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On Sustainability of Biomass for Energy and the Governance Thereof

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Cover: Part of Figure 3 in the appended Paper I: Example of modelling results for the Tapajós watershed in Brazil.

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Abstract

Due to concerns about climate change, energy security, and resource scarcity, non-renewable resources are increasingly being displaced by biomass. As with most human activities, the production of biobased products can be associated with negative impacts. Primarily, this relates to the biomass supply systems, i.e., agriculture and forestry, which currently are major causes of biodiversity loss and degradation of ecosystem services. Developing sustainable production systems when transitioning from non-renewable resources to biomass is imperative. **This thesis** aims to clarify the meaning of sustainability in the context of biomass for bioenergy, and contribute to our understanding of how different forms of governance can promote sustainably sourced biomass for bioenergy. The thesis is based on five appended papers: **Paper I** analyses to what extent, where, and under what conditions oil palm for biodiesel in Brazil can be produced profitably, and what risks and opportunities that can be associated with introducing large-scale oil palm production in Brazil. **Paper II** lays the foundation for understanding how new biomass production can be introduced into landscapes while supporting rather than compromising the ability of the landscape to supply other ecosystem services. **Paper III** describes different forms of governance and shows how these can play different roles in promoting sustainable bioenergy in different countries. **Paper IV** focuses on how short rotation coppice production systems are affected by EU policy and how different governance forms can assist in adapting production systems to conform to the corresponding sustainability requirements. Finally, **Paper V** assesses how sustainability certification (private governance) addresses biodiversity conservation and contributes to our understanding of possible improvements.

Keywords: Bioenergy, biomass production, sustainability, land use, governance, certification, environmental legislation, biodiversity, ecosystem services, biomass resources, GIS, spatial modelling

List of appended papers

In order of appearance in the thesis

I **Englund O**, Berndes G, Persson M, Sparovek G, 2015. Oil Palm for Biodiesel in Brazil – Risks and Opportunities. *Environmental Research Letters,* 10, 044002

OE and GB had the idea. OE designed the study, with contributions from GB, MP, and GS. OE performed the analysis. OE analysed the results, with contributions from GB, MP, and GS. OE wrote the paper, with contributions from GB, MP, and GS.

II **Englund O**, Berndes G, Cederberg C. Methods and Concepts for Mapping and Analysing Ecosystem Services in Landscapes: a systematic review. *Working paper*

GB and CC had the idea. OE designed the study. OE performed the analysis. OE analysed the results, with contributions from GB and CC. OE wrote the paper, with contributions from GB and CC.

III **Englund O**. Berndes G, *In press*. The Roles of Public and Private Governance in Promoting Sustainable Bioenergy. In Le Bouthillier Y, Cowie A, Martin P, and McLeod-Kilmurray H (eds). *The Law and Policy of Biofuels*. Edward Elgar Publishing, UK, Chapter 2

OE and GB had the idea. OE designed the study with contributions from GB. OE performed the analysis. OE analysed the results, with contributions from GB. OE wrote the paper, with contributions from GB.

IV **Englund O**, Berndes G, Fredrikson F, Dimitriou J, 2012. Meeting Sustainability Requirements for SRC Bioenergy: Usefulness of Existing Tools, Responsibilities of Involved Stakeholders, and Recommendations for Further Developments. *Bioenergy Research,* 5(3): 606-620

GB had the idea. OE designed the study, with contributions from GB. OE performed the analysis, with contributions from FF. OE and GB analysed the results. OE and GB wrote the paper, with contributions from FF and JD.

V **Englund O**, Berndes G, 2015. How Do Sustainability Standards Consider Biodiversity? *Wiley Interdisciplinary Reviews: Energy and Environment,* 4(1): 26-50

GB had the idea. OE designed the study. OE performed the analysis. OE analysed the results, with contributions from GB. OE wrote the paper, with contributions from GB.

Other relevant publications by the author

Englund O, Berndes G, Persson M, Sparovek G, 2014. Oil Palm for Biodiesel in Brazil: Potentials and Trade-offs, *World Bioenergy 2014 Proceedings,* Svebio, Stockholm, pp. 140-148.

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Contents

1. Introduction

How to manage our natural resources sustainably is one of humanity's most important challenges. Presently, the global demand for resources is in many cases greater than the supply that can be sustained over time (Rockström et al. 2009, Steffen et al. 2015), and as global population (United Nations 2015), affluence, and resource intensity (Mont et al. 2014) continue to increase, so does the demand for resources. The extensive use of fossil fuels has led us to depend on a finite (declining) resource for energy and has created another great challenge: climate change. To allow future generations sufficient access to energy and to avoid severe climatic effects, the use of fossil resources needs to be replaced.[1] By displacing fossil fuels, bioenergy may contribute to solving both challenges, declining energy resources and climate change.

Bioenergy has several advantages: (1) Biomass is a renewable resource, i.e., sustainably managed, the supply will never expire. (2) In theory, bioenergy is therefore also climate neutral.[2] (3) Bioenergy shares several properties with fossil fuels. For example, solid biomass can replace coal, liquid biofuels can replace petrol and diesel, and biogas can replace natural gas, with only small alterations to current infrastructure and end-use applications. (4) Bioenergy can help increase energy availability and security in countries that currently depend on importing fossil fuels from the world's few oil-exporting countries. (5) Where possible, producing bioenergy products for export can strengthen national economies and bring employment opportunities. (6) Introduction of bioenergy feedstock production with modern technology and knowledge in countries with an underdeveloped agricultural sector can modernize the entire agricultural sector, increasing overall yields and agricultural output.

Bioenergy is therefore a highly interesting option for renewable energy systems. However, the production of bioenergy and other biobased products can be associated with negative environmental and socio-economic impacts (Azar 2011). Primarily, this relates to the biomass supply systems, i.e., agriculture and forestry, which currently are major causes of biodiversity loss and degradation of ecosystem services (Steffen et al. 2007, Ellis 2011, Zalasiewicz et al. 2011). The extent to

[1] It should be noted that the transition from fossil fuels can be delayed if combined with carbon capture and storage (to avoid climate impacts). However, the transition is inevitable since fossil resources are limited.

[2] In practice, bioenergy often causes net positive greenhouse gas (GHG) emissions due to the use of fossil fuels in cultivation, transportation, and processing, as well as emissions from the production and use of agricultural inputs, and from changes in land use. However, combined with carbon capture and storage, bioenergy could also cause net negative GHG emissions.

which land can be used to produce bioenergy feedstock is of particular concern (Nakada et al. 2014, Souza et al. 2015).

The future potential for bioenergy has been studied repeatedly (Nakada et al. 2014, Souza et al. 2015), with highly varying results. For example, studies reviewed by Nakada et al. (2014) report a potential of 0-550 EJ for agricultural residues, 0-220 EJ for forestry products, and 0-130 for bioenergy crops in 2050.[3] This large spread can partly be explained by different assumptions on, e.g., future population and the demand for other biobased products, but is often also due to differences in scope. For example, Beringer et al. (2011) estimated the global bioenergy potential from dedicated lignocellulosic biomass plantations under environmental and agricultural constraints, while Schueler et al. (2013) estimated the global bioenergy potential under sustainability restrictions defined by the EU Renewable Energy Directive. Both these studies are examples of bioenergy crop estimates, but since they assess different kinds of biomass production systems using different kinds of constraints, they are difficult to compare.

In addition, studies can present different kinds of potentials: The *theoretical* potential estimates the potential with no constraints; the *ecological* potential considers environmental constraints; the *technical* potential considers what is technically feasible; and the *economical* potential considers what is economically feasible. Finally, the *market* potential considers all of the above constraints as well as what is socially desirable (*social* potential) and demanded on markets.[4] Except for the theoretical potential, which is always the maximum potential, the other potentials cannot be sorted into a general order. Rather, the order varies between, e.g., different feedstock types, regions, and time frames.[5] From an environmental perspective, the market potential should not be greater than the ecological potential, since the market otherwise would allow more biomass extraction than could be sustained over time.[6]

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[3] For reference: In 2010, the global primary energy demand was 520 EJ; the total bioenergy production about 62 EJ, of which 40 EJ “traditional bioenergy”, 21.5 EJ “modern bioenergy” and 4.2 EJ liquid biofuels (Souza et al. 2015). The gross calorific value of all harvested biomass in the year 2000 was about 300 EJ (Beringer et al. 2011).

[4] The market potential can to a large extent be influenced, or even set, by policy. For example, the EU-RED market for biofuels (see Chapter 4) is set by European law to 10% of the energy use in the respective member states’ transportation sectors.

[5] For example, the ecological potential for forest residues is much higher than the economic potential (at least in a near future), while the ecological potential for oil palm biodiesel is much lower than the economic potential (see Paper I).

[6] Provided the entire market potential is utilized. Note that environmental impacts are not only associated with the total *amount* of biomass extracted or the total *area* that is used but also with *how* it is produced and *where*, as will be elaborated.

In a recent synthesis of bioenergy and sustainability (Souza et al. 2015), land availability was claimed not to be a limiting factor for bioenergy: *'Bioenergy can contribute to sustainable energy supplies even with increasing food demands, preservation of forests, protected lands, and rising urbanization. While it is projected that 50 to 200 million hectares (ha) would be needed to provide 10 to 20% of primary energy supply in 2050, available land that does not compromise the uses above is estimated to be at least 500 million hectares and possibly 900 million hectares if pasture intensification or water-scarce, marginal and degraded land is considered.'* Albeit correct, this can give the impression that land is abundant and that large-scale bioenergy expansion is unproblematic. However, land is a relatively scarce resource for which there are competing demands (Smith et al. 2013). Human societies already use roughly half the planet's land surface (see Figure 1), producing biomass with a total energy content equivalent to about 25% of the total global net primary productivity (Krausmann et al. 2013).[7] Even though there is enough land for substantial bioenergy production in theory, large-scale bioenergy expansion may well compete with other land uses, such as food and biomaterials production, and displace natural vegetation (Smith et al. 2013). Obviously, bioenergy might in some places be, or grow to be, a higher priority than some other biobased products, but with global population expected to increase another 30-80% by 2100 (United Nations 2015), with many people adopting more resource intensive lifestyles (Mont et al. 2014), it is clear that land demand will be high and governance of bioenergy and land use in general will be important. Therefore, in the absence of knowledge of *how* and *where* different kinds of feedstock can be produced to limit direct and indirect environmental and socio-economic impacts, and without effective governance that would steer bioenergy expansion in such directions, land availability is indeed a limiting factor for bioenergy.

Suppose all land on Earth were to be equally divided among all humans (Figure 1). We would each have about two hectares, of which a large share is unproductive (0.7 ha),[8] 0.5 ha is forest and about the same is pasture, 0.2 ha is cropland,[9] 800 m^2 is grassland, and 500 m^2 has been developed.[10]

[7] Including the decrease in NPP caused by conversion of natural ecosystems to less productive production systems.

[8] For example: glaciers, deserts, and mountains.

[9] Of which about a third is used for fodder production.

[10] Based on land classification by Bringezu (2014). Note that it can be difficult to distinguish between different land (use) classes and that other studies can provide differing relations.

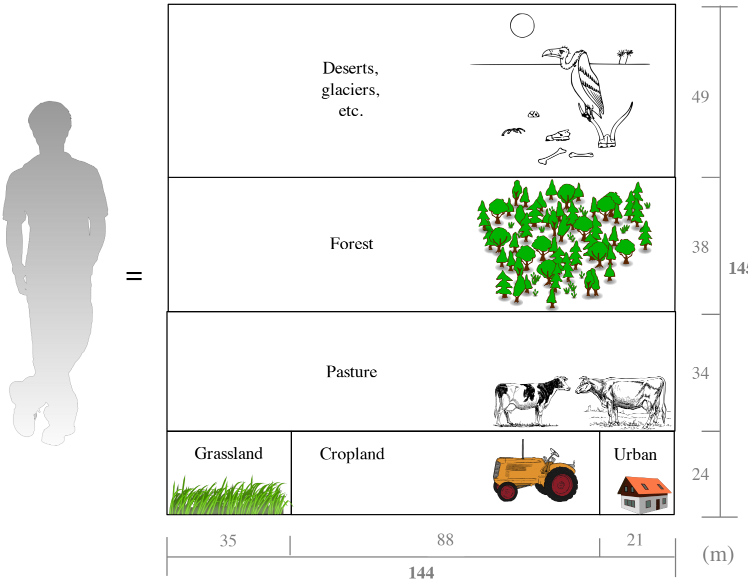

Figure 1: *Global land use per capita. Adapted from Englund (2013).*

If we wanted to produce more biomass for energy, what land would we use? If we use the cropland, it might compromise our ability to produce food. Since nutrition is crucial for survival, we might be reluctant to use this land. The forest may seem like a better alternative. However, it is crucial for providing global and local climate regulation and other *ecosystem services* that we depend on for our survival and well-being.[11] Sourcing biomass from forests thus requires that we not systematically convert or degrade the forest to an extent that it fails to provide the services upon which we depend. What about cultivating grasslands? They are not as important for climate regulation as forests, but they also provide vital services for humans that need to be preserved.[12] What remains of our two hectares is the large pasture. Since meat production[13] requires more land per unit of protein than many vegetarian alternatives (Nijdam et al. 2012), changing our dietary preferences could free up significant amounts of land for new biomass production. Finally, to avoid competition for land, it may be preferable to pursuit ways of incorporating bioenergy production into human and natural systems,[14] but such possibilities are still insufficiently-well understood (Souza et al. 2015). However, intensifying the current production of food and biomaterials may be top priority, as this requires minimal alterations to our lifestyles.

[11] As described in Chapter 3.

[12] For example, grasslands harbour pollinators and species for pest control.

[13] Primarily red meat.

[14] See Chapter 3.

Luckily, as individuals, we need not actually decide how to use "our" two hectares. Land management, all over the world, is influenced by what humans, all over the world, demand. Currently, this demand means that, for instance, more land-efficient, white meat is increasingly replacing red meat, in Europe as well as globally (Henchion et al. 2014), and public concern about climate change is increasing, along with awareness of many other environmental and socio-economic issues. However, despite these trends, it is unlikely that voluntary decisions by individuals will lead to as efficient and sustainable a utilization of the land as possible. For instance, people in developing countries are expected to adopt more resource-intensive lifestyles with increasing affluence (Mont et al. 2014). Further, even if we were all to exhibit much more altruistic behaviour than we currently do,[15] for someone to act in the most "responsible" way requires knowledge of complex interactions between human and natural systems, many of which science has yet to fully understand. However, there is something that *can* effect significant and rapid changes in demand and in how land is managed: effective governance.[16,17] But for this governance to lead to efficient and sustainable land use, it must promote biomass production that is *sustainable.*

1.1 Aim and scope

With the support of the five appended papers, this thesis aims to clarify what sustainability of biomass for bioenergy means as well as contribute to our understanding of how various forms of governance can promote sustainably sourced biomass for bioenergy.

Chapter 2 discusses sustainability of biomass for bioenergy, using the case of oil palm for biodiesel in Brazil (Paper I) as an illustrative example.

Chapter 3 discusses how production landscapes can be managed so as to support biomass production in combination with other ecosystem services, facilitated by methods for mapping ecosystem services at the landscape scale (Paper II).

Chapter 4 describes different forms of governance and shows how these can play different roles in different countries, in promoting sustainable bioenergy systems (Paper III); how different governance forms can assist in adapting short rotation coppice production systems to conform with sustainability requirements introduced

[15] Including altruism benefitting future generations.

[16] See Chapter 4.

[17] Which requires public support. Awareness among the public is thus essential for facilitating ambitious policy targets and for creating markets for sustainably produced products.

by EU policy (Paper IV); and in what ways sustainability certification[18] takes biodiversity conservation into account (Paper V).

[18] A form of private governance, see Chapter 4.

2. Sustainably sourced biomass for energy

Since sustainable development was defined by the World Commission on Environment and Development as *'development that meets the needs of the present without compromising the ability of future generations to meet their own needs'* (WCED 1987), hundreds of studies have provided alternative definitions, theoretical as well as practical. Sustainable development is now perceived as an *'irreducible holistic concept where economic, social, and environmental issues are interdependent dimensions that must be approached within a unified framework'* (IPCC 2007).

Sustainable biomass production commonly refers to practices that are environmentally sound, economically profitable, and socially just. How the three sustainability dimensions are defined and balanced reflects the priorities of societies, and definitions can therefore vary both between societies and over time (Dale et al. 2013). However, the general priorities of societies can differ from those of individuals. If individuals do not agree that biomass produced according to the priorities of society as a whole reflects their view on what is environmentally sound, economically profitable, and socially just, they could constitute markets for biomass produced according to different priorities. In other words, producing biomass sustainably refers to applying practices that avoid environmental and socio-economic impacts that are unacceptable in the eyes of a given society, market, or individual,[19] and since these priorities can differ, sustainability can mean very different things. However, whatever the conception of sustainability, to ensure that biomass production does not cause impacts that are considered unacceptable, it must be made sufficiently clear how biomass should be produced and what land and other resources can be used, i.e., what practices to apply to meet the requirements for sustainability. Such practices can be defined by, e.g., national legislation or farmer guidelines, reflecting a national society's general preferences; sustainability certification standards, reflecting the preferences of customers on a market; or international policies, reflecting the preferences of a regional or global society.[20]

Most standards and guidelines are specific to certain crops and nations, even though there are examples of standards that apply for any feedstock type and in any

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[19] Although it can also mean that biomass production should not only avoid impacts but improve the overall conditions in the landscape. See Chapter 3.

[20] These different *governance forms* are thoroughly described in Chapter 4.

nation.[21] Following the reasoning above, compliance with any sustainability standard that defines clear thresholds and limits and describes specific production methods, would by definition mean that the biomass that is produced is sustainable. However, standards, as is clearly shown in Paper V, differ not only between different feedstock types and regions, but also in scope and in how they prioritize different aspects of sustainability. For example, some may be very focused on the environmental performance of a production system, while others focus more on social aspects. Compliance with a specific standard may thus mean, for instance, that the biomass is sustainable from an ecological, but not from a social, perspective, or vice versa. In addition, even standards that are seemingly similar in scope often differ in stringency, i.e., they have differently strict requirements on how biomass can be produced, and on what land. Biomass produced in accordance with seemingly similar sustainability standards may thus cause varying degrees of environmental and socio-economic impacts.[22]

The ISO 13065:2015, '*Sustainability Criteria for Bioenergy'*, has been under development for several years and was recently published. It provides a *'practical framework for considering environmental, social and economic aspects to facilitate the evaluation and comparability of bioenergy production and products, supply chains and applications'*. This standard does not establish thresholds or limits and does not describe specific bioenergy processes and production methods. Compliance does therefore not determine the sustainability of processes or products,[23] but it does facilitate comparability of various bioenergy processes or products, or even of bioenergy and other energy options, which can be useful given the difficulties in distinguishing between seemingly similar sustainability standards, as discussed above.

Since sustainability of biomass is so ambiguous, science cannot prescribe how to produce sustainable biomass. Rather, science can show society what environmental and socio-economic consequences can be expected from different options. Society can then decide what options to use, given their preferences and priorities, as discussed above. For example, in Paper I, we explore risks and opportunities associated with oil palm production for biodiesel in Brazil. The *risks* that we describe may – depending on what the priorities are – be interpreted as the risks that oil palm production will cause certain unacceptable impacts, while the *opportunities* may be interpreted as opportunities for producing oil palm while

[21] For example, the Roundtable on Sustainable Biofuels (RSB), and the International Sustainability & Carbon Certification (ISCC) - see Paper V.

[22] We discuss the implications of this thoroughly in Paper V.

[23] The standard explicitly states, *'Compliance does not determine the sustainability of processes or products.'*

avoiding such impacts. As scientists we can propose certain sustainability requirements and investigate if these can be met. But societies may have different priorities than the ones expressed through these sustainability requirements. If you interpret sustainability to mean profitable production that does not take place in protected areas or on land where production would reduce the carbon stock, then our oil palm study provides useful information about the potential for *sustainable* oil palm production for biodiesel in Brazil.

2.1 Contributions of the appended papers

Paper I informs the discussion on prospects for oil palm in Brazil, accounting for environmental and economic aspects at a high spatial resolution. It presents a novel approach to estimating biomass potentials using profitability as a prerequisite for potential production.

2.2 Summary of Paper I

Englund O, Berndes G, Persson M, Sparovek G, 2015. Oil Palm for Biodiesel in Brazil – Risks and Opportunities. *Environmental Research Letters* 10, 044002

Introduction

Among cultivated plants, oil palm has the highest known vegetable oil yield and can be a profitable feedstock for biodiesel production (Serraõ 2000, Gui et al. 2008, Butler 2010, Schwaiger et al. 2011). About 90% of global oil palm production takes place in Indonesia and Malaysia, with around six and four million hectares (Mha) of oil palm plantations, respectively. Of these plantations, about 40% were established at the expense of tropical forests (Gunarso et al. 2013) causing negative impacts on, e.g., biodiversity and also greenhouse gas (GHG) emissions associated with the forest conversion and peatland drainage.

The Brazilian government acknowledges the risks of negative environmental impacts associated with oil palm expansion, and the aim is for plantations mainly to be established on degraded agricultural land (Villela et al. 2014). Brazil's *'Agro-Ecological Zoning of Oil Palm in Deforested Areas of the Amazon'* (EMBRAPA 2010) identified 29.7 Mha of land where the Brazilian Investment Bank (BNDES) is allowed to provide credit on favourable terms to support oil palm establishment. About 5 Mha of new oil palm plantations have been authorized so far (Villela et al. 2014). Oil palm may be planted outside the designated areas, but without support from the BNDES. In addition to introducing environmental protection policies, Brazil has launched a number of initiatives that seek to promote and regulate expansion of oil palm, involving, e.g., technical assistance to farmers, agricultural and industrial incentives and credits, sustainability monitoring and evaluation, land titling, traditional people's protection, and social inclusion (Villela et al. 2014).

However, despite the recent policies, large forest areas in Brazil can still legally be converted to cultivated systems (Sparovek et al. 2010).

Here, a spatially explicit model was developed to: (i) determine the net present value (NPV) of establishing new oil palm plantations for biodiesel production under different climate and energy policy regimes in order to map areas in Brazil where production would be profitable; (ii) estimate the associated biodiesel production and land use change (LUC); and (iii) investigate whether pricing of carbon emissions from LUC could make oil palm production unprofitable on lands with high carbon stocks. Finally, we delineate areas where oil palm expansion would minimize LUC emissions and displacement of native ecosystems and avoid impinging on land protected by law.

Methods

The NPV of establishing new oil palm plantations for biodiesel production was calculated (Equation 1) for each hectare in Brazil for a total of 27 scenarios: the main 18 are based on the three energy scenarios from the 2012 World Energy Outlook (WEO) (IEA 2012) – *'Current policies'* (CP), *'New policies'* (NP), and *'450 ppm'* – providing variations in oil, coal, and carbon (C) price developments that affect the willingness to pay for biodiesel and palm oil residues. The WEO scenarios were combined with three different levels of a LUC carbon price to form nine scenarios. Finally, two different establishment years (2013 and 2025) were used for each scenario to analyse how the results differ over time, given the price projections on oil, coal, and carbon. In addition to the 18 main scenarios, all scenarios having an establishment year of 2025 were analysed with both present and prospective (i.e., planned) road infrastructure, to facilitate a complementary analysis of how improvements in road infrastructure would affect the profitability of establishing oil palm plantations. The NPV of establishing oil palm plantations for biodiesel production was estimated for each scenario with a resolution of 100 m.

Equation 1: *Formula for estimating NPV of establishing new oil palm plantations for biodiesel*

$$\begin{aligned} &NPV_{oilpalm}(t) \\ &= Revenue\ from\ timber - Land\ price - Cost\ of\ establishing\ plantations \\ &- Cost\ of\ establishing\ mill - Carbon\ costs/revenues\ from\ LUC \\ &+ \sum_{n=1}^{25}\left[\frac{Revenue\ - Cultivation\ costs - Milling\ costs - Trp\ costs - C\ costs\ (N2O)}{(1+r)^n}\right] \end{aligned}$$

The land price is spatially explicit and based on FNP (2012). Revenue from timber produced when land is cleared to make place for oil palm (in all cells classified as "forest") (Busch et al. 2009) and mill establishment cost are spatially explicit. Cost of establishing plantations is set to be constant (data and references given in the appended supplementary information, SI). Cost of LUC carbon emissions is estimated by multiplying the change in carbon stock in each cell from establishing oil palm plantations by the carbon price in the different scenarios. Here, carbon stocks in natural vegetation are based on Baccini et al (2012), but adjusted using spatial data on current land use (see SI for details). Revenue from palm oil production is spatially and temporally explicit, based on the potential yield in each cell, following a specific yield profile over 25 years (Persson 2012), and the willingness to pay for biodiesel. The latter was assumed to be equal to the willingness to pay for petrodiesel, estimated using projected global oil prices in the different WEO scenarios (IEA 2012), with costs for refining oil into petrodiesel (Li et al. 2012), and the projected EU carbon tax (IEA 2012), added. The willingness to pay for residues (to use for bioenergy) was assumed to be equal to the willingness to pay for coal, calculated using projected coal prices, with a Brazilian carbon tax added in the WEO scenarios that assume such a tax (IEA 2012). Cultivation cost (SUFRAMA 2003) depends on the plantation year. Milling cost per tonne (t) of palm oil and palm kernel oil yield is estimated for each cell on each plantation year (SUFRAMA 2003). Transport cost is calculated using the estimated cost in each cell for transporting one tonne of goods the cheapest way to an export port, multiplied by the palm oil yield in the same cell, depending on the plantation year (see SI). Carbon cost from N_2O emissions is only added in the 450 ppm scenarios, in which Brazil is assumed to have implemented a carbon tax. It is set constant at 0.42 tC/ha a^{-1} multiplied by the carbon price (Forster et al. 2007, IEA 2012, Persson 2012). The discount rate **r** is set at 10% and the plantation lifetime **n** is 25 years (Persson 2012). Spatial NPV calculations, as well as various spatially explicit algebraic and statistical operations on the NPV results, were made using ArcGIS. All costs and prices are expressed in constant (inflation adjusted) USD for the year 2010.

Main findings

The results show that palm oil production for biodiesel can be profitable (positive NPV) over very large areas in Brazil, including areas where oil palm would displace native vegetation and cause LUC emissions. For establishment year 2013, without a price on LUC carbon emissions, results show that it would be profitable to establish oil palm plantations on about 360–390 Mha, corresponding to a biodiesel production almost equal to the present global diesel demand (FAO 2013). The situation for 2025 is similar. These results do not account for the dynamic effects an increase in the biodiesel production of this magnitude would have on global oil prices, and hence on the willingness to pay for biodiesel (Rajagopal et al. 2011). Nevertheless, they give a clear indication of the geographical pattern of exploitation pressure in a situation where biodiesel prices follow the trajectories given in the WEO scenarios (Figure 2).

In the absence of a LUC carbon price, establishment of oil palm plantations would have a positive NPV in almost all forests in Brazil where climate and soil conditions support oil palm cultivation, including rainforests (Figure 2). To illustrate the GHG dimension: if this forest land were converted to oil palm plantations, up to 50 Gt of carbon would be emitted to the atmosphere. This corresponds to over 70 times the emissions from forest conversion and peat oxidation due to oil palm expansion in Southeast Asia in 1990–2010 (Agus et al. 2013) or almost half of the US cumulative emissions from fossil fuels since preindustrial times (Boden et al. 2013). Such forest conversion would also, obviously, cause a number of other impacts, including adverse impacts on biodiversity.

The effects of pricing LUC carbon emissions on the profitability of converting forests to oil palm plantations naturally depends on the carbon price. By 2025, in the 450 ppm scenario, the highest carbon price used ($249/tC) results in oil palm establishment having a positive NPV on 4% of the forest area, compared with 90% in the absence of a LUC carbon price. If this high carbon price is cut in half, oil palm establishment still has a negative NPV on 80–90% of the forest area, but if reduced by two thirds, the NPV would only be negative on about half the forest area. Thus, pricing of LUC carbon emissions may strongly discourage forest conversion to oil palm plantations if the carbon price is sufficiently high, i.e., at least $125/tC.

However, substantial amounts of palm oil may be produced without compromising objectives for GHG emissions reductions and nature conservation. Establishing oil palm plantations on currently unprotected land, where carbon stocks would either increase or be roughly unaffected, would have a positive NPV on 40–60 Mha (Figure 3). The corresponding biodiesel production is estimated at 4–6 EJ/a.[24] Almost all of this land is presently in agriculture, with roughly three-quarters pasture[25] and one-quarter cropland.[26] Conversion of this land would also increase the carbon stock and generate solid biomass fuel from plantation renewal.[27] Taking the *2013, CP, no carbon pricing* scenario as an example,[28] converting all 46 Mha would increase the carbon stock by an estimated 3 $GtCO_2$-eq, corresponding to more than seven times Brazil's current annual emissions of CO_2 from fossil fuel

[24] Equivalent to 40–60 times the current demand for biodiesel in Brazil, 2–3 times the Brazilian demand for petrodiesel and biodiesel combined (Barros 2013), or about 10% of the current global petrodiesel demand.

[25] 15–25% of all pasture in Brazil.

[26] 10–15% of all cropland in Brazil.

[27] However, the net GHG savings that can be obtained by planting on agricultural areas obviously depend on whether such planting indirectly leads to LUC with high GHG emissions elsewhere. The outcome depends on many factors, including governance of land use, food demand development, and productivity development in agriculture, especially concerning meat and dairy.

[28] See Figure 3b for comparisons.

combustion. In addition, it would generate an estimated 2.4 EJ of annual solid biomass fuel from plantation renewal.

The main uncertainties in this study are the discount rate and the oil price projections (the basis for revenues from palm oil). Without a LUC carbon-pricing scheme, using a discount rate of 5% increases the total area with positive NPV by an average 16 and 12% for establishment years 2013 and 2025, respectively. Using instead a discount rate of 15%, the profitable area decreases by 29 and 22%, respectively. The uncertainty of oil price projections has become evident in the light of the recent major oil price decline. Since the willingness to pay for biodiesel is, in principle, positively correlated to the oil price, and since the current oil price is significantly lower than the projections used in our calculations, this would suggest that the NPV is generally overestimated. However, if the oil price would increase to a higher level than projected, the end result may instead be a general *underestimation* of the NPV. In addition, lower oil prices may be accompanied by higher prices for carbon emissions, which would increase the price for petrodiesel and thus compensate for the lower oil prices in the willingness to pay for biodiesel. In scenarios with a LUC carbon-pricing scheme, the estimated carbon content in the assessed carbon pools is another uncertainty. For instance, using the carbon map by Saatchi et al. (2011), instead of the one by Baccini et al. (2012), could yield differing results.

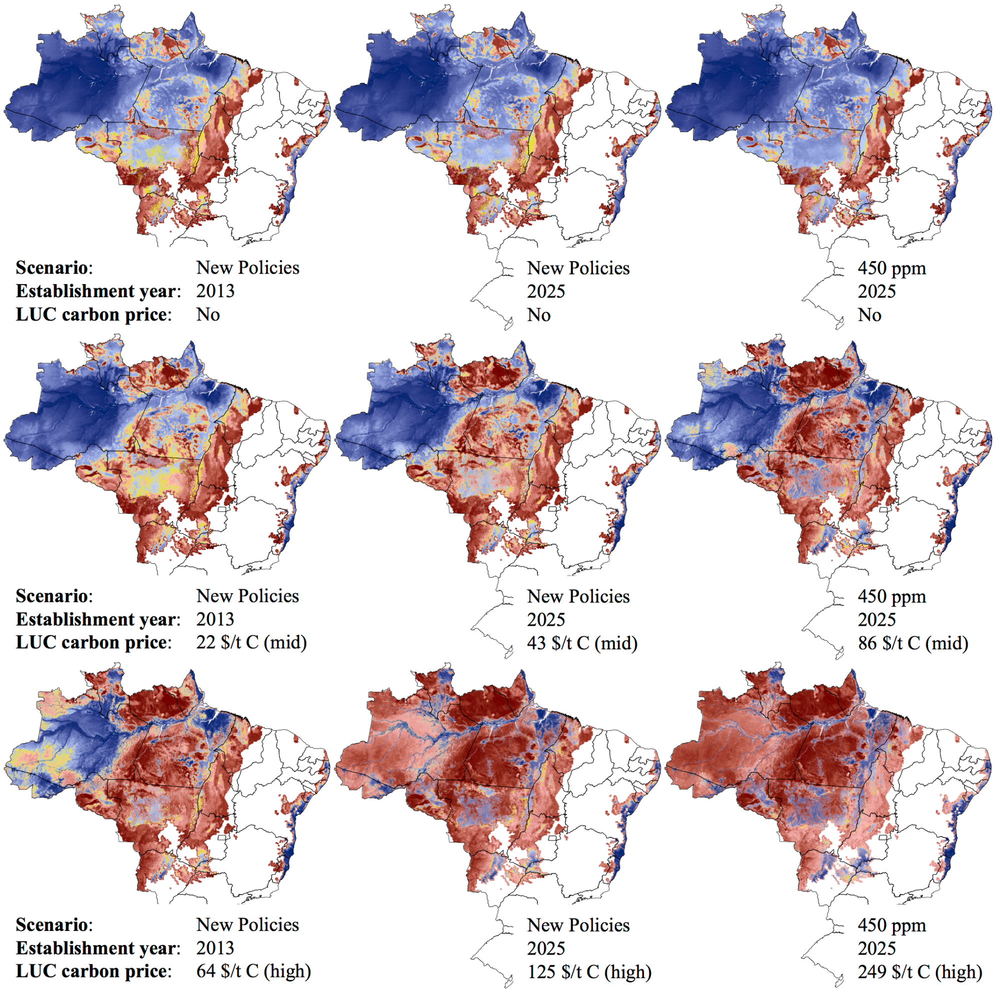

Figure 2: *NPV of establishing new oil palm plantations for biodiesel production in selected scenarios, representative of the variation in results. Red indicates negative, blue positive, NPV. Colours are darkest near the max/min values and lightest near zero. Scenarios: IEA 'New policies' scenario (2013); IEA 'Newt policies' scenario (2025); IEA '450 ppm' scenario (2025). Three levels of LUC carbon prices are shown for each scenario.*

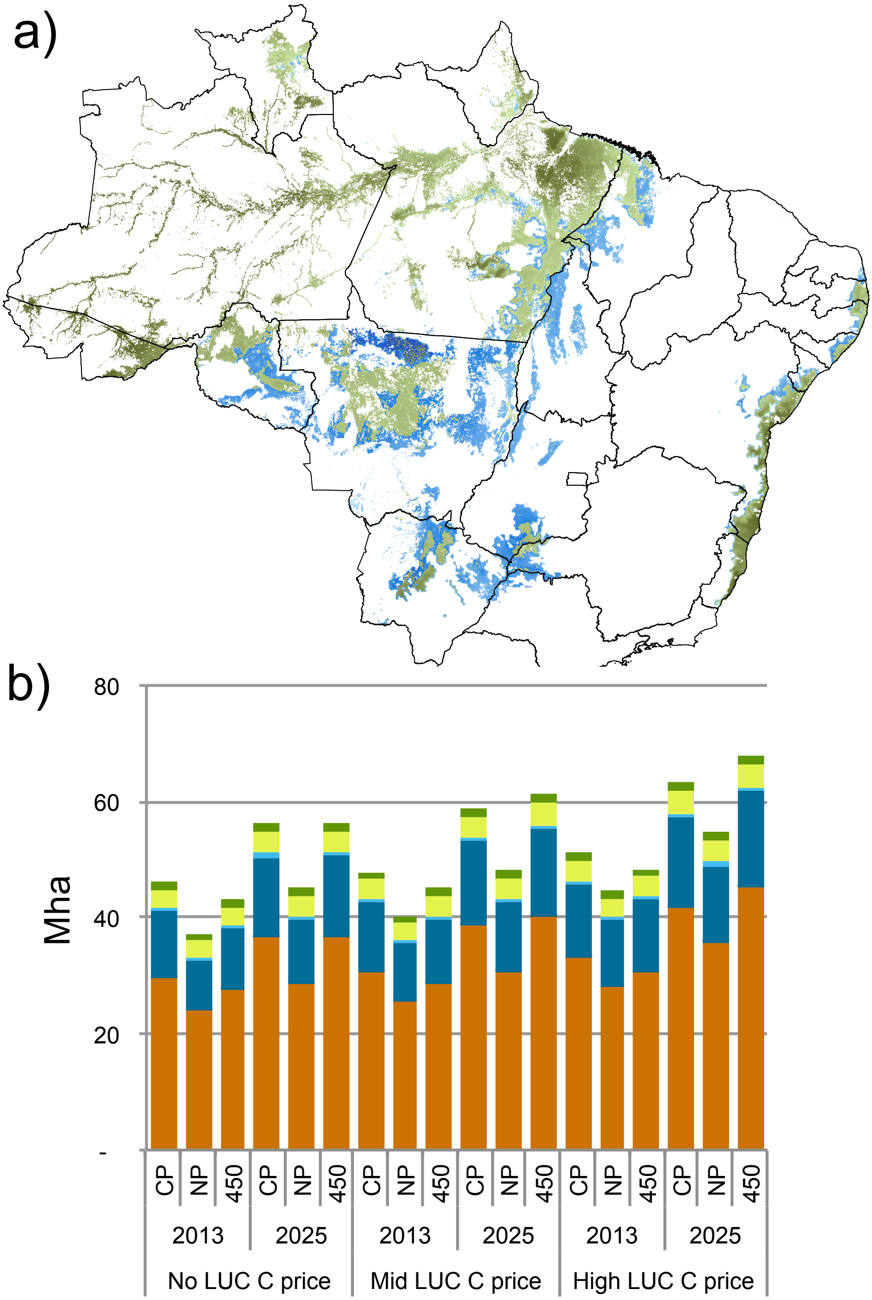

Figure 3: *Areas where establishment of new oil palm plantations would (1) be profitable (NPV > 0); (2) increase carbon stock; and (3) not impinge on land protected by law.* ***(a)*** *Shows the spatial distribution of this land in the scenario with the lowest potential (green) and highest potential (green + blue). Darker colours indicate higher yields;* ***(b)*** *shows quantified results for all scenarios divided into six LULC classes. IEA scenarios: CP='Current policies', NP='New policies', 450='450 ppm'.*

3. Towards multifunctional production systems

Ecosystems provide various goods and services to society, which in turn contribute directly to our survival and well-being. These goods and services are called "ecosystem services" (Daily 1997, MEA 2005). The demand for ecosystem services (ES) is increasing, but a majority of ecosystems are currently being degraded or used unsustainably, with human land use a major cause (Costanza et al. 2014). Loss of biodiversity is of particular concern since it is a major driver of ecosystem change (Hooper et al. 2012). The supply of ES over time is thus at risk.

Biomass for energy[29] is an ES, but as we alter landscapes to obtain biomass, we often alter their capacity to provide other services (Smith et al. 2013).[30] Biomass production that supports biodiversity and enhances rather than degrades the capacity of a landscape to provide other ES could be an attractive option for society (Berndes et al. 2008). However, such possibilities are still insufficiently-well understood (Souza et al. 2015). Designing such *multifunctional production systems* requires a better understanding of how biomass production in landscapes affects ES, which in turn requires a proper understanding of how to assess ES in landscapes.

3.1 Ecosystem services

Some ES have been evident to humans throughout history,[31] but the concept as such started to emerge in the late 1960s and 1970s (Hermann et al. 2011, Portman 2013). Scientists then began to discuss the societal value of nature's functions (King 1966, Helliwell 1969, Dee et al. 1973, Bormann and Likens 1979), and in 1981 the term "ecosystem services" was introduced (Ehrlich and Ehrlich 1981). It was however the Millennium Ecosystem Assessment (MEA) (MEA 2003), following important contributions by, e.g., Daily (1997) and Costanza (1997), that brought global attention to the importance of ES. Today, ES is a significant research and policy topic and there are many modelling and mapping approaches aimed at understanding the stocks, demands and flows of ES on different spatial and temporal scales (Burkhard et al. 2013).

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[29] Or for other purposes.

[30] Environmental consequences, as discussed in the former chapter, can for example be deforestation (a decrease in carbon stock and hence an impact on *climate control*), eutrophication (decreased *habitat suitability* for aquatic species and an impact on *water quality*), and erosion (caused by decreased *regulation of mass flows*). All the italicized items are examples of ecosystem services.

[31] For example, vegetable and animal food products, and wood for heating and construction

Several attempts have been made to construct classification systems for ES (Costanza et al. 1997, Daily 1997, 1999, De Groot et al. 2002, MEA 2003, de Groot 2006, Boyd and Banzhaf 2007, Fisher and Turner 2008, TEEB 2010). It has however been difficult to develop a consistent system suiting all purposes. Costanza (2008) argues that there are many useful ways to classify ecosystem goods and services, and that the goal should not be to have a single, consistent system, but rather a pluralism of typologies that can be useful for different purposes. Even so, the use of multiple classification systems makes comparisons among studies, and the integration of assessments with other data, more difficult (Haines-Young and Potschin 2011). Three of the most commonly used classification systems from the past decades are Costanza et al. (1997), MEA (2003) and The Economics of Ecosystems and Biodiversity (TEEB) (2010). A new classification system is currently underway, the Common International Classification of Ecosystem Services (CICES), developed by the European Environment Agency (www.cices.eu). The aim of CICES is to propose a new standard classification of ES that is both consistent with accepted categorizations and allows easy translation of statistical information between different applications (Haines-Young and Potschin 2011). A comparison of CICES with TEEB (2010), MEA (2003), and Costanza (1997), is provided in Table 2 and 3 in the appended Paper II.

In addition to – and to some extent due to – inconsistent classification, the terminology in ES research has remained inconsistent. It has been argued that definitions of ES are purpose-dependent and should be judged on their usefulness for a particular purpose (Zhang et al. 2007, Lamarque et al. 2011). However – as noted also for classification systems above – coexistence of different terminologies and definitions could impede on-the-ground use of the concept (Lamarque et al. 2011). At present, work is in progress to establish working definitions of commonly used terms (Potschin et al. 2014). This can possibly, along with the advancement of the CICES classification, contribute to harmonization of terminology and make studies more consistent and comparable. The terminology used in this paper (Table 1) is based on Potschin et al. (2014), Crossman et al. (2013), Hermann et al. (2011), Andrew et al. (2015), Mastrangelo et al. (2014), and Bastian et al. (2014).

Table 1: *Definitions of commonly used terms. Adapted from Potschin et al. (2014), Crossman et al. (2013), Hermann et al. (2011), Andrew et al. (2015), Mastrangelo et al. (2014), and Bastian et al. (2014)*

Term	**Definition**
Ecosystem structure	Static ecosystem characteristics: spatial and aspatial structure, composition and distribution of biophysical elements *Example:* land use, standing crop, leaf area, % ground cover, species composition
Ecosystem processes	Dynamic ecosystem characteristics: Complex interactions among biotic and abiotic elements of ecosystems causing physical, chemical, or biological changes or reactions. *Examples:* decomposition, photosynthesis, nutrient cycling and energy fluxes.
Ecosystem functions	The subset of *processes* and *structures* that, if benefiting to human well-being, provide *ecosystem services*. Can be defined as the capacity of ecosystems to provide *ecosystem services*. Example: carbon sequestration
Ecosystem properties	Refers collectively to *ecosystem structure* and *processes*.
Ecosystem services	Direct and indirect contributions of *ecosystem functions* to human well-being. *Example:* climate regulation
Intermediate ecosystem service	*Ecosystem functions* that do not directly benefit to human well-being, but that support other functions that do. Synonymous with 'supporting services'
Ecosystem service providers	The ecosystems, component populations, communities, functional groups, etc. as well as abiotic components such as habitat type, that are the main contributors to specific *ecosystem services*. *Example:* Forest tree communities are ecosystem service providers for global climate regulation.
Human well-being	A state that is intrinsically or instrumentally valuable for a person or society. *Example:* The MEA (2005) classifies components (or drivers) of human well-being into: basic material for a good life, freedom and choice, health and bodily wellbeing, good social relations, security, peace of mind, and spiritual experience.
Ecosystem service supply	The capacity of a particular area to provide specific *ecosystem services* over a given time period.
Ecosystem service demand	*Ecosystem services* used in a particular area over a given time period.
Ecosystem service providing units/areas	Spatial units that are the source of *ecosystem services*. Commensurate with *ecosystem service supply*.
Ecosystem service benefiting areas	The complement to ecosystem service providing areas. *Ecosystem service benefiting areas* may be far distant from respective *providing areas*. Commensurate with *ecosystem service demand*.
Landscape	A mosaic of land cover and land use, viewed at a scale determined by ecological, cultural-historical, social or economic considerations
Landscape services	The contributions of landscapes and landscape elements to human well-being
Landscape multifunctionality	The capacity of a landscape to simultaneously support multiple benefits to society

3.2 Analysing ecosystem services in landscapes

Mapping (in this context referring to spatially explicit quantitative estimates) of ES is essential for many ecosystem service assessments, and has been the subject of several recent reviews (Egoh et al. 2012, Martinez-Harms and Balvanera 2012, Crossman et al. 2013, Andrew et al. 2014, 2015). One cause for concern is that proxy methods, e.g., benefits transfer (Costanza et al. (1997)), are used in the majority of ES assessments (Egoh et al. 2012), possibly indicating that many methods used so far may be unsuitable for landscape scale studies. Proxy methods are much less complex than for example direct mapping with survey and census approaches or empirical production function models, and may thus be an appealing approach for ecosystem service assessments. However, there are several disadvantages with proxy based methods, such as the risk for generalization error, which makes them prone to error (Eigenbrod et al. 2010, Stephens et al. 2015). Since landscapes can typically not be seen as a mere combination of ecosystems, but as the result of interactions between ecosystem structures, -processes and humans (Council of Europe 2000), the use of proxies at the landscape level is particularly sensitive to local conditions. Careful calibration and validation is thus necessary (Stephens et al. 2015), which has typically not been done (Seppelt et al. 2011, Martinez-Harms and Balvanera 2012). Eigenbrod et al. (2010) claim that proxies may be suitable for identifying broad-scale trends in ES, or for global level and rapid assessments (Hermann et al. 2011), but that even relatively good proxies are likely to be unsuitable for identifying hotspots or priority areas for multiple ES.

Land management decisions usually relate to spatially oriented issues (Hermann et al. 2011), especially at the landscape level. In order to use ecosystem service assessment as a basis for spatial planning and decision-making in landscapes, a high level of detail and accuracy is necessary at varying spatial and temporal scales. Since landscapes are spatially diverse, with the service supply unequally distributed across space, changes in service supply must be assessed in spatially explicit ways (Nelson et al. 2009, Willemen et al. 2010, 2012). This may entail direct mapping with survey and census approaches, empirical or rule based models, or proxy based methods (Andrew et al. 2015), depending on, e.g., ecological knowledge and data availability (Hermann et al. 2011, Andrew et al. 2015). However, models, indicators and proxies must be chosen and calibrated carefully, and the results should be validated against empirical data.

3.3 Contributions of the appended papers

Paper II reviews methods for mapping ES in terrestrial landscapes, providing a foundation for assessing the effects on ecosystem services from the introduction of biomass production in landscapes. In addition, the paper clarifies the terminology used in ecosystem services research, as well as the concepts *landscape* and *landscape scale*.

3.4 Summary of Paper II

Englund O, Berndes G, Cederberg C. Methods and Concepts for Mapping and Analysing Ecosystem Services in Landscapes: a systematic review. *Working paper*

Introduction

The aim of this working paper is to identify and qualitatively assess methods for mapping ES in terrestrial landscapes, based on a systematic review of the scientific literature. In addition, it aims to clarify the terminology used in ES research, in particular the concept of *landscape* and *landscape scale*, based on a meta-review of recent literature as well as outcomes from the systematic review.

Methods

In order to clarify the terminology used in ES research, in particular the concept of *landscape* and *landscape scale*, and to develop a proper assessment framework for the systematic review of methods, a meta-review of recent literature was performed. Review articles were identified from keyword searches in the Scopus and Web of Science databases. Other papers were identified by examining both the bibliographies of the papers in the database search and papers that cite them.

For the systematic review, papers reviewed by Andrew et al. (2015) and Crossman et al. (2013), and thus also by Egoh et al. (2012) and Martínez-Harms and Balvanera (2012), were revisited and reviewed on their methods for mapping ES at a landscape scale. An additional literature search was made, that sought to identify relevant papers published after 2012. The full literature selection process is described in Table 2.

The 1112 papers that were identified in the literature search were screened to determine their relevance for this review. There were two criteria that had to be fulfilled for a paper to be regarded as relevant:

1. One or several ES must be mapped. Here only papers that presented spatially explicit results were considered relevant.
2. Studies must be done at a *landscape scale*. Here, studies were considered relevant if they claimed to be made at a landscape scale, for the purpose of landscape planning, or if they referred to the study area as a landscape or as containing landscapes.

A total of 171 papers fulfilled these criteria and were included in the review. See the appended Paper II for a full description of the assessment framework.

Table 2: *Literature selection process for systematic review*

Source	Number of papers	Cumulative number of papers
Papers from Crossman et al. (2013)	108	108
Papers from Andrew et al. (2015)	144	252
Additional search in Scopus[a]	757	1009
Additional search in Web of Science[b]	687	1696
Removing duplicates	-584	1112
Title, abstract, and full text screening	-941	171
Included in review	**171**	

a) TITLE-ABS-KEY("ecosystem service*" OR "landscape service*" OR "ecosystem function*" OR "ecosystem process*") AND TITLE-ABS-KEY("Geospatial" OR "Geographic information system" OR "GIS" OR "map" OR "spatial" OR "indicator*") AND TITLE-ABS-KEY("landscape") AND PUBYEAR > 2012

b) TS=("ecosystem service*" OR "landscape service*" OR "ecosystem function*" OR "ecosystem process*") AND TS=("Geospatial" OR "Geographic information system" OR "GIS" OR "map" OR "spatial" OR "indicator*") AND TS=("landscape") AND PY=(2013 OR 2014 OR 2015)

Main findings

A landscape can be defined as: 'a mosaic of land cover and land use, viewed at a scale determined by ecological, cultural-historical, social or economic considerations'. In the reviewed papers, 94 areas referred to as landscapes were found, varying in size from 24 ha to 122 million ha (Figure 4). This review did not provide any basis for proposing a narrow "typical" landscape area range in ES assessments. Rather, it was observed that there are widely differing views on the meaning of landscape scale.

Of the 347 mapping attempts that were identified (Figure 5), most concerned regulating and maintenance services (165 attempts), followed by cultural (85), and provisioning services (73). Compared with other scales (Egoh et al. 2012, Martinez-Harms and Balvanera 2012, Crossman et al. 2013), cultural services seem to be more frequently mapped at the landscape scale.

Logical models and Empirical models have been most commonly used (86 and 84 times, respectively), followed by Extrapolation (66 times), Simulation/Process models (51 times), Data integration (24 times), and Direct mapping (17 times). Proxy based methods are thus widely used also at the landscape scale. If extrapolation and data integration methods are combined, they constitute the largest method type.

Type of method was in several cases difficult to determine due to a too brief or otherwise insufficient method description, and in nine cases we were unsuccessful in determining what had been done in detail and also what method type that had been used. Several of the reviewed papers failed to facilitate reproduction.

Only 12 percent of the mapping attempts include efforts to validate the results with empirical data. The majority of validation efforts was found in studies that map ES using empirical models, or simulation and process models (fed with empirical data), which indicates that validation is most often done when empirical data must be collected anyway. Different ES can be more or less easy to validate, but validation efforts were found for all the mapped ES.

As Nemec & Raudsepp-Hearn (2013), we find it difficult to generalize about which methods that provide the most credible results. Carefully calibrated empirical or process based models, validated against empirical data, can provide accurate and easily evaluated results, but they might not be relevant for certain ES, study areas, or research groups. Thus, it appears preferable that several methods are considered and that selection is done on the basis of research question and, e.g., competence, data availability, and time frame. It is hoped that this review can serve as a resource for information on how different types of methods can be used to map different ES, and in that way be useful for the design of new studies.

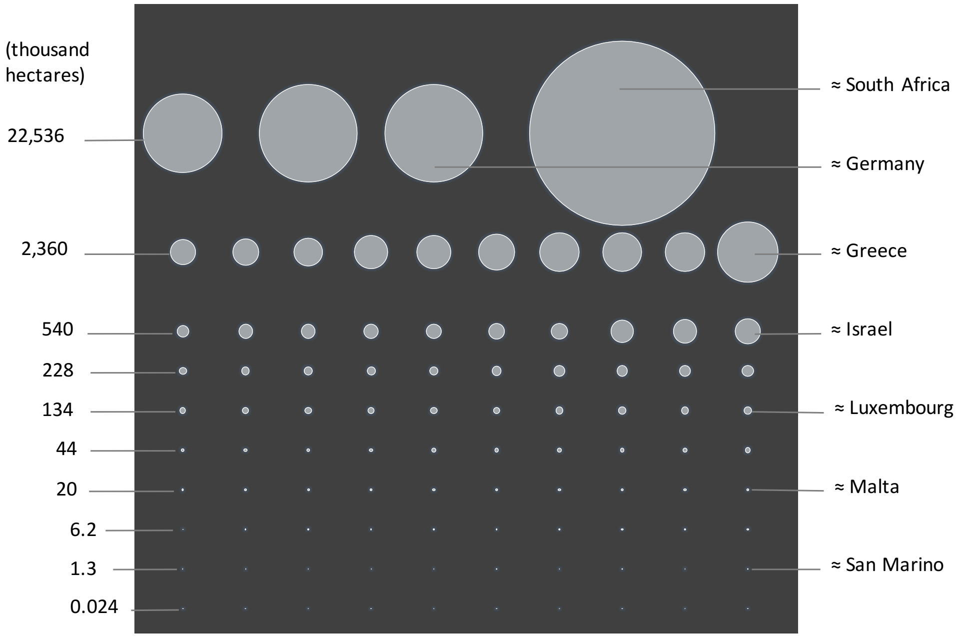

Figure 4: *Size of the 94 areas referred to as "landscape" in the reviewed papers. Size is specified using absolute numbers for the areas at the far left of the figure, and using countries of an approximately equivalent size for the areas at the far right.*

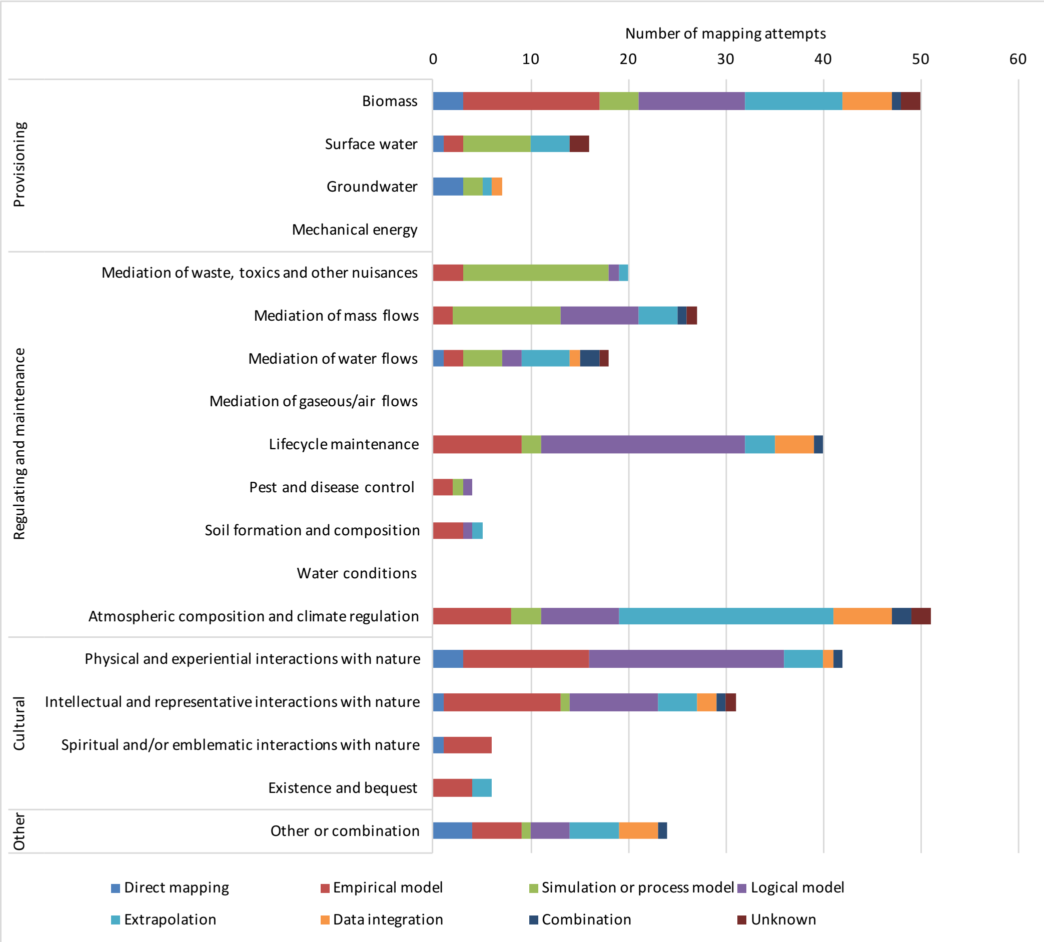

Figure 5: *Number of attempts to map different (groups of) ecosystem services at a landscape scale in the reviewed papers. Divided into different method types (Andrew et al. 2015) used for mapping.*

4. Sustainability governance

Governance is the sum of formal and informal ways actors and institutions, public and private, manage common affairs. It is a continuing process through which diverging interests may be accommodated and cooperative action may be taken (The Commission on Global Governance 1995, Lima and Gupta 2013). Sustainability governance is concerned with promoting the positive effects of production or development processes while avoiding/mitigating their negative impacts (Schut et al. 2014), considering environmental, social, and economic aspects of sustainability. Bioenergy supply chains involve several layers of governance, including mechanisms that specifically address bioenergy (e.g. bioenergy sustainability standards and certification systems) and regulation of sectors involved in bioenergy supply chains. This can involve environmental legislation, labour regulations, environmental codes, best-management agriculture/forestry practices, and international trade standards (Lima and Gupta 2013, Pelkmans et al. 2014, Schut et al. 2014). Here, three forms of governance for the promotion of the sustainable production of biomass and bioenergy are described and discussed: *domestic public governance*, *domestic private governance*, and *international private governance.* Additional tools that can be used to guide biomass production along a more sustainable path are also discussed.

4.1 Public governance

Prior to international integration of many economic activities, *domestic public governance* was the primary form of governance (Schut et al. 2014). In this form of governance, national legislation, formulated and enforced by nation-state institutions, regulates all activities within its jurisdictional limits. Domestic public sustainability governance of biomass and bioenergy[32] is framed by national legislation. Since the comprehensiveness of national legislation varies, production of biomass and bioenergy is governed differently from country to country. In order for domestic public governance to be effective in promoting sustainable production, the legislation needs to be not only sufficiently comprehensive but also effectively enforced. In many countries, insufficient enforcement capacity and/or political will makes legislation ineffective, regardless of its comprehensiveness (Mayer and Gereffi 2010).

Following the definition of domestic public governance, *international public governance* would consider the role of international legislation or policies in governing activities. However, there are limitations to the scope of international public governance. International policies that affect private governance are here

[32] Henceforth referred to as "domestic public governance".

referred to as *policy-driven international private governance*. International public governance can also affect the promotion of sustainable bioenergy either by influencing domestic public governance or by restricting international markets (e.g. using sanctions).[33]

4.2 Private governance

Private governance exists when non-governmental institutions enable or constrain activities within an economy in the public interest (Büthe 2010a, 2010b, Mayer and Gereffi 2010). Domestic private sustainability governance,[34] i.e., where non-governmental institutions govern activities on the basis of sustainability principles, can emerge when domestic consumers demand products meeting more stringent, or different, sustainability requirements than those associated with the public governance system. Producers will provide the products desired by the market by, for instance, adopting codes of conduct and corporate social responsibility schemes.[35] In the case of biomass, producers mainly rely on voluntary sustainability certification for verifying and communicating to consumers that products are produced in accordance with sustainability principles. These principles are defined in a certification standard using criteria and indicators formulated by government or non-governmental organizations, and/or private companies. These may be monitored and verified through third-party independent auditing. Domestic private governance will not be effective in promoting sustainable production of biomass and bioenergy without a sizeable domestic market for certified bioenergy products, although private actors may see other advantages such as promotion of a green company profile or legitimation of the bioenergy sector in general (Huertas et al. 2010, Stupak et al. 2015).

In a global economy, a product often originates in a country other than where it is purchased. Thus, consumers who try to make environmentally conscious purchasing decisions, and regulatory agencies and governments involved in enforcing sustainability standards, need to be concerned with multinational value chains. To a significant extent these are controlled by large private companies rather than nations (Mayer and Gereffi 2010). The scale and complex structure of production and related processes in global economies challenge the capacity of nation-state institutions to govern activities beyond their borders and jurisdiction (Mayer and Gereffi 2010). This, along with an increased interest in neoliberal

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[33] Neither of these are further discussed here.

[34] Henceforth referred to as "domestic private governance".

[35] The creation and growth of markets for sustainable products can also be the outcome of producer push, where producers actively promote their products and in this way contribute to market growth by influencing consumer choices. Companies may therefore not only respond to a demand but also create the demand.

programs of deregulation and privatization, creates the space for alternative forms of private governance extending across national borders (Mol 2010, Abbot 2012).

International private sustainability governance[36] is present where non-governmental institutions govern activities that transcend nations and emerges when there is an international demand for sustainable products. This demand can be either consumer-driven or (public) policy-driven. As with domestic private governance, consumer-driven international private governance can emerge when consumers demand sustainably produced products meeting more stringent, or different, sustainability requirements than those associated with the public governance system in the producing country. Producers will respond by providing, for example, certified products for an international market. Policy-driven international private governance can play a role in countries where there are producers that target export markets. Such incentives may be created by sustainability policies. The EU *Renewable Energy Directive* (EU-RED) (European Council 2009), for example, includes specific sustainability requirements for biofuels with which companies producing for the EU-RED market must comply. Compliance with these requirements can be verified through an approved voluntary certification scheme. This form of public policy-driven international private governance, using third party certification schemes for verification, was considered an effective method for governing bioenergy sustainability in a recent global bioenergy certification survey (Pelkmans et al. 2013).

In response to concerns about unintended consequences of the production and use of biomass for energy, producers of biomass feedstock in the private sector, as well as governmental and non-governmental organizations, have taken initiatives to develop criteria and indicators for sustainable bioenergy supply chains, as a means toward regulating the bioenergy sector. The sustainability certification schemes that are being developed or implemented by a variety of private and public organizations can apply to a variety of feedstock production sectors (notably forest and agriculture sectors) and bioenergy products, ranging from relatively unprocessed forest and agriculture residues to electricity and refined fuels, such as ethanol and biodiesel. They can apply to entire supply chains or only certain segments (O'Conell et al. 2009, van Dam et al. 2010, Junginger et al. 2011, Stupak et al. 2011). In addition, a number of non-operational sustainability standards exist, developed to guide or influence other actors involved in developing operational standards. Such guidelines have been developed by, e.g., the International Tropical Timber Organization (ITTO), for sustainable management of tropical forests; the International Federation of Organic Agriculture Movements (IFOAM), for organic agriculture; and the Global Bioenergy Partnership (GBEP), for sustainable

[36] Henceforth referred to as ‘international private governance’.

bioenergy feedstock production. Many sustainability standards exist, both mandatory and voluntary, with varying scope. They also differ in how they prioritize different aspects of sustainability. For example, some may be very focused on the environmental performance of a production system, while others focus more on social aspects.

Studies show that there are many challenges associated with the current status of sustainability certification and standards (O'Conell et al. 2009, van Dam et al. 2010, Junginger et al. 2011, Stupak et al. 2011, Englund et al. 2012). According to non-certified producers, main barriers include high administrative complexity, high costs, and small market advantages (Goovaerts et al. 2013, Pelkmans et al. 2013). In addition, stakeholders along bioenergy supply chains may need to comply with different standards to maintain market access and to comply with legislative mandates. Consumers who try to make environmentally conscious purchasing decisions, and regulatory agencies and governments involved in enforcing sustainability standards, may find it difficult to manage a wide range of systems that use different criteria/indicators. Thus, the proliferation of schemes and standards has led to confusion among the actors involved, market distortion and trade barriers, an increase in commodity costs, and questions about the adequacy of the systems in place and how to develop systems that are effective and cost-efficient (Buytaert et al. 2011, Magar et al. 2011, van Dam and Junginger 2011, Pelkmans et al. 2013). A recent study undertaken to monitor the actual implementation process of sustainability certification of bioenergy found that there is no global/common definition of how to translate the sustainability concept in practice, i.e., how to measure sustainability and which criteria/indicators to use (Pelkmans et al. 2013).[37] The study called for a globally harmonized approach and establishment of a common language, including terminology, to describe sustainability and specify how to verify and document it.[38]

In addition to certification schemes, certain markets have developed their own rules and requirements that producers have to comply with to gain access. Stakeholders involved with bioenergy that is used within the European Union (EU) have to specifically consider the EU Renewable Energy Directive (RED), which mandates levels of renewable energy use within the EU and also includes a sustainability scheme for liquid biofuels and other bioliquids. However, it is relevant for all types of bioenergy (European Council 2009). In order to ease the process of proving compliance with the sustainability requirements, the EU-RED has approved a set of certification schemes that suffice to verify compliance. This makes it easier for

[37] Which is to be expected, given the reasoning on sustainability in Chapter 2.

[38] Steps towards this have since been taken with the new ISO standard (see Chapter 2).

producers, since they only need to comply with one standard to gain access to several markets.[39]

4.3 Other governance tools

In addition to the governance forms described above, there are additional tools that can be used to guide biomass production along a more sustainable path. For example, producer manuals can be designed to help producers prepare for complying with sustainability standards or at least avoid unnecessary environmental consequences, and environmental impact assessments (EIAs) can be used by funding agencies to verify that a proposed project complies with certain sustainability requirements. Whether such tools should be regarded as a form of public or private governance depends on whether they are required or promoted by governmental or non-governmental institutions. The usefulness of both producer manuals and EIAs is assessed in Paper IV.

4.4 Contributions of the appended papers

Paper III describes different forms of governance and shows how these can play different roles in different countries in promoting sustainable bioenergy production. In particular, it analyses the *de facto* extent to which public governance can suffice to promote sustainable biomass production.

Paper IV focuses on how short rotation coppice production systems are affected by EU policy and how different governance forms and complementary tools can assist in adapting production systems to conform with the corresponding sustainability requirements.

Paper V assesses the ways in which sustainability certification (private governance) takes biodiversity conservation into account and contributes to our understanding of how private governance can be improved in this respect.

In addition, **Paper I** contributes to our understanding of the extent to which a specific policy instrument in Brazil, pricing of carbon emissions from land use change,[40] could make oil palm production unprofitable on lands with high carbon stocks.

[39] That is, both the market for certified goods and the EU-RED.

[40] A form of domestic public governance.

4.5 Summary of Paper III

Englund O, Berndes G, *In press*. The Roles of Public and Private Governance in Promoting Sustainable Bioenergy. In Le Bouthillier Y, Cowie A, Martin P, and McLeod-Kilmurray H (eds). *The Law and Policy of Biofuels*. Edward Elgar Publishing, UK, Chapter 2

Introduction

The role of domestic public governance in the context of international bioenergy supply chains has received little attention in the literature. In particular, there is little information about the relevance and effectiveness of public governance in developing countries with little experience in production and trade of "modern" bioenergy, such as biofuels for transport and pellets for heat and power production (Schut et al. 2014). This paper reports the outcome of a study to address these issues by assessing (i) the extent to which domestic public governance can promote sustainable production of biomass and bioenergy; and (ii) the potentially complementary or substitutive roles of domestic and/or international private governance. First, we propose a framework for identifying the status of domestic public governance, and the potential roles of domestic and international private governance, in promoting sustainable production of biomass and bioenergy. Second, the results of applying this framework to 161 countries are presented, aiming to describe where domestic public governance can successfully promote sustainable production, where the legal and institutional challenges are most critical, and where different forms of private governance can be complementary to, or fill the void of, domestic public governance. Third, a deeper analysis is presented for 13 countries, showing how the countries' environmental legislation covers different aspects of sustainability relevant to bioenergy. This provides insights about the *de facto* extent to which public governance can suffice to promote sustainable biomass production.

Methods

A theoretical framework was outlined for identifying the status of domestic public governance in promoting sustainable production of biomass and bioenergy and the potential roles of international and domestic private governance in different types of countries (Table 3). This framework was applied to 161 countries.

To estimate the potential to produce bioenergy products for export, we used indicators of actual trade dependency and capacity for food self-sufficiency to categorize countries as *more* or *less likely* to produce bioenergy products for export in the near future (“National export potential” in Table 3).

Table 3: *The roles of international and domestic private governance in different countries, or subnational regions having some degree of autonomy, depending on the (i) national export potential; (ii) domestic demand for sustainable products; and (iii) status of domestic public governance.*

National export potential	**Domestic demand for sustainable products**	**Status of the domestic public governance system**	
		Potentially sufficient legal framework	*Challenges with legislation and/or enforcement*
Yes	*Yes*	Both domestic and international private governance can be complementary	Both domestic and international private governance can have an important role
Yes	*No*	International private governance can be complementary	International private governance can have an important role
No	*Yes*	Domestic private governance can be complementary	Domestic private governance can have an important role
No	*No*	Domestic public governance the only possible form	

The demand for sustainable bioenergy products was assumed to reflect the demand for sustainable products in general. This was assessed using three indicators: (i) The Human Development Index (UNDP 2013); (ii) the average value of the World Values Survey 1981-2008 questions *'Would give part of my income for environment'*, *'(Would support) increase in taxes if extra money used to prevent environmental pollution'*, and *'Confidence: Environmental Organizations'* (World Values Survey 2008); and (iii) organic agriculture as a percentage of the total agricultural area (FAO 2013).

The comprehensiveness of environmental legislation was assessed using three indicators: (i) criterion 11 on *'Policies and Institutions for Environmental Sustainability'* in the World Bank IDA Resource Allocation Index (IDA 2011); (ii) indicator on *'pesticide regulation'* in the Environmental Performance Index (Hsu et al. 2014); and (iii) the number of ratified environmental treaties (IUCN 2012), relative to other countries.

The capacity to enforce legislation was assessed using six indicators: (i) The Bertelsmann Transformation Index variable *'Rule of Law'* (Bertelsmann Transformation Index 2014); (ii) the Freedom in the World variable *'Rule of Law'* (Freedom House 2013); (iii) the International Country Risk Guide indicator *'Quality of Government'* (The PRS Group 2013); (iv) the World Bank IDA Resource Allocation Index criterion *'Property Rights and Rule-based Governance'* (IDA 2011); (v) the Worldwide Governance Indicators indicator *'Rule of Law (Estimate)'* (Kaufmann et al. 2013); and (vi) the Economic Freedom Index factor *'Property Rights'* (The Heritage Foundation 2014).

For 13 countries, all legal documents (Σ=1677) available in the ECOLEX database (FAO et al. 2011) were reviewed on how they relate to bioenergy and, if so, how they cover different aspects of sustainability relevant to bioenergy (Table 4).

Table 4: (A) *Areas of bioenergy addressed by environmental laws.* ***(B)*** *Sustainability aspects for which coverage was assessed.*

(A) Areas of bioenergy	Feedstock production	Biofuel feedstock production
		Agriculture
		Forestry
		Nature and biodiversity protection
		Other land-use or land use change
	Processing	Biofuel processing
		Industrial activities
	Other	Other connections
(B) Assessed sustainability aspects	EU-RED requirements	Clearing of forests - (Article 17:3a; 17:4bc)
		Impacts on areas designated for nature protection purposes - (Article 17:3bi)
		Impacts on rare, threatened and endangered species - (Article 17:3bii)
		Conversion of grasslands - (Article 17:3c)
		Drainage of peatlands - (Article 17:5)
		Conversion of wetlands - (Article 17:4a)
	General sustainability aspects	Social sustainability
		Biodiversity
		GHG emissions
		Carbon stock
		Air, water and soil
		Ecosystem services
		Land use

Main findings

In many countries, domestic public governance does not suffice to promote sustainable production of biomass and bioenergy products, due to challenges with both legislation and enforcement (Figure 6). Alternative sustainability governance forms can play an important complementary role (Figure 6). Domestic private governance can rarely fill this role in countries with legal challenges of inadequate legislation and/or enforcement, due to low domestic consumer demand for products meeting certain sustainability requirements (Figure 6). However, in most countries, international private governance can have a role in, and can therefore contribute to, the development of sustainable bioenergy supply chains (Figure 6), especially regarding GHG emissions, air quality, and conversion of wetlands and grasslands, which often are not covered well by national legislation (Figures 7 and 8). However, domestic public governance cannot be fully replaced by national or international private governance. Countries need to address the legal and institutional challenges associated with public governance, which will always be essential in the promotion of sustainable bioenergy. Finally, there is a need for coordination within and between different governance forms. If well-coordinated, the various actors engaged in bioenergy and sustainable development may achieve effective governance of bioenergy supply chains.

Regarding the coverage of national legislation, the results indicate that there may be an overlap between EU-RED requirements and national legislation in relation to *clearing of forests*, *impacts on threatened species* and *impacts on protected areas*. On the other hand, *conversion of wetlands* and notably *drainage of peatlands* and *conversion of grasslands*, appear to be seldom restricted by law (Figure 7). Complementary forms of governance may be required to regulate these activities. At a general level, national legislation may overlap alternative forms of governance in relation to *social sustainability* and *land use*, and, to a lesser extent, *water*, *soil* and *biodiversity*. Requirements related to *ecosystem services*, *carbon stock*, *air*, and, notably, *GHG emissions*, are less likely to exist in national legislation (Figure 8). Complementary forms of governance may be required if these concerns are to be taken into account.

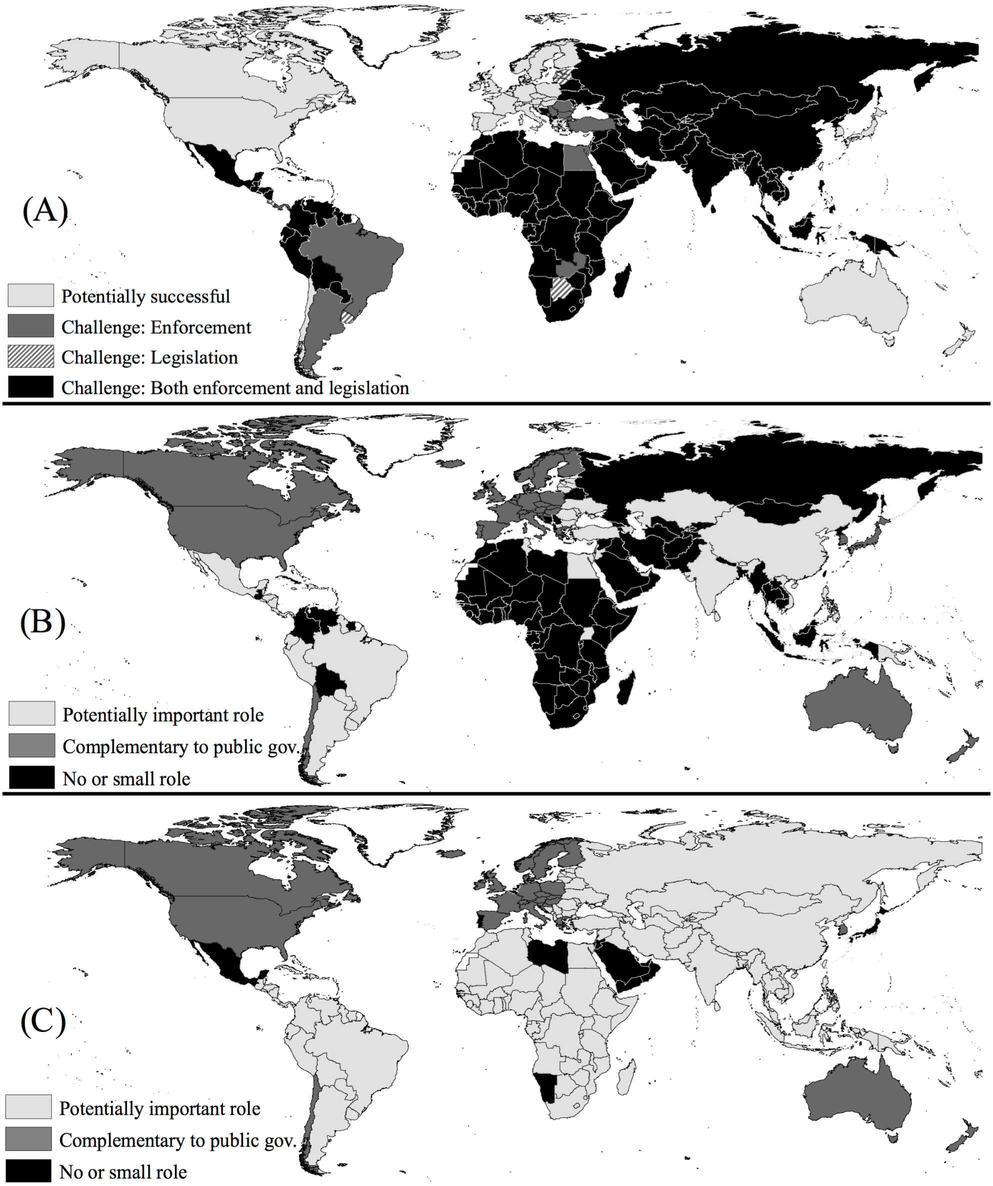

Figure 6: (A) *status of domestic public governance,* ***(B)*** *role of domestic private governance, and* ***(C)*** *role of international private governance, in promoting sustainable biomass production*

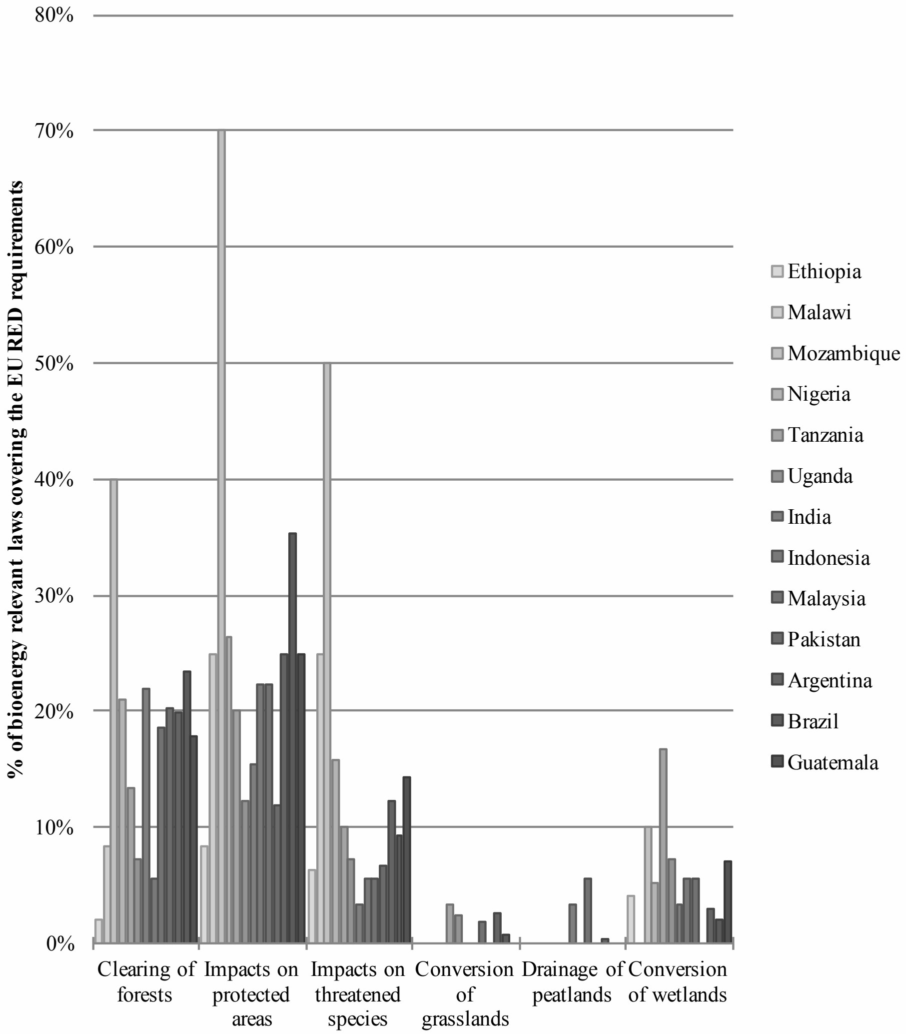

Figure 7: *Percentage of bioenergy-related laws in each country that restrict activities similarly to the EU-RED sustainability requirements*

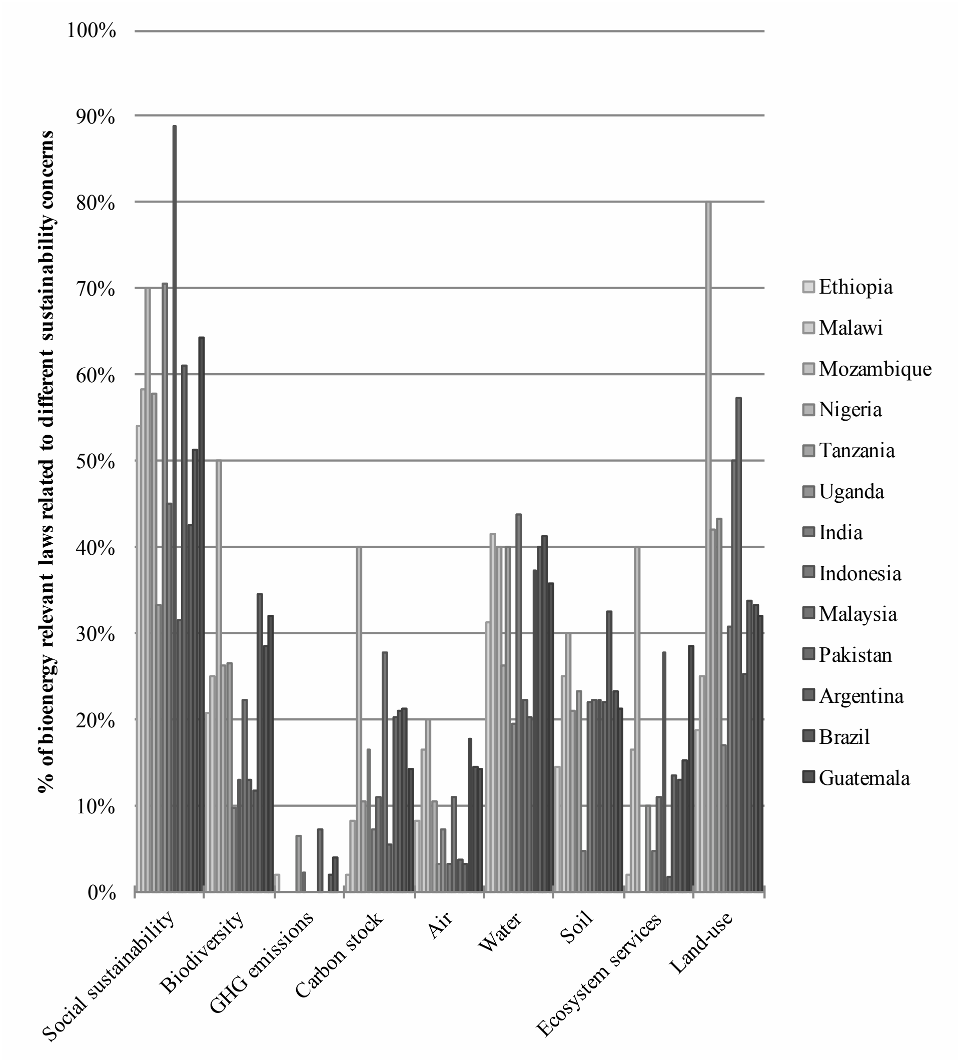

Figure 8: *Percentage of bioenergy-related laws in each country that focus on various general aspects of sustainability*

4.6 Summary of Paper IV

Englund O, Berndes G, Fredrikson F, Dimitriou J, 2012. Meeting Sustainability Requirements for SRC Bioenergy: Usefulness of Existing Tools, Responsibilities of Involved Stakeholders, and Recommendations for Further Developments. *Bioenergy Research*, 5(3): 606-620

Introduction

Short rotation coppice (SRC) (e.g. willow or poplar) is considered an important biomass supply option for meeting the European renewable energy targets (Styles and Jones 2007). An expansion of SRC, especially in agricultural areas near the end user of biomass (e.g. heat and electricity plants for direct biomass combustion), is expected in several European countries.

In this paper we present an overview of existing and prospective sustainability requirements, as well as of Member State (MS) reporting obligations in the EU-RED, and show how these *RED-associated* criteria may affect different stakeholders along the SRC bioenergy supply chain—from feedstock producers to energy consumers. We also attempt to outline a framework for engaging relevant stakeholders in the development of SRC. This framework has two purposes: (1) to facilitate the development of SRC production systems that are attractive from the perspectives of all stakeholders; and (2) to ensure that the SRC production is RED eligible.

Methods

Existing or prospective sustainability criteria relevant for SRC were derived from the EU-RED, as described in Table 5. These *RED-associated* sustainability criteria were then sorted under specific categories to put them into the correct context and finally evaluated on their relevance for SRC bioenergy on a national level.

The stakeholder landscape was investigated using in-house experience and stakeholder consultation, to identify principal stakeholders involved in SRC bioenergy. A general SRC bioenergy supply chain was created (Figure 9), and the stakeholders' roles in meeting RED-associated criteria were discussed.

__Table 5:__ Components of the EU-RED, from which existing and prospective sustainability criteria relevant for SRC were derived

Sustainability requirements for liquid biofuels, or bioliquids	**Monitoring and reporting obligations**	**Methodology for calculating GHG emissions savings**	**Sustainability considerations requiring no particular actions at present**
Currently not mandatory for SRC bioenergy, but may be so in future revisions. EC recommends that they also be included in national sustainability schemes for solid and gaseous biomass used in electricity, heating, and cooling.	Such obligations typically concern impacts due to production and use of bioenergy in general; i.e., no distinctions are made between liquid, solid, or gaseous biofuels.	Considering these in a sustainability framework for SRC bioenergy would support the involved stakeholders in producing bioenergy with high GHG emissions savings.	May be subject to reporting and monitoring obligations in the future or even become additional sustainability requirements.

Producer manuals, environmental impact assessments (EIAs), and certification schemes can all provide guidance as well as contribute to the monitoring and verification of sustainable biomass production. In order to determine whether these tools, individually or combined, can be useful for ensuring that SRC bioenergy is produced in accordance with the RED-associated criteria, they were assessed in terms of their coverage in relation to these criteria.

- Ten producer manuals for willow and/or poplar coppice production, including site selection, planting, and harvesting, were collected and analysed.
- Nineteen EIAs were collected from bioenergy projects that include the establishment of plantations or large-scale agricultural operations, and/or construction of a biofuel processing plant. Depending on the nature of the assessed bioenergy projects, EIAs were sorted into three categories: Plantations, Biofuel plant, and Plantations and biofuel plant.
- A review of international sustainability certification schemes relevant for SRC bioenergy was performed. Based on this, the role of certification in national SRC bioenergy sustainability frameworks was discussed.

Main Findings

Eighteen sustainability criteria associated with the EU-RED were identified as relevant for stakeholders involved in SRC bioenergy (Table 6). These are related to (1) existing and prospective legally binding sustainability requirements; (2)

reporting obligations for MSs; and (3) the methodology for calculating GHG emissions savings.

Table 6: *RED sustainability categories and associated sustainability criteria of national relevance for SRC bioenergy production*

RED-categories	Associated sustainability criteria		Current status
Biodiversity	1.1	Preservation of natural forests	Existing requirement
	1.2	Preservation of areas designated for nature protection purposes or for the protection of rare, threatened, and endangered species	Existing requirement
	1.3	Preservation of highly biodiverse grasslands	Existing requirement
	1.4	Impacts on biodiversity	MS reporting obligation
GHG emissions	2.1	Preservation of peatlands	Existing requirement
	2.2	GHG emissions from extraction or cultivation of raw materials	GHG emissions savings calculation
	2.3	GHG emissions from processing	GHG emissions savings calculation
	2.4	GHG emissions from transport and distribution	GHG emissions savings calculation
	2.5	Carbon capture and replacement	GHG emissions savings calculation
	2.6	Co-generation of electricity, if producing bioliquids	GHG emissions savings calculation
Carbon stock	3.1	Preservation of wetlands	Existing requirement
	3.2	Preservation of continuously forested areas	Existing requirement
	3.3	Restoration of degraded land	GHG emissions savings calculation
	3.4	Restoration of contaminated land	GHG emissions savings calculation
Air, water and soil	4.1	Impacts on air quality	MS reporting obligation / Prospective requirement
	4.2	Impacts on water quality	MS reporting obligation / Prospective requirement
	4.3	Impacts on water availability	MS reporting obligation / Prospective requirement
	4.4	Impacts on soil quality	MS reporting obligation / Prospective requirement

It is important that a sustainability framework is designed so as to facilitate stakeholder interaction to clarify the stakeholders' respective roles and responsibilities and to identify points where conflicts of interests may arise and where there are trade-offs between partially incompatible goals and objectives. Proper consideration of all relevant aspects therefore requires all stakeholders in the SRC supply chain to be engaged in the development of SRC production systems and requires a landscape perspective.

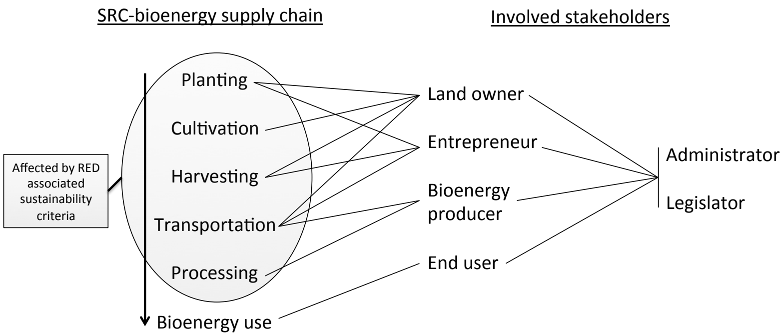

Figure 9: *A typical SRC bioenergy supply chain, specifying the involvement of principal stakeholders in the various supply chain segments*

Producer manuals, EIAs, and voluntary certification schemes can help producers take RED-associated criteria into account, but currently they do not—individually or combined—suffice for this purpose for SRC bioenergy.

Producer manuals need to be complemented to sufficiently cover the RED-associated criteria, and advice on how producers should monitor their activities in order to demonstrate compliance should be provided. *EIAs* also need to be extended to sufficiently consider all criteria, but they also need to be streamlined to become less time consuming and expensive. Regarding *voluntary certification schemes*, national sustainability frameworks for SRC need to be designed so that the producing stakeholders are well informed about the availability and relevance of certification options, which in most cases are likely to vary between countries. The coverage of certain certification schemes in relation to the RED-associated criteria also needs to be assessed on a country level, while considering outcomes from the EC benchmarking process.

Thus, a sustainability framework for SRC bioenergy can have several components. Most importantly, though, a sustainability framework needs to provide landscape-level processes and engage all involved stakeholders. Ensuring that developments progress in line with the interests of all stakeholders requires coordination, supplied by an appropriate institution.

4.7 Summary of Paper V

Englund O, Berndes G, 2015. How Do Sustainability Standards Consider Biodiversity? *Wiley Interdisciplinary Reviews: Energy and Environment,* 4(1): 26-50

Introduction

Biodiversity presents a challenge for sustainability certification. While there is wide support for the objective of conserving biodiversity,[41] operationalizing, through guiding principles, criteria/indicators, and legislation, is complicated. For example, in 2009, the EU-RED established that raw materials used for the production of biofuels and bioliquids may not be produced on land that had the status of highly biodiverse grassland in or after January, 2008 (European Council 2009). However, the European Commission is still in the process of operationalizing elements of the biofuel sustainability criteria, including clarifying some of the requirements that need to be met with respect to the biodiversity criteria, e.g., in relation to highly biodiverse grasslands.

In this paper, we assess how different types of sustainability standards take biodiversity into account. First, biodiversity is defined and strategies for biodiversity conservation discussed. Then, standards for sustainable production of biomass in agriculture and forestry are evaluated based on how they take biodiversity into account, i.e., how they attempt to prevent actions that can threaten biodiversity and support actions that can conserve it. We also assess how sustainability standards address the conversion of certain ecosystem types. Finally, we discuss key barriers to, and challenges for, certification schemes and make recommendations for further development of sustainability standards.

Methods

Four different categories of standards were considered, standards for: (1) certification of sustainable forest management; (2) certification of sustainable agricultural management; (3) certification of sustainable production of specific crops commonly used as biofuel feedstock; and (4) sustainable production of unspecified biofuel feedstock. In addition, guidelines for development or implementation of standards that can be sorted under these categories were also considered. A total of 26 standards were selected for the assessment, including 11 for forest management, 9 for agricultural management, and 6 for biofuels (Table 7). All selected standards include a set of principles and criteria/indicators, or the

[41] For instance, the Convention on Biological Diversity has 193 parties and 168 signatures (CBD 2014).

equivalent,[42] indicating that standard's requirements for production to be considered sustainable or responsible.

Table 7: *Overview of the schemes/organizations for which standards were assessed*

	Scheme/Organization	Abbreviation	Code
Forest management	Forest Stewardship Council (FSC)	FSC	F1
	Sustainable Forestry Initiative (SFI)	SFI	F2
	Finnish Forest Certification System (FFCS)	FFCS	F3
	Malaysian Timber Certification System (MTCS)	MTCS	F4
	Canadian Standards Association (CSA)	CSA-SFM	F5
	Green Gold Label (GGL)	GGLS5	F6
	Naturland	Naturland Forest	F7
	International Tropical Timber Organization (ITTO)	ITTO	F8
	African Timber Organization (ATO) / ITTO	ATO/ITTO	F9
	ITTO / International Union for Conservation of Nature (IUCN)	ITTO/IUCN	F10
	Ministerial Conference on the Protection of Forests in Europe	PEOLG	F11
Agricultural management	Global Partnership for Good Agricultural Practices (GLOBALGAP)	GLOBALGAP	A1
	KRAV - Swedish Organic Agriculture	KRAV	A2
	European Union (EU)	EU Organic	A3
	United States Department of Agriculture (USDA)	USDA-NOP	A4
	Green Gold Label Agricultural Source	GGLS2	A5
	Fairtrade	Fairtrade	A6
	Naturland	Naturland production	A7
	International Federation of Organic Agriculture Movements	IFOAM	A8
	Sustainable Agriculture Network / Rainforest Alliance	SAN/RA	A9
Biofuel-related	Roundtable on Sustainable Palm Oil (RSPO)	RSPO	B1
	Roundtable on Responsible Soy (RTRS)	RTRS	B2
	Bonsucro	Bonsucro	B3
	Roundtable on Sustainable Biofuels (RSB)	RSB	B4
	International Sustainability & Carbon Certification (ISCC)	ISCC	B5
	Greenergy	Greenergy	B6

A general biodiversity-focused *benchmark standard* was developed using seven *principles*, based on threats to, and strategies for conserving, biodiversity, under which 26 *criteria* were defined and sorted. These criteria translate the broadly formulated principles into concrete actions applicable to both agricultural and forest management. The selected standards were then individually compared with the benchmark standard, and for each benchmark criterion it was determined whether a specific standard was *compliant* or not. Based on this, the overall biodiversity *stringency* of a standard was then determined.

Given that land conversion may induce adverse effects on biodiversity, how the standards address conversion of certain types of ecosystems was also investigated for (i) tropical and subtropical forests; (ii) temperate forests; (iii) boreal forests; (iv) wetlands; (v) grass-, shrub- and woodlands; and (vi) degraded land.

[42] Standards often differ in their terminology.

Main Findings

In summary, the assessed biofuel-related standards had the highest level of compliance with the benchmark standard, complying on average with 72% of the benchmark criteria, compared to 61% for the agricultural standards and 60% for the forestry standards. Fairtrade and SAN/RA (agriculture) and RSPO and RTRS (biofuel) were the most stringent, while GGLS5 and PEOLG (forest), GLOBALGAP, EU Organic, NOP, and GGLS2 (agriculture), and ISCC (biofuel) were the least stringent (Table 8).

In general, the assessed standards take *Overexploitation, Habitat destruction and fragmentation,* and *Habitat degradation and modification* well into account, while *Invasive species and GMOs, Research, awareness and education,* and *Energy use and GHGs* are often poorly considered.

There are notably large differences in stringency between some standards having a similar scope. For example, IFOAM, which sets the “norms” for organic agriculture, is significantly more stringent than both EU Organic and NOP. In addition, KRAV endorses EU Organic, even though KRAV is classified as *stringent* and EU Organic is *unstringent*. Further, the SFI standard, a forest industry initiative, shows similar stringency as the FSC standard, which is often regarded as more thorough in its coverage of ecological issues (Clark and Kozar 2011). Furthermore, the high stringency in the Fairtrade standard, and to some extent also SAN/RA, was unexpected, as these were perceived to primarily focus on social aspects.

Regarding ecosystem conversion, forestry standards typically only protect areas that are considered high conservation value (HCV). They also tend to limit the HCV assessment requirements to include forested land only; i.e., they do not prevent conversion of highly biodiverse grasslands or wetlands into certified plantation forests. Agricultural standards cover more ecosystem types and typically do not provide for much flexibility: Specific ecosystem types are no-go areas, or there are no conversion restrictions at all. The inflexibility of several of the agricultural standards may result in the unavailability of areas that could have been beneficially converted into sustainable cultivation, such as some degraded grasslands. The biofuel-related standards are influenced by EU-RED and cover ecosystem conversion comprehensively, using a combination of HCV requirements and strict protection measures. Finally, some standards (EU Organic, NOP, and GGLS2) do not restrict land conversion at all. This may not be a large problem in countries with stringent legislation and sufficient enforcement capacity, but in countries where this is lacking, natural vegetation may be converted into certified agriculture, impacting biodiversity.

All the assessed standards can, to a varying degree, be improved to better consider biodiversity. The benchmark standard presented in this paper could be used to develop more concrete criteria/indicators that fit into the scope of individual standards. Sustainability standards need to be harmonized and made more homogenous, while maintaining relevance for their intended production systems. Stringency and comprehensiveness need to be balanced against feasibility from a biomass-producer perspective. Unnecessary requirements that increase administrative burden and cost without improving conservation outcome should be avoided.

Table 8: *Compliance with benchmark principles. Green (+) indicates "considered"; yellow (+/-) indicates "partly considered"; orange (–) indicates "disregarded". F1-F11 constitute the eleven forestry standards, A1-A9 the nine agriculture standards, and B1-B6 the six biofuel-related standards.*

Principle	F1	F2	F3	F4	F5	F6	F7	F8	F9	F10	F11
1. Endangered species	+/-	+	+	+	+/-	+/-	+	+	+/-	+	+/-
2. Habitat destruction and fragmentation	+	+/-	+	+	+	+/-	+	+	+	+	+/-
3. Habitat degradation and modification	+/-	+	+/-	+/-	–	–	+	+/-	+/-	+/-	–
4. Overexploitation	+	+/-	+	+	+	+	+	+	+	+	+/-
5. Invasive species and GMOs	+	+/-	+/-	+/-	+	–	+	–	+/-	+/-	–
6. Energy use and GHGs	+/-	+/-	–	+/-	+/-	+/-	+	+/-	+/-	–	+/-
7. Research, awareness and education	+/-	+	+/-	+/-	+/-	–	–	–	+/-	+	+

	A1	A2	A3	A4	A5	A6	A7	A8	A9
1. Endangered species	–	+	–	–	–	+	+/-	–	+/-
2. Habitat destruction and fragmentation	+/-	+/-	+/-	+/-	–	+	+	+	+
3. Habitat degradation and modification	+/-	+	+/-	+	+	+	+	+	+
4. Overexploitation	+	+	+	+	+/-	+	+	+	+
5. Invasive species and GMOs	–	+/-	+/-	–	–	+/-	+/-	+/-	+/-
6. Energy use and GHG	+/-	+	–	–	–	+/-	+/-	+/-	+
7. Research, awareness and education	+/-	–	–	–	+/-	+	+/-	–	+

	B1	B2	B3	B4	B5	B6
1. Endangered species	+	+	+	+/-	–	+
2. Habitat destruction and fragmentation	+	+	+	+	+/-	+/-
3. Habitat degradation and modification	+	+	+	+	+	+
4. Overexploitation	+	+	+/-	+	+/-	+
5. Invasive species and GMOs	+/-	+/-	+/-	+/-	–	+/-
6. Energy use and GHG	+	+	+/-	+/-	+/-	+/-
7. Research, awareness and education	+/-	+/-	+/-	–	+/-	+/-

5. Summary and Conclusions

5.1 Sustainability of biomass for energy

Producing biomass sustainably refers to applying practices that avoid environmental and socio-economic impacts that are unacceptable in the eyes of a given society, market, or individual. In practice, this requires prioritising among such impacts, and since priorities can differ among societies, markets, and individuals, *sustainable biomass* can mean very different things. To ensure that biomass production does not cause impacts that are considered unacceptable, it must be sufficiently clear what practices to apply—how biomass should be produced and what land and other resources that can be used—to meet the requirements for sustainability. Such practices can be defined by, e.g., national legislation or farmer guidelines, sustainability certification standards, or international policies. Since sustainability of biomass is so ambiguous, science cannot prescribe how to produce sustainable biomass. Rather, science can show society the expected environmental and socio-economic consequences of different options. Societies can then make informed choices among the options, given their preferences and priorities.

Paper I informs the discussion about prospects for oil palm in Brazil, accounting for environmental and economic aspects at a high spatial resolution. It presents a novel approach to estimating biomass potentials using profitability as a prerequisite for potential production. Oil palm was found to be profitable on extensive areas, including areas under native vegetation where establishment would cause large land use change (LUC) emissions. However, some 40–60 Mha could support profitable biodiesel production corresponding to approximately 10% of the global diesel demand without causing direct LUC emissions or impinging on protected areas. Pricing of LUC emissions could make oil palm production unprofitable on most lands where conversion would impact native ecosystems and carbon stocks, if the carbon price is at the level \$125/tC, or higher.

5.2 Multifunctional production systems

Biomass for energy – or for other purposes – is an ecosystem service, but as we alter landscapes to provide it, we often change the capacity of these landscapes to provide other ecosystem services (Smith et al. 2013). Biomass production that supports biodiversity and enhances rather than degrades the capacity of a landscape to provide ecosystem services could be an attractive option for society. However, designing such *multifunctional production systems* requires an improved understanding of how biomass production affects ecosystem services, which in turn requires an increased understanding of how to assess ecosystem services in landscapes.

Paper II reviews methods for mapping ES in terrestrial landscapes, providing a foundation for assessing the effects on ecosystem services from the introduction of biomass production in landscapes. Of the 347 mapping attempts that were identified, most concerned regulating and maintenance services (165 attempts), followed by cultural (85), and provisioning services (73). Compared with other scales, cultural services seem to be more frequently mapped at the landscape scale. Logical models and Empirical models have been most commonly used (86 and 84 times, respectively), followed by Extrapolation (66 times), Simulation/Process models (51 times), Data integration (24 times), and Direct mapping (17 times). Proxy based methods are thus widely used also at the landscape scale. If extrapolation and data integration methods are combined, they constitute the largest method type. As Nemec & Raudsepp-Hearn (2013), we find it difficult to generalize about which methods that provide the most credible results. Carefully calibrated empirical or process based models, validated against empirical data, can provide accurate and easily evaluated results, but they might not be relevant for certain ES, study areas, or research groups. Thus, it appears preferable that several methods are considered and that selection is done on the basis of research question and, e.g., competence, data availability, and time frame. It is hoped that this review can serve as a resource for information on how different types of methods can be used to map different ES, and in that way be useful for the design of new studies.

5.3 Sustainability governance

Governance is the sum of formal and informal ways actors and institutions, public and private, manage common affairs. Sustainability governance is concerned with promoting the positive effects of production or development processes whilst avoiding/mitigating their negative impacts. Bioenergy supply chains involve several layers of governance, including mechanisms that specifically address bioenergy (e.g. bioenergy sustainability standards and certification systems) and regulation of sectors involved in bioenergy supply chains. This can involve environmental legislation, labour regulations, environmental codes, best-management agriculture/forestry practices, and international trade standards.

Paper III describes different forms of governance and shows how they can play different roles in different countries, in promoting sustainable bioenergy production. In particular, it analyses the *de facto* extent to which public governance can suffice to promote sustainable biomass production. In many countries, domestic public governance does not suffice to promote sustainable production of biomass and bioenergy products, due to challenges with both legislation and enforcement. Alternative sustainability governance forms, primarily international private governance, can play an important complementary role. The results indicate that there may be an overlap between EU-RED requirements and national legislation in relation to *clearing of forests*, *impacts on threatened species* and *impacts on protected areas*. On the other hand, *conversion of wetlands* and notably *drainage of peatland* and *conversion of grasslands*, seldom appear to be restricted by law. At a general level, there may be overlaps between national legislation and alternative forms of governance in relation to *social sustainability* and *land use*, and, to a lesser extent, *water*, *soil* and *biodiversity*. Requirements related to *ecosystem services*, *carbon stock*, *air*, and, notably, *GHG emissions*, are less likely to exist in national legislation.

Paper IV focuses on how short rotation coppice production systems are affected by EU policy and how different governance forms and complementary tools can assist in adapting production systems to conform with the corresponding sustainability requirements. It was found that producer manuals, EIAs, and voluntary certification schemes can all be useful for ensuring RED eligibility. However, they are currently not sufficiently comprehensive, neither individually nor combined, and suggestions for how they can be more complementary are given. Geographical information systems offer opportunities for administrative authorities to provide stakeholders with maps or databases over areas/fields suitable for RED-eligible SRC cultivation. However, proper consideration of all relevant aspects requires that all stakeholders in the SRC supply chain become engaged in the development of SRC production systems and that a landscape perspective is used.

Paper V assesses in what ways sustainability certification (private governance) takes biodiversity conservation into account, and adds knowledge on how private governance can be improved in that respect. Of the 26 assessed standards, the biofuel-related standards demonstrated the highest level of compliance with the benchmark. On average, they complied with 72% of the benchmark's component criteria, compared to 61% for the agricultural standards and 60% for the forestry standards. In general, the assessed standards consider *habitat destruction*, -*fragmentation*, *-degradation*, *-modification* and *overexploitation* well, while *invasive species and GMOs, research, awareness and education,* and *Energy use and GHG* are often poorly considered. Regarding ecosystem conversion, forestry standards typically only protect areas that are considered high conservation value (HCV). They also tend to limit the HCV assessment requirements to include forested land only, i.e., they do not prevent conversion of highly biodiverse grasslands or wetlands into certified plantation forests. Agricultural standards cover more ecosystem types and typically do not provide for much flexibility: specific ecosystem types are either no-go areas or there are no conversion restrictions at all. The biofuel-related standards are influenced by EU-RED and cover ecosystem conversion comprehensively, using a combination of HCV requirements and strict protection measures. Some standards do not restrict land conversion at all. This may not be a large problem in countries with stringent legislation and sufficient enforcement capacity, but in countries where this is lacking, natural vegetation may be converted into certified sustainable agriculture.

5.4 Future research

Further research of the kind exemplified by Paper 1 is required for societies to better understand the environmental and socio-economic consequences entailed by different bioenergy options. Such research can either aim at a better understanding of the current context or build understanding to support the development of future pathways. The challenge is not just to choose among options but also to realize them. This requires further research on how different forms of governance can be effective in promoting options that meet the selected sustainability criteria, while anticipating and avoiding indirect effects that limit the actual benefits. The latter requires a better understanding of land-use and commodity-market dynamics. A systems approach will be needed, and many of the research questions will be interdisciplinary.

I intend to continue to use spatial modelling to assess the risks and opportunities that can be associated with different kinds of bioenergy feedstock and land-use systems. Initially, I will focus on the use of forest residues in Europe—an interesting source of biomass with significant theoretical and ecological potential. I also intend to continue the project initiated in Paper II, assessing how different kinds of biomass production can be integrated in different kinds of landscapes, while supporting biodiversity and other ecosystem services. Finally, I intend to explore novel ways of using *big data,* e.g., geotagged images, to map ecosystem services. Computational power and data availability now offer opportunities for developing methods that scientists could only dream of as recently as a decade ago.

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