

**THESIS FOR THE DEGREE OF LICENTIATE OF PHILOSOPHY**

**Felling Forests from Afar: Quantifying Deforestation  
Driven by Agricultural Expansion and International Trade**

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**Thesis for the degree of Licentiate of Philosophy**

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# Felling Forests from Afar: Quantifying Deforestation Driven by Agricultural Expansion and International Trade

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## **Abstract**

Deforestation is a major source of human-caused greenhouse gas emissions and the largest threat to terrestrial biodiversity. Most forest loss is due to the expansion of agricultural land use increasingly driven by international demand for food, fuel and fibre. However, there is still limited understanding of the extent to which different agricultural commodities are contributing to deforestation. It has therefore also been difficult to evaluate the role of international trade in driving deforestation. This dissertation aims at quantifying the agricultural drivers of tropical deforestation (Papers I and II) and the associated carbon emissions (Paper III). It further assesses the role of international trade, by following the agricultural commodities with embodied deforestation through international supply chains using trade models (Papers II and III). The results show that a few commodity types, primarily cattle meat and oilseed products, account for a large part of tropical deforestation. Much (26–39%) of the embodied deforestation and concomitant emissions were found to be associated with international demand (from products and services). Looking closer at the countries that import embodied deforestation, Paper II finds that many countries that are increasing their forest cover at home, import products associated with deforestation elsewhere, thereby offsetting about a third of their forest gains. Paper III finds that imports of embodied deforestation emissions for many developed countries are similar in size to their national agricultural emissions amounting, e.g., for the EU, to around 15% of the carbon footprint of an average diet. Put together, the results add to the evidence that combating deforestation can benefit from complementing domestic policies with measures that target international demand. The results also indicate that tackling deforestation and its associated impacts at the global level is probably even more challenging than at the national level, although international trade can also provide efficiency gains by optimising land use globally.

**Keywords:** Deforestation, Agriculture, Carbon emissions, International trade, Forestry, Carbon footprints, Land cover, Land use, Forest transitions

## List of publications

**Paper I** Pendrill, F., & Persson, U. M. (2017). Combining global land cover datasets to quantify agricultural expansion into forests in Latin America: Limitations and challenges. *PLOS ONE*, 12(7). doi:10.1371/journal.pone.0181202

UMP conceived the study, with contributions from FP. FP collected and processed the data, carried out the analyses and wrote the paper, the latter with contributions from UMP.

**Paper II** Pendrill, F., Persson, U. M., Godar, J., & Kastner, T. (2019). Deforestation displaced: trade in forest-risk commodities and the prospects for a global forest transition. *Environmental Research Letters*, 14(5). doi:10.1088/1748-9326/ab0d41

UMP conceived the study, with contributions from FP. FP and UMP carried out the deforestation drivers analyses. TK carried out the trade analyses. FP and UMP analysed the results and wrote the paper with contributions from TK and JG.

**Paper III** Pendrill, F., Persson, U. M., Godar, J., Kastner, T., Moran, D., Schmidt, S., & Wood, R. (2019). Agricultural and forestry trade drives large share of tropical deforestation emissions. *Global Environmental Change*, 56, 1-10. doi:10.1016/j.gloenvcha.2019.03.002

UMP conceived the study, with contributions from FP. FP and UMP carried out the deforestation drivers and carbon emissions analyses. TK and SS carried out the trade analyses. FP analysed the results and wrote the paper with contributions from UMP, TK, JG, DM and RW.

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# 1. Introduction

Every year in the tropics, forests amounting to an area larger than the size of the Netherlands (~5–10 million hectares) are cut down or burned (Hansen et al., 2013, FAO, 2016, Curtis et al., 2018). This not only contributes to climate change—deforestation accounts for around a tenth of anthropogenic carbon emissions (Baccini et al., 2017)—but also impacts the livelihoods of people depending on the forest (Sunderlin et al., 2005, Chhatre and Agrawal, 2009). Deforestation also threatens the habitats of a multitude of species, making land-use change the leading driver of terrestrial biodiversity loss (Newbold et al., 2015, Barlow et al., 2016, Tilman et al., 2017, IPBES, 2019). These impacts, along with the attention brought to the persistently high deforestation rates in the tropics, have sparked multiple international commitments, both in the public and private sectors, urgently seeking to reduce pressures on forests. The New York Declaration on Forests, endorsed by many governments and companies alike, aims to halve deforestation by 2020 and halt it by 2030. Even more ambitiously, the United Nations’ Sustainable Development Goals (SDGs) aim to halt deforestation already by 2020. The United Nations Framework Convention on Climate Change (UNFCCC) has developed REDD+ (Reducing Emissions from Deforestation and forest Degradation) as a mechanism to support climate change mitigation, providing results-based financial incentives to developing countries that keep forests standing. The European Commission is currently (2019) looking into ways of stepping up the European Union action on deforestation. In the private sector, multiple companies are making “zero-deforestation” pledges and looking over their supply chains in order to rid them of products causing deforestation (Newton et al., 2013, Grimard et al., 2017, Gardner et al., 2018, Lambin et al., 2018).

Though reducing deforestation on the global scale is clearly an important concern, like all land-use change, deforestation is a matter of trade-offs: When do the benefits provided by the new land use outweigh the downsides of the lost forest? (Of course, the full range of impacts, positive as well as negative, is rarely possible to quantify.) Deforestation may improve the livelihood for the farmer who clears a patch of forest to provide food for their family and community. At the larger scale, much of the world’s agricultural land—which now makes up over 40% of the vegetated land area (FAO, 2017, Poore and Nemecek, 2018)—was at some point in the past deforested and now supplies us with food, feed, fibre, fuel, and more. Instead of seeking to avoid deforestation entirely, addressing deforestation is rather a question of how we can better understand when, where, and under what circumstances the benefits of opening up land for, e.g., agriculture, outweigh the costs—environmental and social—of doing so (DeFries et al., 2004, Foley et al., 2005, Goldstein et al., 2012, Grau et al., 2013, Mehrabi et al., 2018, Meyfroidt, 2018).

To tackle deforestation, it is therefore important to understand why and how it is happening. Currently, most deforestation is due to expanding agriculture and forestry (Hosonuma et al., 2012, Curtis et al., 2018), fuelled by increasing demand for food, energy and fibre (Gibbs et al., 2010, Alexander et al., 2015). And this demand is expected to continue to increase: the world’s population is increasing in both numbers and affluence, and ever more people consume ever more animal-based products (which are particularly land-intensive) (Godfray et al., 2010, Wirsenius et al., 2010, Foley et al., 2011, Tilman et al., 2011, Willett et al., 2019). There is also a growing demand for land to produce bioenergy, in attempts to reduce the greenhouse gas emissions resulting from the reliance on fossil fuels (Miyake et al., 2012, Popp et al., 2014, Creutzig et al., 2015). At the same time as demand is increasing, the patterns of demand are becoming more complex and spatially disconnected from the supply,

as products are increasingly traded internationally, often in multi-stage supply chains (Lambin and Meyfroidt, 2011, Yu et al., 2013, Wood et al., 2018). Such distant interactions—between the demand from human systems and the environmental impacts in natural systems—are sometimes called telecouplings (Liu et al., 2013, Friis et al., 2016). It is therefore vital not only to consider the causes of deforestation at the point of production, but also to understand where—and for what purpose—the demand is arising.

However, despite increasing international trade in agricultural and forestry commodities (Kastner et al., 2011a, Yu et al., 2013, Kastner et al., 2014), the understanding of the patterns of demand for commodities driving tropical deforestation has been limited. A reason for this is the lack of pan-tropical data on which commodities are associated with deforestation. The overarching aim of this dissertation is to identify, in more detail, the causes of deforestation in the tropics. More specifically, the papers in this dissertation aim to:

- Quantify to what extent the expansion of different crops, pastures and tree plantations are contributing to deforestation (and related CO<sub>2</sub> emissions) in tropical and subtropical countries, by examining what land uses follow deforestation, and which agricultural commodities are expanding where deforestation is occurring.
- Determine the relative role of domestic and international demand have in driving deforestation, and further examine the role of international trade in redistributing pressures on forests.

Paper I deals primarily with the first aim, in seeking to use currently available, spatially explicit global datasets (maps) for investigating to what extent forest loss is followed by cropland and pasture expansion. While all the included papers also discuss limitations of current methods and data to represent the drivers of deforestation accurately, this is a main focus of Paper I.

Paper II and III deal with both the aims presented above. In Paper II, deforestation is attributed to agricultural and forestry commodities, while Paper III further quantifies the carbon emissions resulting from the land-use change and attributes these to commodities. Papers II and III then use trade models to trace the impacts (deforestation and deforestation-related CO<sub>2</sub> emissions, respectively) “embodied” in production of these commodities to final consumption. This helps identify which countries are large consumers of products associated with deforestation and determine the relative role of domestic demand compared to international demand (which, among other things, has implications for the potential of different types of measures for preventing deforestation). The use of trade models also makes it possible to take a consumption-based perspective on deforestation and put imports of embodied impacts into the context of consuming countries.

This dissertation introduction begins with a background section, introducing some main concepts relating to deforestation and its causes, as well as ways of advancing our understanding of those causes. This is followed by a synthesis of the results from the included papers, along with a summary of the approach used. A discussion then highlights the key contributions of this dissertation and the relevance of the results. Finally, the papers themselves are included.



## 2. Background

### 2.1 What is deforestation?

Before setting out to determine the causes of deforestation, a key point is to decide what to consider as deforestation in the first place. This, in turn, depends on what we consider to be a forest.

In land system science, it is common to distinguish between land *cover* and land *use*. While land cover is a biophysical description of the properties of the land (e.g., what type of vegetation is there; are there any buildings and roads?), land use describes how, and/or for what purpose(s), the land is used by people (Gregorio and Jansen, 2005, Lund, 2006). This distinction between land cover and land use is also fundamentally important when discussing forests and deforestation: How we define what constitutes a forest will determine where we consider deforestation to have taken place.

So, what is a forest? Not surprisingly, it depends on who you ask: there are several hundred official definitions (Lund, 2006). Some definitions rely on the intended use (the land use), whereas others rely solely on biophysical properties (the land cover), such as degree of canopy cover, tree height and sometimes patch size. One reason for the multitude of forest definitions is the lack of clearly defined natural thresholds for canopy cover threshold and patch size (Sexton et al., 2016). A couple of commonly used forest definitions include:

- the FAO Forest Resources Assessment, which uses: “*land spanning more than 0.5 hectares with trees higher than 5 meters and a canopy cover of more than 10 percent, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use*” (Keenan et al., 2015, FAO, 2016) and
- the UNFCCC, whose definition allows countries to use minimum canopy cover thresholds ranging from 10 to 30% (UNFCCC Conference of the Parties (COP), 2002).

It is also worth noting that a piece of land considered a forest using land-use based definition might not have any actual tree cover (e.g. it might be a recently cleared rotation forest which is intended to regrow). Vice versa, a piece of land considered a forest using a land-cover based definition might not be considered a forest according to some land-use definitions (e.g., despite fulfilling the biophysical criteria, an oil palm plantation is not considered forest by the UN Food and Agriculture Organisation (FAO) Forest Resources Assessment (FRA) (FAO, 2016). Different definitions can yield wildly varying pictures on the extent of forests and serve different purposes; for example, a land-cover based approach will probably give a better insight into the carbon content of the biomass than a land-use based one (Chazdon et al., 2016, Sexton et al., 2016).

Given that there is no clear consensus on what a forest is, there is also no clear consensus on what constitutes deforestation. The question, therefore, again goes back to the intended purpose, and there is no uniquely correct definition (Chazdon et al., 2016).

For defining deforestation, there are several things to consider. As discussed above, a first question is whether to assess deforestation as a change in land cover or land use. These two perspectives are sometimes terminologically distinguished as *forest loss* (for a land-cover change) and *deforestation* (for a land-use change), respectively. This distinction can be

needed to avoid misunderstandings. For example, one of the most widely used spatial datasets on forest loss from Hansen et al. (2013) (available online at Global Forest Watch), uses a pure land-cover based definition and thus shows the loss of tree cover at the level of individual pixels. Some have criticised this (e.g., Tropek et al., 2014), as it includes felling within rotational forest and plantations, which—from a land-use based perspective—is not considered deforestation. Whatever definition of deforestation one chooses to use, it is essential to ensure that any data used are in line with this, or—perhaps more realistically—to at least be aware of where discrepancies occur and account for such limitations when interpreting results.

If using a land-cover based definition of forest loss, other considerations include deciding on a canopy-cover threshold for delineating forests prior to their loss and what type of canopy-cover reduction should count as forest loss. For example, is it forest loss if (a) canopy cover is reduced by a certain amount (such as by 20 percentage points from 50% to 30%); (b) the definition changes from forest to not forest; (c) canopy cover falls below a(nother) specific threshold; or (d) only if the canopy cover is entirely removed? In this dissertation, *forest loss* refers to complete removal of tree cover exceeding 5m height and 25–30% canopy-cover threshold (in the year 2000). *Deforestation* is used to refer to a land-use change, i.e., where forest loss is attributed to a subsequent land use (e.g., cropland, pasture or tree plantations).

When considering impacts that forest changes have on, e.g., carbon stocks or biodiversity, it is also worthwhile remembering that deliberate deforestation is not the sole change that happens to forests. Smaller changes (for example, from a higher to a lower canopy cover not captured by a given definition of forest loss), as well as changes in management, can also have a large impact (see, e.g., Erb et al. (2017a)). On a longer time scale, deforestation to date may itself lead, together with other contributions to climate change, to future forest loss (i.e., a positive feedback): its contribution to global and local climate change may bring about further forest loss or disturbance caused by changing temperature and precipitation patterns, and by more extreme events, e.g., droughts, flooding and fire (Lawrence and Vandecar, 2014, Nobre et al., 2016, Zemp et al., 2017).

## **2.2 What are the causes of deforestation?**

The question of what causes deforestation and other land-use changes can be answered in many ways, and at multiple levels. Ascribing deforestation to a single cause is rarely sufficient. Deforestation, or any land-use change for that matter, typically depend on complex interactions between human/socio-technical and natural/ecological dynamics at multiple levels (Geist and Lambin, 2002).

In the context of deforestation, it is common to distinguish between direct (or proximate) drivers and underlying (or indirect) causes of the land-use change (Geist and Lambin, 2002, Meyfroidt, 2015). At the most immediate level, an actor decides to change the use of land from one purpose to another. For example, a farmer buys a plot of forest to convert it into pasture for grazing cattle. In this case, we might say that the expansion of pasture (or demand for beef) caused the land-use change. This would be an example of direct land-use change (or a direct driver of land-use change). But this is of course only part of a much wider—and far more complex—story, which can span multiple domains. In our simple example above, the reason that our farmer bought a new plot of land might not be an increased demand for cattle after all. It might “simply” be that the farmer needed to find new land for their grazing cattle because their previous grazing land was bought up by, e.g., an agribusiness wanting to meet increased demand for soy or bioenergy (or responding to increased regulation preventing

production somewhere else). This is sometimes called indirect land-use change, or iLUC (Lapola et al., 2010, Ostwald and Henders, 2014, Richards et al., 2014). However this is still not the whole story, and there are lots of unanswered questions: Did changes to the prices of land, or of beef and soy, affect the decisions of the farmer and the agribusiness? What influenced those prices and profitability? Why is the farmer a farmer and not working with something else? Moreover, who wanted the soy and the beef in the first place? In short, the direct land-use change did not happen in a vacuum but was probably influenced by several underlying causes arising from demographic, economic, cultural, policy/institutional and technological factors (Geist and Lambin, 2002, Hertel, 2018). The causes may also have originated far from the deforestation itself. Telecouplings and unintended effects can come into play (Liu et al., 2013, Bruckner et al., 2015, Eakin et al., 2017). An example of this is leakage, where an environmental policy intending to reduce land-use change in one location triggers it in another, reducing the intended effect of the policy (Meyfroidt et al., 2018). The potential underlying causes of deforestation can, therefore, be many, perhaps combining in causal chains (or pathways) (Lambin and Meyfroidt, 2011, Meyfroidt, 2015, Meyfroidt et al., 2018) and the further up the causal chain, the more difficult it is to establish the causality.

The dominating direct driver of tropical deforestation is agriculture; other direct drivers are the expansion of infrastructure, mining and urbanisation, though their contribution is typically minor (Hosonuma et al., 2012, De Sy et al., 2015, Curtis et al., 2018). (In other parts of the world, e.g., Europe, agricultural land is being abandoned, sometimes followed by reforestation (Kauppi et al., 2006, Benayas et al., 2007, Estel et al., 2015).) When it comes to the agricultural drivers of deforestation, a few commodities, such as soy, palm oil, beef and also wood products (including charcoal and firewood), are particularly important and are sometimes referred to as “forest-risk commodities” (Rautner et al., 2013, Henders et al., 2015). (Although demand for wood also leads to expansion of forests/tree plantations, e.g., in Asia (Hyde et al., 1996, Foster and Rosenzweig, 2003, Meyfroidt and Lambin, 2011).) Despite this general picture, for many countries, we do not know in detail which type of commodity production expands into which forests, or how extensively.

This dissertation focusses on identifying the agricultural and forestry land uses expanding at the expense of forests across the tropics. The papers included primarily seek to identify the direct drivers of deforestation in terms of agricultural land uses (Paper I) and agricultural and forestry commodities (Papers II and III), although the model and scale of the data used in Papers II and III do not fully distinguish between direct and indirect land-use change.

### **2.3 How can we learn about the causes of deforestation?**

There are many ways of learning about the causes of deforestation, ranging from local case studies with interviews and surveys in the field, to global-scale studies relying on models, remote sensing and/or other data, such as agricultural statistics (Rounsevell et al., 2012, Kuemmerle et al., 2013, Verburg et al., 2015, Margulies et al., 2016) and initiatives for improving supply chain transparency (Grimard et al., 2017, Gardner et al., 2018). The use of trade models is becoming increasingly popular for learning about the linkages between international demand and distant environmental impacts (Tukker et al., 2018a). Papers II and III in this dissertation are among the first to apply trade models for studying deforestation and deforestation emissions. This section gives a brief overview of remote sensing data and trade models as tools for learning about the causes of deforestation.

### 2.3.1 Satellite remote sensing

The possibilities of satellite remote-sensing data are continuously improving through new sensors and satellites, enhanced processing capabilities, and increased availability and openness of data (Finer et al., 2018). These developments have led to a flurry of new datasets on, for example, land cover, biomass, and, to some extent, land-cover changes. The papers in this dissertation all rely on datasets based on satellite remote sensing, seeking to make use of these advancements to study the causes of deforestation.

Remote sensing typically makes use of electromagnetic radiation, actively or passively. Just looking at something with our own eyes can be considered a simple form of remote sensing. More typically, the term refers to the use of a sensor that, as in the case of satellite remote sensing, detects the intensity of radiation within a narrow range of wavelengths emanating from the Earth (or, indeed, another celestial body) (Campbell and Wynne, 2011). These intensity values are gathered in a collection of pixels, which can be used to create images of the area, such as a true colour composite combining data collected in visible wavelengths. This remote sensing permits surveying larger (or remote) areas of the Earth than would otherwise be feasible. However, the advantages of satellite remote sensing do not stop there: using and combining different wavelengths—and not necessarily only within the visible part of the spectrum—can enhance detection of certain features even better (Campbell and Wynne, 2011). For example, near-infrared wavelengths are useful for distinguishing the chlorophyll of vegetation (Tucker, 1979). When it comes to remote sensing, several types of resolution are therefore relevant: spatial (i.e., the pixel size), temporal (how frequently the satellite passes over and collects data) and spectral (the number and width of the wavelength bands sampled) (Campbell and Wynne, 2011). Typically, trade-offs exist between these properties (or with costs): a better spatial and spectral resolution is generally comes at the cost of poorer temporal resolution (or by using more expensive sensors or multiple satellites).

A common further step in using remote sensing data is to convert the intensity values into data that tell us something further about the area we are interested in, such as the type of land cover, the land surface temperature, the elevation, or the above-ground biomass. For continuous variables, an algorithm could be used to “translate” the intensity values into values of the variable of interest (e.g., based on training data points where the value of the variable is known). For data with categories, like land cover maps, this is done by classifying the data, i.e., identifying and then labelling groups of pixels that are similar in a way relevant for the purpose at hand, such as distinguishing pixels with forests from those with bare ground (Campbell and Wynne, 2011). Somewhat simplified, this entails using an algorithm to label pixels with similar spectral properties (e.g., high intensity values in one wavelength range, low intensity in another). In the resulting classified map (dataset), each pixel is assigned to a single class, although sometimes several classes and percentages are assigned (Campbell and Wynne, 2011). The classification requires that the classes one wants to separate are sufficiently distinct in the set of wavelengths used. This can sometimes be challenging, e.g., in some regions, pasture and cropland have resembling spectral properties, thus making them difficult to differentiate (Müller et al., 2015).

It is good practice to also assess the accuracy of the resulting classification (Olofsson et al., 2014). A set of validation/reference/“ground truth” data, for example from field visits, higher resolution data or crowdsourcing, provides independent classifications for a number of locations (pixels) (Olofsson et al., 2014, Wulder et al., 2018). For each pixel, the classification in the validation set is compared with that of the main map/dataset. The comparison results are compiled in an error/confusion matrix (Table 1 gives an illustrative

example), quantifying how many in each class were correctly classified or not. (In doing this, one assumes the validation set to be entirely “true”, which is generally a simplification (Foody, 2002, Olofsson et al., 2013).) The data in the confusion matrix are commonly summarised to estimate the overall accuracy (i.e., number of correct pixels divided by the total number of pixels), and often also producer’s- and user’s accuracies for each class (Story and Congalton, 1986, Foody, 2002). The producer’s accuracy expresses what share of the pixels was correctly classified; in the example given in Table 1, the validation data had 100 forest pixels, of which the classified map correctly identified only 75, so the producer’s accuracy for forest would be 75%. The user’s accuracy expresses the share of pixels that are what the map classification says they are (Story and Congalton, 1986); in the Table 1 example, only half of the pixels the map classifies as forest were indeed forests (75 of 150), giving a user’s accuracy of 50%. (These concepts are related to errors of omission and commission, respectively). The different types of accuracies can vary quite a lot between each other and between classes (as seen in Table 1), so the reliability of the map can depend considerably on which classes are of interest for the purpose at hand.

**Table 1.** *An illustrative example of a simple error matrix.*

		Validation set			User's
		Forest	Not forest	Total	
Classified map	Forest	<b>75</b>	75	150	50%
	Not forest	25	<b>225</b>	250	90%
	Total	100	300	<b>400</b>	
Producer's		75%	75%		Overall: 75%

Overall accuracies of existing global- and continental-scale land cover and forests datasets, such as the Hansen et al. (2013) and GlobeLand30-2010 data (Chen et al., 2015) used in the papers of this dissertation, commonly lie around 80–90%, but there are often considerable differences between classes and/or biomes.

With remote sensing data, capturing the biophysical properties of the land—the land cover—is more straightforward than determining its intended use(s) (Fritz et al., 2017). Because of this, maps or other data products based on remote sensing generally describe land cover rather than land use (Comber, 2008, Verburg et al., 2011). However, despite land cover and land use being fundamentally distinct concepts, they are frequently confused, used loosely, or used in a combined manner (Fisher et al., 2005, Comber, 2008). Attempts to infer land use from land cover face challenges, as there is not necessarily a clean “translation” between the two concepts. A single land cover class might match several land-use classes (e.g., a land cover of grassland may be used as a park or as a pasture) and vice versa (i.e., there is a many-to-many relationship between land cover and use). A single piece of land can also have multiple uses, as is the case of multifunctional landscapes (e.g., agroforestry); for example, an area with trees may serve simultaneously as a forest, grazing land and a recreational area (Fisher et al., 2005, Comber, 2008). Nevertheless, in many cases there is at least some kind of relationship between land cover and use, and, e.g., cropland (used for cropping) can often be readily identified from satellite data; in other cases, e.g., for land covered by grassland, the use might be more difficult to discern (Cihlar and Jansen, 2001). When it comes to maps and data, there have been attempts at bridging between these concepts to get closer to producing land-use maps, e.g., by combining different types of remotely-sensed data with additional information and assumptions to infer the land use (Cihlar and Jansen, 2001, Joshi et al., 2016, Fritz et al., 2017). Irrespective of whether a pure land-cover or land-use map are used, or

whether a map with a more blurred classification scheme is used, the suitability will depend on the match between the classification used in the dataset and the intended purpose (Verburg et al., 2011).

### **2.3.2 Examples of remote sensing-based datasets relevant to deforestation**

There is an increasing number of potentially useful datasets available for learning about the causes and consequences of deforestation. The most immediate is a dataset by Curtis et al. (2018), classifying five broad drivers of forest loss: commodity-driven deforestation, shifting cultivation, forestry, wildfire and urbanisation. However, in general, there is a shortage of data on *land-use* change at the global level (Kuemmerle et al., 2013, Song et al., 2018). Recent years have seen the release of a few global *land-cover* change datasets (e.g., ESA CCI, 2017, Li et al., 2018)). These are generally constrained to a few land cover classes; for example, the recent, state-of-the-art Song et al. (2018) dataset distinguishes changes between tree cover, short vegetation and bare ground. There are also several spatial datasets on forest loss/change (alongside more “real-time” alerts) (e.g., Hansen et al., 2013, Achard et al., 2014, Kim et al., 2014, Feng et al., 2016), carbon stocks and change (e.g., Saatchi et al., 2011, Tyukavina et al., 2015, Avitabile et al., 2016, Baccini et al., 2017), along with many datasets on land cover at single or several points in time (e.g., Friedl et al., 2010, Chen et al., 2015). Further summaries of available land cover datasets can be found in several review studies (Giri et al., 2013, Congalton et al., 2014, Da Ponte et al., 2015, Grekousis et al., 2015, Pérez-Hoyos et al., 2017).

Data on land use (rather than land cover) are sparse for the tropics (Verburg et al., 2011, Kuemmerle et al., 2013, Müller et al., 2015, See et al., 2015), and most global and continental land-cover datasets do not provide much detail on the type of agriculture, and rarely separate pasture from cropland, and much less between different crops (Pendrill and Persson, 2017). While there are a couple of spatial datasets on livestock (Robinson et al., 2014) and crops (Monfreda et al., 2008, Fritz et al., 2015, You et al., 2017), these are limited to single points in time and given the speed at which agricultural expansion occurs in forest frontiers, these datasets provide limited information on what drives recent deforestation. There are also some ongoing developments, e.g., attempts to map soybean using remote sensing (e.g., Song et al., 2017) and the US Department of Agriculture produces an annual Cropland Data Layer for multiple crops (Boryan et al., 2011), but these have not yet been expanded to global or pan-tropical scales. Thus, for learning more about the relative role of different agricultural commodities in driving deforestation, other data sources are needed. A primary source for this is agricultural statistics, from, e.g., national statistics agencies or the FAO, and which will typically be available only for more or less aggregated spatial units, such as a municipality or a country.

### **2.3.3 Spatial resolution and the drivers of deforestation**

The spatial resolution of the data used in an analysis of deforestation drivers has implications for how to interpret the results. When spatial resolution is high, such as with high-resolution datasets based on remote sensing, the land cover (or use) following forest loss may represent the direct land-cover change. When resolution is low, e.g., changes in the harvested area of a crop at country-level agricultural statistics, the analysis will not necessarily determine reliably what land uses are directly driving deforestation. How representative spatially aggregated land-cover/use change data are of direct land-cover/use changes will ultimately depend on how homogenous land-cover/use patterns are within that aggregation (e.g., if the land covers/uses expanding at national level are the same as those expanding in the places

where the forest loss/deforestation is occurring). In general, a higher spatial resolution gets closer to the direct drivers of land-cover/use change.

Similarly, the spatial resolution of the datasets based on remote sensing has implications for what the data/values represent. For classified variables, such as a land cover map, the value often represents only the dominant land cover class. This entails that an area that appears fully forested at a low spatial resolution might show as a mix of forest and several other land uses at higher spatial resolution (Olofsson et al., 2014). Therefore, the poorer the resolution, the greater the risk of “mixed pixels” consisting of multiple land cover types (Foody, 2002). These can be problematic, both because the spectral characteristics can become muddled, and because the pixel might not adequately represent varied landscapes (Foody, 2002, Verburg et al., 2011). In the papers included in this dissertation, the resolution of most of the spatially-explicit data used is fairly high (30 m), but when using data for more spatially-explicit analyses (e.g., those in in Paper I) mixed pixels can entail a lower accuracy at the boundaries between different types of land cover (Foody, 2002, Brovelli et al., 2015, Tyukavina et al., 2015). This might be particularly pertinent here, as much of forest loss likely occurs where land cover is changing from one type to another (this is discussed further in Paper I).

### **2.3.4 Following embodied deforestation and emissions through trade to consumption**

The previous parts of this chapter have focussed on analysing drivers of deforestation in the vicinity of where forests are being lost. However, there is increasing recognition that only evaluating impacts from the production-side can be limiting, especially when responsibility is to be assigned and where policies do not cover the entire system of interest (Munksgaard and Pedersen, 2001, Peters, 2008, Duus-Otterström and Hjorthen, 2019). A prominent example of this is that countries committing to reduce their carbon emissions under the United Nations Framework Convention on Climate Change and the Kyoto Protocol (sometimes referred to as Annex-I or Annex B countries) report only emissions occurring within their national/territorial boundaries (IPCC, 2006). There are indications that this territorial approach has resulted in (at least weak) carbon leakage: developed Annex-I countries seeking to reduce their domestic emissions tend to be net importers of embodied carbon emissions, thus meeting part of their consumption needs by relying on imports from (primarily developing) countries not covered by the emissions-reduction commitments (non-Annex I/B countries) (Peters and Hertwich, 2008, Peters, 2010, Peters et al., 2011, Kanemoto et al., 2014). (However, this trend may be changing: net emissions transfers between OECD and non-OECD countries peaked and then plateaued after 2006, as did emissions embodied in both production and consumption in OECD countries (Wood et al., 2019). Still, developed countries do remain net importers of embodied emissions.) Also, as the capacity to act towards reducing impacts lies not only at the point of production but potentially also along the supply chain all the way to the point of consumption (e.g., by reducing demand in the first place), there is increasing attention brought to the value of complementing production-based approaches with different perspectives, including consumption-based (“downstream”) ones, as well as those sharing responsibility between producers and consumers (Lenzen and Murray, 2010, Steininger et al., 2016, Duus-Otterström and Hjorthen, 2019).

Trade models enable the evaluation of impacts from consumption-based perspectives, thus complementing production-based assessments (Peters, 2008, Lenzen and Murray, 2010, Wiedmann et al., 2011). They are frequently used for calculating the embodied, upstream impacts associated with the consumption of a product (similar to the goal of a life-cycle assessment), or for looking at the trade of environmental impacts between countries (trade

models, in general, are also widely used outside the environmental domain) (Miller and Blair, 2009, Minx et al., 2009, Kitzes, 2013).

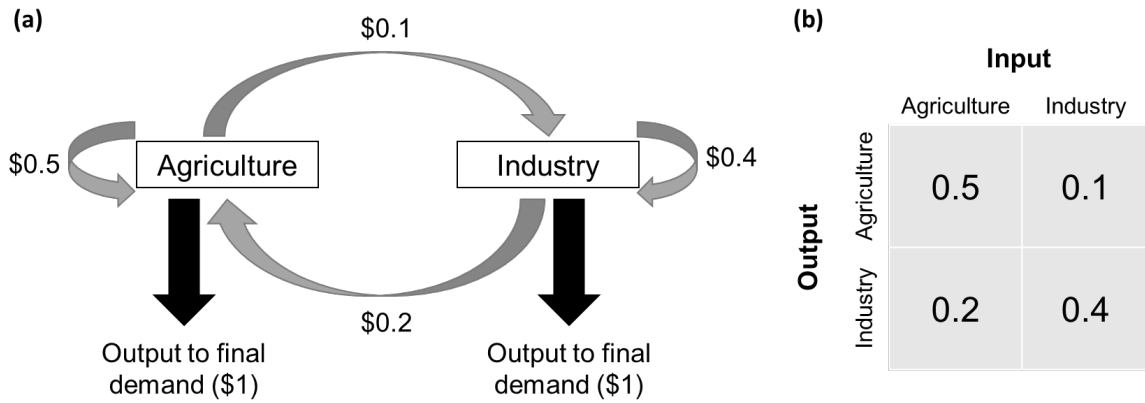
A common use of trade models thus lies in carbon footprinting and consumption-based accounting of CO<sub>2</sub> emissions (e.g., Peters, 2008, Davis and Caldeira, 2010, Peters et al., 2011), environmental (Weinzettel et al., 2014) and material footprinting (Wiedmann et al., 2015), along with a range of other impacts embodied in trade, including water consumption (Lutter et al., 2016), biomass (Erb et al., 2009), species threats (Moran and Kanemoto, 2017) and health impacts of air pollution (Zhang et al., 2017). Closer to the area of interest in this dissertation, there have been several studies assessing land use embodied in trade (e.g., Meyfroidt et al., 2010, Steen-Olsen et al., 2012, Weinzettel et al., 2013). However, there have been very few studies of land-use *change* embodied in trade, and these studies are limited to a few countries and commodities (Saikku et al., 2012, Karstensen et al., 2013, Henders et al., 2015). In this dissertation, Papers II and III seek to fill part of this gap, by quantifying pan-tropical deforestation embodied in trade.

By examining where the demand for the products stems from, trade models can present one way of beginning to “unpack” the sources of the demand behind the direct drivers of deforestation. We might find out from where—geographically—the demand originates. With some types of trade models, we can also gain information on which sector of the economy (e.g., cattle farming, manufacturing, or retail) generates the demand. In this way, we can learn more about the linkages between consumption and a range of environmental impacts associated with the production (Kitzes, 2013, Henders and Ostwald, 2014).

The most common types of trade models used for examining environmental linkages are multi-regional input-output models (MRIO) and (bio-)physical trade models (Kitzes, 2013, Henders and Ostwald, 2014, Bruckner et al., 2015, Schaffartzik et al., 2015, Hubacek and Feng, 2016). Here I will give a brief and simplified introduction to input-output modelling, parts of which also apply to physical trade models. For a more thorough introduction, see, e.g., Kitzes (2013) and Miller and Blair (2009).

In essence, trade models describe interrelationships between sectors and between regions (e.g., countries). Sectors (and regions) are not only producers of goods (outputs), but also consume goods (inputs) while producing their outputs (Miller and Blair, 2009). Thus, in order to meet a given level of final/external demand, the total output from sectors needs to cover not only the final demand but also the demand for intermediary products (as inputs) in various production sectors (Miller and Blair, 2009, Kitzes, 2013). To take an example (illustrated in Figure 1a), imagine a very simple economy consisting of two sectors: agriculture and industry. To produce output, the agriculture sector uses various inputs, both from its own sector (e.g., feed for cattle) and from the industry sector (e.g., machinery and fertiliser). These intermediary products also need to be produced, which in turn require further inputs, and so on, and so forth. Input-output modelling provides a way to summarise all of these “upstream” inputs, without double counting them, to find the total output needed to meet a certain level of final demand (Kitzes, 2013).





**Figure 1.** A simplified example of input-output sector interrelationships. (a) Schematic illustration of two sectors consuming inputs and producing output for both final demand and for use as inputs in other sectors (intermediary products). The dollar values show the amounts of input needed to produce \$1 of output to final demand. In this example, producing \$1 of agricultural output requires \$0.5 of input from agriculture as well as \$0.2 from industry. (b) A corresponding technical coefficients matrix ( $A$ ) shows the same thing. The technical coefficients matrix, in essence, gives a "recipe" of the inputs needed to create \$1 of output from each sector. Viewed in another way,  $A$  also shows the share of output from one sector going to another; here, for example, 10% of the output from agriculture is consumed by the industry sector.

This is accomplished using a set of vectors, matrices and linear algebra equations, at the core of which lies the technical coefficients matrix,  $A$ , which describes the interrelationships between sectors (and regions) mentioned above (Miller and Blair, 2009, Kitzes, 2013) (an example is illustrated in Figure 1b). The technical coefficients matrix shows both where the outputs from each sector go (if reading the rows) and what inputs are needed by the sector (if reading the columns) (Miller and Blair, 2009). So, element  $a_{ij}$  of  $A$  shows the share of output from sector  $i$  that is consumed by sector  $j$ .

The Leontief Input-Output model (named by its creator),

$$\mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{y}$$

can then be used to find the total output vector  $\mathbf{x}$ , depending on the level of final demand expressed in vector  $\mathbf{y}$  (each with one element for each sector) (Miller and Blair, 2009). ( $\mathbf{I}$  is an identity matrix, and  $(\mathbf{I} - \mathbf{A})^{-1}$  is sometimes called the Leontief matrix.) A multi-regional input-output analysis (MRIO) works the same way, but where each element corresponds to *sector-region* combinations, rather than just between sectors (Miller and Blair, 2009). The type of physical trade model used in Papers II and III also uses similar mathematics to describe trade between countries (export shares) by assuming that domestic production and imports are distributed proportionally between consumption as well as exports (Kastner et al., 2011b).

Environmental input-output analysis expands the Leontief model described above by accounting also for some type(s) of environmental impacts associated with the total output needed to meet a level of final demand (or, e.g., for a single product bought by a consumer). This is done by introducing (pre-multiplying by) an intensity vector listing the environmental impacts (e.g., tonnes of carbon emissions) associated with \$1 of sector output for each sector (Wiedmann et al., 2011, Kitzes, 2013).

One central limitation shared by all trade models is therefore an assumption of homogeneity. All products within the same sector (or sector-region combination) are assumed to be exported to the same extent (that specified in **A**). They are also assumed to have the same environmental impact (per unit of measure; e.g., per dollar, in the case of MRIOs, or, e.g., per kilo or calorie of product, in the case of physical trade models) (Kitzes, 2013, Bruckner et al., 2015). These are oversimplifications, further exacerbated by the fact that commodity categories, and sometimes also regions, are in many cases quite aggregated, especially for global MRIOs (Bouwmeester and Oosterhaven, 2013, Majeau-Bettez et al., 2016). For example, EXIOBASE3, the MRIO database used in Paper III—which is one of the global MRIOs with the highest resolution for food sectors—has a joint category for “other crops”. This category mixes cash crops (such as cocoa, coffee and tea) with subsistence products (such as cassava) (Stadler et al., 2018, Weinzettel and Wood, 2018), which clearly differ in the extent to which they are exported and quite likely also in the environmental impact of their production.

Another limitation of trade models lies in the quality of the input data (Kitzes, 2013, Tukker et al., 2018b). The values in matrix **A** describing the interrelationships between sectors and countries/regions are based on observed economic data (and thus limited by the time frames for which such data are available) (Miller and Blair, 2009, Wiedmann et al., 2011, Kitzes, 2013). These data are often compiled from multiple sources, which may not be using the same standards, introducing additional uncertainties. For transactions between sectors within countries, data often come from supply-and-use tables compiled by national statistics offices; for trade between countries, a common data source is the UN Commodity Trade Statistics Database (COMTRADE) (Tukker and Dietzenbacher, 2013). These vary considerably in quality. Those compiling global MRIOs often need to reconcile conflicting data as well as infer data where they are missing, and many MRIOs also have some “rest-of-the-world” regions, grouping together multiple countries which may differ in their trade patterns (Wiedmann et al., 2011, Tukker and Dietzenbacher, 2013, Tukker et al., 2018b).

MRIOs and (bio-)physical trade models have several commonalities and some key differences. They differ particularly in the type of metric used (economic, i.e., dollars, in the MRIOs, versus physical units, e.g., kilograms, in the physical trade models) and in how far down the supply chain the embodied impacts are followed (Kitzes, 2013, Henders and Ostwald, 2014, Bruckner et al., 2015, Schaffartzik et al., 2015, Hubacek and Feng, 2016). While some physical trade models do follow embodied impacts through some re-exports and/or processed products (such as the Kastner et al. (2011b) model used in Papers II and III), MRIOs generally go further along the supply chain and continue to follow embodied impacts in intermediary products (as inputs) through all or most sectors of the economy (Miller and Blair, 2009, Kitzes, 2013). As an example, take timber from Vietnam used to make furniture sold to a factory in China that makes computers sold in Germany. An MRIO would attribute part of the embodied impacts of timber to Germany’s consumption, whereas a physical trade model would attribute the environmental impacts to consumption in China.

The type of trade model, consequently, affects the type/level of driver that the results will describe, and the choice of model should, therefore, be made with consideration for the research questions or policy aims (Bruckner et al., 2015, MacDonald et al., 2015, Hubacek and Feng, 2016). Understanding the country-to-country trade flows of a physical trade model is more relevant for actors such as companies, investors and governments wanting to reduce deforestation through direct supply chain interventions (such as commodity moratoria, zero-

deforestation commitments, and other demand-side measures). For consumption-based accounting, using an MRIO analysis to follow embodied impacts further through the supply chain to the point of final demand is more useful for understanding better the underlying drivers (Peters, 2008, Wiedmann and Barrett, 2013, Hubacek and Feng, 2016). Thus, the choice of methods for the attribution of deforestation drivers and the choice of trade model affects the type of driver/level of causality described.

### **3. Results**

This section synthesises the results from the papers in this dissertation. It begins with the attribution of deforestation to drivers, which is part of all three papers. Then follows a comparison of the relative role of different drivers depending on whether the focus is on deforestation area or deforestation emissions. Finally, results on the role of international trade and on consumption-based perspectives on deforestation are presented.

#### **3.1 Attribution of deforestation to drivers (Papers I and II)**

##### **3.1.1 Approach**

Papers I and II both provide quantifications of deforestation drivers, thus contributing to the aim of examining the causes of deforestation in the tropics. The approach used, however, differs between the papers. Paper I presents a spatially explicit analysis, using global maps on forest loss and subsequent land cover to quantify to what extent forest loss is directly followed by cropland and pastures, respectively. Paper II attributes deforestation to more detailed groups of agricultural commodities, but at a more coarse spatial scale, using a simple land-balance model with input data primarily summarised to the country level. The input data used for Paper II include those based on remote sensing, as well as agricultural statistics. Both papers focus on tropical and subtropical areas (Paper I on Latin America, Paper II on 156 countries across multiple continents). These areas are where most agricultural expansion into native vegetation occurs (Curtis et al., 2018) and where the impacts on biodiversity and carbon stocks are expected to be the greatest (Myers et al., 2000, Saatchi et al., 2011).

##### **3.1.2 Main findings**

Paper II shows that much of the deforestation for agriculture in the tropics and subtropics was associated with just a handful of commodities: cattle meat, palm oil and soybeans together accounted for more than half of the deforestation and expanding tree plantations contributed an additional 15%. The variation between countries is large, however. In the Asia-Pacific region, palm oil and forest products together account for 60% of the deforestation, while in many Latin American countries, cattle meat alone accounted for more than half of the deforestation.

Paper I, with its more spatially-explicit approach, also shows large variations in the dominating land uses following deforestation, between countries as well as within countries. The most common land cover replacing forests in Latin America is pastures. While Paper I does capture some of the overall driver patterns, the main contribution of Paper I lies in highlighting some important limitations of combining global datasets/maps for assessing the drivers of deforestation.

Table 2 shows the amount of deforestation attributed to pasture and cropland, respectively, in Paper I and II. Only countries included in both papers and with the highest deforestation rates are shown. Table 2 also compares the total (cropland plus pasture) area to the area of commodity-driven deforestation found in a recent study by Curtis et al. (2018). The total cropland/pasture area is reasonably similar in the two papers, and also all within the range of values reported by Curtis et al. (2018) for the same period. The differences between Papers I and II are larger when looking at cropland and pasture separately. Generally, less deforestation is attributed to grassland in Paper I than to pasture in Paper II. This difference is probably largely due to the incomplete separation between cropland and pasture in the dataset used in Paper I. (Instead of distinct classes for pasture and cropland, it has classes for Grassland and Cultivated land, which both include certain pasture types.) Overall, therefore,

the papers provide quite a coherent picture of the area of forest loss driven by the expansion of agriculture, but it is likely that Paper II provides a better representation of the share attributed to pasture and cropland (and subsequently even more detailed into commodities).

**Table 2.** Comparison between the attribution of deforestation to agriculture using different approaches (data from Paper I and II in this dissertation, and from Curtis et al. (2018).)

Country	Follow-up land cover/use	Paper I kha yr <sup>-1</sup> (% of deforestation) 2001–2011	Paper II kha yr <sup>-1</sup> (% of deforestation) 2005–2013	Curtis et al. kha yr <sup>-1</sup> (range) commodity-driven deforestation 2001–2013
Brazil	Cultivated/Cropland	589 (30%)	345 (20%)	1,849 (1,040–2,910)
	Grassland/Pasture	1,349 (70%)	1,235 (72%)	
	Both	1,939*	1,580*	
Argentina	Cultivated/Cropland	229 (88%)	199 (54%)	294 (68–499)
	Grassland/Pasture	32 (12%)	169 (46%)	
	Both	261	368	
Paraguay	Cultivated/Cropland	174 (86%)	108 (55%)	285 (127–462)
	Grassland/Pasture	29 (14%)	83 (43%)	
	Both	203	191	
Bolivia	Cultivated/Cropland	105 (72%)	62 (63%)	169 (83–340)
	Grassland/Pasture	40 (28%)	36 (36%)	
	Both	146	98	

\*Deforestation rates in Brazil declined considerably after 2004. As such, differences in the results between Papers I and II here are expected and likely largely due to the difference in time period (as less deforestation occurred in the 2005–2013 time period of Paper II than in the 2001–2011 time period of Paper I), rather than due to the methodological differences between the two papers.

As noted above, the main contribution of Paper I lies primarily in discussing the limitations related to the use of currently available global datasets for quantifying drivers of deforestation, rather than in the quantification of drivers per se. First, Paper I highlights the need to exercise care that the classification of land cover/use matches the purpose at hand. The “Cultivated land” class of the global land cover dataset used (GlobeLand30) in Paper I, which several studies interpret and use as cropland, in fact also contains significant amounts of planted pastures. Second, it shows that forest loss identified by one dataset was frequently still identified as forest in a land cover dataset over the same area a few years later. This indicates that combining datasets to identify land-cover transitions can suffer from limited accuracy (as errors compound when maps are combined (Fuller et al., 2003)), especially in areas with heterogeneous land cover or small-scale forest loss. A comparison with the recent study by Curtis et al. (2018) possibly provides further context: In the countries reported in Paper I to have a large share of forest loss seemingly remaining under forest cover (e.g., Colombia and Mexico), shifting cultivation seems to have caused much of the forest loss. Forest loss was more likely to have resulted in permanent deforestation in countries that were reported to have a larger share of forest loss followed by grassland or cultivated land (e.g., Brazil and Argentina) in Paper I.

Paper II also shows some areas of potential improvements. Increasing the spatial scale of the analysis for Brazil and Indonesia, from national-level (as used for all other countries) to subnational level, led to better representation of the land-use dynamics in the deforestation regions. The subsequent trade analysis would also benefit from improved spatial resolution,

but, in practice, this is currently limited by data availability on subnational trade patterns (Gardner et al., 2018). Coarser-grained analyses will, therefore, likely be needed to prioritise where acquiring additionally detailed data is warranted (Godar et al., 2016).

## **3.2 Attributed deforestation compared to attributed deforestation emissions (Papers II and III)**

### **3.2.1 Approach**

While the deforestation area associated with different agricultural drivers indicates the main causes of deforestation, the further consequences of these land-use changes also matter. Paper III seeks to identify further carbon emissions associated with the land-cover change from forest to something else.

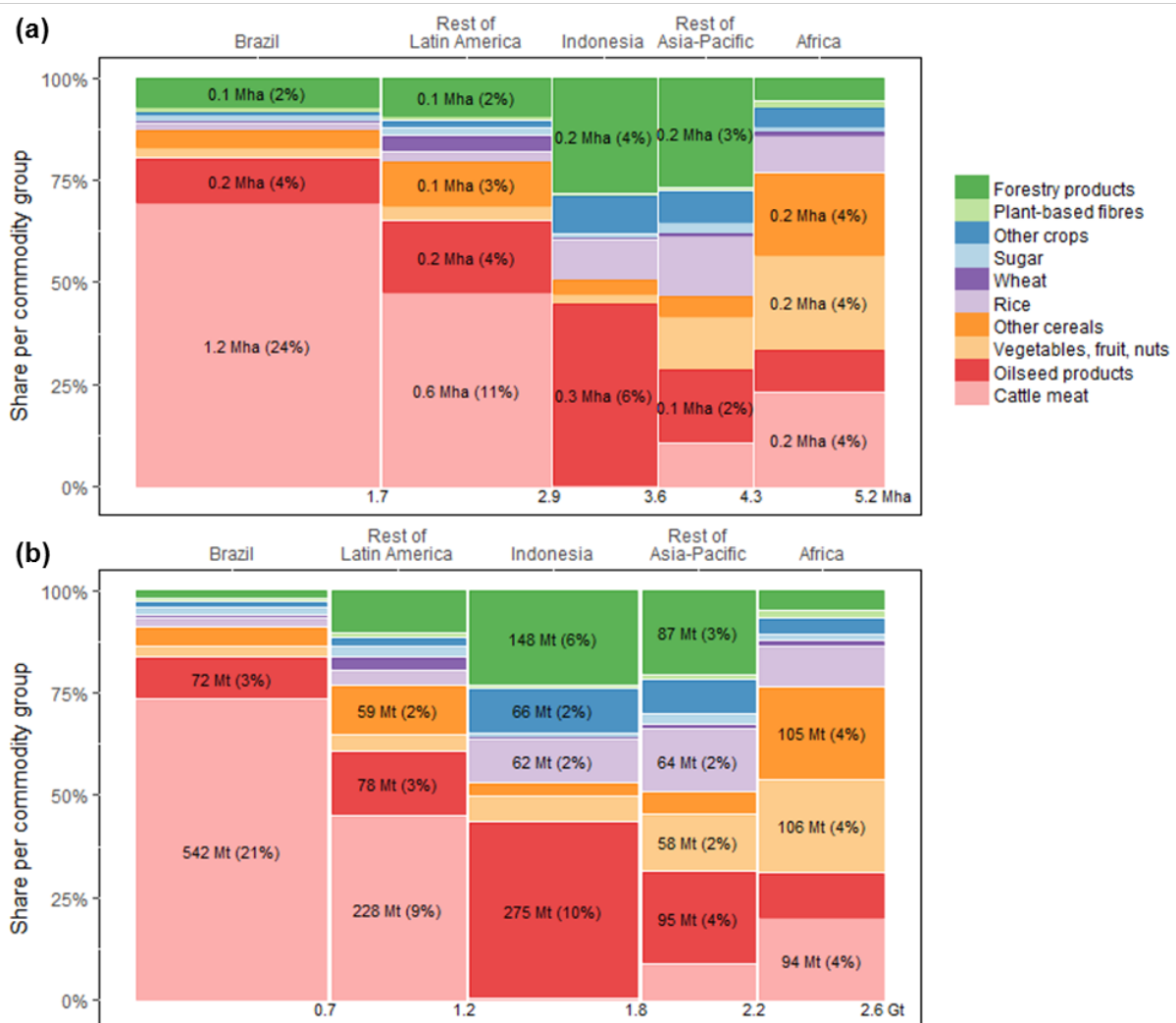
In Paper III, the carbon emissions are quantified by estimating the changes to carbon stocks as a piece of land is changed from one type to another. Carbon is stored in biomass above ground (above-ground biomass, or AGB) as well as below ground (below-ground biomass, or BGB) and also in organic matter in the soil (soil organic carbon, or SOC). There can also be significant additional carbon emissions resulting from peatland drainage and burning in places where peatlands are abundant, such as Indonesia and other parts of Southeast Asia.

The knowledge and data available for different carbon reservoirs vary; the best described is generally AGB, while there is considerable uncertainty when it comes to BGB, and even more so for SOC. For changes in AGB, there are multiple spatially-explicit estimates based on remote sensing (e.g., Saatchi et al., 2011, Baccini et al., 2012, Harris et al., 2012, Achard et al., 2014, Tyukavina et al., 2015, Avitabile et al., 2016, Baccini et al., 2017), which agree reasonably well at the continental level but diverge more at the regional level (Mitchard et al., 2013, Avitabile et al., 2016). For BGB, most studies follow an approach similar to the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) combining ratios between AGB and BGB with typical values from the literature. In Paper III, therefore, carbon stock change is estimated with different levels of precision and with varying levels of uncertainty, ranging from 30-m remote-sensing based data on the AGB stocks prior to forest loss (Zarin et al., 2016) to estimates from the literature on typical SOC changes when a broad land-use category changes to another broad category (Don et al., 2011). Thus, while the results indicate the carbon emissions associated with deforestation, significant improvements to the estimates of carbon stock (or carbon-stock changes) would be needed to quantify the actual emissions associated with each land-cover change.

### **3.2.2 Main findings**

The results from Paper II and Paper III can be used to investigate how the choice of indicator—hectares of deforestation versus CO<sub>2</sub>-emissions resulting from the land-use change—can influence the result. While the main commodities associated with deforestation-related carbon emissions in Paper III are similar to the commodities associated with deforestation area in Paper II, the relative importance of countries and commodities differ somewhat (Figure 2). In the deforestation-area attribution, Brazil is clearly the dominating country (33% of the total). In the attribution of the associated carbon emissions, Brazil is still dominant; however, its relative role is smaller (around a quarter), while the relative role of Indonesia is notably larger (from around a seventh of the deforestation-area attribution, to almost a quarter for the carbon-emission attribution). Similarly, cattle meat is the dominant commodity irrespective of the indicator, but less markedly so for the deforestation carbon

emissions than for deforestation area. Conversely, the relative role of oilseeds is greater from an emissions perspective, especially for Indonesia and the rest of the Asia-Pacific region.



**Figure 2.** Attribution of deforestation hectares (a) and carbon emissions (b) to commodities and regions from Papers II and III for the period 2010–2014. The width of a region on the x-axes corresponds to the total deforestation area/emissions attributed to that region, whereas the y-axes show the relative proportion within each country/region attributed to each commodity group. The area of the rectangles in the figure thus represent the deforestation area/emissions attributed to each region-commodity group combination, and the percentages within them indicate each combination’s proportion of the embodied deforestation area/emissions. (The data for Paper II have been modified to match the commodity groups and set of countries used in Paper III.)

In short, while the broad picture is similar concerning drivers of deforestation and its concomitant carbon emissions, there are some notable differences in the relative role of both countries and individual commodities. As such, deforestation area is an incomplete indicator of the carbon emissions associated with deforestation. For other impact categories, such as biodiversity and changes to local climate and hydrology, the differences can potentially be larger (and almost certainly different), making it important to be aware that single indicators give but a limited view on the full impacts of deforestation.

### 3.3 Role of international trade (Papers II and III)

#### 3.3.1 Approach

To examine further where the demand for these commodities stem from, Papers II and III use trade models to trace the commodities with embodied deforestation (Paper II) and concomitant emissions (Paper III) through international supply chains to consumers across the world.

In both Papers II and III, a physical trade model by Kastner et al. (2011a), (2011b) was used to follow country-to-country trade flows to where commodities were physically consumed as food or in industrial processes. Paper III additionally used a state-of-the-art multi-regional input-output model (MRIO), EXIOBASE3 (Wood et al., 2015, Stadler et al., 2018). The MRIO provides a complementary perspective by following the embodied deforestation and deforestation emissions further through monetary trade flows in all sectors of the economy (although with somewhat less detailed regional and commodity resolution). The use of trade models permits an examination of the relative roles of domestic and international demand, as well as the identification of major consumer countries and regions.

#### 3.3.2 Main findings

Commodities with embodied deforestation area and emissions were primarily consumed domestically. Still, a substantial share, especially of some commodities, was destined for export markets and eventually consumed outside the producing country, and thus outside the country where the deforestation impact took place (Table 3).

*Table 3. The share of embodied deforestation and concomitant emissions embodied in international trade varies between countries and commodity groups (results from a physical trade model).*

	Average share of deforestation embodied in international trade	Average share of deforestation emissions embodied in international trade
<b>Average for all included countries and commodities</b>	26%	29%
<b>By region</b>		
Latin America average	24%	23%
Asia-Pacific average	39%	44%
Africa average	10%	9%
<b>By commodity group</b>		
Crops	41%	40%
Cattle meat	10%	10%
Forestry products	25%	24%

On average, 26% (of 5.3 Mha yr<sup>-1</sup>) of the deforestation area and 29% (of 2.6 Gt CO<sub>2</sub> yr<sup>-1</sup>) of the deforestation-related emissions were attributed to international demand, using a physical trade approach. The somewhat lower share of deforestation area (compared to carbon emissions) embodied in international trade, is mostly because of cattle meat and oilseeds. Cattle meat, which is a relatively more important commodity from the area perspective, is



primarily for domestic demand (on average only 10% is exported), while oilseeds, which are a relatively more important commodity from the emissions perspective, are primarily for export (on average 62%).

The share of embodied deforestation and concomitant emissions attributed to international demand varies significantly not only between commodities but also between countries. For countries in the Asia-Pacific region, near half of the embodied deforestation and deforestation emissions were attributed to international demand (on average, 39% and 44% exported, respectively), whereas for countries in Africa, the demand for commodities with embodied deforestation and deforestation emissions was primarily domestic (on average, 10% and 9% exported, respectively).

The average share attributed to international demand increases to 39% if the embodied carbon emissions are followed further through the economy using an MRIO trade model (rather than the physical trade model above), as it includes more “indirect” exports in additional sectors. This share is substantially higher than the equivalent share in MRIO studies assessing fossil carbon emissions (23–26%; Davis and Caldeira (2010), Peters et al. (2011), health impacts of air pollution (22%; Zhang et al. (2017)), nitrogen pollution (25–28%; Oita et al. (2016)), water consumption (24–32%; Lenzen et al. (2013)), global species threat (30%; Lenzen et al. (2012)), land footprint (24%; Weinzettel et al. (2013)). This is a key result of Paper III, which is further pronounced if looking solely at crops, where the share exported is 48%. Paper II shows that this high export share of embodied deforestation stems mainly from Brazil, Indonesia and Argentina, which are high-deforestation countries, and export much of their crops (27–74%) and especially those with embodied deforestation (49–76%).

A few trade flows were found to account for a large share of the total embodied deforestation and deforestation emissions, and the main consumers of both deforestation and related emissions embodied in international trade were found to be countries in Europe and Asia, in particular, China, India and Russia.

### **3.4 Consumption of traded deforestation (Papers II and III)**

#### **3.4.1 Approach**

Looking at deforestation embodied in international trade also permits widening the focus to include a consumption perspective on deforestation, in addition to more conventional production-side perspectives.

For deforestation, in Paper II, we relate imports of embodied deforestation to trends and changes in forest cover within the importing countries. In particular, we do this in light of their stage of *forest transition*. Forest transitions is a concept used to describe where regions (or countries), as they develop, tend to shift from decreasing their (net) forest area to increase it instead (Mather, 1992, Rudel et al., 2005). Thus, we first distinguish between countries in different stages of forest transition, depending on their rate of forest change and current forest cover. We then assess whether countries that are increasing their forest cover or reducing their deforestation rates (i.e., countries that have gone through or are undergoing a forest transition), also tend to import commodities that are contributing to deforestation in other countries. We also calculate to what extent such imports of embodied deforestation offset the forest gains made by countries that have undergone a forest transition.

For deforestation emissions, in Paper III, we make several consumption-side comparisons. We compare the imports of embodied deforestation by developed (Annex I) countries to the size of domestic (territorial) agricultural emissions as reported to the UNFCCC. We also calculate deforestation/land-use change carbon footprints: per capita, for countries' food consumption, and per kilogram of product, for key forest risk commodities.

### **3.4.2 Main findings**

The results in both Papers II and II show that, from a consumption perspective, imports of embodied deforestation can be sizeable.

In Paper II, we found that a majority (79%) of exported deforestation is consumed in countries that are currently increasing their forest cover (post-forest transition countries), thus partly offsetting some of the net forest gains made by these countries. For many countries, imports of embodied deforestation rivalled or exceeded domestic gains in forest area: for example, for India, deforestation abroad offset almost 60% of forest gains. On average, imports of embodied deforestation offset around a third of the net forest gains made in the countries that have undergone a forest transition. The net forest area saved in the post-forest transition countries is potentially larger, though, as imports may also have prevented domestic deforestation (in addition to enabling forests to expand).

In Paper III, we found that, for many developed countries, the imports of embodied deforestation emissions are of a similar order of magnitude as the domestic emissions from agriculture. On average, for the Annex I countries to the UNFCCC, deforestation emissions embodied in imports amount to 17–31% of the reported territorial agricultural emissions. For key forest-risk commodities, such as palm oil and beef, we found carbon footprints from deforestation and peatland drainage to be in the same order of magnitude as non-land-use change emission footprint. The per capita deforestation carbon footprint of food consumption was found to be highest for Brazil ( $2.6 \text{ tCO}_2 \text{ cap}^{-1} \text{ yr}^{-1}$ ; primarily due to domestic beef consumption). For emerging economies (China, India, South Africa), the footprints are generally low ( $< 0.1 \text{ tCO}_2 \text{ cap}^{-1} \text{ yr}^{-1}$ ), while for most developed countries they lie around  $0.3 \text{ tCO}_2 \text{ cap}^{-1} \text{ yr}^{-1}$ . For the EU, this implies that deforestation emissions constitute a substantial share (around 15%) of the total carbon footprint of an average EU diet.

## 4. Discussion

The papers in this dissertation all provide insights into the agricultural drivers of deforestation, and Papers II and III further shed light on roles of international trade in redistributing deforestation pressure arising from agricultural commodity demand.

The main contribution of Paper I lies in exposing some limitations in using available datasets on land cover and land-cover change to quantify the direct drivers of deforestation. It shows that an approach to quantify deforestation drivers that combines different datasets (a) will require careful attention to the classification schemes of the datasets and (b) is currently not accurate enough to identify the land-cover transitions in a satisfactory way, especially in places where the land cover or forest loss varies over small areas. For future research, the availability of spatially-explicit datasets is an area bound to progress. New satellites and sensors supply data with higher spatial, spectral and temporal resolution and there is continuous development of processing techniques (Goetz et al., 2015, Gómez et al., 2016, Finer et al., 2018) as well as crowdsourcing (Estes et al., 2016, Fritz et al., 2017). These advancements will likely aid the determination of the direct drivers of deforestation and forest degradation. Adequately evaluating the trade-offs of deforestation would also require knowledge and data to estimate economic, social and environmental effects, e.g., changes to carbon stocks and emissions, to ecosystem services and biodiversity, and on food security. After all, forest conversion can have rather different impact on the environment if it takes place in a highly-biodiverse or carbon-dense biome, compared to one that is less so (the impact will of course also depend on the type and intensity of the subsequent land use (Phalan et al., 2011, Kehoe et al., 2015, Newbold et al., 2015, Erb et al., 2017b)). This is also important in light of the lack of agreement of what constitutes a forest, and thus deforestation: Getting closer to the reasons for caring about land-use changes will not only improve the relevance but potentially also bring a clearer agreement on what is measured (Sexton et al., 2016).

Papers II and III make several contributions. While it has previously been shown that international trade has a large role in displacing multiple types of environmental impacts, Papers II and III are among the first to do so for pan-tropical deforestation and deforestation emissions. A key contribution of Paper III is that it shows that the share of deforestation emissions embodied in international trade is substantially higher (39%, rising to 48% if only crops are considered) than for most other environmental impacts (which typically lie around 20–30%). This indicates that it is particularly important for efforts aiming to reduce deforestation and concomitant emissions at the global level to consider the role of international demand and trade in driving deforestation. That said, domestic demand still constitutes more than half of the attributed deforestation and deforestation emissions, implying that to successfully reduce deforestation on the global scale, demand-side efforts targeting international supply chains must be complemented by efforts at the national level (Lambin et al., 2014, Bush et al., 2015, Lambin et al., 2018).

Even though the resolution of the results of Papers II and III improves upon previous studies, it is still quite coarse; nonetheless, the results can help point at areas to prioritise, in terms of commodities, regions and trade flows. A finding with implications for initiatives aiming to reduce deforestation is that large part of the deforestation is attributed to a handful of commodities (confirming findings of previous studies, e.g., Henders et al. (2015)) and trade flows. This indicates that supply-chain efforts to reduce deforestation that focus on specific commodities and/or trade relationships (such as Brazil's Soy Moratorium and Cattle agreement (Nepstad et al., 2014, Gibbs et al., 2015a, Gibbs et al., 2015b) and zero-

deforestation commitments targeting commodities such as palm oil, beef and high-value crops (Donofrio et al., 2017, Lambin et al., 2018, Garrett et al., 2019) may address a relatively large part of the deforestation and/or deforestation emissions. That said, unless underlying demand is addressed, there will be a risk of leakage (le Polain de Waroux et al., 2017, Lambin et al., 2018).

Given the increasing role of non-state actors and the recent focus on supply-chain sustainability policy as a tool for reducing deforestation (Newton et al., 2013, Lambin et al., 2014, Bush et al., 2015, Lambin et al., 2018), furthering the understanding of how global supply chains link consumers across the world to deforestation and forest degradation in the tropics is key to advance both science and policy addressing tropical deforestation. Future research could further help understand more directly the causal linkages between trade and land-use change, though this is currently limited by the spatial coarseness of trade data and lack of insight into the identity of the actors involved (Godar and Gardner, 2019).

Papers II and III further show some ways in which international trade redistributes pressures on forests. This can be relevant for global goals seeking to halt deforestation or boost reforestation, such as the UN SDGs, the New York Declaration on Forests, and the Bonn Challenge, as well as for commitments seeking to reduce greenhouse gas emissions. For example, Paper II shows that while some countries have successfully begun to increase their forest cover (or reduce their deforestation), this has partly been enabled by importing products from abroad, where their production is associated with deforestation (in the producing countries). In this case, the offset deforestation generally seems to be lower than the forest gains made in the consuming countries, potentially pointing to some efficiency gains. Evaluating the net effects of international trade would require further research; for example, a country relying on domestic production instead of imports may require a larger area to meet the same commodity demand. Further research would also be needed to draw any further conclusions about whether leakage is occurring, but there does appear to be a general need for displacing land-demanding production elsewhere in order to accomplish the gains in forest cover. If the goal is to reduce deforestation and increase forest cover globally, shifting deforestation from one country to another will not be an adequate option.

Similarly, Paper III shows that the deforestation-related carbon emissions embodied in imports are sizeable: for many developed countries, they are comparable in size to the emissions within the country's agricultural sector. For example, for the EU, imported deforestation carbon emissions amount to a sixth of the carbon footprint of an average EU diet. These imported emissions can thus be relevant to account for, e.g., when assessing the environmental impacts of national consumption.

International trade clearly has a role in redistributing pressures on forests, though it is worth noting that international trade is not inherently a bane for forests. Indeed, international trade also brings possibilities of efficiency gains by optimising the use of land between countries and continents and may thus have an important role to play in reaching the global goals seeking to minimise the adverse impacts of deforestation (Lambin and Meyfroidt, 2011, Grau et al., 2013). Further research would be needed to assess the net effects of international trade, evaluating the relative merits of import compared to domestic production. However, it is clear that achieving goals to reduce deforestation (or to reduce other environmental impacts through reducing deforestation) at the international level presents additional challenges to accomplishing them at the national level. Aside from the challenges inherent in reaching agreement and enforcement at the international level, the results presented in this dissertation

indicate that the forest gains and concomitant greenhouse-gas emissions reductions achieved in certain countries have partly been accomplished by displacing these impacts abroad.

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