strategy.

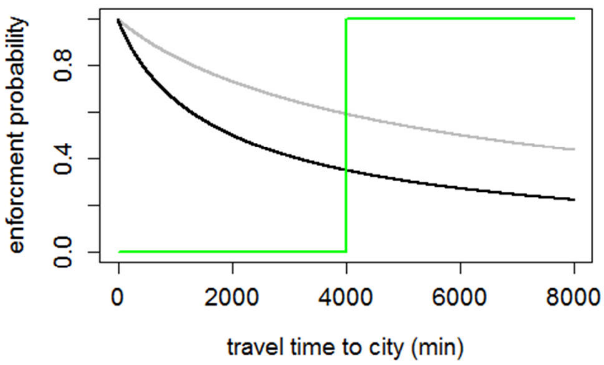

Figure 4: Enforcement probability as a function of travel time to cities. Enforcement scenarios: "least cost" (gray & black) and "flagship biome" (green).

Multiplying conversion probabilities with the inverse of the enforcement probabilities results in a shift of cropland expansion to remoter areas of a country, i.e. the land use spillover effect.

To assess the environmental impact of native vegetation loss due to cropland expansion we overlay converted spatial units with global maps of ecosystem service indicators (see Table 1 for all data sources). Differences between losses and gains in the 'with' and 'without enforcement' scenarios of cropland expansion are considered net-effects of land use spillovers.

Table 1: Data sources and use in cropland land allocation model and impact assessment

Data Type (global maps)	Purpose	Source
Agricultural revenue potential [USD/ha]	Estimation of conversion probability	Naidoo and Iwamura (2007)
Global land cover	Estimation of conversion probability	Arino et al. (2012)
Accessibility (travel time for enforcement missions) [hours to nearest urban center]	Estimation of enforcement probability	Nelson (2008)
Protected Areas	Estimation of enforcement probability	UNEP-WCMC and IUCN (2017)
ABG biomass carbon [C in above ground biomass / ha]	Impact assessment	Ruesch and Gibbs (2008)
Ecosystem service value [USD/ha]	Impact assessment	Costanza et al. (2014)
Species equivalents / area unit	Impact assessment	Kier et al. (2009)

4 Results

We present simulations of spillover effects only for developing and emerging economies that export relevant amounts of the energy crop products we have focused on (Figure 5). Industrialized economies are excluded, because they generally exhibit high levels of compliance with land use regulations with illegal conversion of native vegetation being the exception of the norm. Most of the countries included in simulations dispose of significant amounts of carbon and/or biodiversity rich tropical and dry forests and projected cropland expansion lies in the range of 100 to 1000 sqkm.

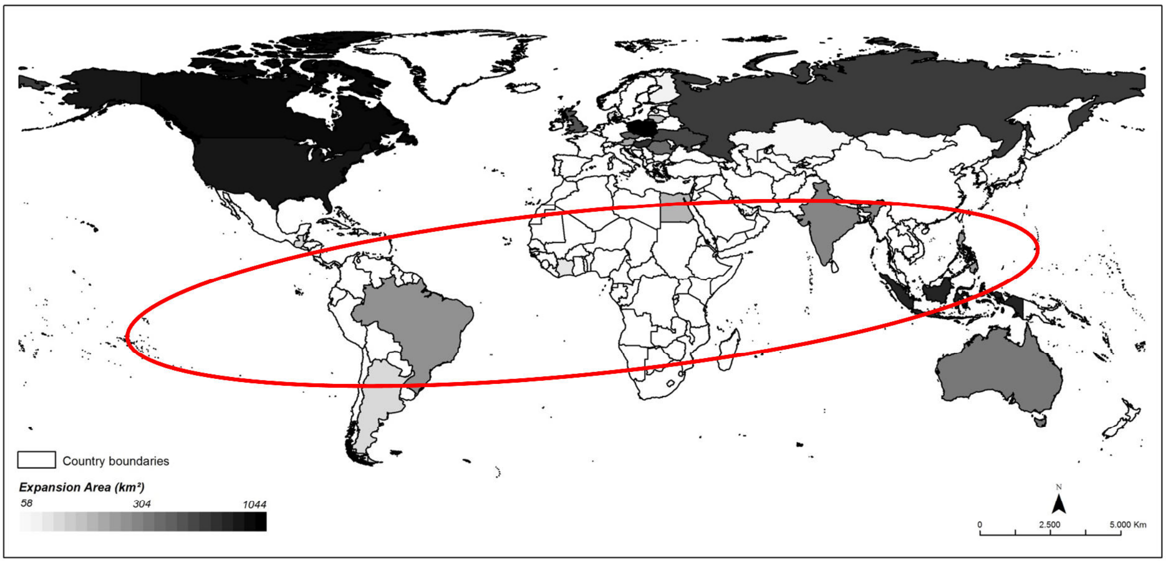

Figure 5: mrPIOT predicted cropland expansion due to projected EU demand for vegetable oil and biofuels in 2020. The red circle identifies countries included in spillover simulations.

As expected we find that net-effects of enforcement induced land use spillovers differ between enforcement strategies and countries.

Results of the 'least cost' enforcement scenario (see Figure 4, black line) are summarized in Figure 6. Effect sizes are not necessarily proportional to area expansion, but also depend on the spatial location of cropland expansion and the corresponding spatial distribution of ecosystem service quality. For example, projected cropland expansion is larger in Brazil than in Argentina (Figure 5), but this difference is reflected only in spillover-induced above ground carbon losses. Ecosystem services losses in monetary terms are approximately equal in Brazil and Argentina and larger in Argentina than in Brazil for endemic species equivalents. This is explained by the difference between the major biomes in which cropland expansion is predicted to expand in the two countries. In Brazil, spillover expansion takes place in the carbon rich Cerrado and Amazon biomes, whereas in Argentina it is concentrated in biomes with lower levels of above-ground biomass, but high endemic species diversity, such as the Pampa and the Chaco. All ecosystem service indicators exhibit large losses for Indonesia, both due to the large amount of projected cropland expansion, but mainly because of high biomass carbon densities and endemic species richness in spillover expansion areas.

India and Guatemala are special cases. In India, imperfect enforcement is beneficial as it shifts land use to regions with generally lower levels of ecosystem service provision than in the baseline scenario without enforcement. In Guatemala on the other hand, spillovers lead to net carbon gains, but overall losses in ecosystem service value and endemic species equivalents.

Lower levels of enforcement pressure, such as illustrated by the gray line in Figure 4, lead to almost proportionally smaller net-effects.

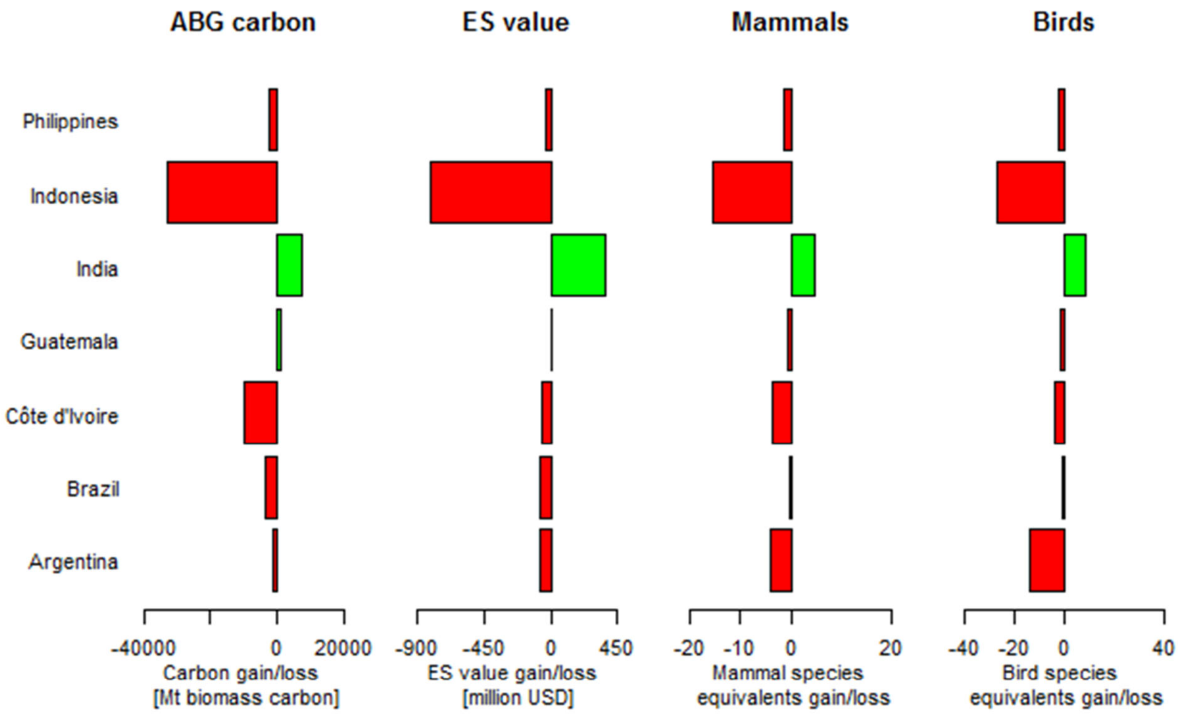

Figure 6: Spillover effects of 'least cost' enforcement in terms of above-ground carbon (ABG), ecosystem service (ES) value as well as Mammal and Bird species equivalents.

The 'flagship biome' enforcement scenario (Figure 4, green line) produces smaller and distinctly different net spillover effects (Figure 7). Net-effects are generally smaller because well protected areas remove an important share of environmentally sensitive land from the total area available for spillover expansion in every country. The size of effects then depends on the remaining amount of productive and at the same time environmentally valuable land. Hence, spillover losses are still comparatively large in Indonesia where the protected area network is distributed such that considerable amounts of unused land with high ecosystem service value remains available for spillover expansion. In the case of Brazil, enforcement spillover effects turn from net losses in the 'least cost' scenario into gains in the 'flagship biome' scenario, because of the relatively dense protected area network covering carbon dense forest landscapes in the Amazon region. The opposite happens in India, whereas spillover effects in most other countries are negligible. From an environmental perspective, the 'flagship biome' approach thus comes to be a strategy that is less prone to negative environmental spillover effects of land use than the 'least cost' enforcement strategy.

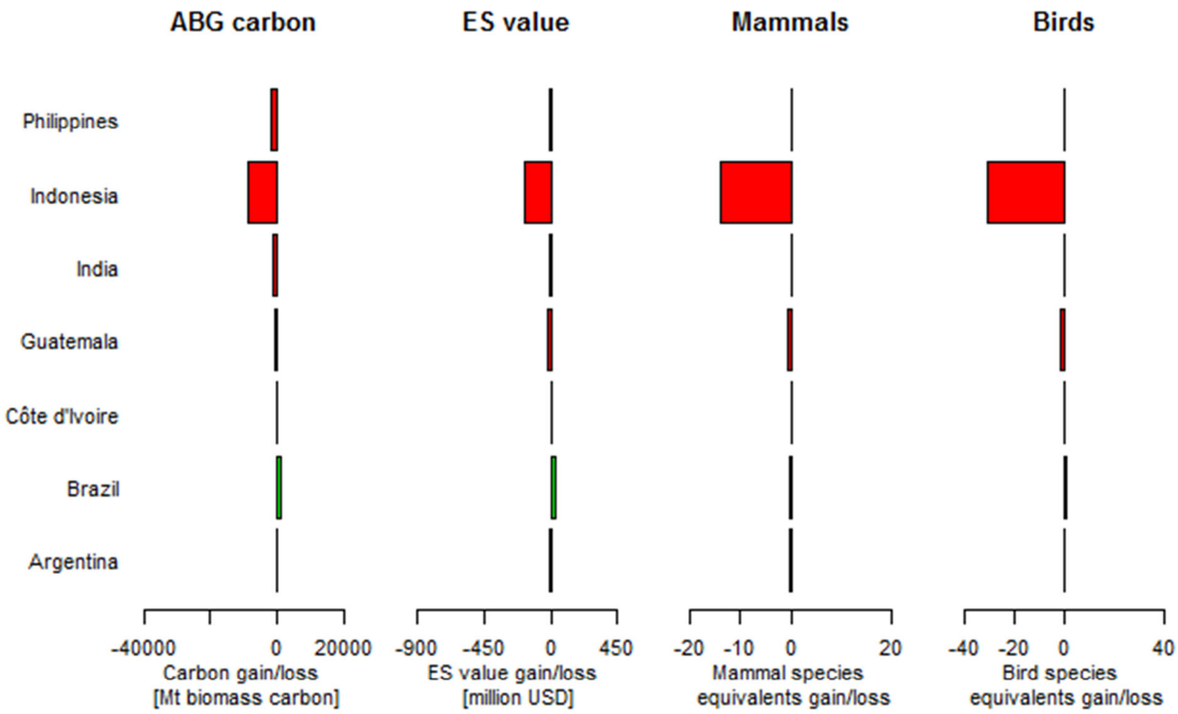

Figure 7: Spillover effects of 'flagship biome' enforcement in terms of above-ground carbon (AGB), ecosystem service (ES) value as well as Mammal and Bird species equivalents.

In our third scenario, we look at a combined 'least cost' and 'flagship biome' enforcement strategy, where enforcement probability is equal to one in protected areas and equal to the black line in Figure 4 otherwise. Results are presented in Figure 8 and, perhaps surprisingly, suggest that enforcement spillover effects are not unequivocally lower than in any of the two single strategy solutions despite overall higher enforcement probabilities. The reason is that flagship biome protection does not remove all potentially productive remote land from countries' pools of land for agricultural expansion. Combined with 'least cost' enforcement, flagship biome protection thus shifts cropland expansion to the remaining remotely located productive land units producing outcomes that are similar to the results in Figure 6. One exception is Argentina, where the combined enforcement scenario produces net gains in ecosystem service value and above ground carbon, probably because of lower ecosystem service provision in dryer parts of the country.

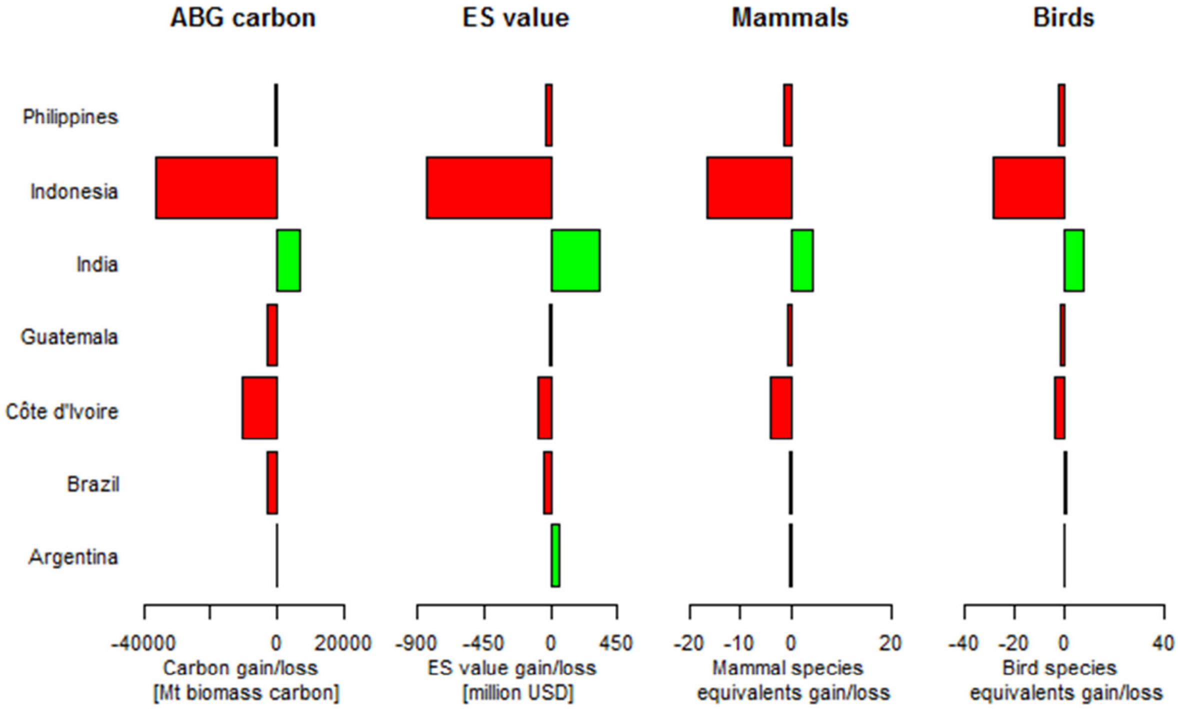

Figure 8: Spillover effects of combined 'least cost' and 'flagship biome' enforcement in terms of above-ground carbon (AGB), ecosystem service (ES) value as well as Mammal and Bird species equivalents.

In sum, these simulations indicate that land use spillover effects due to imperfect enforcement of environmental land use regulations are country-specific and can be negative and non-trivial. Comparing alternative enforcement strategies suggests in addition that scope exists to minimize spillover effects if enforcement effectiveness can be increased in landscapes with high ecosystem service values. A number of caveats apply given the simplicity of our modelling framework and will be discussed below.

5 Discussion

As indicated at the end of section 2, a number of well-known economic mechanisms are so far excluded from both our theoretical framework and the stylized modelling exercise. Since we are generally not worried about positive environmental spillovers from imperfect enforcement, such as in most of our simulations for India, we focus our discussion about how these mechanisms would alter our findings in country settings where we expect predominantly negative spillover effects.

First, the environmental policy incentive increases the cost of converting native vegetation to cropland. Since our model ignores the possibility of substituting capital and labor for land, it likely overestimates, all else equal, the environmental spillover effects. Even considering that the production of the most important crops for biofuel and vegetable oil production (i.e., sugar cane, soy, and oil palm) tends to occur at relatively high levels of capital intensity, some degree of capital and labor substitution in response to policy enforcement is expected and would reduce the potential amount of land use expansion, and thus, spillover outcomes.

Second, significant enforcement-induced production shortfalls, however, may affect output prices depending on the elasticity of demand for globally traded commodities. Such upward adjustment of global market prices for biofuels and vegetable oil products could partially neutralize the effect of the above-mentioned intensification response at farm-level on spillover outcomes (Sá et al. 2012).

Third, our conceptual considerations in section 2 led us to assume that deforestation incentives relocate toward the agricultural frontier in response to imperfect enforcement pressure. Such relocation, of course, requires mobility of capital assets and labor, which may be sticky for institutional and cultural reasons. Immobility of assets and labor would result in lower spillover effects that our simulation model suggests.

Fourth, relatively little empirical information exists about the actual enforcement strategies adopted by environmental protection agencies in developing and emerging countries (Robinson et al. 2010). Our stylized implementation of enforcement scenarios thus necessarily represents a generalization that may over or underestimate actual spillover effects on the ground.

Finally, our simulations are only as good as the underlying data sets. There is a tradeoff with respect to country level accuracy and detail versus consistency of data sources at the global level. The global data sets used to simulate land use allocation and impacts on ecosystem service provision have limitations with implications for the analysis of spatial spillover effects (Verburg et al. 2011). For example, spatial aggregation of locally heterogeneous levels of soil quality in the agricultural revenue data set may lead us to over or underestimate the relocation of land use depending on whether aggregate spatial units are biased towards low or high quality of land. The same essentially applies for the global data sets of ecosystem service indicators. Especially the ecosystem service value and biodiversity indicators are provided at very high levels of aggregation.

6 Conclusion

We have conceptually identified a potential mechanism for land use, or more specifically, illegal deforestation spillovers from imperfect enforcement at agricultural frontiers in forest rich developing and emerging countries. Such spatial spillovers can occur when the distribution of enforcement pressure along land rent gradients affects the spatial probability distribution of deforestation, such that forest conversion becomes more likely closer to the agricultural frontier. In landscapes where the value of ecosystem services is higher at and around such frontiers than in the proximity to markets and urban centers, the enforcement spillover can produce net-losses in ecosystem service provision vis-à-vis a no enforcement scenario.

We find this pattern in countries like Brazil and Indonesia (see Figure 6), globally relevant producers of soy, sugar cane ethanol, and palm oil, where development has historically spared large tracts of dense tropical forests with high levels of biomass carbon and species endemism. But, effects can go the other way as shown for Argentina and India (Figures 6 and 6), when imperfect enforcement pushes illegal land use change towards dry or previously degraded landscapes with lower ecosystem service values than in regions where agriculture dominates in the baseline. Aggregate effects at global level where not analyzed and would require a coupled trade and land use model that captures equilibrium effects.

Beyond country contexts, spillover outcomes are driven by enforcement strategies. We find that a ‘least cost’ enforcement strategy that prioritizes enforcement in easily accessible agricultural landscapes produces more pronounced environmental spillovers than ‘flagship biome’ protection strategy that targets environmentally valuable landscapes at and around the agricultural frontier. This particular finding is supported by the growing empirical literature on the effectiveness of protected area networks (Geldmann et al. 2015).

A key policy message arising from our findings is that more enforcement is not necessarily better enforcement. It matters how enforcement efforts are allocated in space, because illegal land use change is a moving target driven by (1) changes in global demand for bio-based products, (2) local economic incentives affecting the profitability of forest conversion, such as infrastructure development, and (3) the spatial distribution of policy incentives.

Implications arise for both national environmental policy implementation and value-chain-based governance schemes that have recently received great attention by both state and non-state actors across the world (Lambin et al. 2014). For example, investments in land use monitoring technology and flexible on-the-ground enforcement action can increase detection probability uniformly across space thus reducing incentives for illegal deforestation also at the agricultural frontier (Börner et al. 2015b). International funding under UN climate policy, such as the for the REDD+ mechanism, should be targeted to such national scale measures, because their benefits accrue primarily in terms of global public goods. Improved real-time forest cover monitoring is also likely to make conservation incentives, such as payments for environmental services, more effective in complementing law enforcement (Börner et al. 2015c). And, last but not least, certification and labelling schemes as well as voluntary commitments must collaborate closely with national environmental enforcement agencies in so called ‘hybrid governance’ arrangements (Viana et al. 2016) to counteract undesirable spillover outcomes at agricultural frontiers.

Publication bibliography

Albers, H. J. (2010): Spatial modeling of extraction and enforcement in developing country protected areas. In *Resource and Energy Economics* 32 (2), pp. 165–179.

Altenburg, Tilman; Agrawal, Nand Kishor; Faust, Joerg; Goltz, Nicolaus von der; Schloms, Michael; Kattermann, Dieter et al. (2007): From Project to Policy Reform Experiences of German Development Cooperation. In *Deutsches Institut fuer Entwicklungspolitik*.

Angelsen, Arild (2007): Forest cover change in space and time. Combining the von Thünen and forest transition theories: World Bank Publications.

Angelsen, Arild (2010): Policies for reduced deforestation and their impact on agricultural production. In *Proceedings of the National Academy of Sciences* 107 (46), pp. 19639–19644. DOI: 10.1073/pnas.0912014107.

Arino, Olivier; Ramos Perez, Jose Julio; Kalogirou, Vasileios; Bontemps, Sophie; Defourny, Pierre; van Bogaert, Eric (2012): Global Land Cover Map for 2009 (GlobCover 2009): PANGAEA.

Aviso, Kathleen B.; Tan, Raymond R.; Culaba, Alvin B.; Cruz, Jose B. (2011): Fuzzy input–output model for optimizing eco-industrial supply chains under water footprint constraints. In *Journal of Cleaner Production* 19 (2-3), pp. 187–196. DOI: 10.1016/j.jclepro.2010.09.003.

Barrett, John; Scott, Kate (2012): Link between climate change mitigation and resource efficiency. A UK case study. In *Global Environmental Change* 22 (1), pp. 299–307. DOI: 10.1016/j.gloenvcha.2011.11.003.

Benatti, Jose Heder; da Cunha Fischer, Luly Rodrigues (2017): New trends in land tenure and environmental regularisation laws in the Brazilian Amazon. In *Regional Environmental Change*, pp. 1–9.

Börner, J.; Marinho, E.; Wunder, S. (2015a): Mixing Carrots and Stickstoff Conserve Forests in the Brazilian Amazon. A Spatial Probabilistic Modeling Approach. In *PLoS ONE* 10 (2), pp. e0116846.

Börner, J.; Wunder, S.; Wertz-Kanounnikoff, S.; Hyman, G.; Nascimento, N. (2014): Forest law enforcement in the Brazilian Amazon. Costs and income effects. In *Global Environmental Change* 29, pp. 294–305.

Börner, Jan; Kis-Katos, Krisztina; Hargrave, Jorge; König, Konstantin (2015b): Post-Crackdown Effectiveness of Field-Based Forest Law Enforcement in the Brazilian Amazon. In *PLoS ONE* 10 (4), pp. e0121544 EP -. DOI: 10.1371/journal.pone.0121544.

Börner, Jan; Marinho, Eduardo; Wunder, Sven (2015c): Mixing Carrots and Sticks to Conserve Forests in the Brazilian Amazon. A Spatial Probabilistic Modeling Approach. In *PLoS ONE* 10 (2), pp. e0116846. DOI: 10.1371/journal.pone.0116846.

Bruckner, Martin; Fischer, Günther; Tramberend, Sylvia; Giljum, Stefan (2015): Measuring telecouplings in the global land system. A review and comparative evaluation of land footprint accounting methods. In *Ecological Economics* 114, pp. 11–21. DOI: 10.1016/j.ecolecon.2015.03.008.

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

Ceddia, Michele Graziano; Bardsley, Nicholas Oliver; Gomez-y-Paloma, Sergio; Sedlacek, Sabine (2014): Governance, agricultural intensification, and land sparing in tropical South America. In *Proc. Natl Acad. Sci. USA* 111 (20), pp. 7242–7247.

Chomitz, Kenneth M. (2006): At Loggerheads? Agricultural Expansion, Poverty Reduction, and Environment in the Tropical Forests. Washington D.C.: World Bank ((None)).

Costanza, Robert; Groot, Rudolf de; Sutton, Paul; van der Ploeg, Sander; Anderson, Sharolyn J.; Kubiszewski, Ida et al. (2014): Changes in the global value of ecosystem services. In *Global Environmental Change* 26, pp. 152–158.

Cruz, Jose B.; Tan, Raymond R.; Culaba, Alvin B.; Ballacillo, Jo-Anne (2009): A dynamic input–output model for nascent bioenergy supply chains. In *Applied Energy* 86, S86-S94. DOI: 10.1016/j.apenergy.2009.04.007.

Deininger, Klaus (2013): Global land investments in the bio-economy. evidence and policy implications. In *Agricultural Economics* 44 (s1), pp. 115–127.

Ehrlich, P. R.; Holdren, J. P. (1971): Impact of population growth. In *Science* 171 (3977), pp. 1212–1217.

FAO (2001): Food balance sheets. Rome: Food and Agriculture Organisation of the United Nations.

FAO (2003): Technical conversion factors for agricultural commodities. Rome: Food and Agriculture Organization of the United Nations.

FAOSTAT (2017): FAO Statistical Databases: Agriculture, Fisheries, Forestry, Nutrition. Rome: Statistics Division, Food and Agriculture Organization of the United Nations.

Ferng, Jiun-Jiun (2009): Applying input–output analysis to scenario analysis of ecological footprints. In *Ecological Economics* 69 (2), pp. 345–354. DOI: 10.1016/j.ecolecon.2009.08.006.

Geldmann, Jonas; Coad, Lauren; Barnes, Megan; Craigie, Ian D.; Hockings, Marc; Knights, Kathryn et al. (2015): Changes in protected area management effectiveness over time. A global analysis. In *Biological Conservation* 191, pp. 692–699.

Graham, Victoria; Laurance, Susan G.; Grech, Alana; Venter, Oscar (2017): Spatially explicit estimates of forest carbon emissions, mitigation costs and REDD+ opportunities in Indonesia. In *Environmental Research Letters* 12 (4), p. 44017.

Guan, Dabo; Hubacek, Klaus; Weber, Christopher L.; Peters, Glen P.; Reiner, David M. (2008): The drivers of Chinese CO2 emissions from 1980 to 2030. In *Global Environmental Change* 18 (4), pp. 626–634. DOI: 10.1016/j.gloenvcha.2008.08.001.

Hubacek, Klaus; Sun, Laixiang (2001): A scenario analysis of China's land use and land cover change. Incorporating biophysical information into input–output modeling. In *Structural Change and Economic Dynamics* 12 (4), pp. 367–397. DOI: 10.1016/S0954-349X(01)00029-7.

Kalkuhl, Matthias; Edenhofer, Ottmar (2017): Ramsey meets Thünen. The impact of land taxes on economic development and land conservation. In *International Tax and Public Finance* 24 (2), pp. 350–380.

Kerschner, Christian; Hubacek, Klaus (2009): Assessing the suitability of input–output analysis for enhancing our understanding of potential economic effects of Peak Oil. In *Energy* 34 (3), pp. 284–290. DOI: 10.1016/j.energy.2008.07.009.

Kier, Gerold; Kreft, Holger; Lee, Tien Ming; Jetz, Walter; Ibisch, Pierre L.; Nowicki, Christoph et al. (2009): A global assessment of endemism and species richness across island and mainland regions. In *Proceedings of the National Academy of Sciences* 106 (23), pp. 9322–9327.

Kronenberg, Tobias (2009): The impact of demographic change on energy use and greenhouse gas emissions in Germany. In *Ecological Economics* 68 (10), pp. 2637–2645. DOI: 10.1016/j.ecolecon.2009.04.016.

Lambin, Eric F.; Meyfroidt, Patrick; Rueda, Ximena; Blackman, Allen; Börner, Jan; Cerutti, Paolo Omar et al. (2014): Effectiveness and synergies of policy instruments for land use governance in tropical regions. In *Global Environmental Change* 28, pp. 129–140.

Leontief, Wassily W. (1936): Quantitative Input and Output Relations in the Economic Systems of the United States. In *Review of Economics and Statistics* 18 (3), p. 105. DOI: 10.2307/1927837.

Liu, Yi Y.; van Dijk, Albert IJM; Am Jeu, Richard de; Canadell, Josep G.; McCabe, Matthew F.; Evans, Jason P.; Wang, Guojie (2015): Recent reversal in loss of global terrestrial biomass. In *Nature Climate Change* 5 (5), pp. 470–474.

Malik, Arunima; Lenzen, Manfred; Ely, Rômulo Neves; Dietzenbacher, Erik (2014): Simulating the impact of new industries on the economy. The case of biorefining in Australia. In *Ecological Economics* 107, pp. 84–93. DOI: 10.1016/j.ecolecon.2014.07.022.

Meyer, Rolf (2017): Bioeconomy Strategies. Contexts, Visions, Guiding Implementation Principles and Resulting Debates. In *Sustainability* 9 (6), p. 1031.

Naidoo, Robin; Iwamura, Takuya (2007): Global-scale mapping of economic benefits from agricultural lands. implications for conservation priorities. In *Biological Conservation* 140 (1), pp. 40–49.

Nelson, A. (2008): Travel time to major cities. A global map of Accessibility. Ispra Italy. Available at http://gem.jrc.ec.europa.eu/ (Global Environment Monitoring Unit - Joint Research Centre of the European Commission.).

Nkonya, Ephraim; Anderson, Weston; Kato, Edward; Koo, Jawoo; Mirzabaev, Alisher; Braun, Joachim von; Meyer, Stefan (2016): Global cost of land degradation. In : Economics of Land Degradation and Improvement–A Global Assessment for Sustainable Development: Springer, pp. 117–165.

OECD-FAO Agricultural Outlook 2016-2025 (2016): OECD Publishing (OECD-FAO Agricultural Outlook).

Popp, Alexander; Calvin, Katherine; Fujimori, Shinichiro; Havlik, Petr; Humpenöder, Florian; Stehfest, Elke et al. (2017): Land-use futures in the shared socio-economic pathways. In *Global Environmental Change* 42, pp. 331–345.

Pouzols, Federico Montesino; Toivonen, Tuuli; Di Minin, Enrico; Kukkala, Aija S.; Kullberg, Peter; Kuusterä, Johanna et al. (2014): Global protected area expansion is compromised by projected land-use and parochialism. In *Nature* 516 (7531), pp. 383–386.

Robinson, Elizabeth J. Z.; Kumar, Ajay M.; Albers, Heidi J. (2010): Protecting Developing Countries' Forests. Enforcement in Theory and Practice. In *Journal of Natural Resources Policy Research* 2 (1), pp. 25–38.

Ruesch, Aaron; Gibbs, Holly K. (2008): New IPCC Tier-1 global biomass carbon map for the year 2000.

Sá, Saraly Andrade de; Palmer, Charles; Engel, Stefanie (2012): Ethanol production, food and forests. In *Environmental and Resource Economics* 51 (1), pp. 1–21.

Slade, Raphael; Saunders, R.; Gross, R.; Bauen, A. (2011): Energy from biomass. the size of the global resource. In *Imperial College Centre for Energy Policy and Technology and UK Energy Research Centre, London*.

Staffas, Louise; Gustavsson, Mathias; McCormick, Kes (2013): Strategies and Policies for the Bioeconomy and Bio-based Economy. An analysis of official national approaches. In *Sustainability* 5 (6), pp. 2751–2769.

Tan, Raymond R.; Aviso, Kathleen B.; Barilea, Ivan U.; Culaba, Alvin B.; Cruz, Jose B. (2012): A fuzzy multi-regional input–output optimization model for biomass production and trade under resource and footprint constraints. In *Applied Energy* 90 (1), pp. 154–160. DOI: 10.1016/j.apenergy.2011.01.032.

Tilman, David; Balzer, Christian; Hill, Jason; Befort, Belinda L. (2011): Global food demand and the sustainable intensification of agriculture. In *Proceedings of the National Academy of Sciences* 108 (50), pp. 20260–20264. DOI: 10.1073/pnas.1116437108.

Tukker, Arnold; Goldbohm, R. Alexandra; Koning, Arjan de; Verheijden, Marieke; Kleijn, René; Wolf, Oliver et al. (2011): Environmental impacts of changes to healthier diets in Europe. In *Ecological Economics* 70 (10), pp. 1776–1788. DOI: 10.1016/j.ecolecon.2011.05.001.

UNEP-WCMC and IUCN (2017): Protected Planet: [insert name of component database; The World Database on Protected Areas (WDPA)/The Global Database on Protected Areas Management Effectiveness (GD-PAME)] [On-line], October 2017. Cambridge, UK: UNEP-WCMC and IUCN. Available online at http://www.protectedplanet.net/, checked on 10/2017.

Verburg, P. H.; Neumann, K.; Nol, L. (2011): Challenges in using land use and land cover data for global change studies. In *Global Change Biology* 17 (2), pp. 974–989. DOI: 10.1111/j.1365-2486.2010.02307.x.

Viana, Cecilia; Coudel, Emilie; Barlow, Jos; Ferreira, Joice; Gardner, Toby; Parry, Luke (2016): How does hybrid governance emerge? Role of the elite in building a green municipality in the Eastern Brazilian Amazon. In *Environmental Policy and Governance* 26 (5), pp. 337–350.

Walker, Robert (2004): Theorizing land-cover and land-use change. The case of tropical deforestation. In *International Regional Science Review* 27 (3), pp. 247–270.

Washizu, Ayu; Nakano, Satoshi (2010): ON THE ENVIRONMENTAL IMPACT OF CONSUMER LIFESTYLES – USING A JAPANESE ENVIRONMENTAL INPUT–OUTPUT TABLE AND THE LINEAR EXPENDITURE SYSTEM DEMAND FUNCTION. In *Economic Systems Research* 22 (2), pp. 181–192. DOI: 10.1080/09535314.2010.488218.

Wehkamp, Johanna; Aquino, André; Fuss, Sabine; Reed, Erik W. (2015): Analyzing the perception of deforestation drivers by African policy makers in light of possible REDD+ policy responses. In *Forest Policy and Economics* 59, pp. 7–18.

Wilting, Harry C.; Faber, Albert; Idenburg, Annemarth M. (2008): Investigating new technologies in a scenario context. Description and application of an input–output method. In *Journal of Cleaner Production* 16 (1), S102-S112. DOI: 10.1016/j.jclepro.2007.10.017.