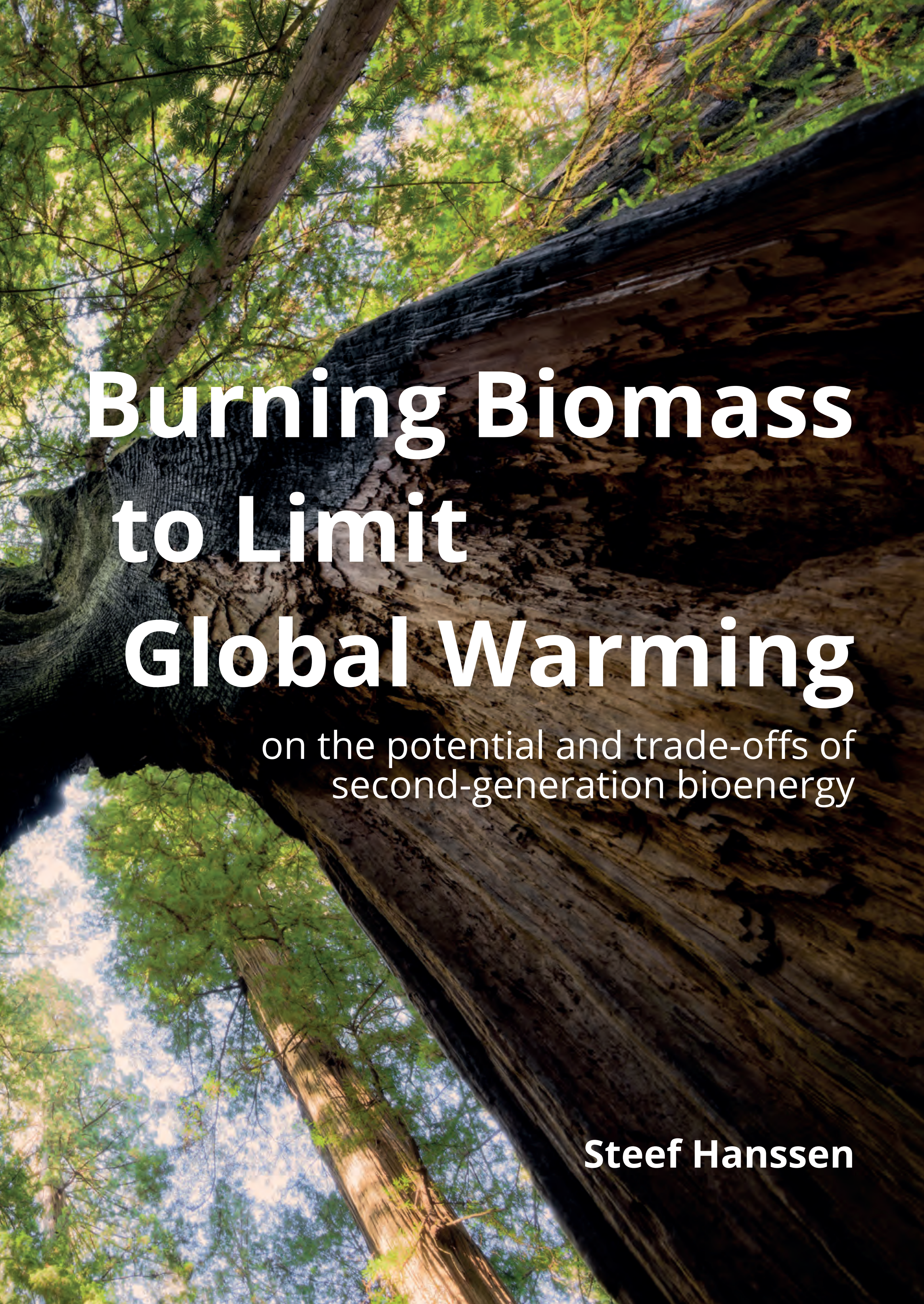

Burning Biomass to Limit Global Warming

on the potential and trade-offs of second-generation bioenergy

Steef Hanssen

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Burning Biomass to Limit Global Warming

on the potential and trade-offs of second-generation bioenergy

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Contents

1 Introduction

1.1 Bioenergy

Over the last million years, thousands of generations of humans have *burned biomass* to stay warm, cook food, shed light, and repel danger (Berna et al., 2012; Mallol & Henry, 2017; Brittingham et al., 2019). Bioenergy in fact predates our species and shaped its evolutionary path, enabling our loss of fur, smaller guts and bigger brains (Wrangham & Carmody, 2010). With the mastery of fire at its origin, bioenergy can in a broader sense be defined as any form of energy outside our bodies derived from 'biomass', i.e., recently living biological materials (US DOE, 2020). It has formed a central part of life across all human cultures until the advance of industrialisation. And even as the world industrialised, it was not until the beginning of the twentieth century that bioenergy was overtaken by coal as the largest global source of primary energy (Figure 1.1). Currently, bioenergy provides around 10% of the global primary energy supply (IEA, 2019). This encompasses traditional bioenergy, i.e., the burning of firewood and animal dung for cooking and heating that is still common in developing countries, but also *modern bioenergy*: biomass processed into modern energy carriers like electricity and transport fuels. It is this modern variant of bioenergy, and its potential to mitigate the 21st century threat of climate change that forms the focus of this thesis.

Modern bioenergy (henceforth: 'bioenergy') can be based on a wide range of biomass feedstocks. So-called first-generation bioenergy is based on food crops with high carbohydrate or oil contents, such as grains or rapeseed and are suitable for bioethanol and biodiesel production, respectively (Turkenburg et al., 2012). Second-generation bioenergy feedstocks include: i) woody or herbaceous ('lignocellulosic') biomass from purpose-grown trees or grasses, ii) residual lignocellulosic biomass from agriculture (e.g., straw or corn stover), forestry (tree tops and branches or sawdust) or landscape management (mown grass or cleared trees), or iii) various 'waste' streams, such as animal manure, waste cooking oils, and organic waste (Antizar-Ladislao & Turrion-Gomez, 2008; IEA, 2010; Sims et al., 2010; Turkenburg et al., 2012). These feedstocks are processed in an array of conversion pathways to ultimately form fuels, electricity and heat (Figure 1.2). The production of third-generation biofuel from algal biomass still *requires* large amounts of energy (Raheem et al., 2018; De Bhowmick et al., 2019) and is not considered in this thesis.

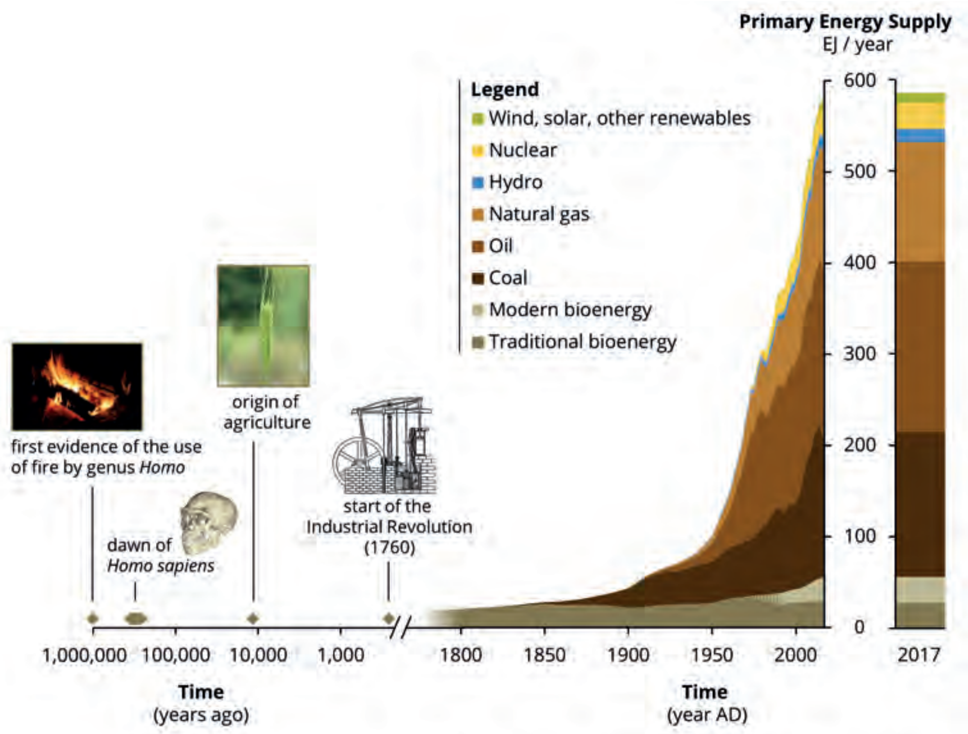

Figure 1.1 | A brief historical overview of (bio)energy. On the left are several key developments in the history of mankind and its use of bioenergy. The right side of this graph shows the global primary energy supply per energy source from the year 1800 onwards. Note that due to data gaps, modern bioenergy statistics before 1990 form an estimate only. This figure is based on Schlebusch et al. (2017), Hublin et al. (2017), Colledge et al. (2004), Smil (2017), Our World in Data (2019), BP (2018), and IEA (2017ab, 2018a, 2019), as explained in detail in appendix I.

The use of bioenergy has several advantages over other renewable energy sources, such as solar, wind and hydropower. The diversity of energy carriers produced (fuels, electricity, heat) means that bioenergy can be used in many different applications and often within existing infrastructure. Electricity based on biomass can for instance provide base load power, but also balance out electricity systems that are heavily dependent on intermittent renewables such solar and wind energy. Biofuels can supply sectors that are difficult to electrify or decarbonise in other ways, such as aviation and shipping (Sims et al., 2010; Wang & Ling, 2016). Bioenergy can also provide a new market for farmers and a model for economic growth in developing countries, while increasing energy security (Campbell et al., 2008; Chum et al., 2011), or be a means to more effectively deploy and add value to residues and waste streams (Carriquiry et al., 2011; Smeets et al., 2015; Creutzig et al., 2015). Moreover, when bioenergy is combined with carbon capture and storage (BECCS), it has the unique ability to sequester atmospheric CO_2 and geologically store it while producing energy (Obersteiner et al. 2001; Gough & Upham, 2011; Kemper et al., 2015). This can result in so-called negative CO_2 emissions, as detailed below.

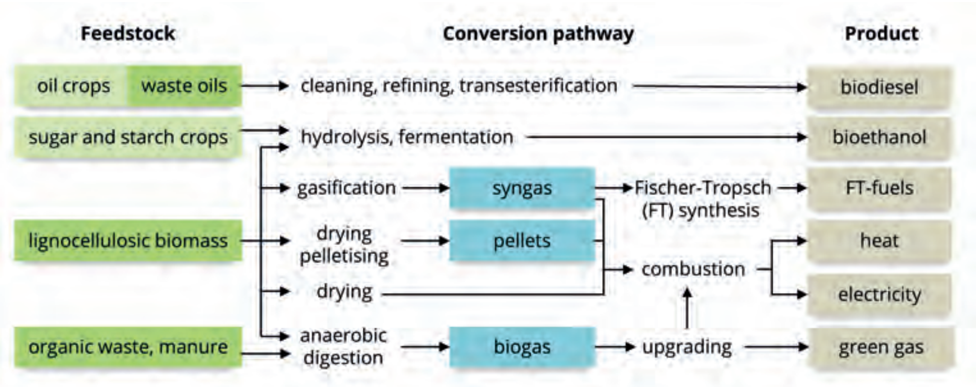

Figure 1.2 | Several main bioenergy conversion pathways. First and second-generation feedstocks are indicated in light and dark green, respectively. Based on: Twidell & Weir (2004), Hoefnagels et al. (2010), Turkenburg et al. (2012) and Creutzig et al. (2015).

There are, however, also large concerns on the sustainability of bioenergy. The cultivation of food crops and lignocellulosic crops for bioenergy requires fertile land and fresh water. This can lead to competition with food production (Doelman et al., 2018; Hasegawa et al., 2018; Fujimori et al., 2019) or entail the conversion of natural land, resulting in biodiversity loss (Heck et al., 2018; Chaudhary & Brooks, 2018; Núñez-regueiro et al., 2019) and high GHG emissions (Searchinger et al., 2008; Gibbs et al., 2008; Gerssen-Gondelach, 2017). Cultivation of bioenergy crops can in some cases lead to soil improvement and enhanced soil carbon, but also to soil depletion and erosion (Hill, 2009; Dale et al., 2011; Creutzig et al., 2015; Whitaker et al., 2018). Fertilisers are used to partially mitigate these issues and, more generally, to increase yields, but cause eutrophication and the emission of the greenhouse gas N_2O (Stehfest & Bouwman, 2006). Similarly, pesticides increase yields, but affect human health and biodiversity (Damalas & Eleftherohorinos 2011). Environmental impacts are typically lower when using residual biomass ('residues') to produce bioenergy, as no additional land, water or chemicals are required for their production (Smith et al. 2014; Creutzig et al. 2015), though residue removal may still negatively impact soil quality and biodiversity (e.g., Bouget et al., 2012; Liska et al., 2014; Raffa et al., 2015).

1.2 Bioenergy & Climate Change

In terms of climate change, the uptake of atmospheric CO_2 in growing biomass means that bioenergy can have relatively low *net* greenhouse gas (GHG) emissions. Bioenergy can thus contribute to climate change mitigation when replacing fossil fuels, which only emit GHGs (Creutzig et al., 2015). However, beside the uptake of CO_2 and emissions from combustion, there are more flows of GHGs to consider for bioenergy (Figure 1.3a). The land conversion towards bioenergy crop plantations can lead to land-use change (LUC) emissions in the form of CO_2, methane (CH_4) and nitrous oxide (N_2O) from burning or rotting of original vegetation or from soil carbon losses (e.g., Kim & Kirschbaum, 2015). This can also encompass the lost capacity of natural vegetation to sequester CO_2, so-called 'foregone sequestration'. Furthermore, LUC can occur indirectly (iLUC) when bioenergy crop production on cultivated land leads to bringing pristine land under cultivation elsewhere (Searchinger et al., 2008; Gerssen-Gondelach et al., 2017; Daioglou et al., 2020). Because of these i(LUC) emissions the GHG intensity of bioenergy is strongly dependent on cultivation location (e.g., Elshout et al., 2015; Daioglou et al., 2017; Harper et al., 2018). Cultivation on locations with previously large natural carbon stocks can lead to a supply of energy that is more GHG-intensive than coal and strongly *contributes* to climate change (Searchinger et al., 2008; Fargione et al., 2008; Gibbs et al., 2008). Further GHG emissions occur during the cultivation of biomass, e.g. from fertiliser use (N_2O) and the operation of machinery, as well as during transport and processing of biomass along the bioenergy supply chain (Figure 1.3a). These 'supply chain' GHG emissions again depend on location, but also on the type of biomass feedstock used, and its conversion pathway to a final energy carrier (e.g., Hoefnagels et al., 2010; Elshout et al., 2015; Hanssen et al., 2020; Figure 1.2). Lastly, stochastic events like wildfires, pests and windfall may lead to additional GHG emissions and loss of cultivated biomass (e.g., Dalin et al., 2011; Buchholz et al., 2015).

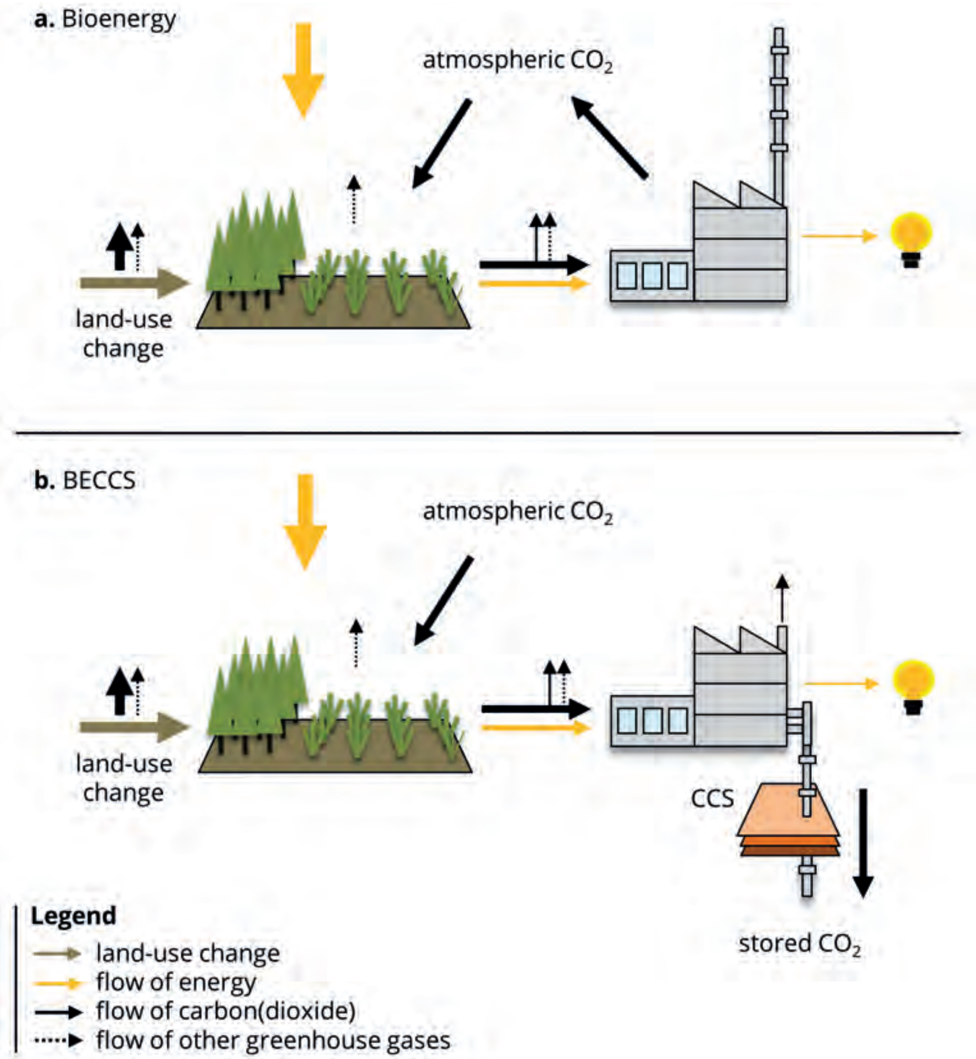

Figure 1.3 | Greenhouse gas and energy flows in bioenergy systems. a. In a conventional bioenergy system, plant biomass captures solar energy and atmospheric CO_2 during growth. At a processing facility (e.g., a power plant) energy is generated from biomass, and carbon dioxide is emitted back to the atmosphere. Further GHG emissions occur during land-use change and along the supply chain of biomass. **b.** In a bioenergy with carbon capture and storage (BECCS) system, the same GHG flows take place, except that CO_2 emissions at the processing facility are largely captured and geologically stored. If this carbon sequestration outweighs total GHG emissions, net negative emissions are achieved.

When bioenergy is combined with carbon capture and storage (abbreviated as BECCS) the same flows of GHGs occur, except that CO_2 emissions at the processing facility (e.g., a power plant or biorefinery) are largely captured and geologically stored, rather than being released back into the atmosphere (Figure 1.3b). If this sequestration of CO_2 outweighs total GHG emissions of the bioenergy system, net *negative* GHG emissions are achieved. While net negative emissions can be achieved, it is important to consider that not all CO_2 can be captured and that capturing and storing CO_2 requires energy, which reduces the biomass to final energy carrier conversion efficiency (Al-Qayim et al., 2015; Schakel et al., 2014).

Beside the size of these GHG flows, their timing also matters. A substantial part of the GHG emissions of a bioenergy system occur as an initial pulse, for instance most of the (i)LUC emissions. Remaining GHG flows, such as supply chain emissions and the uptake and release of biogenic CO_2 in the cultivated biomass, are more spread out and occur over the lifetime of the bioenergy system. Time is therefore a key element in the evaluation of bioenergy, and this is reflected in the two main *types* of metrics that are commonly used to assess the climate change impacts or mitigation potential of a bioenergy system:

- *GHG emission factors* (e.g., kg CO_2-eq./MJ) are determined by summing the GHG flows of a system over a fixed 'evaluation' period of time, and dividing resulting net emissions by the total amount of energy generated over this same evaluation period (e.g., Creutzig et al., 2015; Daioglou et al., 2017).

- *GHG payback times* on the other hand explicitly focus on the temporal dimension and also compare bioenergy to an often fossil, benchmark technology. GHG payback times (in years) express how long it takes before an incurred "debt" of initial GHG emissions (e.g. from LUC) is "paid back" by the GHG emission reductions of replacing fossil fuels with bioenergy (Gibbs et al., 2008, Elshout et al., 2015).

Multiple varieties exist of both metrics, and it is partially subjective which exact GHG flows are accounted for and how (e.g., Lamers & Junginger 2013; Creutizg et al., 2015; Buchholz et al., 2015). In this thesis, both GHG emission factors and GHG payback times are used. They describe the GHG impacts of bioenergy, but exclude the effects of other climate forcers, such as changes in albedo (the reflectivity of the Earth's surface). These effects are usually limited for bioenergy systems, but can be significant in boreal areas, where plantation forests are for instance less reflective than snow covered cropland (Smith et al., 2016).

1.3 Problem setting

From the previous sections it becomes clear that bioenergy is a broad term for a diverse range of energy systems. The wide variety of biomass sources and conversion pathways, the resulting differences in GHG balances, and the array of methodological decisions required to integrate these balances into policy-relevant metrics, make the climate impact of bioenergy a complex topic with a large field of research. This thesis focuses on two lines of research in particular. The first line of research considers alternative uses of biomass when determining the climate impact of bioenergy from existing biomass flows. It concerns biomass from residues and existing bioenergy plantations, in the short to medium term (0-30 years) and therefore often with a local or regional focus. The second line of research assesses the potential global contribution that bioenergy could have towards mitigating climate change in the 21st century, and has a longer-term perspective (30-80 years).

The climate impact of bioenergy in view of alternative biomass uses.
The climate change impact or benefits of bioenergy are typically determined for a bioenergy system in isolation (Creutzig et al., 2015), or as compared to a benchmark of not producing biomass and using land in a different way (e.g., Mitchell et al., 2012). However, this potentially overlooks two additional important aspects of bioenergy systems. First, there are usually multiple alternative uses or fates of biomass (Figure 1.4). These alternative uses could have different climate benefits or impacts and is therefore useful to compare them to the bioenergy option. In situations where it can be assumed that biomass would already be produced, bioenergy can in fact *only* be assessed in view of alternative fates or uses of biomass, as not producing the biomass is not an option. Assuming biomass would be produced anyway is bold, but is often closely approached for residual biomass from for instance agriculture and forestry. Biomass from existing plantations can in the short term also be viewed in this way, especially

if its traditional markets are in decline, while biomass supply continues through system inertia.

A second aspect to consider is whether the production of bioenergy would replace a so-called *counterfactual*, for instance whether it would avoid the use of fossil fuels. Counterfactuals are implicitly or explicitly always used in determining climate payback times of bioenergy, as they usually express the period of time until a bioenergy system has net climate benefits *compared to* a fossil fuel-based system (Gibbs et al., 2008; Lamers & Junginger, 2013; Elshout et al., 2015). The use of counterfactuals also has similarities to consequential life-cycle assessment (LCA). In consequential LCA, the environmental impacts of a product are determined over its life cycle, while including the environmental *consequences* of this product's substitution of any other products or processes in the conventional economy (Ekvall, 2019). Because counterfactuals make the consequences of bioenergy production explicit, they can make for a more comprehensive assessment, and their use is increasingly encouraged (Millward-Hopkins & Purnell, 2019). Assessing the climate impact of bioenergy in light of alternative biomass uses, may thus benefit from considering counterfactuals for each alternative use of biomass (Figure 1.4). Examples of this approach are, however, still sparse. Gerssen-Gondelach et al. (2014) found that various bioenergy options often have lower GHG emissions than biomaterial options, when including what these products replace. More recently, Thonemann & Pizzol (2019) applied a similar methodology to evaluate various products produced from CO_2 and also found that environmental impacts strongly depend on their counterfactuals. For bioenergy, especially from existing biomass flows, this approach requires additional understanding based on case studies as well as methodology formalisation.

Bioenergy's potential global contribution to climate change mitigation.

Bioenergy, and in particular bioenergy with carbon capture and storage (BECCS), forms a key component of many global energy supply scenarios and climate change mitigation pathways that limit global warming to 1.5 or 2 °C over the 21st century (e.g., Chum et al., 2011; Bruckner et al., 2014; Rose et al., 2014; van Vuuren et al., 2016; Rogelj et al., 2018; Figure 1.5). These scenarios and pathways are typically created using so-called integrated assessment models (IAMs). These IAMs model the economy, biosphere and atmosphere in conjunction and allow exploring what the global energy system and economy would have to look like to achieve climate targets. The main reasons why bioenergy plays such a large role in these IAM-generated scenarios and mitigation pathways are: i) the possible (temporary) limitations in the supply of other renewables, ii) the wide range of bioenergy applications, iii) the potentially relatively low costs of bioenergy, and importantly iv) the negative CO_2 emissions that can be achieved with BECCS.

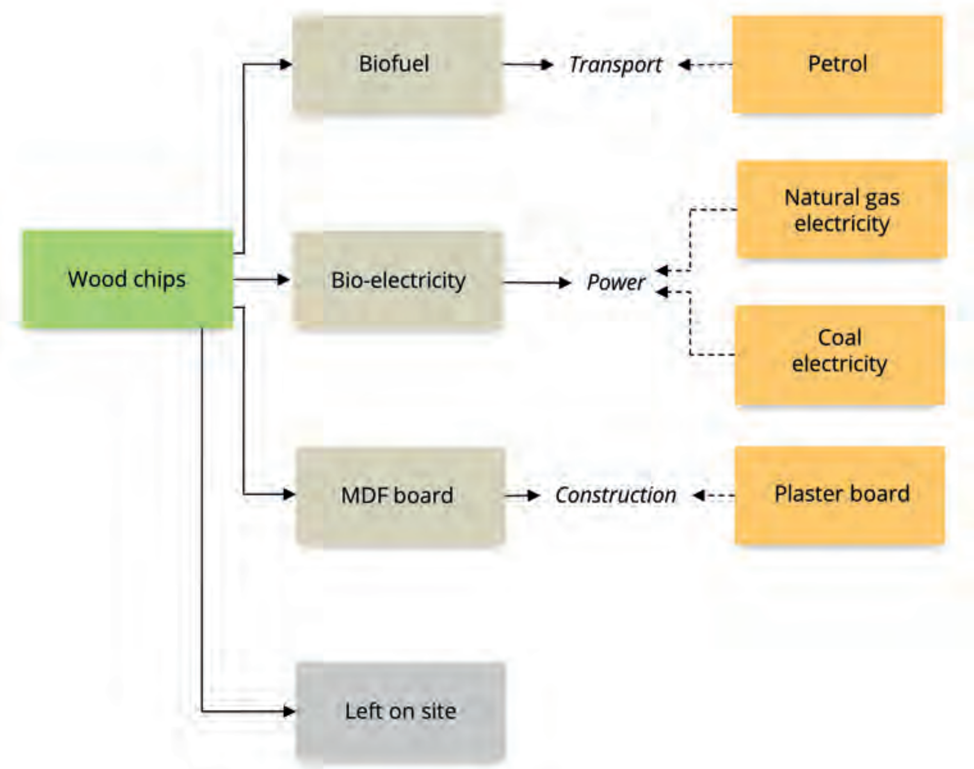

Figure 1.4 | Hypothetical example of a biomass feedstock, its alternative uses and their counterfactuals. The alternative uses of a biomass feedstock would have different functions in the economy and potentially replace (combinations of) different *counterfactual* products. Some biomass fates, such as leaving biomass on site, may not have a direct economic counterfactual, though biomass could still have a function here, for instance enhancing soil quality and biodiversity.

However, while featuring prominently in model scenarios, the exact contribution that bioenergy could have towards meeting climate change mitigation targets depends on many factors that are subject of ongoing research. One key factor is the future availability of biomass for energy (Dornburg et al., 2010; Beringer et al., 2011; Daioglou et al., 2015a,b; Creutzig et al., 2015). The availability of bioenergy from purpose-grown crops depends on crop yields (Kato et al., 2015; Elshout et al., 2015; Daioglou et al., 2017) and on the amount of land that can be allocated to growing these crops, alongside competing land uses like agriculture and natural areas. Residues from agriculture and forestry do not require additional land, but their availability for energy is still quite uncertain, as it depends on future population size and diet, associated levels of agriculture and forestry, and competing uses of biomass (Daioglou et al., 2015a,b; Van Zanten et al., 2018). The amount of residues available for bioenergy therefore varies widely across IAMs and requires additional research.

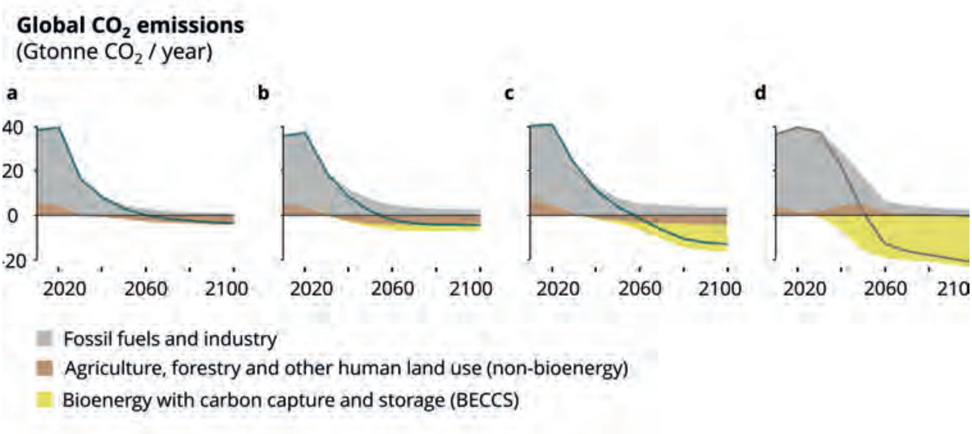

Figure 1.5 | The contribution of bioenergy with carbon capture and storage (BECCS) to global climate change mitigation in four illustrative mitigation pathways that limit global warming to 1.5°C. Annual global CO_2 emissions are shown per overarching sector and the green line indicates their combined effect. Emissions are presented for the following pathways: **a**. a low energy demand pathway (LED), **b**. a sustainability pathway (SSP1), **c**. a middle-of-the-road pathway (SSP 2), and **d**. a fossil-fuelled development pathway (SSP5). Negative CO_2 emissions from BECCS have a prominent role in the latter three pathways. This figure was adapted from the IPCC special report on 1.5 °C (IPCC, 2018); note that it does not include non-CO_2 greenhouse gases.

A second factor in the climate change mitigation potential of bioenergy is the effectiveness of bioenergy in reducing GHG emissions. As explained in section 1.2, this effectiveness depends on overall GHG balance of the bioenergy itself, including (i)LUC emissions, life-cycle supply chain emissions, and CO_2 sequestration. The GHG emission reduction potential of bioenergy also depends on the counterfactual energy replaced, though in long term counterfactuals may become less relevant when fossil fuels are phased out (e.g., in the second half of the century). When bioenergy is specifically combined with CCS to achieve negative emissions, it is crucially important to know what *net* negative emissions could be achieved. Previous work has shown that BECCS can result in both negative and positive GHG emissions, depending on LUC emissions and the efficiency of the bioenergy supply chain (Fajardy & Mac Dowell, 2017; Harper et al., 2018). However, spatially-explicit, full life-cycle GHG emissions for BECCS have not been determined before, and are essential to evaluate the potential contribution of BECCS in reaching mitigation targets.

Third, many concerns have been raised on the environmental effects of large-scale crop-based bioenergy and BECCS deployment, due their intensive land, water and nutrient use (Kemper et al., 2015; Bonsch et al., 2016; Fajardy & Mac Dowell, 2017; Heck et al., 2018; Stoy et al., 2018; Kato & Yamagata, 2014). This may limit the desirable amount of climate change mitigation via bioenergy and BECCS. The large land requirements for crop-based bioenergy and BECCS may in particular form a large threat to biodiversity (Heck et al., 2018; Núñez-regueiro et al., 2019). However, the exact trade-off between BECCS-based negative emissions and global species extinctions due to the required LUC is still largely unknown. Furthermore, it is not clear yet how extinctions via LUC for BECCS compare to the potential positive effects that climate change mitigation by BECCS might have on biodiversity.

1.4 Aim and outline

The main aim of this thesis is to assess the potential and trade-offs of using second-generation bioenergy for climate change mitigation. I specifically looked at two research questions:

1. *What are the climate impacts or benefits of current regional bioenergy production in view of other uses of biomass?*

2. *What can bioenergy globally contribute to climate change mitigation over the 21st century, considering biomass supply, negative emission potential and biodiversity trade-offs?*

These questions and their lines of research differ in scope with regards to: spatial and temporal scale, biomass supply, consideration of alternative biomass uses, counterfactuals, carbon capture and storage, and policy relevance, as summarised in Table 1.1.

To answer the first research question:

- *Chapter 2* looks at GHG payback (parity) times of wood pellet-based electricity from low-value biomass from US pine plantations versus counterfactual fossil fuel-based electricity, while accounting for the alternative uses of this biomass feedstock and their counterfactuals.

- *Chapter 3* considers a second regional case study and compares the climate impacts and benefits of using residual biomass from Dutch river floodplains for various bioenergy and biomaterial options, accounting for their counterfactuals.

- *Chapter 4* builds upon chapters 2 and 3 and proposes a general methodology to assess the environmental benefits of utilising residual (biomass) flows, by including their alternative uses and counterfactuals.

To answer the second research question:

- *Chapter 5* looks at the global amount of agricultural and forestry residues that can be supplied for bioenergy over the 21st century based on eight integrated assessments models. These findings are compared to bottom-up estimates of residue availability in literature.

- *Chapter 6* focuses on the global amount of negative emissions that can biophysically be achieved using crop-based bioenergy with carbon capture and storage (BECCS). This assessment is performed spatially-explicitly for multiple crops and types of bioenergy, over different evaluation periods. Outcomes are compared with negative emission requirements in climate change mitigation pathways.

- *Chapter 7* looks in detail at the trade-off between BECCS-based negative emissions and biodiversity conservation. It includes the negative effect on biodiversity of land conversion for bioenergy crops, as well as the potential positive effect of partly mitigating climate change with BECCS.

Table 1.1 | Scope of the two lines of research in this thesis.

	Climate impacts and benefits of bioenergy in view of alternative biomass uses *Chapters 2-4*	**Potential global contribution of bioenergy to climate change mitigation** *Chapters 5-7*
Spatial scale	Regional	Global
Temporal scale	Short to medium term (0-30 years)	Longer term (30-80 years)
Biomass supply	Largely fixed due to inertia: current residues, existing plantations	Changeable: future residue availability, new plantations
Alternative uses of biomass	Comparison of various material and energy uses of biomass	Not explicitly included, other uses prioritised over energy
Counterfactuals to bioenergy/biomaterials	Integrally included in the assessment of bioenergy	Not included, or only used as benchmark (chapter 6)
Carbon capture & storage (CCS)	Not included	Focus on bioenergy with CCS (BECCS) in chapter 6 & 7
Policy relevance	Current regional decisions on bioenergy	Long-term global climate change mitigation strategy

2

Wood pellets, what else? Greenhouse gas parity times of European electricity from wood pellets produced in the south-eastern United States using different softwood feedstocks

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Anna Duden
Martin Junginger
Virginia Dale
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ABSTRACT

Several EU countries import wood pellets from the south-eastern United States. The imported wood pellets are (co-)fired in power plants with the aim of reducing overall greenhouse gas (GHG) emissions from electricity and meeting EU renewable energy targets. To assess whether GHG emissions are reduced and on what timescale, we construct the GHG balance of wood-pellet electricity. This GHG balance consists of supply chain and combustion GHG emissions, carbon sequestration during biomass growth, and avoided GHG emissions through replacing fossil electricity. We investigate wood pellets from four softwood feedstock types: small roundwood, commercial thinnings, harvest residues, and mill residues. Per feedstock, the GHG balance of wood-pellet electricity is compared against those of alternative scenarios. Alternative scenarios are combinations of alternative *fates* of the feedstock material, such as in-forest decomposition, or the production of paper or wood panels like oriented strand board (OSB). Alternative scenario composition depends on feedstock type and local demand for this feedstock. Results indicate that the GHG balance of wood-pellet electricity equals that of alternative scenarios within 0-21 years (the GHG parity time), after which wood-pellet electricity has sustained climate benefits. Parity times increase by a maximum of 12 years when varying key variables (emissions associated with paper and panels, soil carbon increase via feedstock decomposition, wood-pellet electricity supply chain emissions) within maximum plausible ranges. Using commercial thinnings, harvest residues or mill residues as feedstock leads to the shortest GHG parity times (0-6 years) and fastest GHG benefits from wood-pellet electricity. We find shorter GHG parity times than previous studies, for we use a novel approach that differentiates feedstocks and considers alternative scenarios based on (combinations of) alternative *feedstock fates*, rather than on alternative land-uses. This novel approach is relevant for bioenergy derived from low-value feedstocks.

2.1 Introduction

The EU aims to increase the share of renewable energy in its gross final energy consumption to 20% by the year 2020 to mitigate climate change and improve energy security of supply (EU directive 2009/28/EC). Wood pellets, a type of solid biofuel, form one such renewable and accounted for 0.47% of EU gross inland energy consumption in 2014 (Aebiom, 2015; Eurostat, 2016). The EU is the largest global producer, consumer and importer of wood pellets used for both electricity production and for residential and district heating (Sikkema et al., 2011; Lamers et al., 2012; Goh et al., 2013; Eurostat, 2015). The United Kingdom, the Netherlands, Belgium and Denmark have been the main importers of wood pellets from outside the EU that are used for (co-)firing in power plants to (partly) replace fossil fuels (Sikkema et al., 2011; Lamers et al., 2012, 2015; Goh et al., 2013; Goetzl, 2015). These wood pellet imports increased more than fourfold between 2009 and 2014 (Eurostat, 2015).

The United States is the largest global exporter of wood pellets (EIA, 2014). Production is largest in the US Southeast (US SE; as defined by Wear & Greis, 2012, see Figure S1) and nearly all wood pellet exports from the US SE go to the EU (Pinchot institute, 2013; Abt et al., 2014; EIA, 2014). Primarily driven by EU demand (Abt et al., 2014), US SE wood pellet production and export have *doubled* since 2011 (Prestemon et al., 2015; Eurostat, 2015), making the region one of the largest global wood pellet suppliers to the EU (Hoefnagels et al., 2014). Even so, the wood pellet market is small relative to that other forest products (e.g. saw timber or paper), with wood pellets having comprised <1% of total US forest products by weight and about 1% of total US forest products exports by value in 2014 (FAO, 2016).

There are several concerns regarding the sustainability of electricity production from wood pellets, including biodiversity loss, soil degradation and climate change (e.g. Pinchot Institute, 2013; Lamers et al., 2013; Thiffault et al., 2015, Olesen et al., 2016). This study considers the climate change impact of wood-pellet electricity. This impact is usually assessed by constructing the greenhouse gas (GHG) balance of wood-pellet electricity, and then determining the (time to) GHG emission savings (i.e. GHG benefits) compared to a reference system or scenario. Standardised guidelines for GHG-accounting currently do not exist (Buchholz et al., 2015; Galik & Abt, 2015). However, there is wide agreement that regardless of the GHG-accounting method applied, larger and faster GHG emission savings (GHG benefits) from wood-pellet electricity are achieved:

- when replacing higher GHG-intensity fossil fuels (e.g. Cherubini et al., 2009; Walker et al., 2010; Colnes et al., 2012; Zanchi et al., 2012; Jonker et al., 2013; Lamers & Junginger, 2013),

- when forest productivity is high; production depends on climate, soil and other biophysical site characteristics, as well as tree species and forest management (e.g. Marland & Schlamadinger, 1997; Cherubini et al., 2011a; Zanchi et al., 2012; Jonker et al., 2013; Lamers & Junginger, 2013),

- when GHG emissions along the wood pellet supply chain are low; emissions depend on forest management, transport and processing (e.g. Schlamadinger & Marland, 1996a; Magelli et al., 2009; Sikkema et al. 2010; Mitchell et al., 2012; Jonker et al., 2013).

GHG footprinting studies that use a life-cycle assessment approach (in which forest carbon sequestration is accounted for by considering biogenic emissions carbon neutral) show that European electricity generated using softwood wood pellets from the US SE causes 50-75% less GHG emissions than fossil fuel-derived grid electricity (Dwivedi et al., 2011, 2014a; personal communication G.-J. Jonker, October 23, 2015). Other studies point out that GHG benefits of wood-pellet electricity are often not immediate but depend on: i) the time lag between GHG emission (harvesting of forest biomass and burning of resulting biofuels) and GHG sequestration during forest regrowth (Zanchi et al., 2010, McKechnie et al., 2011; Lamers & Junginger, 2013); ii) the potential (in)direct GHG emissions or sequestration from the conversion of a previous land- or forest use to production forests (Fargione et al., 2008; Searchinger et al., 2008, 2009; Berndes et al., 2013; Lamers & Junginger, 2013; Wang et al., 2015); and iii) whether or not the GHG balance of wood-pellet electricity is compared against a (dynamic) *counterfactual scenario*, in which forestland or pellet feedstock is used differently and electricity is produced from fossil sources (Schlamadinger & Marland, 1996b; Mitchell et al., 2012; Lamers & Junginger, 2013, Lamers et al., 2014; Stephenson & Mackay, 2014; Buchholz et al., 2014).

The first two effects can lead to initial GHG emissions and an initial dip in the GHG balance of forest bioenergy that is ambiguously refered to (Matthews et al., 2014) as the *carbon debt* (Zanchi et al., 2010; reviewed by Lamers & Junginger, 2013). Carbon debt payback times (i.e., the time until forest regrowth and avoided fossil GHG emissions compensate the carbon debt) have been estimated to be 1-27 years for Dutch electricity from US SE wood pellets (Jonker et al., 2013). The third consideration has led to the calculation of so-called *GHG (or carbon-) parity times:* the time to the point at which wood-pellet electricity (usually with higher initial emissions) and the counterfactual have the same cumulative net GHG emissions (explained in detail by Mitchell et al., 2012; Lamers & Junginger, 2013). Beyond the GHG parity time wood-pellet electricity leads to GHG emissions savings compared to the counterfactual. For European electricity from US SE wood pellets, estimated GHG parity times range 2-80 years (in most cases: 20-50), when compared to commonly used counterfactuals of continued forest growth or natural regrowth after one harvest (Jonker et al., 2013; Colnes et al., 2012).

The choice of counterfactual[1] greatly influences the GHG benefits of wood-pellet electricity (Jonker et al., 2013; Lamers & Junginger, 2013, Lamers et al., 2014; Stephenson & Mackay, 2014). So far however, the counterfactuals for wood-pellet electricity considered in previous studies have been limited- and can be improved in three ways. First, earlier work on counterfactuals has focused on alternative land or forest uses (Mitchell et al., 2012; Colnes et al., 2012; Jonker et al., 2013; Lamers & Junginger, 2013, Lamers et al., 2014). However, decisions on land- or forest use are more often driven by saw timber and paper markets (Forest2market, 2016; Wear & Greis, 2013) or external pressures like urbanisation (Wear & Greis, 2013) than by wood pellet markets. Instead of alternative land-use, counterfactuals for wood-pellet electricity should therefore focus on alternative fates of wood-pellet feedstock material, i.e. what would have happened to the feedstock material had it not been used to produce wood pellets. Examples include the production of other products or in-forest decomposition of feedstock material (similar to counterfactuals by Stephenson & Mackay, 2014).

[1] Note that the term 'counterfactual' is used differently in this paper than in the rest of this thesis. Throughout this thesis, the term counterfactual refers to what would be replaced by a product or service. For example the counterfactual of wood-pellet electricity can be fossil fuel-based electricity. In this paper, however, the term counterfactual is used for alternative fates of biomass such as material production or decomposition in the field, and multiple of these alternative fates are together referred to as 'alternative scenarios'. This difference in terminology is caused by new insights after the publication of this chapter, it does not influence any of the findings. When discussing the findings of this paper in the synthesis and summary, the term counterfactual is used as in the rest of this thesis, that is "what would be replaced by a product or service".

Second, the exact type of wood-pellet *feedstock* likely affects its alternative-fate counterfactual. Counterfactuals should therefore be determined per feedstock type. This approach enables more accurate parity time calculation as well as intercomparison of GHG benefits of wood-pellet electricity from different feedstocks. Wood-pellet feedstock types derived from softwood plantations in the US SE generally include (saw) mill residues, small roundwood (including pulpwood traditionally used for the pulp and paper industry), forest thinnings, and harvest residues (Lamers & Junginger, 2013; Dwivedi et al., 2014a, Dwivedi & Khanna, 2014c, 2015; Stephenson & MacKay, 2014; Buchholz & Gunn, 2015). Previous work on the effect of feedstock type on GHG emissions has been limited to showing that using harvest residues as wood-pellet feedstock leads to largest GHG benefits, while assuming that residues would otherwise decompose or be burnt (e.g. McKechnie et al., 2011; Zanchi et al., 2012; Bernier & Paré, 2013; Lamers & Junginger, 2013, Lamers et al., 2014; NRDC, 2015).

Third, multiple counterfactuals, i.e. multiple alternative fates, for each wood-pellet feedstock type are feasible (e.g. some material is used in the paper industry, while the remainder decays on site). To our knowledge there has been no attempt to create a mix of counterfactuals and analyse its effect on GHG benefits of wood-pellet electricity or forest bioenergy in general.

In this study, we calculate GHG parity times of wood-pellet electricity from different feedstocks originating from existing US SE softwood plantations to determine if and when GHG benefits of wood-pellet electricity occur. We use a new approach that *for each feedstock* compares the GHG balance of wood-pellet electricity against alternative scenarios that are a *combination* of individual *feedstock fate-based counterfactuals*. Alternative scenario composition is also made to depend on demand for the different feedstock materials for production of alternative products. Our research focuses on pellets from softwood plantations, as more than 60% of US SE wood pellets are produced from softwood material (Forest2Market, 2016), and softwood plantations form 19% of US SE total forest cover (Wear & Greis, 2013).

2.2 Methods

The GHG balance of forest bioenergy is often assessed using a forest carbon accounting model like GORCAM (Schlamadinger & Marland, 1996b; Jonker et al., 2013), LANDCARB (Harmon, 2012; Mitchell et al., 2012) or FORCARB2 (Heath et al., 2010). However, the exact calculations behind these models are often not transparent and/or US SE-specific parameterisations are not available. We therefore calculated GHG parity times with a set of equations tailored for comparing wood-pellet electicity to alternative scenarios, and parametrised for the US SE (Table S1). Before discussing these calculations in detail, we first define different wood-pellet feedstock types. We then describe feedstock production, wood-pellet electricity and individual counterfactuals and their combination into alternative scenarios (Figure 2.1). Then lastly, we set out our GHG accounting assumptions and explain our calculations and sensitivity analyses.

Wood-pellet feedstock definition

A softwood plantation yields several products; most valuable are saw logs (diameter at breast height [DBH] of >35 cm) and chip 'n saw wood (DBH of 25-35 cm; SC forestry commission, 2015). These two categories were lumped here as saw wood, which is sawn into lumber at a (chip-n-) sawmill and is too expensive for use as wood-pellet feedstock (Dwivedi et al., 2014b), except for the (chip-n-) saw mill's residues. We defined the following wood-pellet feedstocks derived from softwood plantations, in consultation with various local scientists and wood pellet stakeholders (personal communication, K. Kline, A. Taylor, B. Abt, D. Hazel, K. Abt [US SE based forest/bioenergy scientists], B. Wigley, R. Miner [US National Council for Air and Stream Improvement], M. Jostrom, B. Emory [repr. largest US SE corporate foresters] and Parrish, B. [repr. a pellet mill], April 7 - December 15, 2015; Enviva, 2015), and in line with previous studies (Dwivedi et al., 2014a, Dwivedi & Khanna, 2015; Stephenson & MacKay, 2014; Buchholz & Gunn, 2015; NRDC, 2015):

- *Small roundwood*: wood harvested at final cut, including: stemwood (10-25 cm DBH), larger tops and limbs (10-25 cm diameter), and stemwood >25cm DBH that is damaged or otherwise unsuitable for saw wood. The category includes pulpwood (i.e., wood that is traditionally used in the pulp and paper industry).

- *Commercial thinnings*: wood that is harvested during mid-rotation plantation thinning and is merchantable (usually as pulpwood). Pre-commercial thinning (at an earlier stage of rotation) is not commonly practiced in the US SE (B. Parrish, personal communication, June 17, 2015) and was excluded.

- *Collectible harvest residues*: woody material left behind after the final cut that is still economically collectible (typically 70% of total harvest residues, Dwivedi et al. 2014a; leaving 30% required for ecological services, Daioglou et al., 2015a). The category includes wood of <10 cm diameter, coarse woody debris, and in-wood chips (i.e., chipped harvest residues).

- *Saw mill residues*: woody material that is a co-product of sawing saw logs and chip ’n saw wood into lumber, i.e. clean wood chips including chips from chip-n-saw wood (67%), wood shavings (15%) and sawdust (18%; Aebiom, 2013).

Wood-pellet feedstock production

We estimated biomass growth (Figure S2) and associated carbon sequestration (Table S1) of medium- to highly intensively managed softwood plantations using the Carbon OnLine Estimator (COLE; NCASI, 2016). COLE uses empirical data from the US Forest Service’s Forest Analysis & Inventory data base (FIA, 2016) and estimates stored carbon in live tree biomass, among other ecosystem carbon pools (which are fairly constant on both landscape and plot level; Smith et al. 2006, NCASI, 2016; for details see Table S1). Based on COLE and a plantation rotation period of 25 years (Markewitz, 2006; Colnes et al., 2012; Jonker et al., 2013; Dwivedi et al., 2014b, Dwivedi & Khanna, 2014c, 2015; Table S1), plantation yield was estimated at 197 dry tonne tree biomass per hectare after 25 years, including thinnings (in line with Dwivedi et al., 2011, 2014a; Dwivedi & Khanna, 2015; Jonker et al., 2013 [140-232.5 dtonne · ha^{-1} · 25 yr^{-1} at medium-high intensity management]). Plantation thinning was included by harvesting 1/3 of live tree biomass 15 years after planting (based on Markewitz, 2006; Jonker et al., 2013). We estimated that the *enhanced* growth of the remaining trees after thinning compensates for 50% of the biomass taken out during thinning (for details see Table S1; Figure S2). This estimate is conservative, as some studies indicate near 100% compensation (e.g. Gonzalez-Benecke et al., 2010, 2011; Jonker et al., 2013).

Mass fractions of the different products orginating from softwood plantations (saw wood, small roundwood, etc. – at medium to high intensity forest management) and saw mills (lumber, residues, bark) were estimated from literature (Table S2).

GHG emissions of medium- to high intensity forest management (including: site preparation, planting, fertiliser and herbicide use, and thinning) and harvesting were obtained from literature (Table S1). Emissions were allocated to the different forest products, according to their mass (or equivalently: embodied carbon – as all feedstocks were assumed to have the same moisture- and carbon contents [0.5 and 0.25 respectively], Table S1). Similarly, sawmill GHG emissions were mass-allocated over different sawmill products, including mill residues. No forest management GHG emissions were allocated to non-collectible harvest residues (twigs, needles; 3.7% of total live tree biomass produced).

Wood-pellet electricity

To get from wood-pellet feedstock to electricity requires transport of feedstock and pellets (truck, train, transatlantic shipping), pelletising, handling, and combustion, which lead to GHG emissions in the form of biogenic CO_2, fossil CO_2, CH_4 and N_2O emissions. These supply chain emissions were assumed to be equal for all feedstocks and were based on literature (Table S1). It was assumed that wood-pellet feedstock material is dried at the pellet mill using heat from burning biomass (Magelli et al., 2009; Sikkema et al., 2010; McKechnie et al., 2011; Dwivedi et al., 2011, 2014a; Jonker et al., 2013); in this study: bark (in case of commercial thinnings and small roundwood, which are debarked at the pellet mill) and/or part of the feedstock material itself (Table S1). Feedstock and pellet material that is lost along the supply chain is assumed to decompose quickly (Table S1). Based on this set of assumptions and parameterisation, overall supply chain efficiency (including losses) was 2.56 tonne of wet feedstock per tonne pellets combusted (Table S1), in line with Dwivedi et al. (2011) [2.32] and Jonker et al. (2013) [2.65].

Wood-pellet electricity was assumed to replace EU fossil grid electricity (JRC, 2014), thereby avoiding emissions from fossil grid electricity. Since wood pellets from all feedstocks are dried to the same moisture level, they have the same energy density and lead to the same (gross) avoided emissions per tonne of pellets combusted.

Counterfactuals and alternative scenarios

If wood-pellet feedstock is not used to produce wood pellets, there are three main *counterfactuals* for the US SE *(*personal communication, B. Abt, K. Abt, D. Hazel, M. Jostrom, R. Miner, A. Taylor, B. Wigley, May 28 – December 15, 2015): i) wood-pellet feedstock is used for alternative products, i.e. pulp and paper and panels (including feedstock use for process heat), ii) wood-pellet feedstock remains in the forest and decomposes, and iii) (for the commercial thinnings feedstock category specifically) softwood plantations are *not* thinned in the first place.

In the alternative products counterfactual wood-pellet feedstock material is used to produce the following alternative products (on landscape scale, on average): 80% pulp and paper, 19% oriented strand board (OSB), and 1% other wood panels like medium density fibreboard (MDF) (based on Matthews et al., 2014), including biomass for process heat. The counterfactual includes the GHG emissions of production and disposal of these alternative products, as well as avoided GHG emissions of the alternative products (Table S1). Avoided emissions were based on what the (wood-based) alternative products replace and consist of the GHG emissions associated with the replaced products (recycled paper, blockwork external wall cladding, plasterboard partition wall, see Table S1 note ab; based on Matthews et al., 2015). The alternative products' use phase (between production and disposal) does not lead to significant GHG emissions and was excluded (in line with Matthews et al., 2015). Disposal is assumed to occur via incineration (or quick decomposition of uncollected waste), incineration with electricity production, or landfilling (based on Smith et al., 2006). Disposal patterns were based on Smith et al. (2006; see Table S3) and are specific to US SE softwood pulpwood products. As landfilled material decomposes, it releases CO_2 and CH_4. Landfill decomposition was modeled as exponential decay. Part of the produced CH_4 is flared or is used for electricity production (Table S1). Pulp and paper products can also be recycled. Carbon then remains embedded in products for a longer time – effectively delaying final disposal; this was investigated in the sensitivity analysis.

Based on previous studies (Naesset, 1999; Palosuo et al., 2001; Liski et al., 2002; Palviainen et al., 2004; Zanchi et al., 2012; Russell et al., 2014), the in-forest decomposition counterfactual was modeled as exponential decay with the majority of carbon in the feedstock being released as CO_2, part as CH_4 and part of the carbon being stored in the soil (Table S1).

In the third counterfactual, plantations are not thinned, meaning that the commercial thinnings are not produced and any (avoided/reduced) GHG emissions associated with their use no longer exist. Not thinning was therefore considered to cause zero GHG emissions. Not thinning *does* result in lower plantation management GHG emissions (Table S1) and larger landscape wide carbon stocks (Figure S2). However, these effects reduce the GHG emissions of this counterfactual by less than one percent compared to wood-pellet electricity (based on default parameterisation, see Table S1) and were excluded from the analysis.

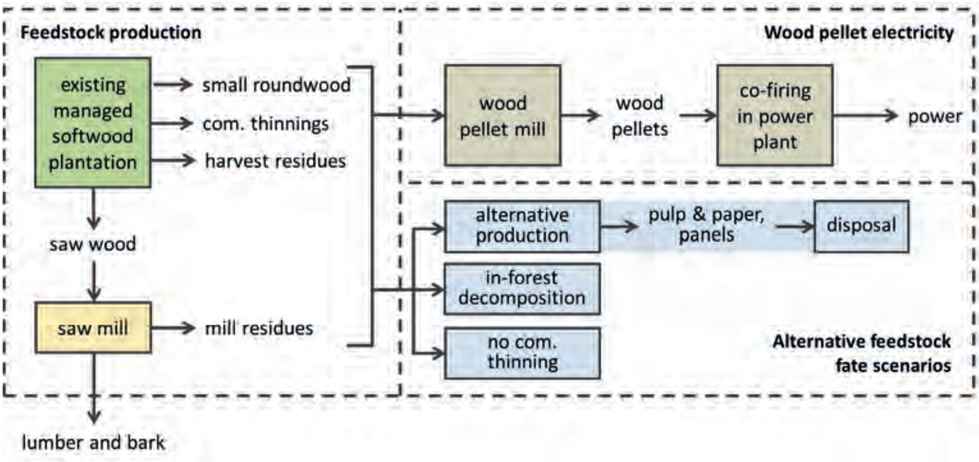

Figure 2.1 | Overview of feedstock production (on the left), wood-pellet electricity production (top right) and alternative scenarios (bottom right). Alternative scenarios consist of multiple individual feedstock-fate based counterfactuals. After feedstock material is produced it either goes to wood-pellet electricity or the alternative scenario. All definitions are explained in detail in the main text. Note that in contrast to life cycle assessment, the compared systems yield different products here: power vs. paper and panels (e.g. OSB). Avoided GHG emissions of these products were included in our analysis.

Which counterfactual is relevant for which feedstock and to what extent is a hypothetical matter that likely varies over time and space and is subject to large uncertainty. Therefore, we investigated a wide range of combinations of these counterfactuals into *alternative scenarios* for each feedstock type (Figure 2.2). Alternative scenario composition was determined in consultation and conversation with local experts (personal communication, B. Abt & D. Hazel, K. Abt, M. Jostrom, A. Taylor, B. Wigley & R. Miner, May 28 – December 15, 2015). Scenario composition was based on feedstock properties, e.g. only a limited share of harvest residues can be used for alternative products, mill residues tend to be fully allocated in the market, and decomposition of commercial thinnings is infrequent, because economic use is what makes them 'commercial'. Alternative scenario composition was also made to be dependent on the demand for alternative products (pulp and paper, panels). Higher demand means that in the absence of wood pellet production, less feedstock is left to decompose and more is used to produce alternative products (K. Abt, personal communication, November, 23, 2015; Stephenson & Mackay, 2014, p. 11). In this study, demand for feedstock material to produce alternative products was considered at three levels: low, US SE average or high. Feedstock properties and levels of demand were translated to fractions that each counterfactual contributes to the alternative scenarios of each feedstock (see Figure 2.2).

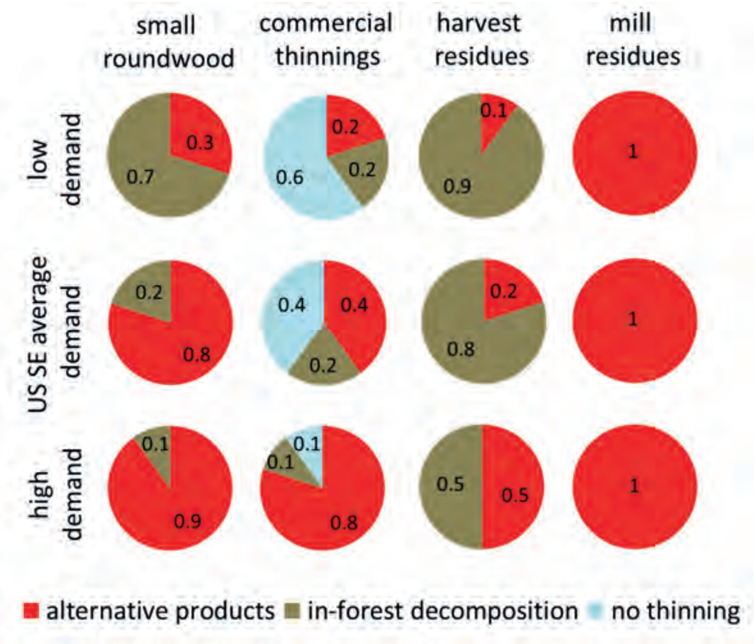

Figure 2.2 | Alternative scenario composition from individual alternative feedstock fates. Each pie diagram represents one alternative scenario. The fractions that alternative products (f_{APi}), in-forest decomposition (f_{DCi}), and no thinning (f_{NTi}) contribute to each alternative scenario are indicated. Scenario composition depends on feedstock type and the demand for feedstock for alternative products.

GHG accounting assumptions

Three main assumptions underlie our approach. First, biogenic CO_2 emissions were considered equal to non-biogenic CO_2 emissions, and carbon sequestration during growth was explicitly modeled. Second, the time lag between GHG emission and sequestration was accounted for by applying a landscape-level approach, in which temporal dynamics of individual forest plots are averaged out geographically across all plots in the landscape (see Jonker et al., 2013; Lamers & Junginger, 2013), resulting in constant annual carbon sequestration and GHG emission associated with (constant) wood-pellet feedstock production. Third, potential GHG emissions caused by the conversion of a previous land- or forest use to a softwood plantation were not included, as only existing plantations were considered (see discussion).

GHG parity time calculations

GHG parity times were determined as the number of years it takes until the initially lower GHG balance of wood-pellet electricity (Equation 2.1) becomes equal to or larger than that of the alternative scenario (Equation 2.3). GHG balances were determined in time steps of one year by calculating *cumulative* GHG emissions and sequestration associated with a *constant* feedstock use of one tonne per year (for either wood-pellet electricity or the alternative scenario). GHG emissions are negative on the GHG balance, while sequestration and avoided emissions are positive. The equations are the same for all feedstocks. Parameter values can be found in Table S1.

Equation 2.1 describes the GHG balance of wood-pellet electricity (B_{WP}). It consists of a constant feedstock production and use (u) over time (t) to produce wood-pellet electricity. Furthermore it consists of GHG sequestration (SQ), various GHG emissions (e) including biogenic CO_2 emissions, and avoided GHG emissions (ae) associated with wood-pellet electricity. GHG sequestration and (avoided) emissions are expressed per tonne pellets and are therefore divided by the feedstock-to-wood-pellet conversion efficiency (H_{WP}).

$$B_{WP}(t) = u \cdot t \cdot \left(\frac{SQ - e_{MH} - e_{TH} - e_{SM} - e_{PM} - e_{PP} - e_{LO} + ae}{H_{WP}}\right) \qquad \textbf{Equation 2.1}$$

Where:

$B_{WP}(t)$	*= cumulative GHG balance of wood-pellet electricity over time (kg CO_2-eq.)*
u	*= constant feedstock use (1 tonne feedstock · year^{-1})*
t	*= time (years)*
SQ	*= carbon sequestration (kg CO_2-eq. · tonne pellets^{-1})*
e	*= GHG emissions (kg CO_2-eq. · tonne pellets^{-1}); Subscripts: MH = plantation management and harvesting, TH = thinning, SM = sawmill, PM = pellet mill (incl. biogenic CO_2 emission from drying), pp = power plant (incl. biogenic CO_2 emissions from combustion), LO = transport losses (incl. biogenic CO_2 emission from lost biomass)*
ae	*= avoided GHG emissions of wood-pellet electricity (kg CO_2-eq. · pellets^{-1})*
H_{WP}	*= overall conversion efficiency (tonne feedstock · tonne pellets^{-1})*

Equation 2.2 describes the avoided GHG emissions of wood-pellet electricity (*ae*) with a pellet-to-electricity conversion efficiency η. Avoided emissions arise through replacing fossil fuel-based electricity and avoiding its emissions (*EF*).

$$ae = \eta \cdot EF$$ **Equation 2.2**

Where:

η	*= wood pellet to electricity conv. efficiency (MWh· tonne pellets^{-1})*
EF	*= GHG emssion factor of EU fossil grid electricity (kg CO_2-eq. · MWh^{-1})*

Equation 2.3 describes the GHG balance (*B*) of alternative scenarios (*i*). In the first term feedstock is produced at a constant rate, in the same way as in Equation 2.1. In the next three terms, feedstock is divided over the three counterfactuals (alternative products, in-forest decomposition, not thinning) according to the alternative scenario-specific fractions of each counterfactual (f_{APi}, f_{DCi}, and f_{NTi}, for the three counterfactuals respectively, see Figure 2.2). The alternative products counterfactual leads to GHG emissions associated with production(ε_{APp}), avoided GHG emissions from replacing other products ($\alpha\varepsilon$), and disposal GHG emissions ($E_{APd}(t)$). Decomposition and not-thinning counterfactuals also lead to GHG emissions ($E_{DC}(t)$ and ε_{NT}, respectively).

$$B_i(t) = u \cdot t \cdot \left(\frac{SQ - e_{MH} - e_{TH} - e_{SM}}{H_{WP}}\right) + f_{APi}\left(u \cdot t \cdot \left(\alpha\varepsilon - \varepsilon_{APp}\right) - E_{APd}(t)\right) - f_{DCi} \cdot E_{DC}(t) - f_{NTi} \cdot u \cdot t \cdot \varepsilon_{NT}$$

Equation 2.3

Where:
$B_i(t)$ = *cumulative GHG balance of alternative scenario i over time (kg CO_2-eq.)*
f = *fraction (dimensionless)*
$\alpha\varepsilon$ = *avoided GHG emissions of alternative products (kg CO_2-eq. · tonne feedstock^{-1})*
ε = *GHG emissions (kg CO_2-eq. · tonne feedstock^{-1}); subscripts: APp = alternative products production; NT= not thinning*
$E(t)$ = *cumulative GHG emissions over time (kg CO_2-eq.); subscripts: APd = alternative products disposal, DC = decomposition*

Equation 2.4 shows the cumulative GHG emissions over time from the in-forest decomposition counterfactual ($E_{DC}(t)$). Annually produced feedstock (u) decomposes via exponential decay (with half-life $t_{1/2DC}$). When also considering that feedstock that is produced earlier has decayed more than recently produced feedstock, the cumulative amount of feedstock that has decomposed at time step t can be represented as shown in the first part of Equation 2.4 (up to *cc*). Carbon in wet feedstock ($cc \cdot (1-mc)$) that has decomposed is emitted as CO_2 ($f_{DC\ CO2}$) or CH_4 ($f_{DC\ CH4}$), or is stored in the soil ($f_{DC\ soil}$), where it is GHG neutral (explaining the *zero*).

$$E_{DC}(t) = u \cdot \sum_{j=1}^{t}\left((t+1-j)\cdot\left(1-\frac{1}{2}^{j/t_{1/2DC}}\right)\right)\cdot cc\cdot(1-mc)\ \cdot 1000\cdot\left(f_{DC\ CO2}\cdot\frac{44.01}{12.01}+f_{DC\ CH4}\cdot\frac{44.01\cdot 34}{12.01}+f_{DC\ soil}\ \cdot 0\right)$$

Equation 2.4

Where:
j = *year of emission since start of decomposition (years)*
$t_{1/2DC}$ = *half-life of exponential decay during in-forest decomposition (years)*
cc = *carbon content dry feedstock (kg C · kg dry feedstock^{-1})*
mc = *moisture content of wet feedstock (kg H_2O · kg wet feedstock^{-1})*

Equation 2.5 shows the cumulative GHG emissions over time from the *disposal* of alternative products ($E_{APd}(t)$). The summations over k (the year of disposal since production) in Equation 2.5 multiplied by the alternative product supply (u/H_{AP}) represent the cumulative amount of disposed alternative product at time t. Part of disposal of alternative products takes place through incineration (with and without energy recapture, f_{IWk} and f_{IEk} respectively), which causes net GHG emissions ($\bar{e}_{IW}$ and $\bar{e}_{IE}$ respectively). Another part of disposed alternative products are landfilled (f_{LFk}). Landfilled products are assumed to decompose via exponential decay according to half-life $t_{1/2LF}$ releasing GHG emissions ($\bar{e}_{LF}$). The cumulative nature of these emissions is expressed as the summation over *l* (the year of emissions, since initial disposal; similar to *j* in Equation 2.4). Note that disposal fractions (f_{IWk}, f_{IEk}, f_{LFk}) are dependent on the year of disposal since the product was formed (*k*), see Table S3.

$$E_{APd}(t) = \frac{u}{H_{AP}} \cdot \left(\sum_{k=1}^{t} \left((t+1-j) \cdot (f_{IWk} \cdot \bar{e}_{IW} + f_{IEk} \cdot \bar{e}_{IE}) \right) + \sum_{k=1}^{t} \sum_{l=1}^{t+1-k} \left((t+2-k-l) \cdot \left(1 - \frac{1}{2}^{l/t_{1/2LF}} \right) \cdot f_{LFk} \cdot \bar{e}_{LF} \right) \right)$$

Equation 2.5

Where:

H_{AP} = *conversion efficiency alternative product production (tonne feedstock · tonne alternative product*$^{-1}$*)*
k = *year of disposal since production of alternative product (years)*
$\bar{e}$ = *GHG emissions (kg* CO_2*-eq. · alternative product*$^{-1}$*); subscripts: IW = incineration without electricity production, IW = incineration with electricity production, LF = landfill*
l = *year of emission since initial disposal (years)*

Equation 2.6 shows the overall lifetime landfill GHG emissions per tonne disposed alternative products ($\bar{e}_{LF}$). Methane that is produced in the landfill (*MP*) is partly released to the atmosphere ($f_{LF\ CH4}$), partly flared ($f_{LF\ flare}$), and partly burned for electricity production (f_{LFel}). The latter is considered GHG neutral, as emissions from natural gas-based electricity are avoided by using landfill methane. Part of CO_2 production in the landfill (*CP*) is emitted to the atmosphere ($f_{LF\ CO2}$), while the remainder remains in the landfill.

$$\bar{e}_{LF} = MP \cdot \left(f_{LF\ CH4} + f_{LFflare} \cdot \frac{44.01}{16.04 \cdot 34} + f_{LFel} \cdot 0 \right) + CP \cdot f_{LF\ CO2}$$ **Equation 2.6**

Where:
MP = *overall landfill* CH_4 *production (kg* CO_2*-eq. · t alternative product*$^{-1}$*)*
CP = *overall landfill* CO_2 *production (kg* CO_2*-eq. · t alternative product*$^{-1}$*)*

Lastly, to allow for comparison with GHG footprinting studies (e.g. Dwivedi et al., 2011, 2014a), the percentages of GHG emission reduction of wood-pellet electricity compared to EU fossil grid electricity (*ER*) were calculated as well (Equation 2.7). In GHG footprinting biogenic CO_2 emissions are considered GHG neutral and no alternative scenarios are included.

$$ER = \left(1 - \frac{(SQ - e_{MH} - e_{SM} - e_{PM} - e_{PP} - e_{LO})}{ae}\right) \cdot 100\% \qquad \textbf{Equation 2.7}$$

Sensitivity analyses

A sensitivity analysis was performed on the GHG parity times of wood-pellet electricity for parameters that most affect parity times (as determined by trying all parameters), and for parameters whose values are uncertain based on literature. The variation in parameter value of the selected parameters was based on literature (Table 2.1). Two further sensitivity analyses were performed. First, economic allocation was applied to feedstock production GHG emissions, instead of mass-based allocation (see Table S2). Second, the timing of alternative product disposal was investigated to test the sensitivity of GHG parity times both to alternative product composition (as some products have longer use phases than others) and to uncertainty in disposal patterns in general (Smith et al., 2006, see Table S3), including delayed final disposal due to recycling. The analysis consisted of delaying disposal of half of the alternative product produced by an additional 50 years compared to default values.

Table 2.1 | Parameter sensitivity.

parameter	minimum	maximum	notes
	% of default value		
GHG emissions of plantation management and harvesting ($\mathbf{e_{MH}}$)	75%	125%	a
enhanced growth of thinned forest (affects e_{MH})	0%	200%	b
GHG emissions of wood-pellet electricity supply chain ($\mathbf{e_{PM} + e_{PP} + e_{LO}}$)	75%	125%	a, c
GHG emissions of production of alt. products ($\boldsymbol{\varepsilon_{APp}}$)	77%	128%	d
GHG emissions disposal of alt. products ($\mathbf{E_{APd}(t)}$)	50%	200%	e
half-life of carbon during (exponential) in-forest decomposition ($\mathbf{t_{1/2DC}}$)			
small roundwood and commercial thinnings	27%	136%	f
harvest residues	69%	123%	g
fraction of decomposed carbon stored in forest soil ($\mathbf{f_{DC\ soil}}$)	50%	200%	e, h
fraction of CH_4 (and N_2O) released during in-forest decomposition ($\mathbf{f_{DC\ CH4}}$)	50%	200%	e
softwood plantation yield (affects e_{MH})	75%	125%	a, i

Notes: a: A wide range of literature is available for these parameters – with relatively little variation among studies (see main text). Therefore uncertainty was limited to 75-125%. **b:** The growth rate of thinned plantations was varied such that final biomass stocks on a thinned plantation were reduced by either 0% or 100% of the amount of biomass taken out during thinning; the default setting was a reduction of 50% (see main text). Growth linearly affects yield, which affects e_{MH} (see note i)**. c.** Excluded are: GHG emissions from feedstock production, sequestration and any CO_2 emissions from biogenic carbon. **d:** Matthews et al. (2015). **e:** Few studies on these parameters exist and uncertainty was therefore deemed high at 50-200%, i.e. doubling or halving parameter values. **f:** Based on: Palosuo et al., 2001; Liski et al., 2002; Radtke et al., 2009; Zanchi et al., 2012; Dunn & Bailey, 2012; and Russell et al. (2014, 2015). **g:** Based on: Liski et al. (2002) and Mobley et al. (2013). **h:** Decomposition GHG emissions ($f_{DC\ CH4}$ and $f_{DC\ CO2}$) change accordingly. **i:** Plantation yield inverse linearly affects GHG emissions of plantation management and harvesting. In terms of GHG balance, yield sensitivity analysis is therefore essentially the same as for e_{MH}.

2.3 Results

Our results show how the GHG balances of wood-pellet electricity from different feedstocks compare to the balances of individual counterfactuals (Figure 2.3) and of alternative scenarios (Figure 2.4). GHG parity times that result from this comparison form the main results of this study (Figure 2.5, Tables S4 and S5). GHG footprinting outcomes are included as well, for comparison with previous studies. Lastly, the sensitivity of all results to parameterisation is shown (Figure 2.6).

Wood-pellet electricity

The GHG balance of wood-pellet electricity from all feedstocks is positive (i.e., wood-pellet electricity results in reduced GHG emissions compared to the EU fossil grid electricity it replaces; Figure 2.3), because the avoided fossil electricity emissions are higher than net emissions from wood-pellet electricity itself. The GHG balance is *immediately* positive because of the landscape-level approach applied (which is considered appropriate for the US SE; Jonker et al., 2013), and because only existing softwood plantations are considered – meaning that land- or forest use change emissions were excluded. The GHG balance of wood-pellet electricity differs little among feedstocks (Figure 2.3). The only deviations are caused by thinning and saw milling emissions, which slightly lower the GHG balance of wood-pellet electricity from commercial thinnings and from mill residues, respectively.

Wood-pellet electricity vs. individual counterfactuals

The GHG balance of the alternative product counterfactual is determined by manufacturing emissions, temporary carbon storage in the product and product disposal emissions. Temporary carbon storage has a positive effect on the GHG balance. However, the average GHG emissions from the production of alternative products are higher than the average avoided emissions of the alternative products, having a (strongly) negative effect on the GHG balance. The avoided emissions of the alternative products were determined as the GHG emissions of the products they replace. The alternative products (i.e., pulp and paper, OSB and other panels) in this study are more GHG-intensive than the products they replace (i.e. recycled paper, blockwork external wall cladding, plasterboard partition wall, see Table S1 note ab and Matthews et al., 2015). Overall, wood-pellet electricity (from all feedstocks) has GHG parity times of one year when compared to the alternative product counterfactual (Table S4). Product disposal only has a minor effect on parity time, as most alternative products are still in use after this first year (see Table S3). After parity is reached, wood-pellet electricity has larger and increasing GHG benefits (Figure 2.3). Ultimately, despite alternative products embedding carbon, their GHG balance becomes negative after about 40 years (Figure 2.3), because alternative product production GHG emissions are larger than avoided emissions (as explained above), and because methane is emitted from an increasing amount of disposed material.

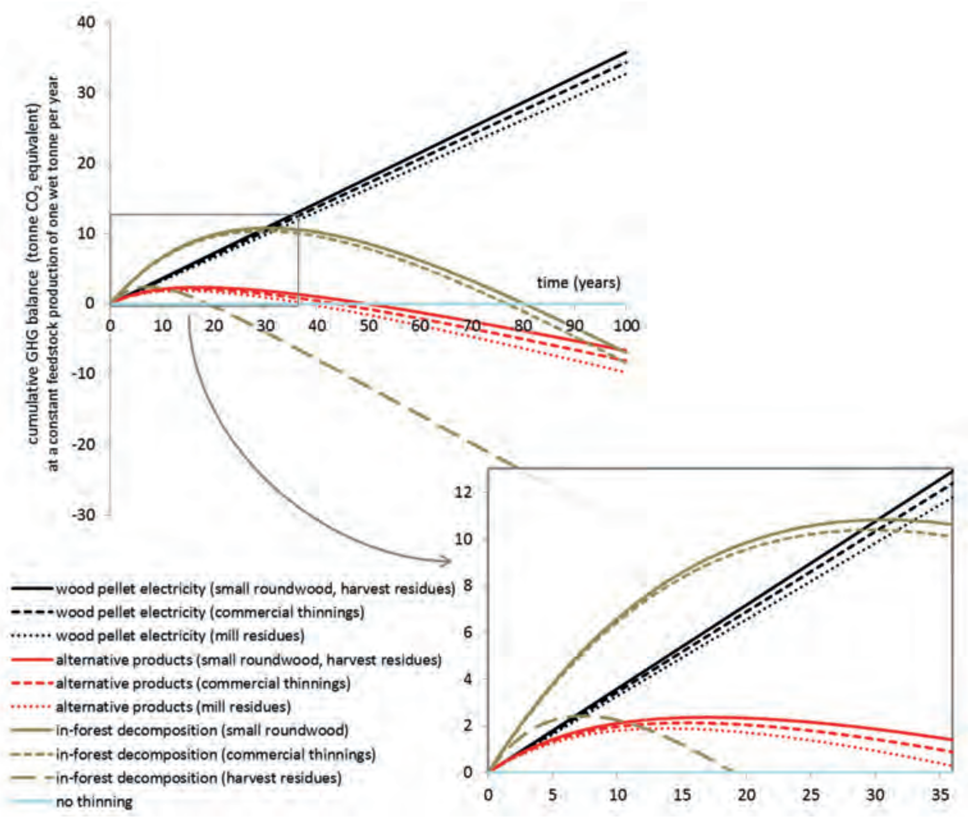

Figure 2.3 | Cumulative GHG balance of wood-pellet electricity and this study's three individual counterfactuals (alternative products, in-forest decomposition, and not thinning) shown per feedstock type at constant feedstock production of one wet tonne per year.

GHG parity times of wood-pellet electricity as compared to the in-forest decomposition counterfactual are 6 years for harvest residues, and a substantially longer 30 years for small roundwood and commercial thinnings (Figure 2.3, Table S4). The latter two decompose more slowly and hence store carbon for longer period of time, resulting in a more positive GHG balance. GHG emissions (including methane) from an accumulating amount of decomposing material eventually become larger than the GHG benefits of carbon stored in decomposing matter, causing a negative GHG balance after 18 years or about 80 years for harvest residues, and for small roundwood and commercial thinnings, respectively (Figure 2.3). In the long run, the GHG balance of the decomposition counterfactual becomes more negative than that of the alternative product counterfactual. This means that in-forest decomposition may cause larger absolute GHG emissions than alternative products, despite the fact that decomposition results in longer GHG parity times. This result is especially relevant for harvest residues, where decomposition is relatively fast (Figure 2.3). The counterfactual of not thinning was assumed to be GHG neutral (as explained in counterfactual section of the methods), resulting in immediate GHG parity, when compared to wood-pellet electricity, and in accumulating GHG benefits in the long run (Figure 2.3, Table S4).

Wood-pellet electricity vs. alternative scenarios

The largest differences among feedstocks are found in the GHG balance of their alternative scenarios (Figure 2.4), which consist of combinations of individual counterfactuals' GHG balances (Figure 2.3). Using small roundwood results in the longest GHG parity times for wood-pellet electricity, of 3-21 years (Figure 2.4 and 2.5, Table S5), for the alternative scenario (especially at low feedstock demand for alternative products) consists of a large share of in-forest decomposition. Due to the feedstock's relatively large size, decomposition is relatively slow, and carbon is stored for a long time. At higher demand, more roundwood is used for alternative products (rather than being left to decompose), which is a worse alternative in terms of GHG emissions, hence shortening GHG parity times of wood-pellet electricity.

The alternative scenarios for commercial thinnings have similar GHG balances to those of small roundwood (Figure 2.4). However, since part of all alternative scenarios for commercial thinnings is *not thinning*, which was considered GHG neutral (as explained in the methods section), the alternative scenarios' GHG balances are lowered. This means that, for commercial thinnings, wood-pellet electricity has near-instant GHG benefits (GHG parity times of 0-1 year) over the alternative scenarios at all levels of feedstock demand (Figure 2.5, Table S5).

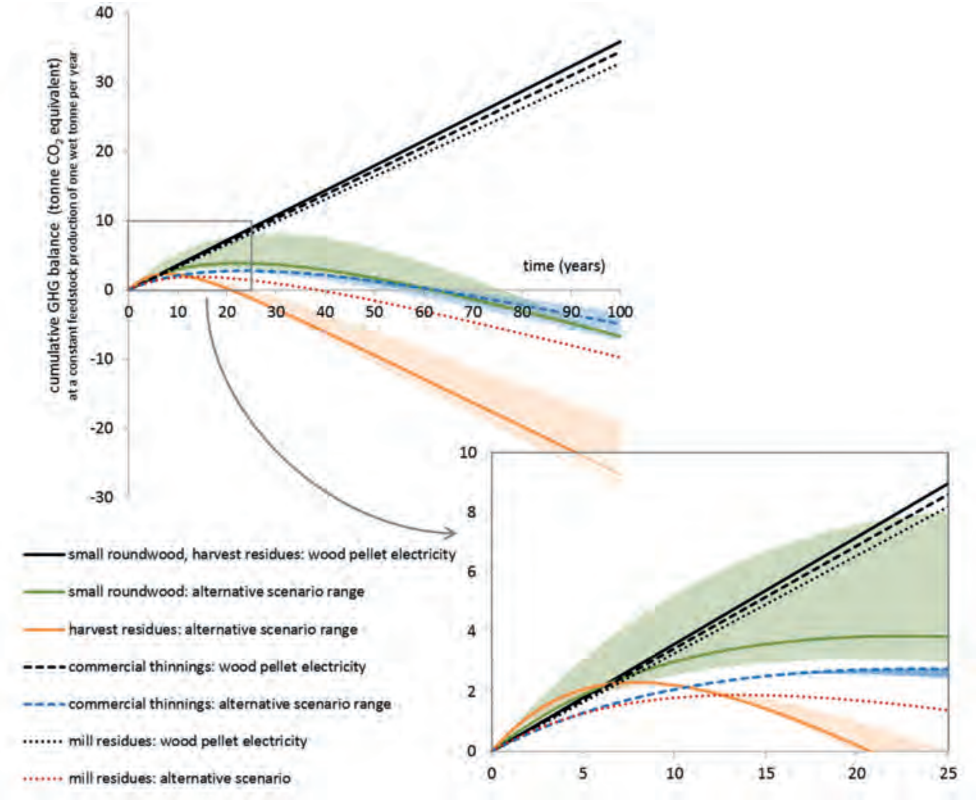

Figure 2.4 | Cumulative GHG balances of wood-pellet electricity and alternative scenarios per feedstock type at constant feedstock production of one wet tonne per year. Lines indicate alternative scenarios with average demand for feedstock to produce alternative products. Shaded areas indicate the range of alternative scenario outcomes from low feedstock demand (upper end of shaded area) to high feedstock demand (lower end of shaded area); harvest residues form an exception, for low demand leads to the lowest GHG balance (lower end of shaded area). The GHG balances of all alternative scenarios for mill residues are equal, due to equal scenario composition (Figure 2.2). Note that a negative balances indicates net GHG emissions and that sensitivity analyses are not included in this figure.

Wood-pellet electricity from harvest residues has short GHG parity times (5-6 years; Figure 2.5, Table S5) at all levels of feedstock demand for alternative products. This result is caused by the fact that the alternative scenarios for harvest residues largely consist of decomposition, which is relatively fast for harvest residues due to their small size, meaning that GHG benefits of temporary carbon storage are small (Figure 2.4). In the long run, using harvest residues for wood-pellet electricity causes relatively large absolute GHG savings, as the alternative scenario (largely fast decomposition) leads to large GHG emissions.

The alternative scenario for mill residues consists entirely of the production of alternative products, at all levels of feedstock demand (Figure 2.2). As explained in the previous section, the GHG balance of alternative products quickly becomes lower than that of wood-pellet electricity, resulting in GHG parity times of one year.

Demand for feedstock to produce alternative products only had a strong effect on GHG parity times of small roundwood. For small roundwood, a larger demand means replacing more (GHG-intensive) alternative products and less (slow) decomposition. Alternative scenario composition of mill and harvest residues was not or minimally influenced by demand (Figure 2.2). For commercial thinnings, the alternative scenarios are dependent on demand, but consist mostly of either the not-thinning counterfactual or the alternative products counterfactual, which both cause a lower GHG balance.

GHG footprinting

When applying a GHG footprinting approach (i.e. considering biogenic CO_2 emissions GHG neutral and not including alternative scenarios), GHG emission reductions of wood-pellet electricity compared to fossil EU grid electricity are 71% (for small roundwood and harvest residues), 69% (for commercial thinnings) or 65% (for mill residues), as shown in more detail in Figure S3. The GHG reduction percentage of wood-pellet electricity from mill residues was also calculated using JRC methodology (JRC, 2014), which considers mill residues a pure by-product and excludes plantation management, harvesting and saw milling emissions; this leads to a 75% GHG emission reduction (Figure S3).

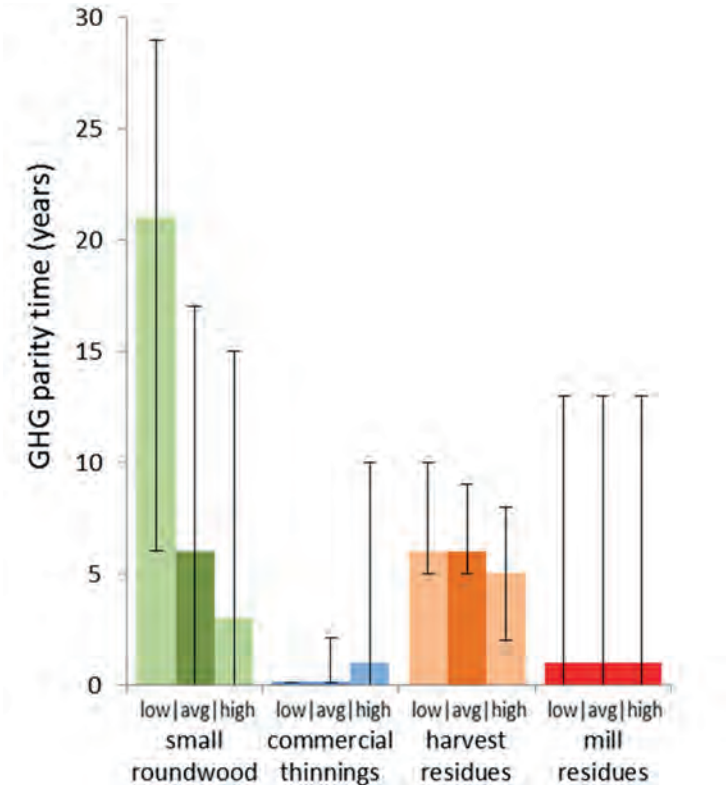

Figure 2.5 | GHG parity times of wood-pellet electricity from different feedstocks. Parity times are determined as compared to the feedstocks' alternative scenarios, at low, average and high demand for feedstock for alternative products. Error bars indicate the range of GHG parity times found in the sensitivity analysis.

Sensitivity analyses

GHG parity times of wood-pellet electricity are sensitive to five of the investigated parameters (Figure 2.6, Table S6). First, sensitivity is in most cases highest for GHG emissions of the production of alternative products. For alternative scenarios with a large alternative products component (mill residues, other feedstocks at high demand for feedstock to produce alternative products), halving production of GHG emissions increases GHG parity time by up to twelve years (Figure 2.6). Furthermore, GHG parity times of harvest residues become shorter than those of commercial thinnings (at high demand) and mill residues in general. Second, by doubling or halving the emissions from alternative product disposal, GHG parity times are respectively reduced or extented by a maximum of five years

(Figure 2.6). When varying this or any of the remaining parameters, the order of feedstocks in terms of GHG-benefits does not deviate from the ranking under default parameterisation. Third, when in-forest decomposition forms a large component of the alternative scenario (i.e. small roundwood at low demand and harvest residues at all levels of demand), doubling or halving the fraction of decomposed carbon that is stored in the forest soil, increases or decreases GHG parity times by a maximum of eight years (Figure 6). Fourth, when in-forest decomposition forms a large component of the alternative scenario, varying the half-life value of (exponential) in-forest decomposition of feedstocks proves another sensitive parameter. GHG parity times for harvest residues change by up to 3 years (Figure 2.6). For small roundwood, assuming the shortest half-life (5 years) even reduces GHG parity times of wood-pellet electricity from 21 years to 6 years. Fifth, variation in wood-pellet electricity supply chain emissions (Table 2.1) changes parity times by one to three years (Figure 2.6). Lastly, delaying half of the GHG emissions from alternative product disposal by 50 years, causes a maximum GHG parity time increase of six years (Table S6).

Varying other investigated parameters (CH_4 and N_2O emissions from decomposition, plantation productivity, effect of thinning on growth, plantation management emissions) over their estimated parameter range (Table 2.1), or applying economic allocation to feedstock production GHG emissions, affects GHG parity times of wood-pellet electricity by less than one year.

Overall, the sensitivity analysis shows that results are robust across our wide range of alternative scenarios. The changes in GHG parity times through varying input parameter values are limited. Parity times of wood-pellet electricity range 0-21 years for default values, and 0-29 years in the sensitivity analyses, excluding interaction effects. The order of feedstocks in terms of GHG parity times *only* changes when substantially varying alternative product production GHG emissions.

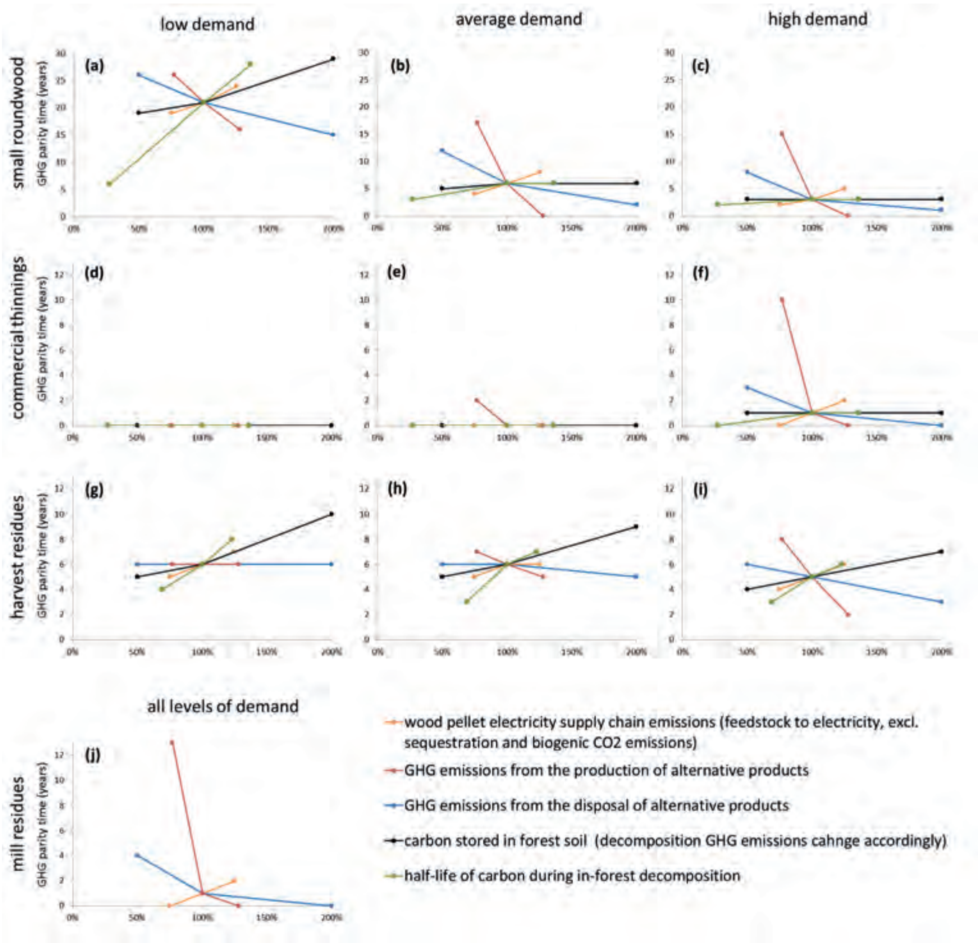

Figure 2.6 | Sensitivity analysis of GHG parity times of wood-pellet electricity from different feedstocks as compared to three alternative feedstock-use scenarios. Sensitivity analyses are performed for different feedstocks: **a-c**. small roundwood, **d-f**. commercial thinnings, **g-i**. harvest residues, **j.** mill residues, and for low-, average- and high demand for feedstock to produce alternative products, respectively. Parameter variation is shown on the x-axis. Note that sensitivity analyses of all alternative scenarios for mill residues are equal, due to equal scenario composition.

2.4 Discussion

Comparison with previous studies

In our study, GHG parity times of wood-pellet electricity from US SE softwood plantation-derived feedstocks range 0-21 years under default parameterisation and 0-29 years in sensitivity analysis. Previous studies with similar assumptions (reference electricity formed by an average fossil fuel mix for electricity, medium-intensity forest management) yielded different results, because different alternative scenarios were assumed. Jonker et al. (2013) compared wood-pellet electricity to the alternative scenarios of *protection* (no trees harvested) and *natural regrowth* (trees harvested once, followed by natural regrowth), which resulted in GHG parity times of wood-pellet electricity of 55 and 41 years, respectively. Colnes et al. (2012) used a *business-as-usual* alternative scenario (harvest for traditional products only) that resulted in GHG parity times (*avant la lettre)* of about 40 years.

We argue that our new feedstock-fate based alternative scenarios are more relevant for wood-pellet electricity than these land- or forest-use based alternative scenarios. Land- or forest use-based alternative scenarios assume a single end use for all forest products and implicitly assume that wood-pellet markets are the *main* driver of forest- and/or land use. Decisions on land- or forest use are, however, more likely influenced by saw timber and paper markets (Forest2market, 2016; Wear & Greis, 2013), landownership changes (Forest2market, 2016) and external pressures like urbanisation (Wear & Greis, 2013). Feedstock fate-based alternative scenarios, on the other hand, differentiate feedstocks and fates of different forest products. Moreover, they consider the more relevant question of what happens to the (lower-value) feedstock once it is produced (rather than whether it is produced). This question is highly relevant, because wood-pellet feedstocks are co-products of more valuable forest products (like saw timber), whose production largely determines feedstock availability.

Previous work indicates that wood-pellet electricity from residues (harvest- and mill residues) leads to fast and/or large GHG benefits, as this feedstock would otherwise be burnt or decompose (McKechnie et al., 2011; Colnes et al., 2012; Zanchi et al., 2012; Bernier & Paré, 2013; Lamers & Junginger, 2013, Lamers et al., 2014; Stephenson & MacKay, 2014; Dwivedi et al., 2016). We come to similar conclusions regarding harvest residues (GHG parity times of 5-6 years and relatively large long-term GHG savings), for their alternative scenario largely consists of rapid decomposition. For mill residues we also found short GHG parity times (approximately 1 year), but for a different reason: the alternative scenario consists of the production of relatively GHG-intensive alternative products (as discussed in the next section).

Commercial thinnings and small roundwood are often not separately considered in previous work, but are lumped in the wider category of “whole trees” (which also includes saw wood). GHG benefits of wood-pellet electricity from this category are low and/or slow, as additional tree felling is required, reducing carbon stocks (McKechnie et al., 2011; Colnes et al., 2012; Zanchi et al., 2012). However, except for culled trees, whole-tree usage for pellets is unlikely, as other industries pay more for larger diameter parts of straight stems (see Table S2). In contrast to these studies, we found that commercial thinnings (0-1 year GHG parity times) and small roundwood at medium- and high demand for feedstock (3-6 years GHG parity times) lead to rapid GHG benefits, as the alternative is either not thinning at all or usage for relatively GHG-intensive alternative products. At low feedstock demand, the alternative scenario for small roundwood largely consists of decomposition, which delays GHG benefits of wood-pellet electricity (in this study: a 21 year GHG parity time), in line with Gustavsson et al. (2015). Since decomposition rates vary significantly and locally (Russell et al., 2014, 2015), GHG parity times of small roundwood at low demand may also be substantially shorter (down to 6 years in the most extreme case). The default 21 year GHG parity time can be considered a conservative estimate.

GHG footprinting showed that GHG emissions of wood-pellet electricity from different feedstocks are 65%-75% lower than the EU fossil electricity mix (without considering alternative scenarios or temporal dynamics). This estimate is in line with the 50-75% reduction found in previous studies on EU electricity from US SE wood pellets (Dwivedi et al., 2011, 2014a; personal communication G.-J. Jonker, October 23, 2015).

Robustness of our approach

Sensitivity analysis showed that our wood-pellet electricity GHG parity times are robust for all studied alternative scenarios. What exact combination of counterfactuals is relevant to a wood-pellet feedstock remains a more hypothetical and to some degree subjective matter. This issue was largely negated by considering a wide range of alternative scenarios for each feedstock (at different levels of feedstock demand for alternative products) and by the outcome that for each feedstock GHG parity times are similar accross these alternative scenarios (except for small roundwood at low demand). Saw wood demand may also influence alternative scenario composition, as it is an important driver of forest management and harvesting decisions (Aebiom, 2013). However, considering the range of alternative scenarios already studied here, we do not expect substantial changes in overall outcomes.

We captured the most important counterfactuals for wood-pellet feedstocks from softwood plantations via consultation with local experts (see 'Methods'). Other, less frequent counterfactuals may include the following: i) burning feedstock material as waste (which is common on non-plantation private forests), resulting in immediate GHG benefits of wood-pellet electricity; or ii) using feedstock material for local energy (beyond processing heat), which may cause fewer GHG emissions than electricity from long-distance transported wood pellets. As these counterfactuals are not frequent in softwood plantations, they would unlikely affect our conclusions.

The alternative product counterfactual showed a relatively low and eventually negative GHG balance, because the alternative products were relatively GHG-intensive. This result is somewhat counterintuitive, as most wood-based products are relatively GHG-unintensive; lumber for instance can replace more GHG-intensive products like steel or concrete. However this relationship does not hold for the wood-based alternative products of wood-pellet feedstocks: OSB (19% of alternative products) and other panels (1%) replace products with similar associated GHG emissions (based on Matthews et al., 2015). Pulp and paper products (80% of alternative products) are even three times *more* GHG intensive than the product they replace, that is, recycled pulp and paper (with both virgin and recycled pulp and paper starting from dry feedstock; based on Matthews et al., 2015). Taking *GHG-unintensive* recycled paper as replaced product, may seem to lead to an (overly) optimistic estimate of the GHG benefits of wood-pellet electricity (as the alternative product to wood pellets, i.e. virgin pulp and paper, becomes relatively *GHG-intensive*). However, recycled paper is in many applications the only real alternative to virgin paper (as also assumed by Matthews et al., 2015).

Increasing the share of recycled paper in the US seems feasible, as the EU paper recycling rate is for instance 7% higher than the US rate (EPA, 2013; ERPC, 2015). Moreover, when pulp and paper replace products other than recycled paper, these other replaced products are often also less GHG-intensive than virgin pulp and paper. Plastic packaging for example is about three times less GHG-intensive than paper packaging, due to lower weight requirements and a less GHG-intensive production process (Cadman et al., 2005; NIAR, 2011; Franklin Associates, 2014).

We explicitly looked at wood pellets derived from *existing* softwood plantations. Results may be different for new plantations, as GHG emission or sequestration from converting previous land- or forest uses to plantations should be accounted for (e.g. Fargione et al., 2008; Searchinger et al., 2008, 2009; Berndes et al., 2013; Lamers & Junginger, 2013), as well as potential associated albedo changes and other biogeophysical climate forcings (e.g. Cherubini et al., 2012; Bright, 2015). Current availability of wood-pellet feedstock material will likely continue to suffice for wood pellet exports towards 2030 (Fingerman et al., 2016). This implies that a large share of wood-pellet feedstock will continue to be derived from existing softwood plantations, highlighting the importance of our study. In case demand for alternative products (pulp and paper, panels) increases, our high demand scenarios will be more relevant. When demand for alternative products does not increase and/or when pellet, paper and OSB mills avoid local competition for feedstock, our low demand scenarios may be more relevant.

We included direct wood-use change (WUC) effects by considering counterfactuals. We also accounted for avoided emissions of both wood-pellet electricity and of alternative products. These assumptions are internally consistent and account for indirect wood use change (iWUC; e.g. using woody feedstock for pellets, means that less virgin paper and more recycled paper is produced). Since we considered existing plantations, no direct land-use change (LUC) effects had to be accounted for. However, indirect land-use change (iLUC) effects could still be caused by WUC. As an hypothetical example, increased feedstock use for pellets could mean that more feedstock has to be produced on other land to meet demand from paper mills. This iLUC through WUC effect may not be large, as pellet mills produce the least valuable product (see Table S3) and tend to have lower buying power than the competing industries, but requires further research nonetheless.

Implications of our findings

Based on robust results, we conclude that wood-pellet electricity from *existing* US SE softwood plantations reduces GHG emissions compared to EU fossil grid electricity within 0-29 years for all investigated wood-pellet feedstocks while taking feedstocks' alternative fates into account. The climate change mitigation potential of wood-pellet electricity can be maximised by sourcing wood pellets from commercial thinnings, mill- and harvest residues, leading to GHG benefits within several years, substantially faster than was found in previous work (e.g. Jonker et al., 2013; Colnes et al., 2012). However, the GHG balance of wood-pellet electricity from non-plantation forests or from newly created plantations, as well as sustainability concerns beyond climate change, need to be addressed separately.

We also find that allocating the studied feedstocks, i.e. lower-value forest materials, to wood-pellet electricity rather than to paper and wood panels (e.g. OSB) reduces GHG emissions. Electricity and these products serve very different purposes and are not interchangeable. Therefore, whether (feedstock use of) wood-pellet mills will replace paper or OSB mills ultimately depends on market dynamics of the different products. Whether it is desirable that they replace paper or OSB mills in terms of GHG emissions also depends on potential iLUC emissions. Nonetheless, our findings do imply that the climate change mitigation paradigm of prioritising materials over bioenergy (e.g. Ellen-MacArthur Foundation, 2013; Vis et al., 2016) does not hold in all circumstances and should in some cases be reconsidered.

Finally, we argue that for wood-pellet electricity from the studied feedstocks, alternative feedstock fates form a more relevant alternative scenario than alternative land- or forest use scenarios. The reason being that the latter implicitly and (likely) inaccurately assume wood pellets are the main driver of forest- and land-use change and assume a single end use for all forest feedstocks. More generally, feedstock-fate based analyses may be highly relevant for all bioenergy from lower value co-products of existing industries. The discussion on land- or forest use for bioenergy vs. carbon storage or traditional uses (e.g. Schlamadinger & Marland, 1996a,b; Marland & Schlamadinger, 1997; McKechnie 2011; Berndes et al., 2013) should therefore also include trade-offs between using feedstock for bioenergy vs. alternative fates.

2.5 Acknowledgements

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3

Life cycle greenhouse gas benefits or burdens of residual biomass from landscape management

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ABSTRACT

The use of residual biomass for the production of bioenergy and biomaterials is often suggested as a strategy to avoid the negative effects associated with dedicated biomass production. One potential source is biomass from landscape management. The goal of this study was to find the lowest net greenhouse gas (GHG) emissions of various applications of residual biomass from landscape management. GHG balances of thirteen residual biomass applications were calculated and compared to their respective conventional counterfactuals. As a case study, the potential contribution to climate change mitigation of biomass from vegetation management in floodplains of the Dutch Rhine delta was quantified. The greatest GHG benefits are achieved when using woody biomass to produce heat (-132 kg CO2-eq./ tonne wet biomass) and grassy biomass to produce growth media (-229 kg CO2-eq./tonne wet biomass). In contrast, composting grassy biomass for fertiliser replacement on agricultural fields results in the largest GHG burdens of 62 kg CO2-eq. / tonne wet biomass. The findings imply that residual biomass from landscape management can contribute to both GHG benefits and burdens, depending on the application. Higher benefits were found for bioenergy than for biomaterial applications. Biomass applications should be chosen with care and consideration of their counterfactuals.

3.1 Introduction

Bioenergy and biomaterials may contribute to a reduction in fossil fuel use and the mitigation of climate change (Creutzig et al., 2015). The dedicated production of biomass requires significant amounts of land and water, which can lead to an increase in water scarcity and both direct and indirect effects of land-use change. In many cases, greenhouse gas (GHG) emissions caused by land-use change outweigh the GHG savings of bioenergy production for years to decades (Elshout et al., 2015) or even longer (Searchinger et al., 2008). The use of residual biomass, rather than dedicated biomass production, can avoid negative effects associated with land-use change and water use Creutzig et al., (2015) and is recommend to policymakers Dornburg et al. (2010). Residual biomass includes harvesting and processing residues from agriculture and forestry, animal manure, biogenic waste streams from industry and consumers, and residues of landscape management (Smith et al., 2014). Landscape residues include biomass released during vegetation management in various types of landscapes, for example roadside vegetation, pastures and semi-natural landscapes such as floodplains (Pfau, 2015).

Various publications have addressed the GHG emissions of bioenergy produced from residual biomass reporting potential GHG savings in comparison to reference systems, for example woody biomass residues from Italian orchards (Boschiero et al., 2016), forest residues in the UK (Whittaker et al., 2011) and cattle manure (de Azevedo et al., 2017). Several studies compare the climate impacts of biomass usage for different forms of bioenergy or biomaterials. For example, Gerssen-Gondelach et al. (2014) analysed a variety of feedstocks, pre-treatment technologies and applications. The authors calculated avoided GHG emissions and found beneficial results for almost all routes analysed. Kim and Song (2014) compared the recycling of wood waste into either energy or materials and reported GHG savings for both. Recchia et al. (2010) analysed the environmental benefits of energy derived from riparian vegetation in Italy and Boscaro et al. (2018) calculated GHG impacts of using grass obtained from landscape management of riverbanks for biogas production in Italy. Both studies report significant GHG benefits and are discussed further in section 3.4. No previous studies have investigated the optimal use of residual biomass from riparian vegetation, or from landscape management in general, comparing various bioenergy and biomaterial applications from a GHG emission perspective.

This study quantified the potential contribution of residual biomass available from vegetation management in floodplains of the Dutch Rhine delta to climate change mitigation through bioenergy and biomaterial production. The Dutch Rhine delta is densely populated and has a relatively high flood risk due to expected increases in peak river discharges as a result of climate change (Middelkoop et al., 2001). This has led to extensive and ongoing flood risk management (Kabat et al., 2005), including frequent riparian vegetation management to increase the water conveying capacity of floodplains (Straatsma and Kleinhans, 2018). Vegetation management based on cyclic rejuvenation can be applied to achieve optimal biomass removal (Baptist et al., 2004), while at the same time yielding a continuous biomass supply (Koopman et al., 2018). Vegetation management is costly, giving rise to the idea of residual biomass usage to (partly) repay management costs, while providing a valuable resource for sustainable products.

The goal of this study was to find the lowest net GHG emissions from various applications of residual biomass derived from landscape management (such as energy, material and feed uses). The GHG benefits or burdens of such applications are calculated in comparison with the emissions of their respective conventional energy and material counterparts, which are referred to as *counterfactuals* (cfl.). The consideration of counterfactual emissions, as proposed in this study, enables the comparison of net GHG emissions across different types of applications (e.g. energy *vs.* material applications), and can be applied to any source of residual biomass. This study demonstrates how landscape management residues can contribute to climate change mitigation, focusing on thirteen applications of residual biomass from Dutch floodplain management.

3.2 Methods

Biomass applications and counterfactuals

Residual biomass harvested during vegetation management was categorised into: i) woody biomass from forests and shrubs, and ii) grassy biomass from reeds, herbaceous vegetation and natural grassland (adapted from Koopman et al., 2018). Information on current applications for both types of biomass was collected through semi-structured interviews with water management organisations involved in the management of vegetation in publicly owned areas of Dutch floodplains. These include the executive part of the Dutch Ministry of Infrastructure and Water Management, the state forestry service, and several water boards. Some of these interviews were conducted during a parallel study (Bout et al., Unpublished data).

This inventory revealed a total of thirteen biomass applications that are realised in current practice and can be subdivided into four categories: i) left or ploughed on site, ii) grazing, iii) energy production and iv) material production. Figure 3.1 shows the applications, transport and processing steps and counterfactuals. Table 3.1 provides short descriptions of the applications. An extensive description and rationale for the choice of counterfactuals is included in the supplementary information.

Greenhouse gas emissions

The GHG emissions in kg CO_2-eq / tonne wet biomass (t_{wb}) of the different applications were calculated as the difference between emissions linked with the biomass application and avoided emissions of counterfactuals (ε_C), following Equation 3.1.

$$\varepsilon_{VM} + \varepsilon_T + \varepsilon_P + \varepsilon_B + \varepsilon_D + \varepsilon_R - \varepsilon_C \qquad \textbf{Equation 3.1}$$

Emissions of biomass applications included vegetation management activities (ε_{VM}), transport of biomass to processing location (ε_T), processing (ε_P), biogenic CO_2 emissions (ε_B), decomposition emissions (ε_D) and ruminant CH_4 emissions (ε_R). Input parameters for calculations were based on literature, data from Ecoinvent v3 LCI database using the IPCC 2013 GWP100a method (Wernet et al., 2016), personal communication with stakeholders and own calculations. Default values for parameters for which ranges were found in literature were calculated as the geometric mean of all available data. For skewed distributions, as is the case for the applied input parameters, the geometric mean describes the central tendency of the data. Specific calculations for each application are shown in the supplementary information. All input parameters and their sources are shown in Tables S1 and S2.

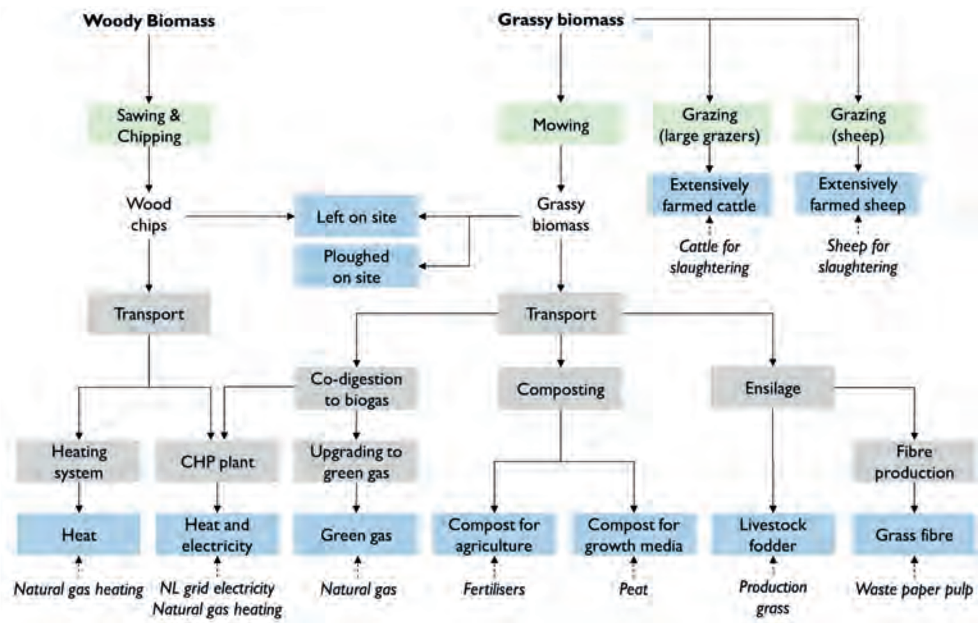

Figure 3.1 | Schematic presentation of biomass applications and counterfactuals analysed in this study. Vegetation management activities are shown in green, transport and processing steps in grey and applications in blue. Counterfactuals are indicated in italic. Both woody and grassy biomass may be left on site or applied in combined heat and power (CHP) installations (grassy biomass after conversion to biogas), resulting in 13 applications.

GHG emissions from vegetation management were calculated following Equation 3.2.

$$\varepsilon_{VM} = \sum_{MU} HP \times F_{MU} \times E_{MU} \quad \textbf{Equation 3.2}$$

Where: *HP* is the harvesting pace for woody or grassy biomass (h / t_{wb} harvested), F_{MU} the fraction of machine use for each type of machine (dimensionless) and E_{MU} the emission factors for each type of machine used (kg CO_2-eq. / h), including construction and fuel consumption. Data on machine use and fuel consumption were based on reports from contractors conducting vegetation management in the Netherlands (see supplementary information, Table S1).

Transport GHG emissions were calculated following Equation 3.3.

$$\varepsilon_T = 2 \times TD \times E_T$$ **Equation 3.3**

Where: *TD* is the biomass transport distance (km) for each application and E_T is the emission factor for transport with lorries (kg CO_2-eq. / tkm). E_T is derived from Ecoinvent and based on average load factors from the Tremove model v2.7b (De Ceuster et al., 2009) and EcoTransIT (Knörr et al., 2011) report. The emission is based on partial loading (83% of capacity) and empty return trips. The one-way transport distances were doubled to account for the distance covered by lorries to the floodplain and from the processing locations. For *TD* the minimum transport distance driving routes were determined for lorries to transport biomass from floodplains to biomass processing locations. In total, 95 processing locations in the Netherlands were identified from several sources (details in Table S3) and subsequently manually geocoded. Minimum transport distances for driving routes were calculated by means of the Google maps programming interface. The 179 floodplain sections in the study area, described below, provided the starting points and the 95 biomass processing locations gave the destination points, giving a total of 17,005 routes. Subsequently, the shortest route was selected for each floodplain section to each processing location with a specific biomass application (example shown in Figure 3.2). Transport distances were summarised by calculating the mean over all floodplain sections.

Table 3.1 | Description of biomass applications and counterfactuals. Includes the acronyms used in the text, the name of each application, a short description and the counterfactuals. More extensive description and rationale for the choice of counterfactuals is included in the supplementary information.

Acronym	Application	Description	Counterfactual
Biomass left on site and ploughed on site			
WLS	Woody biomass left on site	Biomass left at vegetation management location; natural decomposition	None: non-productive land; no fertiliser replacement
GLS	Grassy biomass left on site	Biomass left at vegetation management location; natural decomposition	None: non-productive land; no fertiliser replacement
GPoS	Grassy biomass ploughed on site	Biomass ploughed on fields to improve soil quality	None: fresh biomass applied additionally; no fertiliser replacement
Grazing			
GLG	Grassy biomass grazing large grazers	Vegetation management by year-round grazing, 70% cattle	Conventionally farmed cattle: grazers provide small amounts of organic meat
GGS	Grassy biomass grazing sheep	Vegetation management by herds of sheep	Conventionally farmed sheep: grazers provide small amounts of organic meat
Energy production			
WH	Woody biomass heat	Wood chip incineration producing heat	Conventionally produced heat
WCHP	Woody biomass CHP	Wood chip incineration producing heat and power in combined heat and power (CHP) plants	Conventionally produced heat and grid-electricity
GCHP	Grassy biomass CHP	Co-digestion of biomass with manure and subsequent CHP application of biogas	Conventionally produced heat and grid-electricity
GGG	Grassy biomass green gas	Co-digestion of biomass with manure and subsequent upgrading to green gas	Natural gas
Material production			
GCA	Grassy biomass composting for agriculture	Composting of biomass and application on agricultural fields to improve soil quality	Artificial fertilisers
GCG	Grassy biomass composting for growth media	Composting of biomass and use in production of growth media	Peat
GFo	Grassy biomass fodder	Ensilage of biomass and use as livestock fodder	Organic production grass
GFi	Grassy biomass fibres	Extraction of fibres and application in cardboard production	Pre-treated waste paper pulp

Processing GHG emissions were derived following Equation 3.4.

$$\varepsilon_P = \sum_p A_p \times E_p \qquad \textbf{Equation 3.4}$$

where A_P is the amount of each product *P* produced (e.g. kg / t_{wb} or MJ / t_{wb}) and E_P is the emission factor for production of product *P* (e.g. kg CO_2-eq. / kg or kg CO_2-eq. / MJ). These emissions can include both upstream emissions (e.g. construction of processing installations) and processing emissions (e.g. energy consumption of processing installations and emissions occurring during processing), depending on the application (see A2).

Biogenic carbon emissions were derived following Equation 3.5.

$$\varepsilon_B = E_B \times GWP_{bio} \qquad \textbf{Equation 3.5}$$

Where: E_B is the biogenic CO_2 emission of woody or grassy biomass (kg biogenic CO_2 / t_{wb}) and GWP_{bio} the global warming potential of CO_2 emissions from biomass combustion (kg fossil CO_2-eq. / kg biogenic CO_2), as developed by Cherubini et al. (2011b). A one-year rotation time was assumed for grassy biomass, based on the annual vegetation management required by flood safety regulations, resulting in a GWP_{bio} and ε_B of zero for all grassy biomass applications. Rotation times for woody biomass vary according to location: five years for high flow zones and 20 years for low flow zones. The GWP_{bio} of woody biomass was calculated based on the proportion of woody biomass increments in both flow zones, as described below.

Decomposition GHG emissions were determined following Equation 3.6.

$$\varepsilon_D = E_{N2O} \times GWP_{N2O} + E_{CH4} \times GWP_{CH4} \qquad \textbf{Equation 3.6}$$

Where: E_{N2O} and E_{CH4} are N_2O and CH_4 emissions occurring during natural decay of biomass (kg / t_{wb}) and GWP_{N2O} and GWP_{CH4} the global warming potentials of N_2O and CH_4 (kg CO_2-eq. / kg CH_4). For woody biomass, E_{N2O} and E_{CH4} were calculated based on the fractions of N emitted as N_2O and C as CH_4.

Ruminant emissions were determined following Equation 3.7.

$$\varepsilon_R = E_R \times AR \div BMP_G \times 365\ days \times GWP_{CH4}$$ **Equation 3.7**

Where: E_R are the ruminant CH_4 emissions of grazers (kg CH_4 / head /day), AR is the number of animals required to maintain one hectare for a year (head / ha), BMP_G is the grassy biomass production per ha (t_{wb} / ha) and the GWP_{CH4} the global warming potential of CH_4 (kg CO_2-eq. / kg CH_4). The grassy biomass production per ha was calculated by dividing the grassy biomass produced in each section, as described below, by the surface areas of the same section. Subsequently, the average for all sections was calculated.

Counterfactual emissions were calculated following Equation 3.8.

$$\varepsilon_C = \textstyle\sum_C A_C \times E_C$$ **Equation 3.8**

Where: A_C is the amount of each counterfactual C avoided (e.g. kg / t_{wb}) and E_C is the emission of the production of each counterfactual (e.g. kg CO2-eq. / kg). See supplementary information for further details on the counterfactual GHG emission calculations.

Study area and biomass production

The overall climate mitigation potential of residual biomass was calculated over the terrestrial floodplain area of the three Rhine river distributaries in the Netherlands (Figure 3.2). The total embanked area amounts to 440 km^2, of which 62% is vegetated. Meadows dominate the land cover, but recent nature rehabilitation programmes have led to an increase in areas with herbaceous vegetation, shrubs and forests.

Biomass from publicly owned areas was distinguished from those that are owned privately. The public areas are managed by water management or other governmental organisations. These organisations are becoming increasingly interested in using landscape residues sustainably. Biomass from privately-owned areas was included to give an impression of the overall potential on a landscape scale.

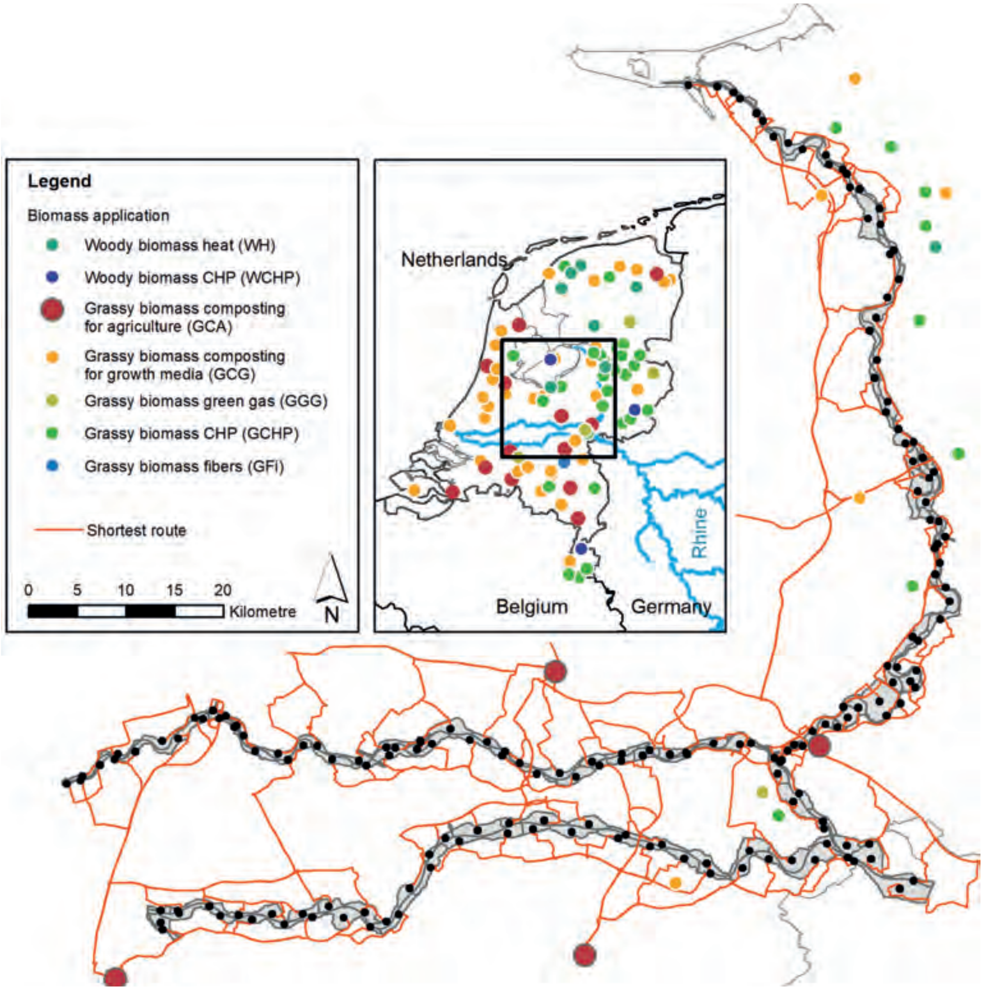

Figure 3.2 | Schematic map of the study area. Indicated are: the floodplain sections of the Dutch Rhine distributaries Waal, Nederrijn-Lek and IJssel (grey), the processing locations for different biomass applications (coloured dots), and an example of the shortest driving routes, given here between floodplains and grassy biomass composting sites for agriculture (orange lines).

The mean biomass production values per floodplain section were calculated based on three spatial datasets. Firstly, the entitled person per cadastral parcel ([dataset] Kadaster, 2017) was classified as public, or private based on the name. Secondly, vegetation limitation data (Rijkswaterstaat, 2014) divided the floodplain area into hydrodynamic flow zones defining the conveyance capacity. In high flow zones, the vegetation is limited to types with a low hydrodynamic roughness, e.g. meadows and agriculture. Shrubs, reeds and forests are allowed in low flow zones. Thirdly, ecotope data provided definitions for vegetation classes. Ecotopes are homogeneous landscape units based on specific hydro-morphological, geomorphological, ecological and land-use characteristics (Van der Molen et al., 2003). A schematic map of the 179 floodplain sections provided the spatial aggregation units (Figure 3.2). The biomass production was calculated according to Koopman et al. (2018). Four biomass production values were determined for each floodplain section using spatial overlays: i) public-low flow, ii) public-high flow, iii) private-low flow, and iv) private high flow. The four biomass production values were summed over all floodplain sections to determine the total biomass production for each combination in tonne dry matter (tDM). A final conversion was applied to wet biomass (t_{wb}) based on the dry matter (DM) fraction of woody and grassy biomass (Table S1).

Sensitivity analysis

A sensitivity analysis on the GHG emissions of different biomass applications was performed. Table 3.2 shows the parameters analysed in the sensitivity analysis. Calculations and sources for all parameters are presented in supplementary Table S1. The total GHG emission in kg CO_2-eq. / t_{wb} of each application was calculated separately for the default, minimum and maximum values of each parameter. The resulting GHG emission outcomes were then plotted against the parameter variation expressed as a percentage, where the default represents 100%.

The sensitivity of the following parameters was considered:

1. The harvesting pace of both woody and grassy biomass shows large variations in literature and has a large influence on harvesting emissions, which are part of almost all applications.
2. Biomass transport distances were by default based on the current minimum distance between floodplains and processing locations. Distances could change when roads or processing locations are altered or added. Variations of a factor 0.5 and 2 were investigated.
3. The ploughing required to apply one tonne of wet biomass on agricultural soils has a large variability in practice and documentation is limited. Variations of a factor 0.5 and 2 were explored.
4. Biogas yields during co-digestion of grassy biomass strongly influence results and are variable due to different feedstock mixtures and fermenter conditions.

5. The calorific value of wood varies with moisture content, which depends on field and (passive) drying conditions. Calorific values for 40-50% moisture contents were analysed.
6. The default electric conversion efficiency of woody biomass CHP installations is based on the current situation. However, larger-scale electricity production can result in higher efficiencies and greater avoided emissions. A scenario of CHP with higher electricity output and higher efficiency was explored.
7. CH_4 and N_2O emissions relating to natural decomposition of biomass are highly variable and little data is available. Because this study considered non-piled wood with aerobic decomposition, default woody biomass decomposition emissions were based on minimum emissions of piled wood. This assumption was tested by applying a typical value for piled wood as a maximum value. Similar variation is expected for decomposition of grassy biomass (GLS and GPoS). Variations of a factor of 0.5 and 2 were investigated.
8. Both the number of grazers required to maintain one ha of land and the CH_4 emissions per grazer affect the GHG emissions and have a substantial natural variability. The maximum and minimum calculated for the parameter based on different sources was analysed.
9. Large variability was observed in literature for data concerning N fertiliser replacement of compost, so the overall range described by different sources was analysed.
10. Regarding GCG, large variations were described in literature for both the amount of peat replaced per t compost and the GHG emissions of the counterfactual (growth media produced using peat). Both are influential parameters.
11. The GHG emission of the GFi counterfactual (fibre produced from waste paper) is uncertain due to lack of data. The actual GHG emissions of fibre production (including waste paper collection, sorting and re-pulping) are unknown. The GHG emission of recycled paper minus the electricity for the papermaking step was used but this could be a conservative estimate. The geomean of both parameters was used as default value and the overall range of values was explored here.
12. The WCHP and GCHP counterfactuals apply the current state of grid-electricity in the Netherlands. Changes in avoided emissions were quantified by applying gas electricity (minimum value) and coal electricity (maximum value), rather than the Dutch grid mix (default).

Table 3.2 | Parameters analysed in sensitivity analysis. Default, minimum and maximum values are given per parameter. It also indicated in what equation(s) each parameter is used. Calculations and sources for all parameter values can be found in Table S1.

	Parameter	Equation	Unit	Default	Min	Max
1.	Harvesting pace woody biomass	2; *HP*	h / t_{wb} harvested	0.91	0.31	2.67
	Harvesting pace grassy biomass	2; *HP*	h / t_{wb} harvested	0.57	0.42	0.77
2.	Biomass transport distance	3; *TD*	km	Table S1	50%	200%
3.	Ploughing required for GPoS	4; via A_P	ha / t_{wb}	0.2	50%	200%
4.	Biogas yield during co-digestion	4; via A_P	m^3 / t_{wb}	70	60	77
5.	Calorific value woody biomass as received	4; via A_P 8; via A_C	MJ / t_{wb}	$8.0 \cdot 10^3$	$7.4 \cdot 10^3$	$1.0 \cdot 10^4$
6.	WCHP electric conversion efficiency	4; via A_P 8; via A_C	dimensionless	0.16	0.16	0.3
7.	CH_4 emissions of WLS decomposition; fraction of C emitted as CH_4	6; via E_{CH4}	dimensionless	0.01	0.01	0.022
	N_2O emissions of WLS decomposition; fraction of N emitted as N_2O	6; via E_{N2O}	dimensionless	0.01	0.01	0.016
	N_2O emissions of GLS and GPoS decomposition	6; E_{N2O}	kg N_2O / t_{wb}	0.070	50%	200%
8.	CH_4 emissions per sheep	7; E_R	kg CH_4 / grazer / d	0.019	0.014	0.024
	CH_4 emissions per large grazer	7; E_R	kg CH_4 / grazer / d	0.19	0.13	0.27
	Sheep required to maintain one ha	7; *AR*	grazers / ha	5.2	3.8	7.2
	Large grazers required to maintain one ha	7; *AR*	grazers / ha	1.4	0.40	2.0
9.	Fertiliser replacement of GCA	8; via A_C	kg N/ t_{wb}	0.89	0.50	1.9
10.	GHG emissions of GCG counterfactual growth media from peat	8; E_C	kg CO_2-eq. / t peat	$8.1 \cdot 10^2$	$5.5 \cdot 10^2$	$1.2 \cdot 10^3$
	Peat replacement of GCG	8; via A_C	t peat / t compost	0.67	0.2	1
11.	GHG emissions of GFi counterfactual fibre from waste paper	8; E_C	kg CO_2-eq. / t pulp	$2.1 \cdot 10^2$	$1.3 \cdot 10^2$	$3.0 \cdot 10^2$
12.	GHG-intensity of counterfactual electricity WCHP and GCHP	8; via E_C	kg CO_2-eq. / MJ	0.15	0.12	0.29

3.3 Results

Greenhouse gas emissions and avoided emissions of residual biomass applications

Figure 3.3 shows the GHG emissions and savings for each application in kg CO_2-eq. / t_{wb} and the total net GHG emissions, representing the overall GHG burden or benefit that can be achieved with each tonne of residual biomass. Biomass left or ploughed on site and biomass removal by grazing animals both result in net GHG burdens. All energy applications provide GHG benefits, ranging from -132 to -112 kg CO_2-eq. / t_{wb} for woody biomass (WH and WCHP), and from -56 to -0.5 kg CO_2-eq. / t_{wb} for grassy biomass (GCHP and GGG). Note that the conversion of biogas to green gas, which more than doubles the processing emission, appeared not to be particularly worthwhile from a GHG perspective because the use of biogas in CHP installations achieves much higher GHG benefits. Material applications of grassy biomass for fibre and fodder achieve GHG benefits of -43 and -3 kg CO_2-eq. / t_{wb}. Depending on the final product, composting results in both the greatest GHG benefit and the highest GHG burden for grassy biomass with values of -229 and 62 kg CO_2-eq. / t_{wb} (GCG and GCA). This is mainly due to the large difference in counterfactual emissions. Replacing peat in growth media with compost achieves great GHG benefits. Applying compost in agriculture replaces only moderate amounts of fertilisers, which results in small GHG savings from avoided fertiliser production and application. In practice, each tonne of biomass delivered to a composting installation will contribute to both products. Assuming 18% GCG and 82% GCA application (based on BVOR, 2016), the combined outcome will be 9 kg CO_2-eq. / t_{wb}. Biogenic CO_2 emissions contribute significantly to woody biomass application emissions, averaging 40%. Transport and vegetation management emissions each contribute an average of 21% to all applications featuring these emissions.

Climate change mitigation potential of residual biomass use

The overall potential for residual biomass derived from the Rhine floodplains to contribute to climate change mitigation differed widely (Figure 3.4). It was calculated that 49 and 93 kilotons (kt) of woody biomass, and 322 and 583 kt of grassy biomass are produced per year on publicly-owned areas and over the whole study area. 86% of all residual biomass is grassy biomass and as a result, grassy biomass applications with overall GHG benefits achieve a higher climate change mitigation potential in comparison to woody biomass applications at landscape scale.

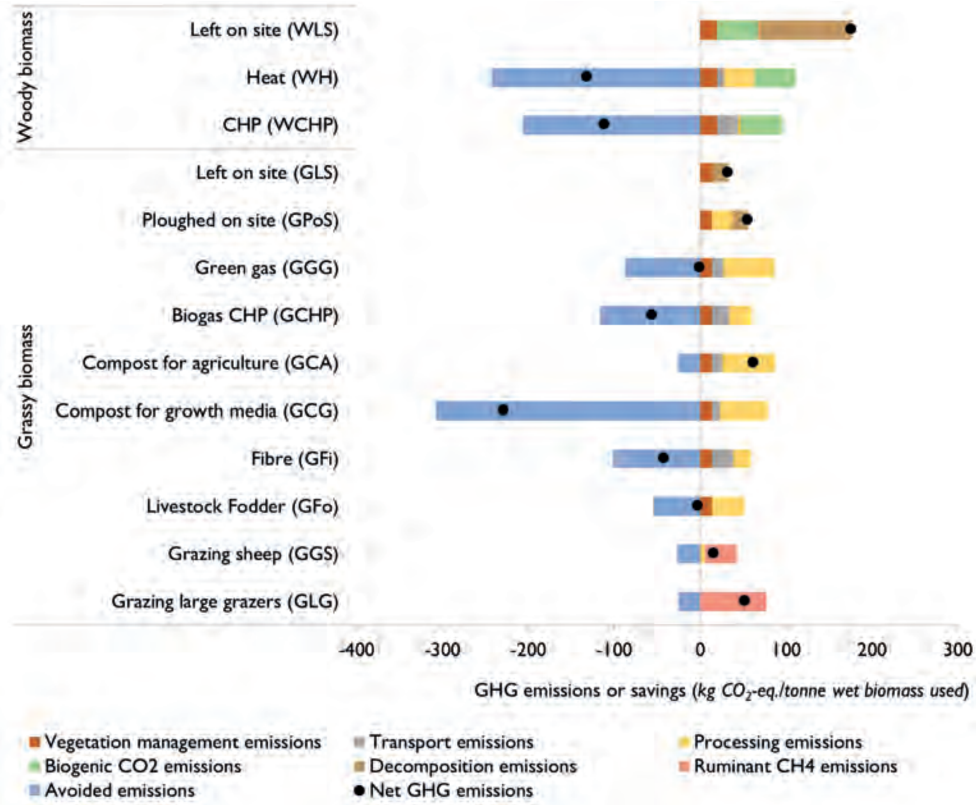

Figure 3.3 | GHG emissions and savings of current residual biomass applications at biomass scale. GHG emissions from various sources are presented as positive values. GHG savings, achieved through the replacement of counterfactuals, are presented as negative values. Net GHG emissions are the sum of emissions and savings and are presented as black dots.

The overall climate change mitigation potential depends not only on the amount of GHG emissions saved by beneficial applications, but also on their processing capacities. Table 3.3 shows the current processing capacities of the five applications resulting in clear GHG savings and the overall potential for processing biomass from the study area, based on a combination of the current capacity and the available residual biomass in the study area. Constraints resulting from current workload of these installations are not considered, assuming in the future additional capacity could be added if more landscape residues were to be processed. Table 3.3 shows that the total amount of residual grassy and woody biomass available annually would not exceed the maximum processing capacity of the most GHG-beneficial applications, WH and GCG. If public organisations ensured that their biomass was processed for the most GHG beneficial applications, a maximum contribution to climate change mitigation of 6.4 and 73.6 kt CO_2-eq. / y could be achieved for woody and grassy biomass. If all biomass from the whole study area were applied for the most GHG beneficial applications, a maximum saving of 145 kt CO_2-eq. / y could be achieved. These maximum savings are based on the usage of all available woody and grassy biomass for the most GHG-beneficial applications at their maximum processing capacities. A comparison of applications featuring the highest GHG benefits with those with the highest GHG burdens reveals a difference of 15.0 kt CO_2-eq. / y for woody biomass and 28.5 kt CO_2-eq. / y for grassy biomass from publicly-owned areas and 93.5 and 169 kt CO_2-eq. / y for the whole study area.

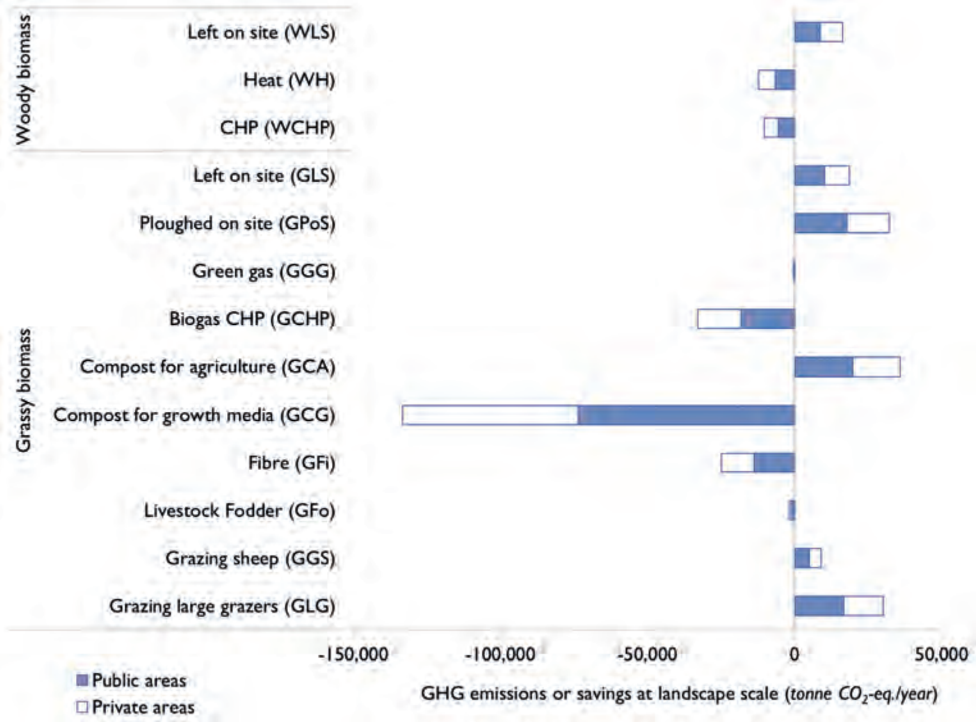

Figure 3.4 | GHG emissions and savings of residual biomass application at landscape scale. The total GHG emissions or savings are shown that can be achieved when using the entire annually available amount of biomass from management of the Dutch river Rhine floodplains. Emissions are split out based on land ownership.

Table 3.3 | Current processing capacities of the five applications with clear GHG savings in the Netherlands. Capacities are based on data from existing installations, see Table S3. The potential to process biomass from the study area is determined from the current capacity of the applications and the available residual biomass in the study area. The lowest of these values defines the potential to process. The last two columns show the maximum product output from the study area and a comparison with reference markets.

Application	**Current capacity** kt wet biomass/ yr	**Biomass availability** kt wet biomass / year	**Maximum output**		**Market comparison**
WH (Heat)	141[a]	93	674	TJ_{th} / year	16,042 Dutch households[e]
WCHP (CHP)	57[a]	57	25	TJ_{el} / year	2,323 Dutch households[e]
			242	TJ_{th} / year	5,762 Dutch households[e]
GCG (growth medium)	642[c]	583	218	kt peat replaced /year	91% of peat in growth media production in NL[c]
GCHP (CHP)	14[b]	14	8	TJ_{el} / year	790 Dutch households[e]
			12	TJ_{th} / year	290 Dutch households[e]
GFi (Fibre)	60[d]	60	29	kt fibre / year	0.5% of recycled paper use in NL[f]

Notes: a. Calculation based on the identified processing locations (described in Table S3) and data from RVO (2018); **b**. Calculation based on data from personal communication with several companies running biogas CHPs; **c**. Calculation based on market data from BVOR (2016); **d**. Calculation based on data from personal communication with a grass fibre producing company; **e**. Calculation based on household energy consumption data from milieu centraal (2018); **f**. Calculation based on data on recycled paper products in the Netherlands (Stichting PRN, 2016), assuming 1 tDM fibre replaces 1 t of recycled paper.

Table 3.3 shows that WH has the highest potential product output of all energy applications despite the limited availability of wood. WCHP and GCHP are limited by current processing capacity because there are only few WCHP installations and most biogas installations are not equipped to process grass as a co-product. Potential for GCG is large, but the large volumes of garden and kitchen wastes currently processed will limit the capacity to process landscape residues in practice.

Sensitivity to parameter variability and data uncertainties

The sensitivity analysis (Figure 3.5) shows that the results of this study are robust, except in four cases where a relatively large sensitivity is observed. Firstly, GHG emissions from biomass decomposition are highly sensitive to the share of decomposition taking place under anaerobic conditions, releasing CH_4. Under maximum anaerobic conditions, woody biomass decomposition (WLS) could lead to 67% higher overall GHG emissions per tonne of biomass (Figure 3.5a). Grassy biomass is thinner and more spread out, and is assumed to decompose aerobically. Secondly, CHP applications are sensitive to CHP efficiency and the level of GHG emissions of the counterfactual electricity production (Figure 3.5b). When replacing coal-based electricity rather than replacing the default counterfactual (current Dutch grid electricity mix) GHG emission savings increase by 44% and 54% for grassy (GCHP) and woody biomass (WCHP). For WCHP, higher efficiencies achieved through upscaling could double GHG emission savings. Thirdly, while the variability in calorific value of wood is low (the minimum value is 8% lower than the default, the maximum value is 26% higher), it is highly influential on GHG emissions of WH and WCHP: dryer wood can increase emission savings by 40% (Figure 3.5b). Fourthly, net GHG emission savings of GCG are sensitive to the amount of peat replaced and to the GHG-intensity of the replaced peat (Figure 3.5c), both of which are uncertain. GHG savings could be 67% larger, but also strongly reduced. It is unlikely that GHG savings would become smaller than those of other investigated grassy biomass applications.

The sensitivity of the results to variation in other parameters is more limited. Harvesting pace and transport distance can for instance vary substantially (200-300%), but change overall emissions per t_{wb} by less than 30%. Only one application, GGG, may alter from slightly GHG-beneficial to a small GHG burden when transport distance increases. The number of grazers and their enteric CH_4 emissions have a natural variability which affects the net GHG emissions of the grazing applications to a larger degree. Even when considering this variation, net GHG emissions remain relatively stable compared to other applications (Figure 3.5c).

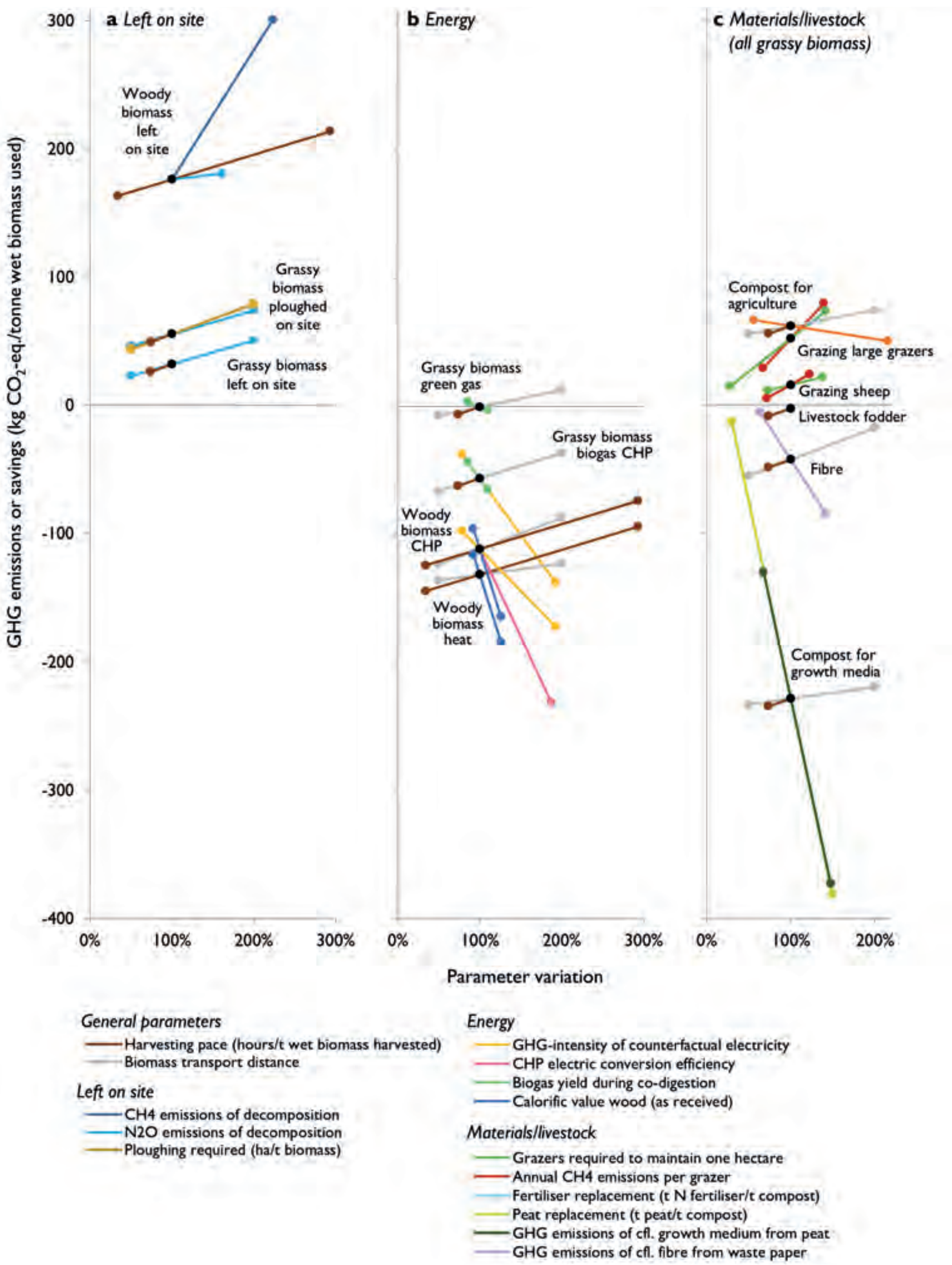

Figure 3.5 | Sensitivity analysis of total GHG emissions of various residual biomass applications. Sensitivity to parameter variations is shown based on the percentage of change in the parameter range (x-axis) and the related GHG emissions or savings (y-axis). Parameter ranges are presented in Table 3.2.

3.4 Discussion

This study compared the GHG emissions of different applications of residual biomass released during landscape management and provided relevant information on the overall climate change mitigation potential of residual biomass. The approach presented facilitated a comparison between a variety of both energy and material biomass applications through the consideration of counterfactuals. The sensitivity analysis showed that, although variation in some parameters may influence the GHG outcome, the calculated GHG benefits or burdens of applications are robust.

Higher GHG benefits were found for bioenergy than for biomaterials, an observation also described by Hanssen et al. (2017) for woody biomass. An exception is the replacement of peat as a growth medium, which results in large CH_4 emissions. Other authors have applied approaches similar to the comparison with counterfactuals in this study. These authors consider the indirect effects of products and often focus on fossil fuel replacement. For example, How et al. (2018) developed a simplified optimisation method for selecting processing technology and transport designs for residual biomass, including the replacement of fossil fuels in their environmental impact assessment. Similarly, Čuček et al. (2012) developed an approach to optimise supply chains considering various footprints and analyse the bioenergy applications of different biomass resources by considering the indirect effect of replacing fossil energy. These studies describe methodologies for the optimisation of supply chains in established biomass applications with the aim of maximising profits while minimising environmental impacts. The current study provides a novel comparison of currently feasible and practiced applications, highlighting the environmental impacts of using a particular set of biomass resources.

Two earlier studies report the impacts of applications using residual biomass from landscape management in riverine areas. Recchia et al. (2010) analysed the environmental benefits of energy derived from riparian vegetation. These authors conducted a lifecycle analysis on woody biomass burnt in a 300kW heat boiler reporting CO_2-eq. emission reductions of between 78 and 83% in comparison with fossil energy production from natural gas. This type of energy generation is similar to the WH application in the current study, which would result in an equivalent 54% emission reduction. It should be noted that Recchia et al. (2010) did not include biogenic CO_2 emissions in their analysis, while it accounted for 40% of emissions in this study (ε_B, based on GWP_{bio}). Excluding ε_B from the current calculations results in a reduction of 74%, which is close to the range described by Recchia et al. (2010), demonstrating the importance of considering biogenic CO_2 emissions. Other differences are the assumed transport distance and harvesting machinery, and the use of a different LCI database. Differences in harvesting machinery parameters are due to different landscape characteristics of the study area (mainly woody biomass as opposed to mostly grassy biomass in the current study). Boscaro et al. (2018) analysed the GHG impacts of grass obtained from riverbank landscape management in biogas production. The authors calculated the GHG balance as the difference between the emissions of biogas production from grass and the fossil fuel emissions saved as a result of heat and electricity production with biogas. This is comparable to the GCHP application. The authors calculated GHG savings of between -67 and -86 kg CO_2-eq. / t_{wb}, based on different harvesting practices and logistical scenarios, both of which differed from the approach presented in this study. When using their reported transport distances of 5 and 10 km in the current calculations, emissions of -74 kg and -73 CO_2-eq. / t_{wb} result, which fall well within the range reported by Boscaro et al. (2018).

The contribution that residual biomass from vegetation management in river floodplains makes to climate change mitigation is an important ecosystem service (Koopman et al., 2018), but this residual biomass can also provide other services. Some of the applications discussed in this paper may have costs or benefits other than their GHG impact which may play a role in choosing a particular biomass application. Natural vegetation management with grazing animals, for example, may also provide cultural ecosystem services (Oteros-Rozas et al., 2014) and contribute to biodiversity recovery during river restoration (Straatsma et al., 2017). Removal of biomass for applications outside of the riparian area may result in carbon and nutrient losses. Carbon sources remain and decompose slowly under natural conditions but certain management practices result in their active removal and a rapid release of CO_2. This has been described as a potentially problematic aspect in the harvest of stumps and logging residues (Lindholm et al., 2011), whole tree harvesting practices (Whittaker et al., 2011) and the removal of crop residues (Cherubini and Ulgiati, 2010). Leaving at least a part of the biomass on site may be advantageous for soil quality under certain conditions but is not always feasible due to flood safety regulations and disadvantageous from a GHG perspective. GCA demonstrated the highest GHG burden but can contribute to an increase in the organic matter content of agricultural soils. Soil quality is becoming increasingly important due to ongoing soil depletion in agriculture. Other factors may influence the choice of biomass applications and ideal combinations based on net GHG benefits alone may not be feasible in practice. For example, composting depends on inputs of woody biomass. The compost mixture would be too dense if only grassy biomass were composted, hindering aerobic processing. In practice, it may not be realistic to apply only residual woody biomass for energy production and only grassy biomass for composting to provide growth media.

Results of this study are based on calculations using carefully selected parameters. Limitations result from lack of data and simplifications that could be addressed in future research. For example, transport emissions could be specified considering optimisation under capacity constraints (How et al., 2016) and current workload of processing installations could be analysed to further define maximum current processing capacities. Future research could also extend to analysing additional impacts other than GHG emissions and compare new applications that are currently under development.

3.5 Conclusions

Removal and application of landscape biomass can contribute to climate change mitigation if GHG beneficial applications are chosen. This is true if landscape biomass can be removed without negative ecological consequences or has to be removed for other reasons, for example where riparian vegetation is removed to reduce flood risk. Producing heat or combined heat and power from woody biomass and growth media from compost of grassy biomass achieve the greatest GHG benefits, although the impact of growth media from compost is uncertain. Several other applications demonstrate GHG burdens and should be avoided from a climate change perspective.

In current river management practice the choice between different residual biomass applications depends on various factors including price, contribution to different ecosystem services, processing capacities of applications, and actors responsible for vegetation management (water management organisations, contractors or private land owners). It is essential that GHG benefits and burdens of different applications and their counterfactuals are considered to ensure that residual biomass makes a positive contribution to climate change mitigation.

3.6 Acknowledgements

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Assessing the environmental benefits of utilising residual flows

4

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ABSTRACT

To reduce the environmental impacts of consumption and promote a more circular economy, residual material and energy flows are increasingly utilised. In this paper we show how the environmental benefits of utilising such residual flows can be determined. Residues can generally be used to produce several final products. Moreover, the prime motivation to start utilising residual flows often is to obtain environmental or economic benefits. Therefore, the question at hand is not what the impact of the final product is, but rather what the most environmentally beneficial use of the residual flows would be. Answering this question requires i) shifting from a functional unit based on the final product towards a functional unit based on utilising a residual flow, and ii) estimating what the residual flow-based product would replace in the conventional economy, i.e. defining its *counterfactual*. In studies on residue utilisation, one or both of these adjustments have been made and they have separately been flagged as important. Here, we argue that these two adjustments to conventional LCA should always be combined and we advocate a systematic implementation of this approach in the environmental assessment of residual flows. We demonstrate the relevance of this combined approach and formalise its four-step methodology using an example on energy carriers from agricultural crop residues. We believe this formalised approach provides a useful framework to maximise the environmental benefits of residual flow utilisation.

Assessing the environmental benefits of utilising residual flows

To reduce the environmental impacts of consumption and promote a more circular economy, residual material and energy flows that were previously considered waste or surplus are increasingly utilised. Examples include the use of agricultural residues for bioenergy, the use of industrial waste heat for residential heating, or the capture and use of CO_2 for the production of chemicals. The prime motivation to start utilising such residual flows usually is to achieve environmental or economic benefits. Therefore, the question at hand is often not what the impact of a specific residues-based product is, but rather what the most environmentally beneficial use of the residual feedstock would be. This question can be answered using lifecycle assessment (LCA), but requires: i) a shift from a functional unit based on the residues-based product towards a functional unit based on utilising a residual flow, and ii) estimating what the residues-based product would replace in the 'conventional' economy, i.e. defining its counterfactual. Together, these two adjustments allow for a systematic comparison of the environmental benefits (or burdens) of alternative uses of a residual flow, based on LCA data.

This approach is well illustrated by two recent studies that look at the climate change mitigation potential of residual flows. Pfau et al. (2019) compared different options for utilising residual biomass from landscape management in Dutch river floodplains. It was found that energy applications have larger climate benefits per tonne of residual biomass utilised than most other applications, as the counterfactual energy generation is carbon-intensive. Thonemann and Pizzol (2019) looked at various options of utilising captured CO_2 from industrial flue gas. They found the production of polyols from waste CO_2 to be most climate-beneficial, predominantly because the counterfactual, conventional polyol synthesis causes significant greenhouse gas emissions.

Over the last decade, various other studies on residues have independently shifted towards a residue-utilisation based functional unit, and often included some form of counterfactuals to allow for intercomparison of different residue-based products. In fact, setting the functional unit at the relevant level in the supply chain has separately been flagged as a key issue in the LCA of multi-output systems (Ahlgren et al., 2015), as has the need for including counterfactuals to assess environmental impacts within a circular economy context (Millward- Hopkins and Purnell, 2019).

Here, we argue that these two adjustments to conventional LCA should always be combined and we advocate a systematic implementation of this approach in the environmental assessment of residual flows. Below, we demonstrate the relevance of this combined approach and formalise its four-step methodology using an example of the climate impact of energy carriers from agricultural crop residues (Figure 4.1). In LCA terminology, the methodology can be interpreted as a consequential approach with complete and explicit substitution of the main product.

The first step in this method is to identify the residual flow-based feedstock and define its functional unit, in this example 'the utilisation of 1 dry tonne of agricultural crop residues'.

The second step is to identify the potential products that could be produced from the selected feedstock and to quantify their lifecycle environmental impacts. Products included in our example are automotive biofuels and bio-electricity. Their impacts were based on median impacts reported by Creutzig et al. (2015), while considering biogenic CO_2 emissions from this annually re-growing biomass flow as GHG-neutral. Importantly, the impacts should be calculated at the level of the new functional unit, i.e., per tonne of agricultural residue utilised. For this calculation, the biomass energy content (17 GJ/dry tonne) was based on the Phyllis2 database (phyllis.nl) and energetic conversion efficiencies were set at 30% and 40% for bio-electricity and biofuels. Optionally, the environmental impacts of what would happen to the residual flows, if not utilised, can be determined as a benchmark. In our example, we looked at leaving residues on the field. We based decomposition emissions on Pfau et al. (2019) and assumed fertiliser requirements are unaltered.

The third step is to determine what the residue-based products would replace in the conventional economy, i.e., to identify the counterfactuals, and to determine their lifecycle environmental impacts at the level of the new functional unit. In our example the assumed counterfactual for biofuel is petrol. Counterfactuals are, however, not always unequivocal. For the bio-electricity option we assumed a counterfactual of (Dutch) natural-gas based electricity, but also explored a second potential counterfactual of hard coal-based electricity. The lifecycle climate change impacts of petrol and fossil electricity were estimated using ecoinvent (ecoinvent.org).

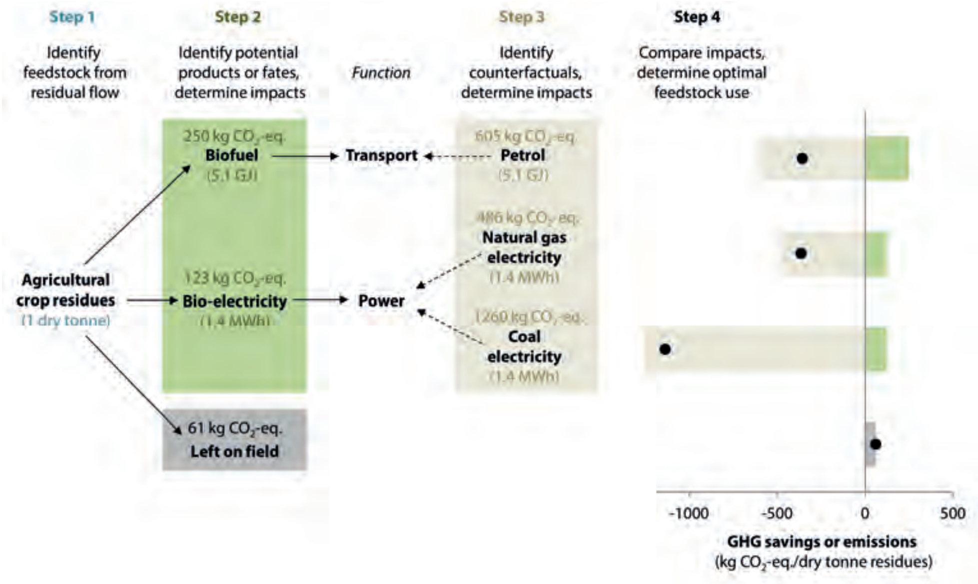

Figure 4.1 | Four steps proposed to systematically assess the environmental impacts of residual flow utilisation. The four steps are illustrated with an example of the climate impacts of utilising agricultural crop residues. Negative emissions (light grey) represent the avoided GHG emissions of replacing a counterfactual, i.e., emission savings. Black dots indicate overall GHG savings or emissions.

The fourth step is to determine the environmental benefits or burdens for each option, by taking the lifecycle impacts of the new (residue- based) product minus the impacts of its counterfactual, and to identify the environmentally optimal option. In our example, utilising agricultural crop residues to produce biofuels or bio-electricity resulted in similar net greenhouse gas (GHG) savings of approximately -3.6 × 10^2 kg CO_2-eq./dry tonne residue (the top two black dots in Figure 4.1; negative values indicate GHG savings). Note that without considering counterfactuals, the production of bio-electricity would have been strongly preferred, as the GHG emissions to produce electricity from agricultural crop residues are lower (-1.2 × 10^2 kg CO_2-eq/dry tonne residue) compared to the production of biofuels (-2.5 × 10^2 kg CO_2-eq/dry tonne residue; green bars in

Figure 4.1). Clearly, erroneous conclusions would have been drawn when using a functional unit based on the final product without considering what is replaced.

What exact counterfactual is chosen for each product can strongly influence results. In our example, when bioelectricity has a different counterfactual and replaces coal-based electricity, much larger GHG savings are achieved (-1.1 × 10^3 kg CO_2-eq/dry tonne residue; third row in Figure 4.1). What counterfactual is most realistic can be difficult to determine and changes with available technologies over time and location. Therefore, besides using a residue-utilisation based functional unit, it is essential to use case-specific and (where required) dynamic counterfactuals to accurately determine environmental impacts of residue utilisation.

A final consideration is how the absolute environmental impacts of residue utilisation should be determined. In our example, we determined impacts as compared to an absolute and hypothetical zero of not producing the residual flow in the first place. In practice it can also be informative to assess the impacts of residue utilisation against a benchmark of no residue utilisation (e.g., burning, flaring or decomposition). In our example, leaving agricultural crop residues on the field emits GHGs (61 kg CO_2-eq./dry tonne residue; fourth row in Figure 4.1). GHG savings of bio-electricity and biofuels would thus increase by this amount when compared to leaving residues on the field.

The combined approach outlined in this perspective allows for an intuitive and explicit evaluation of the system consequences of utilising a residual flow, which is imperative to draw comprehensive conclusions on its environmental impacts. We believe this systematic approach can provide a useful framework to maximise the environmental benefits of residual flow utilisation.

5

Biomass residues as twenty-first century bioenergy feedstock — a comparison of eight integrated assessment models

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ABSTRACT

In the 21st century, modern bioenergy could become one of the largest sources of energy, partially replacing fossil fuels and contributing to climate change mitigation. Agricultural and forestry biomass residues form an inexpensive bioenergy feedstock with low greenhouse gas (GHG) emissions, if harvested sustainably. We analysed quantities of biomass residues supplied for energy and their sensitivities in harmonised bioenergy demand scenarios across eight integrated assessment models (IAMs), and compared them to literature-estimated residue availability. IAM results vary substantially, at both global and regional scales, but suggest that residues could meet 7-50% of bioenergy demand towards 2050, and 2-30% towards 2100, in a scenario with 300 EJ/year of exogenous bioenergy demand towards 2100. When considering mean literature-estimated availability, residues could provide around 55 EJ/yr by 2050. Inter-model differences primarily arise from model structure, assumptions, and the representation of agriculture and forestry. Despite these differences, drivers of residues supplied and underlying cost dynamics are largely similar across models. Higher bioenergy demand or biomass prices increase the quantity of residues supplied for energy, though their effects level off as residues become depleted. GHG emission pricing and land protection can increase the costs of using land for lignocellulosic bioenergy crop cultivation, which increases residue use at the expense of lignocellulosic bioenergy crops. In most IAMs and scenarios, supplied residues in 2050 are within literature-estimated residue availability, but outliers and sustainability concerns warrant further exploration. We conclude that residues can cost-competitively play an important role in the 21st century bioenergy supply, though uncertainties remain concerning (regional) forestry and agricultural production and resulting residue supply potentials.

5.1 Introduction

Model-based projections show that modern bioenergy could become one of the largest sources of energy over the course of the 21st century, replacing fossil fuels and hence contributing to climate change mitigation (Clarke et al., 2014; IRENA, 2014; Rose et al., 2014; Smith et al., 2014; Creutzig et al., 2015; Van Vuuren et al., 2016; Bauer et al. 2018; Rogelj et al., 2018). At present, modern bioenergy provides about 24 EJ per year or 4.2% of the global primary energy supply (IEA, 2018b). This share may increase to 10-35% (75-245 EJ/yr) of the global primary energy supply by 2050 and to 10-50% (70-325 EJ/yr; with low agreement on 300+ EJ/yr) by 2100 (Chum et al. 2011; Rose et al., 2014; Smith et al., 2014; Creutzig et al., 2015; Van Vuuren et al., 2016; Bauer et al., 2018).

Two generations of modern bioenergy are distinguished. The first generation of bioenergy is based on food crops. Second-generation bioenergy feedstocks include lignocellulosic bioenergy crops (i.e., cultivated fast-growing grasses or trees), residues, and wastes (Antizar-Ladislao & Turrion-Gomez, 2008). Second-generation bioenergy is projected to supply a large share of future bioenergy use through advanced biofuels, electricity and heat (Rogner et al., 2012; Rose et al., 2014; Van Vuuren et al., 2016). Growing food or lignocellulosic crops for bioenergy can lead to competition for land with agriculture or natural areas, thus potentially threatening food security (Hasegawa et al., 2015) and biodiversity (Evans et al., 2015), and can increase net GHG emissions as a result of deforestation, foregone sequestration or fertiliser use (Elshout et al., 2014; Creutzig et al., 2015; Albanito et al., 2016; Daioglou et al., 2017). Agricultural and forestry residues on the other hand, are widely considered a promising and inexpensive bioenergy source (Carriquiry et al., 2011) with no or limited allocated land-use and therefore generally low climate change, biodiversity and other environmental impacts (Smith et al., 2014; Creutzig et al., 2015), if residue removal rates are low enough to sustain carbon stocks, soil fertility and other ecological functions (Raffa et al., 2015; Repo et al., 2015). Hence, residue use as a bioenergy feedstock is commonly encouraged (e.g., EU Directives 2009/28/EC and 2015/1513).

Agricultural residues include harvest and processing residues, while forestry residues include i.a., logging, thinning and processing residues (for an overview see Smith et al., 2014; Creutzig et al., 2015). Using a range of methodologies (explored in detail in section 5.3), the mean estimated residue availability[2] in 2050 for primary energy is 36 EJ/yr for agricultural residues (10-55 EJ/yr minimum-maximum range; excl. animal dung), 25 EJ/yr (5-50 EJ/yr) for forestry residues, and 61 EJ/yr (12-76 EJ/yr) combined (Fischer & Schrattenholzer, 2001; Hoogwijk et al., 2003; Smeets & Faaij, 2007; Smeets et al., 2007; Hakala et al., 2009; Gregg & Smith, 2010; Haberl et al., 2010, 2011; Cornelissen et al., 2012; Rogner et al., 2012; Lauri et al., 2014; Searle & Malins, 2015; Daioglou et al., 2015a). The future economic and ecological *availability* of residues as primary energy source, and its drivers and sensitivities have thus been extensively studied in previous work. However, it has not been explored what amount of residues can cost-competitively be *supplied* as primary bioenergy feedstock when competing against other bioenergy feedstocks to meet a given bioenergy demand. Nor what drivers determine the quantity of residues supplied for bioenergy and via what mechanisms. All of which are essential in understanding what role residues could have as an energy source, including in climate change mitigation pathways (Clarke et al., 2014; Rogelj et al., 2018).

In this study, we explore the quantity of biomass residues supplied (i.e., dispatched) for energy use, and their share in the total bioenergy supply over the course of the 21st century, in eight integrated assessment models (IAMs; see section 5.2). We compare model structure, assumptions and outcomes across the IAMs. Residues here constitute agricultural and forestry residues. Supply potential is subject to ecological and economic constraints. We use diagnostic scenarios with exogenous bioenergy demand or prices to analyse how the supplied quantity of residues and share of residues within total bioenergy supplied, depend on four drivers that could be directly or indirectly influenced by climate and energy policy: i) the demand for modern bioenergy, ii) pricing GHG emissions, iii) land protection efforts, and iv) the price of biomass. We also compare the quantity of residues supplied in IAMs with estimates of residue availability in literature, to evaluate if the role of residues as 21st century energy source in different IAM scenarios matches expected availability.

[2] The technical/available potential, i.e., accounting for ecological constraints (preserving soil quality, carbon storage, and biodiversity) and economic constraints (i.e., alternative uses of residues), see section 5.2.

5.2 Methods

Model selection

The IAM projections used in this study were developed within the context of the Energy Modelling Forum 33 Bioenergy Study (EMF-33). The EMF-33 study aims to understand, analyse and improve modelling of biomass supply and demand within IAMs. We analysed projections of all eight IAMs within EMF-33 that reported quantities of biomass residues supplied for primary energy, viz.: the AIM, BET, DNE21+, GCAM, GLOBIOM, GRAPE, IMAGE and NLU models (see table S1). The NLU and GLOBIOM models are not IAMs *sensu stricto*, but rather economic land-use models that focus on agriculture and forestry, respectively.

Model description

IAMs are designed to explore different future energy and land-use consumption and production patterns and their associated environmental impacts. For bioenergy, they describe both demand and supply. This study focuses on the supply side. We use scenarios in which either the demand for modern bioenergy or the price of biomass is exogenously set and harmonised across all the IAMs and determine the cost-optimal quantity of residues supplied. To meet an exogenous bioenergy demand, bioenergy feedstocks - including residues - compete with each other based on costs. At an exogenous biomass price, feedstock is supplied for bioenergy if the feedstock's costs are lower than the exogenous biomass price.

Key characteristics of the individual IAMs in terms of their representation and modelling of residue supply and residue costs are given in Table 5.1. All models include both agricultural and forestry residues, except GLOBIOM, which only models forestry residues for energy use. IMAGE and GRAPE also include municipal solid waste as residues. Besides residues, second-generation bioenergy feedstocks include lignocellulosic bioenergy crops in all models, as well as managed forests and plantations in the BET, GLOBIOM, and NLU models. Residue supply potential is determined endogenously in most IAMs, based on agricultural and forestry production, while it is exogenously set for DNE21+ and GRAPE. Bioenergy demand does not stimulate residue production as residues are considered by-products and not co-products in all studied IAMs, except GLOBIOM in which demand for residues can (co-)incentivise additional roundwood harvesting.

Table 5.1 | Key characteristics of the integrated assessment models included in this study.

	Residue types	Residue supply potential	Residue supply constraints[a]	Residue supply curve	Residue cost components	Conversion restrictions[b]
AIM	AR FR	endogenous; via input/output structure	ecological, economic; combined effect: 50% of total residues available	endogenous; via input/output structure	processing	electricity, biofuels
BET	AR FR MSW O	endogenous; via GLUE model[c] component	ecological, economic; accounted for via exogenous supply curve[d]	exogenous[d]	collection	electricity, biofuels, biogas
DNE21+	AR FR	exogenous[c]	ecological, economic; accounted for via economic potential in GLUE[c]	exogenous[c]	collection, transport, processing	electricity, biofuels, hydrogen, solids
GCAM	AR FR	endogenous; based on agricultural and forestry production	ecological; unspecified fraction remains on land	endogenous	collection, transport, processing	none
GLOBIOM	FR[e]	endogenous; based on forestry production	ecological: min. 50% of residues is left on the field economic: competition with fibre	endogenous	collection, transport [f]	none
GRAPE	AR FR	exogenous[g]	ecological, economic; accounted for via exogenous supply potential[g,h]	exogenous[g]	collection, transport, processing	electricity, heat
IMAGE	AR FR MSW	endogenous[i]; based on agricultural production and timber demand	ecological: 30% of residues is left on the field[j] economic: feed and traditional bioenergy first	endogenous[i]	collection, transport, processing	electricity, heat, hydrogen
NLU	AR FR	AR: endogenous based on agricultural production FR: exogenous[k]	ecological, economic; combined effect: max. 30% of AR and 50% of FR available for energy[l]	none; available residues used for bioenergy[l]	transport	none

Abbreviations: AR: agricultural residues (incl. processing/secondary residues; excl. animal dung); FR: forestry residues (incl. processing/secondary residues); MSW: municipal solid waste; O: other (kitchen refuse, sewage sludge); **Notes**: **a** ecological constraints (i.e., leave residues on land to maintain soil fertility, stability and/or carbon stocks) and economic constraints (i.e., alternative non-energy uses of residues); **b** restrictions that limit energy-use of residues to certain sectors; **c** based on Yamamoto et al. (2001); **d** based on exogenous supply costs derived from Daioglou et al., 2015a; **e** GLOBIOM includes beyond harvesting and processing residues: recycled wood, stump removal, and additional roundwood extraction for bioenergy. **f** harvest costs 5-40US$/m3 based on G4M, transport costs via price elasticity function; **g** based on exogenous supply costs derived from Rogner et al., 2012; **h** residues supplied for modern bioenergy further constrained by competition with residue use for traditional bioenergy; i residue supply potential and supply curves are endogenous for forestry and agricultural residues in IMAGE, but are exogenous for MSW, see c; **j** mass constraint per hectare, which globally aggregates to 30% of residues left on the field; **k** Smeets et al., 2007; **l** in NLU agricultural residues are first used to meet feed demand, of the remaining residues 40% stays on the field, 30% goes to pulp and construction materials, and 30% is (always) used as bioenergy feedstock. In NLU 50% of forestry residues is used for pulp and construction material and 50% is (always) used as bioenergy feedstock. Residue costs are determined endogenously.

In all models supply potential is constrained by ecological constraints (i.e., requirements to leave residues on agricultural land or forestland to maintain soil fertility and/or carbon stocks, and/or to prevent erosion), as well as economic constraints (i.e., alternative residue uses, for non-energy purposes; for details per model see Table 5.1). Residue costs are based on collection, transport and/or processing and are related to the supply of different bioenergy feedstocks, through supply curves (except in NLU, see section 5.3). These residue supply curves set what quantity of residues can be supplied at what costs. Together with the price of biomass used for energy they determine what quantity of the available residue potential is supplied for bioenergy. Most models have endogenous residue supply curves that are determined as part of the model run. However, BET, DNE21+ and GRAPE have fixed, exogenous supply curves derived from literature. All models assume residues cause no GHG emissions other than supply chain emissions, because biogenic carbon emissions are considered GHG neutral, residue extraction is ecologically constrained, and no direct land-use change emissions are allocated to residues.

Scenario selection & model comparison

The studied IAMs were all run according to EMF-33 scenarios; for an overview of all EMF-33 biomass supply scenarios, specification details, and rationale see Bauer et al. (2018). The scenario subset used here includes: i) scenarios with an exogenous demand for second-generation primary bioenergy that increases linearly from the modelled demand in 2010 to 100, 200, 300 or 400 EJ/yr by 2100 (scenario B100/200/300/400; see Table S2), either with or without GHG pricing, and either with or without land protection; and ii) scenarios with an exogenous fixed biomass price of 3, 5, 9 or 15 $US\$_{2005}$ /GJ at farm gate/roadside (scenario PB3/5/9/19; see Table S2), either with or without GHG pricing. GHG pricing means that emitting GHGs has a price of 20 $US\$_{2005}$/tonne CO_2-eq. in 2020 with a 3% annual increase. This GHG price is applied to all major GHGs (CO_2, CH_4 and N_2O) including all land-use related GHG emissions. Land protection means that on top of default model constraints on land availability (e.g., current natural protected areas), further areas are to remain in or transform to a natural state and are not available for human land uses such as agriculture. All scenarios are based on reference socioeconomic assumptions, i.e., socio-economic and technological parameterisation of the scenarios is based on SSP2 (Popp et al., 2017), and are run from 2005 to 2100.

The purpose of these diagnostic scenarios is not to make definite projections of future residue use. Rather, these scenarios allowed us to compare the supplied quantity of residues across models when “forced” to supply bioenergy under exogenous bioenergy demand or biomass prices. These scenarios help to determine what role residues could play in meeting total bioenergy demand, alongside purpose-grown feedstocks like lignocellulosic bioenergy crops, and to show what dynamics underlie the quantity of residues supplied in the different models. The scenarios also allow assessing the sensitivity of the quantity of residues supplied to four main drivers: i) bioenergy demand, ii) GHG emission pricing, iii) land protection, and iv) biomass prices. We analysed absolute quantity of residues supplied and the share that residues form in the total amount of second-generation bioenergy supplied.

Additionally, we used variance decomposition analysis to provide an indication of how individual drivers contribute to the modelled quantity of residues supplied. Taking the quantity of residues supplied across scenarios as dependent variable, we performed an ANOVA to derive the sum of squares (SSQ) for the factors Bioenergy Demand and GHG Pricing, and for the residuals, which represent model variability between the included IAMs. The SSQs of both factors and the residuals were divided by the total SSQ, yielding the variance attributable to these factors and residuals. Log-transformed supplied residue values, or logit-transformed shares of residues in the total bioenergy supply, were used to minimise the influence of outliers.

Literature analysis of future residue availability

We compared the *quantity of residues supplied* in IAMs to the *expected availability of residues* estimated in literature, to determine if the role of residues in IAM scenarios fits within the expected availability of residues. We define residue availability here as the technical potential (IPCC terminology; Chum et al. 2011) or equivalently the available potential (Daioglou et al. 2015a) of residues, which accounts for ecological constraints (i.e., preserving soil quality, carbon storage, and sometimes biodiversity) and economic constraints (i.e., alternative uses of residues) on residue supply. Our literature analysis includes all peer-reviewed studies published since 2000 that estimate the global available/technical potential of forestry and/or agricultural residues over the course of the 21st century, and specifically in 2050. We consider the default available/technical potential reported in these studies. If no default is defined, we use the mean of reported values. We look at the minimum-maximum range per study, based on the lowest and highest reported estimates of residue availability in 2050 across sensitivity tests and scenarios.

We distinguish two types of studies. First, studies with a bottom-up approach that directly estimate residue availability from expected trends in population size, diet and consumption patterns, and ultimately agricultural and/or forestry production. And second, studies with top-down macro-economic drivers that estimate residue availability based on macro-economic, IAM, or IAM-component model results. While some of these latter estimates are based on the same or similar models that were used in this study, it is relevant to compare our diagnostic scenario-based results against their *projected* residue supply. This comparison also serves as a further plausibility check.

5.3 Results

The importance of residues as bioenergy feedstock

Figure 5.1a shows the quantity of residues supplied in two scenarios: i) a scenario with an exogenous demand for second-generation primary bioenergy that increases linearly from 2010 levels to 300 EJ/yr by 2100, and ii) the same scenario including a price on emissions (scenarios B300 and B300C in Table S2, respectively). Figure 5.1b presents the share of residues as part of total supplied second-generation bioenergy for these same scenarios. Analogous figures with exogenous bioenergy demands of 100, 200 and 400 EJ/yr can be found in the supplementary materials (Figure S1-S3). Both the quantity of residues supplied and the share of second-generation bioenergy covered by residues vary widely across the studied IAMs at a given exogenous bioenergy demand level. In the 300 EJ/yr demand scenario for example, the quantity of residues supplied in 2100 ranges from 7 to 91 EJ without GHG pricing, and up to 151 EJ with GHG pricing. GHG pricing effects are limited in most models, as detailed below.

Inter-model consensus is highest among IAMs with endogenous supply curves (i.e., AIM, GCAM, GLOBIOM, IMAGE) and the NLU model with 25-90 EJ supplied by 2100, covering 10-30% of bioenergy demand (Figure 5.1). Meanwhile, the BET and GRAPE models have exogenously derived supply curves based on low exogenous residue costs, and show the largest amounts of residues supplied for energy. DNE21+ on the other hand, has an exogenous supply curve based on higher costs and projects the lowest quantity supplied. DNE21+ and GRAPE also have exogenous residue supply potentials, which may further add to the more extreme outcomes of these models.

Beside model structure, two other sets of factors add to the observed variation in model outcomes. First, in models with endogenous residue supply potential (all models, except DNE21+ and GRAPE, see Table 5.1), agricultural and forestry production affect residue supply potential. We find that agricultural production varies around 20% across models (variation is reported here as the maximum percentage above and below inter-model mean in 2100; Figure S4). The variation in agricultural production is caused by 50% inter-model variation in livestock produced, 15% variation in food demand, and 25% variation in food crop yields (Figure S5), as well as by variation in the type of food crops produced. Forestry production and associated residue production even varies by a factor of ten among the models, with IMAGE on the low end, and GLOBIOM and BET on the higher end (Figure S4). However, while crop yields, diet, agricultural production and forestry production vary across models, and while these variables affect the residue supply potential in IAMs with endogenous supply potential, they are not consistently related to the quantity of residues supplied in these IAMs.

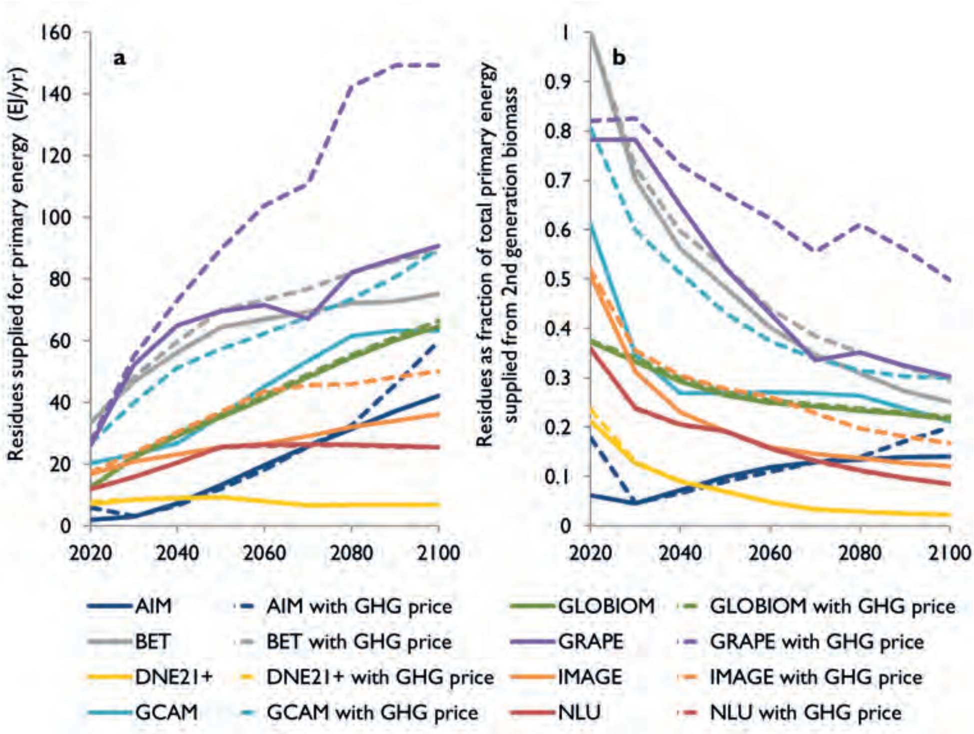

Figure 5.1 | Residues supplied for primary energy under exogenous demand for second-generation bioenergy. a. Quantity of residue supplied for primary energy (EJ/year) at an exogenous demand for second-generation bioenergy that increases linearly from 2010 levels to 300 EJ/yr by 2100, with and without GHG pricing. **b**. Residues as share of total second-generation biomass use for primary energy under the same scenarios. Dotted lines may underlie their respective solid line.

Second, definitions of residues, constraints and costs vary between models (Table 5.1). GLOBIOM excludes agricultural residues, but reports outcomes that are in the middle of the inter-model range. GRAPE and IMAGE include municipal solid waste (MSW) as residues, which add about 10% to the supply potential in GRAPE, but a smaller amount in IMAGE. Ecological and economic constraints are present in all models, but vary, for instance concerning the percentage of residues that should remain on the field, or the competing alternative uses of residues. The IAMs also vary with regard to the types of residue costs they include, i.e., collection, processing, and/or transport costs, and what economic sectors use residues for energy. While adding variation, these effects do not show a consistent effect across models on the quantity of residues supplied.

Despite variability between IAMs and across different exogenous demand levels, IAM outcomes show that residues generally form an important bioenergy feedstock, meeting 7-50% of bioenergy demand towards 2050, and 2-30% towards 2100, in the 300 EJ/year in 2100 scenario (B300). The absolute quantity of residues supplied grows over time, mostly driven by increasing exogenous bioenergy demand over time, and eventually levels off. The share of residues in the total amount of second-generation bioenergy supplied nevertheless decreases over time as residue supply cannot keep up with increasing bioenergy demand. Remaining demand is met by lignocellulosic bioenergy crops and managed forest.

Model drivers of residue supply

Figure 5.2 shows the global quantity of residues supplied in IAM projections in the year 2050, across four scenarios with increasing exogenous second-generation *bioenergy demand*. A higher exogenous bioenergy demand leads to a larger quantity of residues supplied (i.e., dispatched for energy) in most models[3] (Figure 5.2a). Bioenergy demand does not, however, directly stimulate the production of residues. Rather, a higher bioenergy demand increases bioenergy prices, which leads to more residues being taken off the field and/or more residues being diverted from other sectors towards bioenergy, leading to increased quantities of residues supplied for energy.

Nevertheless, the share of residues in second-generation bioenergy *decreases* with bioenergy demand (Figure 5.2b). While residues are a relatively cheap feedstock and thus form a large share of total bioenergy at low demand, residue supply is more constrained than that of other feedstocks such as lignocellulosic bioenergy crops and does not keep up with demand. Constraints include the total volume of

[3] All models except NLU, which does not contain a residue supply curve and in which all residues available for energy are always used, and DNE21+, in which the amount of residues available for energy is limited and already depleted at low demand.

residues, which does not increase with bioenergy demand, as well as the amount of residues that can be diverted from the field (ecological constraints) and from other sectors (economic constraints; see Table 5.1). These patterns observed for 2050 are the same in other years (Figure S6).

In BET, GCAM, GRAPE and IMAGE *pricing GHG emissions* (20 $US\$_{2005}/tCO_2$-eq. in 2020 plus a 3% annual increase) increases both the quantity of residues supplied and the share of residues in the bioenergy mix for the exogenous demand scenarios (Figures 5.1, 5.2, S1-3). The reason being that in the IAMs residues lead to no or low (allocated) GHG emissions, as compared to other bioenergy feedstocks like lignocellulosic bioenergy crops or wood from managed forests, which can require land-use change and lead to larger supply chain/lifecycle emissions. Therefore, GHG pricing does not increase residue costs much, but it does increase the costs of other bioenergy feedstocks. This makes residues a more favourable feedstock and incentivises taking residues off the field or diverting them from other sectors, thus expanding the energy-use of residues. These dynamics are, however, limited or absent in AIM, DNE21+ GLOBIOM and NLU.

Our *variance decomposition analysis* shows that the majority of variation in the quantity of residues supplied across IAMs and scenarios is attributable to differences in IAMs (82-93% variance explained; Table S4). Sources of this variation across IAMs were presented above and are further discussed in section 5.4. Substantially less variation is, however, attributable to the scenario components of exogenous bioenergy demand (5-16%) and GHG pricing (0-3%; Table S4). The ranges shown include analyses throughout the 21st century, for both the absolute quantity supplied and the share of residues in total bioenergy supplied (see Table S4 for examples for the years 2050 and 2100). When DNE21+ and NLU, which do not respond to exogenous bioenergy demand or GHG pricing, are excluded from this analysis, a larger part of the variation in residues supplied is explained by bioenergy demand (12-25%) and GHG pricing (0-12%), though inter-model differences still account for the majority of variation (63-88%).

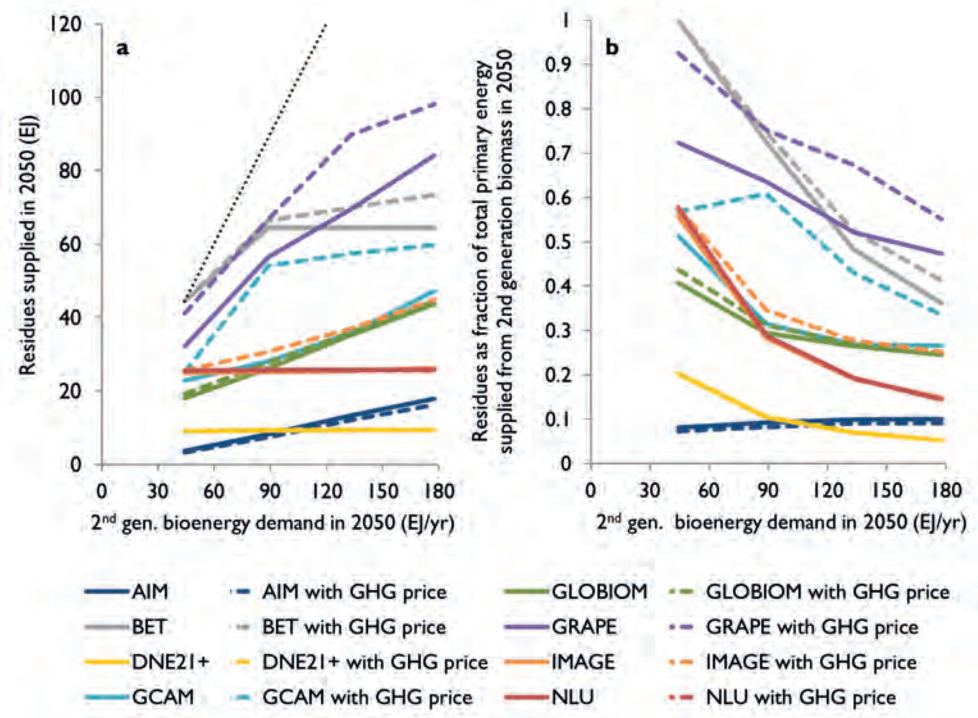

Figure 5.2 | Residues supplied for bioenergy in 2050 at different levels of exogenous bioenergy demand. a. Quantity of residues supplied in the studied IAMs for the year 2050, across four scenarios with increasing exogenous bioenergy demand (to 100, 200, 300 and 400 EJ/yr by 2100), with and without GHG pricing. The black dotted line indicates residues meeting 100% of exogenous bioenergy demand. **b**. Residues as share of total second-generation biomass use for primary energy across the same scenarios in 2050.

Land protection, which is only modelled in IMAGE and GCAM, excludes economic activity from certain areas, making remaining land more expensive. This disproportionally increases the costs of land-intensive lignocellulosic bioenergy crops and increases overall bioenergy prices. Meanwhile, residue costs are *less* affected and residues thus become the more cost-optimal feedstock. The supplied quantity of residues to meet a given bioenergy demand therefore increases (Figure S7), as more residues are taken off the field or diverted from other sectors, incentivised by the increased bioenergy prices. This increase in energy-use of residues is co-facilitated by higher yields and residue production on the scarcer and therefore more intensively managed agricultural/forestland. Ultimately, the effect of land protection on residue supply is similar and complementary to that of GHG emission pricing, until available residue supply levels off (Figure S7).

The effect of the *price of biomass* on the quantity of residues supplied is simulated in the GLOBIOM and IMAGE models using exogenous biomass price scenarios. In these models, the quantity of residues supplied is a consequence of complex relationships that beside the biomass price, include residue costs, competition with other feedstocks and food and timber market dynamics. Higher prices of second-generation biomass lead to larger quantities of residues supplied for energy (Figure S8), as there is incentive to take more residues off the field or divert them from non-energy sectors. This happens independently from the supplied quantity of lignocellulosic bioenergy crops, which also increases. The increase in residues supplied for bioenergy levels off at higher prices, as the maximum residue supply is reached under ecological and economic constraints (Table 5.1). These dynamics are hardly influenced by GHG pricing and land protection (Figure S8).

Regional differences in the quantity of residues supplied

Figure 5.3 shows the absolute (Figure 5.3a) and relative (Figure 5.3b) quantities of residues supplied per region across the studied IAMs for 2050 in the 300 EJ/yr exogenous bioenergy demand scenario. In most models, Asia supplies most residues for energy (24-60%), followed by the OECD90 countries, which form the largest supplier in NLU and AIM, and Africa (Figure 3b; for region definitions see Table S3). There are however, large inter-model differences, for instance the exact share of Asian supply, or the large role of African supply in BET and South American supply in GLOBIOM, GRAPE and NLU. These differences are even larger in absolute terms (Figure 5.3a), with for example Asian-supplied residues in BET and GRAPE equalling 80-320% of *global* supplied quantities in the other models. These patterns per model stay approximately the same when including GHG pricing, or considering different years or levels of exogenous bioenergy demand (Figure S9-11).

The large disagreement among IAMs on quantities of residues supplied per region can be explained by: i) differences in model structure and assumptions (see also Table 5.1), ii) large inter-model differences in regional agricultural and forestry production (Figure S12), and iii) residue definition, specifically: GLOBIOM only includes forestry residues.

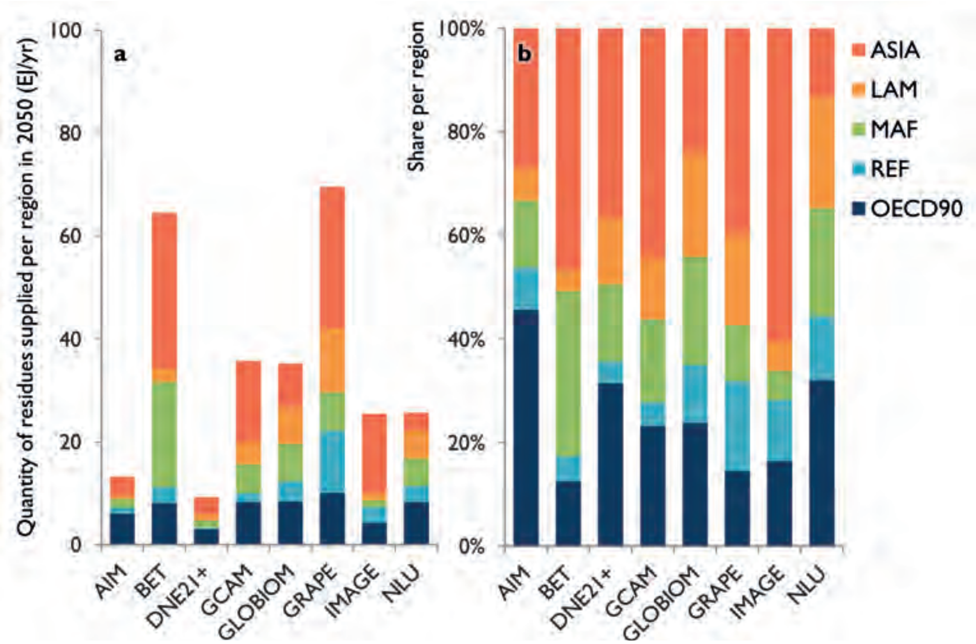

Figure 5.3 | Residues supplied for bioenergy per region for the year 2050 at 300 EJ/yr exogenous bioenergy demand. a. Quantity of biomass residues supplied for energy per region in 2050 in the scenario with an exogenous primary bioenergy demand of 300 EJ/yr by 2100. **b**. The share of residues supplied for energy per region in 2050 in the same scenario. Abbreviations: LAM= Latin America, MAF= Middle East and Africa, REF = reforming economies (former Soviet Union and Eastern Europe), OECD90 = OECD member countries in 1990; for regional definitions see Table S2.

Residues supplied in IAMs versus residue availability in literature

Figure 5.4 shows a comparison of the modelled quantity of residues *supplied* in 2050 in all studied IAMs and exogenous bioenergy demand scenarios, against the expected residue *availability* in 2050, as estimated in literature. This comparison serves to determine if the role of residues in IAM scenarios fits within expected residue availability. Comparison against bottom-up estimates of availability is especially useful here, since these estimates are based on a different approach, independent of top-down or IAM modelling effects. Comparison against top-down macro-economic/IAM modelled residue availability serves as a further plausibility check. While both IAMs and the literature estimates include ecological and economic constraints on residue supply potential, it is important to note that IAMs determine the cost-optimal, "competitive" quantity of residues supplied, while literature estimates consider total residue availability, under the constraints set. Supplied quantities can therefore certainly be lower than availability, but higher projections indicate that such residue use is infeasible.

Estimates of residue availability in bottom-up studies range 12-76 EJ/yr in 2050, with a mean of 55 EJ/yr, which is determined as the sum of mean agricultural and mean forestry residue availability. The wide range in residue availability can be explained by different methodologies, as well as differences in the definition of economic and ecological constraints. Early work by Hoogwijk et al. (2003) indicated a residue availability of around 34 EJ/yr. Smeets & Faaij (2007), Smeets et al. (2007), Hakala et al. (2009) and Haberl et al. (2010), whose results were also part of the literature assessment by Rogner et al. (2012), reported larger availability of both agricultural and forestry residues. These studies look at the maximum realistically possible availability of residues for energy. In contrast, Searle & Malins (2015), and to a substantially lesser extent Cornelissen et al., (2012), include stricter sustainability constraints (i.e., no residue extraction from natural forests, 70% of agricultural residues unavailable) and estimate residue availability in 2050 to be 12 EJ/yr - 80% below this study's literature average.

Studies based on top-down macro-economic modelling and IAMs report high residue availability estimates, with a mean of 68 EJ/yr in 2050. However, there is large variation in this group of studies as well. Gregg & Smith (2010) report the largest residue availability, which can be explained by the fact that they estimate ecological potential only and do not include economic constraints (i.e., alternative uses of residues). Yamamoto et al. (2001) also report relatively large residue availability, but include animal dung, which is excluded in other studies. Excluding these two studies lowers the mean of residue availability estimates in top-down studies to 56 EJ/year in 2050, similar to the 55 EJ/yr mean across bottom-up studies.

The quantity of residues supplied in the studied IAMs varies between and within scenarios, due to model assumptions and structure, but also scenario-specific differences in bioenergy demand, biomass price, and GHG pricing. In most cases, the quantity of residues supplied in 2050 in IAMs is lower than the literature-estimated availability. This means that the relatively large role that the IAMs attribute to residues in meeting the potentially large future bioenergy demand, generally seems possible based on our current understanding of future residue availability (including ecological and economic constraints).

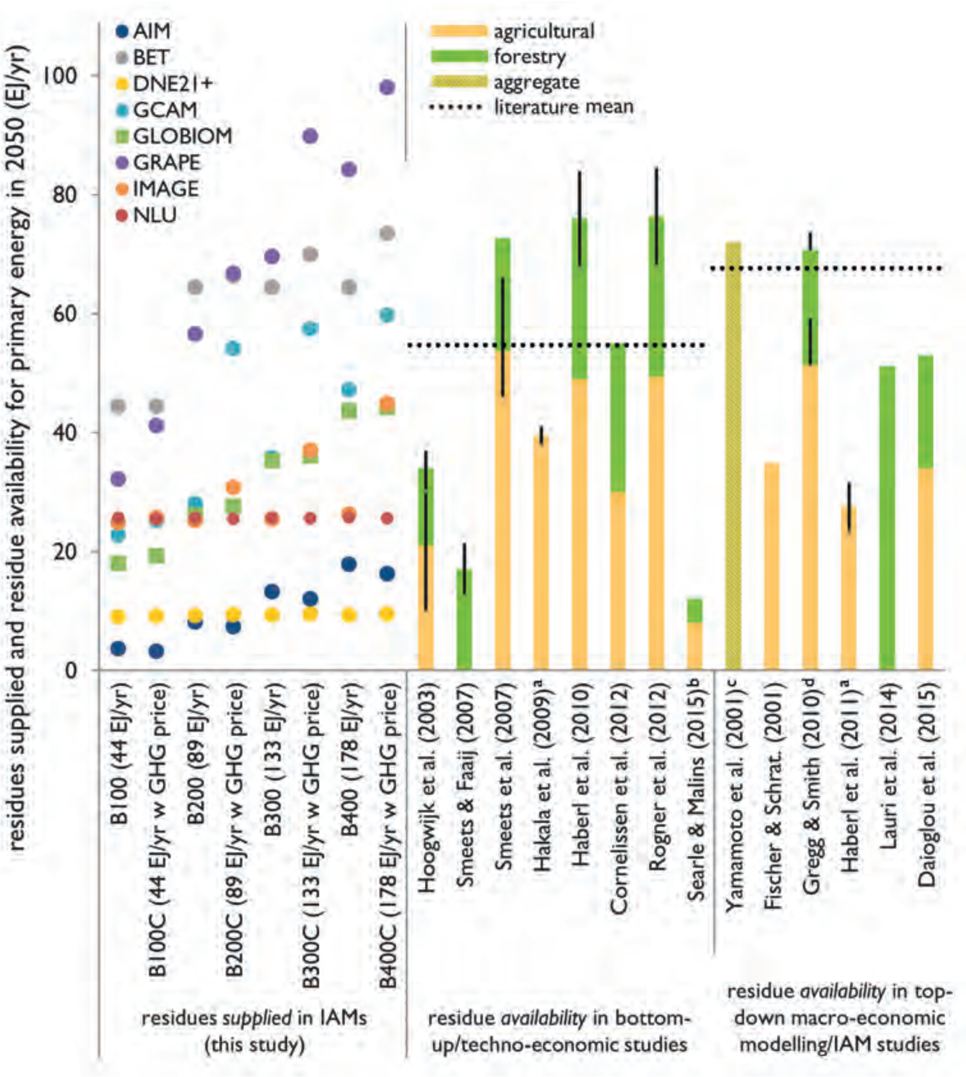

Figure 5.4 | Comparison of the quantity of residues *supplied* for energy in 2050 in IAMS versus expected residue *availability* for 2050 in literature. GLOBIOM projections (in green *squares*) only include forestry residues. Literature means are calculated as the mean availability of agricultural residues plus the mean availability of forestry residues (both including processing/secondary residues). Error bars indicate minimum and maximum values where provided in literature. Notes: a. excludes processing residues; b. very strict sustainability criteria; c. includes animal dung; d. only subject to ecological constraints (i.e., no economic constraints).

5.4 Discussion

Model interpretation

We found that the quantity of residues supplied for bioenergy in the studied IAMs varies substantially, but meets anywhere from several percent to up to half of total second-generation bioenergy demand by 2050, and up to around 30% by 2100. As future bioenergy use is expected to be significant (Chum et al., 2011; Bruckner et al. 2014; Clarke et al., 2014; IRENA, 2014; Rose et al., 2014; Smith et al., 2014; Creutzig et al., 2015; Van Vuuren et al., 2016), biomass residues may play a large role in the 21st century energy supply. In terms of drivers of residue use, we found that a higher bioenergy demand or biomass price increases the quantity of residues supplied, though their effects level off at higher demand or prices. GHG pricing and land protection increase the costs of land, which in most models leads to increased residue use, at the expense of lignocellulosic bioenergy crops. These patterns and drivers of residue supply were similar across models and are well-understood, as IAMs allowed for explicit analysis of the cost dynamics that underlie them. Specific IAM results, however, differed markedly - with model differences explaining 82-93% of the variation in results, as discussed below. Lastly, we found that in most IAMs and scenarios, supplied residue quantities in 2050 were found to be within literature estimates of residue availability. The large role IAMs attribute to residues in meeting bioenergy demand thus seems plausible. The feasibility of such large-scale residue use is discussed below.

Variability in IAM results

As summarised by our variance decomposition analysis, IAM outcomes varied significantly within scenarios at the global *and* regional scale, despite a shared storyline and scenario assumptions. Several factors contribute to the observed differences. First, models with exogenous residue supply potential and supply curves showed more extreme outcomes in the quantity of residues that is supplied for bioenergy. Models with endogenous supply potential and curves better captured residue supply dynamics and lead to more similar results in this study.

Second, structural inter-model differences in agricultural and forestry production and the assumed allocation of production over different end-uses affected the supplied quantity of residues, and along with residue costs (see below) affected the calibration of supply curves. Crop yields, diets, agricultural production and forestry production were, however, not correlated with the quantity of residues supplied across models. Previous work (Daioglou et al., 2015a) showed that these variables may have counteracting effects. Modelling may be improved here by adding scenarios with harmonised assumptions on food and forestry product (i.e., timber and fibre) demand to deduce their influence. The dynamic relationship between agricultural/forestry production and residue supply, which is often assumed to be linear, could also be modelled in more detail, e.g. through crop-specific relationships. Furthermore, the role of residues in the total amount of bioenergy supplied can be investigated in sensitivity scenarios with different crop yields and diets, which are of key importance to the amount of land available for bioenergy crops (Gerssen-Gondelach et al., 2015; Stehfest et al., 2009).

Third, residue definitions and the assumed level and nature of ecological and economic constraints varied among models. As did residue costs, since residues form a diverse feedstock with a wide range of costs that depends strongly on local circumstances. These differences in constraints and costs contribute to inter-model variation, but do not show a consistent effect on residue supply dynamics. More detailed and harmonised constraints on residue supply[4] and detailed cost components would advance our understanding of residue supply dynamics and the range of IAM outcomes.

The feasibility of large-scale residue use

Quantities of residues supplied in IAMs that lie within estimated availability may be possible, but more general aspects of their feasibility still need addressing. First, IAMs assume that all residues -subject to ecological and economic constraints - are usable and substitutable, regardless of the exact type or origin of the residue. Quality and logistical constraints may, however, reduce the quantity of residues that can be used in reality or increase transport and processing costs. If transport distances to processing locations are for instance too large, high transport costs may render residue use infeasible, while transport emissions could make residue use undesirable (Portugal-Pereira et al., 2015).

[4] For example more alternative uses of residues that compete with utilisation for bioenergy, or including the effects of residue removal on soil fertility and carbon stocks.

Second, biomass residues have low or no additional land requirements and associated GHG emissions or competition with food (Smith et al., 2014; Creutzig et al., 2015). Nevertheless, for residues to be a truly sustainable feedstock it is critical that (enhanced) residue extraction does not lead to erosion or losses in soil fertility, biodiversity or carbon stocks (Lal, 2005; Janowiak & Webster, 2010; Lemke et al. 2010; Bouget et al., 2012; Lamers et al. 2013; Liska et al. 2014; Raffa et al., 2015; Poeplau et al. 2015; Repo et al. 2015). IAMs, as well as most studies on residue availability, include this ecological constraint via an unavailable residue fraction that is left on-site for ecological functions. The required size of this fraction has been investigated (Daioglou et al., 2015a), but is dependent on local circumstances and requires additional understanding. The unavailable fraction currently used in IAMs or residue availability estimations may thus be insufficient to guarantee sustainability and lead to overestimation of sustainable residue supply potential (Searle & Malins, 2015).

Several potential effects and implications of large-scale residue use for bioenergy also require further research. First, the life-cycle environmental impacts of collecting, processing, transporting and using biomass residues for energy may be significant. Biogenic carbon emissions from bioenergy, which are considered GHG neutral in IAMs, should be included here, for instance using time-integrated metrics (Cherubini et al, 2011b, 2016). Furthermore, it would be interesting to analyse the environmental consequences of using residues for bioenergy rather than for other potential purposes, including feed, fibre, construction materials and bio-char, or letting them decompose (as studied for forestry residues by Repo et al., 2012, Gustavsson et al. 2015, and Hanssen et al. 2017). Similarly, the economic consequences of taking residues off the field or diverting them from other sectors towards bioenergy require further exploration. Utilising available residues may increase agricultural/forestry profitability (e.g., Smeets et al., 2015) and production. Diverting residues from other sectors may come with significant opportunity costs (Carriquiry et al. 2011). From a more theoretical perspective, increased utilisation and valorisation of residues could justify shifting part of the environmental burden of agricultural and forestry products (i.e., food or timber) towards agricultural and forestry residues, for instance through economic allocation.

5.5 Conclusions

We conclude the following:

- Based on the results of eight IAMs, this study shows that residues might cost-competitively play a large role in the 21st century bioenergy supply. At high bioenergy demand, which was exogenously forced in this study, residues could meet 7-50% of bioenergy demand towards 2050, and 2-30% towards 2100. When also considering (mean) literature-estimated residue availability, residues could provide around 55 EJ/yr by 2050.

- IAM results vary widely at the global scale, and especially the regional scale. Inter-model variation arises mainly from: i) model structure, where endogenous supply potential and curves better capture residue supply dynamics, ii) modelling of agricultural and forestry production, which can be further harmonised to match scenario storylines, iii) definitions of residues, and iv) residue supply constraints and residue cost components, which can be modelled in more detail.

- Despite inter-model variation, the patterns and drivers of residue supply and underlying cost dynamics are similar across IAMs. Residues supply the majority of bioenergy at low bioenergy demand. With higher demand or biomass prices, the quantity of residues supplied for energy increases. However, as available residues are depleted, the share of residues in total bioenergy still decreases.

- In the studied IAMs, GHG emission pricing and land protection can increase the costs of using land for lignocellulosic bioenergy crop cultivation, leading to a disproportional increase in the costs of (land-intensive) lignocellulosic bioenergy crops and therefore increased residue use and a larger share of bioenergy being covered by residues.

- The important role of residues in IAM projections of bioenergy use largely fits within current estimates of residue availability. However, logistic and sustainability *constraints*, as well as economic and environmental *implications* of large-scale residue use for bioenergy need to be addressed in future research.

5.6 Acknowledgements

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The climate change mitigation potential of bioenergy with carbon capture and storage

6

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ABSTRACT

Bioenergy with carbon capture and storage (BECCS) can act as a negative emission technology and is considered crucial in many climate change mitigation pathways that limit global warming to 1.5-2°C. The negative emission potential of BECCS has, however, not been rigorously assessed. Here, we perform a global spatially-explicit analysis of life-cycle GHG emissions for lignocellulosic crop-based BECCS. We show that negative emissions strongly depend on biomass cultivation location, treatment of original vegetation, final energy carrier produced, and evaluation period considered. We find a global potential of 28 EJ/year for electricity with negative emissions, sequestering 2.5 Gtonne CO_2/year when accounting emissions over 30 years, which increases to 220 EJ/year and 40 Gtonne CO_2/year over 80 years. We show that BECCS sequestration projected in IPCC SR1.5°C pathways can biophysically be approached, but considering its potentially very large land requirements, recommend substantially lower and earlier deployment of BECCS.

6.1 Introduction

Most climate change mitigations pathways that limit global warming to 1.5°C or 2°C rely on negative emission technologies (NETs), in particular bioenergy with carbon capture and storage (BECCS) (Azar et al., 2013; Tavoni and Socolow, 2013; Clarke et al., 2014; Fuss et al., 2016; Smith et al., 2016; van Vuuren et al., 2018; Rogelj et al., 2018). BECCS has the benefit of combining energy generation based on existing technologies with the geological storage of sequestered atmospheric carbon (Obersteiner et al., 2001; Gough and Upham, 2011; Kemper, 2015).

Concerns have, however, been raised on the biophysical feasibility, environmental effects and biodiversity impacts of large-scale BECCS deployment, stemming from its intensive land, water and nutrient-use (Smith et al., 2016; Kemper, 2015; Bonsch et al., 2016; Fajardy et al., 2018; Heck et al., 2018; Stoy et al., 2018; Kato and Yamagata, 2014). Moreover, BECCS cost estimates vary widely (Smith et al., 2016; Fuss et al., 2018) and BECCS implementation may prove to be socio-politically difficult (Fridahl and Lehtveer, 2018), among others due to the challenge of accounting and rewarding negative emissions (Torvanger, 2019; Bednar et al., 2019; Daggash & Mac Dowell, 2019).

Given that BECCS is considered a crucial technology in many mitigation pathways, but also has major drawbacks, it is essential to assess its effectiveness as climate change mitigation strategy. Two previous studies report that BECCS electricity can result in both net negative and positive greenhouse gas (GHG) emissions, mainly depending on the required land-use change and the efficiency of the bioenergy supply chain (Fajardy & Mac Dowell, 2017; Harper et al., 2018). Earlier work stresses that the climate change mitigation potential of bioenergy is highly dependent on biomass cultivation location and conversion technology (Harper et al., 2018; Elshout et al., 2015; Daioglou et al., 2017), and that bioenergy crop yields may not suffice to achieve ambitious carbon sequestration targets via BECCS (Kato and Yamagata, 2014). However, spatially-explicit GHG emissions for bioelectricity and liquid biofuels with CCS have not been estimated yet, despite being essential in evaluating the contribution of BECCS in mitigation pathways.

Emission factors (EF) express the amount of GHG emissions per unit bioenergy produced. Here, we quantified spatially-explicit EFs and determined the global potential supply of BECCS at increasing EF levels, producing so-called emission-supply curves. EFs and supply potentials were calculated using the global vegetation model LPJml combined with full life-cycle GHG emission data. The EFs include emissions from land-use change (LUC), the lost carbon sequestration capacity of natural vegetation ('foregone sequestration'), bioenergy supply chain emissions including fertilisers, and CO_2 sequestered through CCS, over a set evaluation time. Agricultural areas (cropland and pastures), including projected additional land requirements, are excluded from our analysis, as employing them could lead to indirect land-use change (iLUC) effects (Searchinger et al., 2008; Gerssen-Gondelach et al., 2017) or threaten food security (Hasegawa et al., 2018; Doelman et al., 2018; Fujimori et al., 2019). We assessed both bioelectricity and liquid biofuels (Fischer-Tropsch diesel and bioethanol) produced with CCS, and considered lignocellulosic biomass from fast-growing grasses (*Miscanthus* and switchgrass) and woody bioenergy crops (short-rotation poplar, willow and *Eucalyptus*), as well as sugarcane (for bioethanol only), with all crops being rainfed. We used a 30 year evaluation time, reflecting typical plantation lifetimes and short to medium-term mitigation without carbon budget overshoot, as well as an 80 year evaluation time, corresponding with mitigation pathways towards 2100. Biomass present before plantation establishment ('initial biomass') was assumed to be burned, consistent with previous analyses (Elshout et al., 2015; Daioglou et al., 2017; Creutzig et al., 2015), but we also quantified EFs and energy supply potential under the assumption that initial biomass is used to produce bioenergy or biomaterials. Our emission-supply curves provide new insights into the amount of BECCS energy that can be produced with negative emissions or at EFs below those of alternative energy generation, allowing evaluation of BECCS' climate change mitigation potential.

6.2 Bioelectricity

For a 30 year evaluation period, the global lignocellulosic crop-based BECCS electricity potential with negative emissions is 28 EJ_{elec}/year (Figure 6.1a), which equals around 32% of the current global electricity production (IEA, 2018c) and would entail net sequestration of 2.5 Gtonnes of CO_2-eq./year (Table S5), based on a 90% carbon capture rate (Table S1). At EFs above zero, BECCS electricity does not result in net negative emissions, but GHG emissions would be reduced when replacing electricity generation technologies with higher EFs. BECCS electricity typically achieves lower EFs on agricultural lands that are abandoned or are projected to be abandoned ('abandoned lands'), but electricity supply potential with negative EFs on these abandoned lands is limited to around 6 EJ_{elec}/year. EFs are higher on natural forest and grasslands, and on managed and degraded forests that have recently been logged or burnt and are re-growing ('managed and degraded forests'; see Methods). Net negative EFs are furthermore typically achieved in sub-tropical and warmer temperate areas (Figure 6.1b), which often sustain high yields (Figure S1), but do not have the large carbon stocks and associated initial LUC emissions of natural tropical and boreal forests. In large parts of the globe, however, purpose-grown biomass use for BECCS electricity would result in (significant) positive EFs over this 30 year evaluation period, stressing that BECCS' mitigation potential is highly dependent on the location of biomass cultivation. The geographical pattern we observe is in line with earlier geo-spatially explicit results on biofuels without CCS (Elshout et al., 2015; Daioglou et al., 2017), though Elshout et al. (2015) do deem boreal areas suitable, based on more optimistic estimates of both high crop yields and limited soil carbon losses in these regions.

Longer evaluation times lead to substantially higher BECCS energy potential at low EFs (Figures 6.1c, S8), predominantly because initial LUC emissions are amortised over longer time periods, and to a lesser extent due to projected yield increases and the levelling off of foregone carbon sequestration in the natural vegetation benchmark scenario. Therefore at an 80 year evaluation time (2020-2100), almost the entire global BECCS electricity potential, i.e., 220 EJ_{elec}/year has EFs below zero (Figure 6.1d), which entails large sequestration potential (40 Gtonne CO_2-eq./year; Table S5). The increase in BECCS' electricity supply potential is predominantly realised on natural forests and grasslands. On abandoned lands and managed and degraded forests, electricity supply potential with negative emissions is limited to 12 and 31 EJ_{elec}/year, respectively. Care should be taken in drawing conclusions based on longer evaluation times, for BECCS capacity that is installed later in the century may only achieve net negative emissions beyond the target year 2100. The results shown here represent lignocellulosic crops in general; grass and woody crop-specific results are provided in the supplementary information (Figures S5-7). Furthermore, we also investigated a shorter, 20 year evaluation period, which reduces electricity potentials by about 60% compared to 30 year evaluation time results (Figure S9).

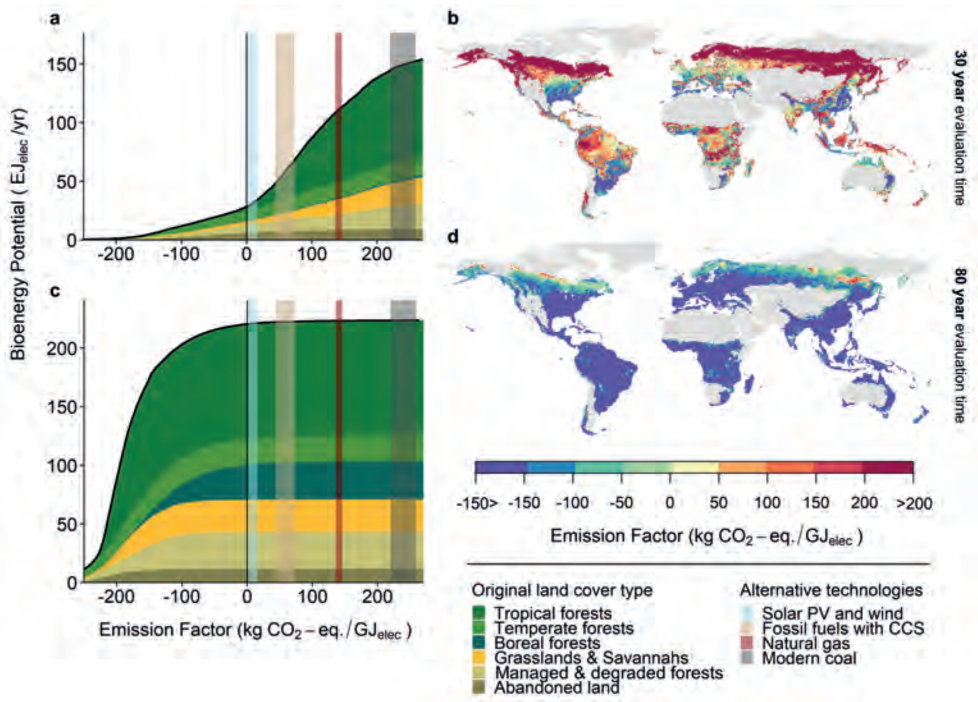

Figure 6.1 | Global emission-supply curve and emission factor map of bioelectricity with CCS. a. Emission-supply curve of bioelectricity with CCS over a 30 year evaluation time (black solid line), split over different original land cover types and excluding agricultural land (coloured areas). Shaded columns indicate EF ranges for alternative electricity generation technologies (Hertwich et al., 2015; Bruckner et al., 2014). **b**. Emission factor map of bioelectricity with CCS over a 30 year evaluation time. **c-d**. Emission-supply curve and emission factor map of bioelectricity with CCS over an 80 year evaluation time.

6.3 Liquid biofuels

Lignocellulosic FT-diesel with CCS has the highest energy and sequestration potentials of the investigated liquid biofuel routes. Over a 30 year evaluation time, however, the FT-diesel supply with negative emissions is minimal (Figure 6.2a, Table S5). Since there is substantial supply potential at EFs below fossil diesel (67 E_{fuel}/year), replacing the entire current global diesel consumption of 60 EJ_{fuel}/year incl. gas oil (UN, 2019), could theoretically result in GHG emission savings of approximately 5.5 Gtonne CO_2-eq./year, though this is not the same as net sequestration. Savings could also be achieved if FT-diesel and FT-synthetic kerosene (Blakey et al., 2011) are used to replace fossil shipping and aviation fuels. At an 80 year evaluation time, the global supply potential of lignocellulosic FT-diesel with negative emissions is large (282 EJ_{fuel}/year; Figure 6.2d), but the resulting global net sequestration potential of 4.8 Gtonne CO_2-eq./year is about eight times lower than for BECCS electricity over the same evaluation period (Table S5), predominantly due to FT-diesel's lower carbon capture rate of 52% (Table S1). The relative geographic and crop-specific patterns for EFs of FT-diesel with CCS are, however, similar to those of BECCS electricity for both evaluation times (Figures S6, S10). Over both a 30 and 80 year evaluation time, the bioethanol pathways with CCS do not result in net negative emissions (Figure 6.2b-f). This is primarily due to their low carbon capture rates (12%-24% for lignocellulosic and sugarcane ethanol, respectively, see Methods and Table S1).

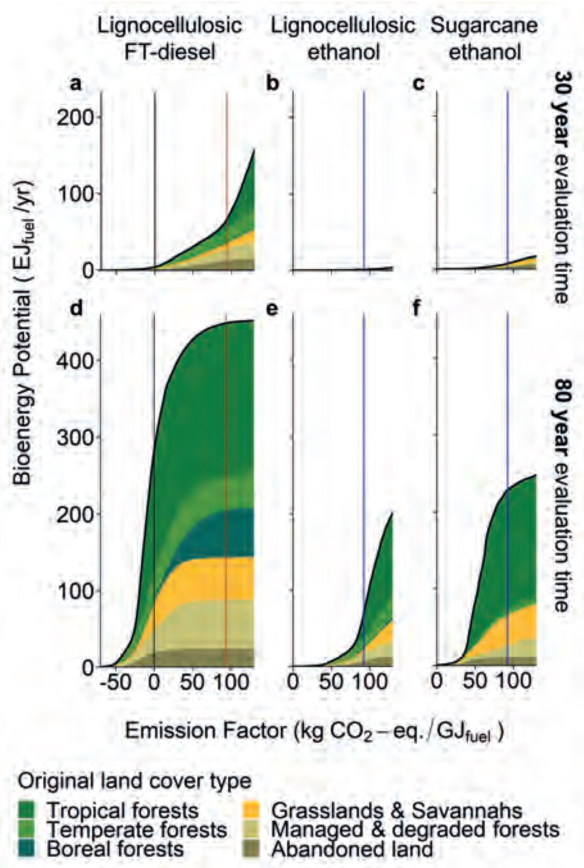

Figure 6.2 | Global emission-supply curves of liquid biofuels with CCS. a-c. Global emission-supply curves of FT-diesel, lignocellulosic ethanol and sugarcane ethanol, all with CCS, over a 30 year evaluation time. **d-f**. Global emission-supply curves of these liquid biofuels with CCS over an 80 year evaluation time. Orange and blue lines indicate the EFs of fossil diesel (94 kg CO_2-eq./GJ_{fuel}) and petrol (92 kg CO_2-eq./GJ_{fuel}; JRC, 2014), respectively. Note that electricity, FT-diesel and bioethanol potentials cannot be summed, as they are based on overlapping locations.

6.4 Initial biomass

In line with previous work (Elshout et al., 2015; Daioglou et al., 2017), we conservatively assumed that the original vegetation is burned when a bioenergy crop plantation is established, releasing all carbon in initial biomass to the atmosphere as CO_2. However, part of this initial biomass could also be used to produce bioenergy (Figure 6.3a). Using initial biomass for bioenergy increases overall BE(CCS) energy potential and sequestration, as also suggested by Harper et al. (2018), and decreases EFs, as emissions are allocated over more energy generated. If 80% (Hanssen et al., 2017) of all initial stem biomass is used and 90% of its carbon content is captured, BECCS electricity potential becomes approximately 4.5 times larger at EFs below zero, increasing from 28 to 125 EJ_{elec}/year over a 30 year evaluation time (Figure 6.3b). Carbon sequestration increases from 2.5 to 5.9 Gtonne CO_2-eq./year (Table S5).

Alternatively, the initial biomass can be used in other sectors to create more valuable products such as timber and paper (Hanssen et al., 2017). In this scenario, part of the initial carbon is stored in these products and when ultimately emitted, allocated to these products. Under this assumption, initial LUC emissions of BE(CCS) are lower, thus lowering EFs. If 80% of initial stem biomass is used in other sectors, the potential of BECCS electricity increases from 28 to 129 EJ_{elec}/year at EFs below zero (Figure 6.3c), while sequestration increases sharply from 2.5 to 11 Gtonne CO_2-eq./year (Table S5).

It is evidently better to use initial biomass for energy or materials rather than burning it, as is also reflected in lower EFs in both cases. However, the increased energy and sequestration potential of BECCS at negative EFs would also come from converting additional natural forests and savannahs, which have significant initial stem biomass. At longer evaluation times, the influence of using initial biomass for bioenergy or other products is limited (Figure S11), as emissions from initial biomass are amortised over longer time periods and have a smaller effect on EFs. Patterns for FT-diesel with CCS are similar to those of bioelectricity with CCS (Figure S12).

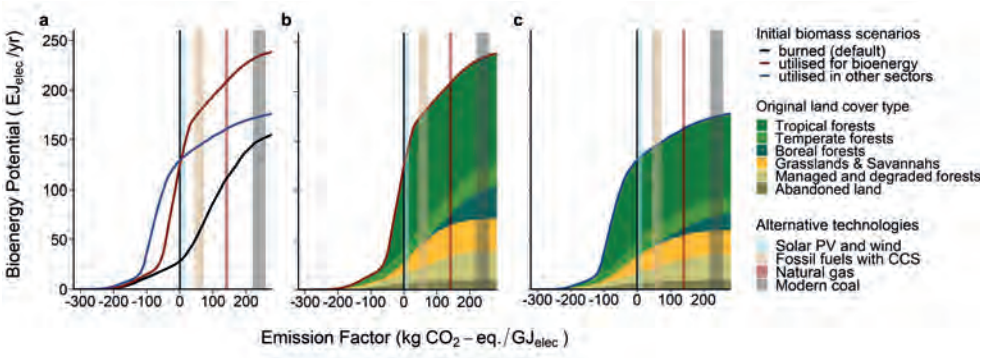

Figure 6.3 | Global emission-supply curves of BECCS electricity with different initial biomass use scenarios over a 30 year evaluation time. a. Overview of emission-supply curves for three initial biomass scenarios. **b**. Emission-supply curve of BECCS electricity with 80% of initial stem biomass used to produce additional BECCS electricity (red solid line), split over different original land cover types. **c**. Emission-supply curve of BECCS electricity with 80% of initial stem biomass used in other sectors (blue solid line). Shaded columns indicate EF ranges for alternative electricity generation technologies (Hertwich et al., 2015; Bruckner et al., 2014).

6.5 BECCS in mitigation pathways

We used our spatially-explicit EFs and energy and sequestration potentials for BECCS to analyse global carbon sequestration up until 2100 following the phased deployment of BECCS in two illustrative mitigation pathways of the IPCC SR1.5°C report: the S2 middle-of-the-road pathway and the S5 fossil fuel and BECCS-intensive pathway (Rogelj et al., 2018; Huppman et al., 2019; see Methods). In our analysis, we deployed land starting with the best locations (lowest EFs; excluding agricultural land) and we matched prescribed BECCS deployment rates either in terms of pathway-prescribed energy generation or pathway-required sequestration. We used a dynamic evaluation time up until 2100 for the installed BECCS capacity (e.g., a 40 year evaluation time for capacity installed in 2060) and assumed initial vegetation is burned.

Because we determine EFs from a full life-cycle perspective and include foregone sequestration, we typically find less carbon sequestration per unit BECCS energy than in mitigation pathways. Following energy-based BECCS deployment rates thus resulted in lower carbon sequestration than projected in the pathways (Figure 6.4a). Following pathway-required annual sequestration, BECCS electricity from lignocellulosic crops only can keep up net sequestration until the year 2066 for S2 and the year 2050 for S5 (Figure 6.4a), after which additional land conversion does not provide negative emissions over the remaining period to 2100. When first deploying all biomass residues available for energy (based on IMAGE SSP2, see Methods) to BECCS before using lignocellulosic crops, these points are postponed to the year 2076 and 2058 for S2 and S5 (Figure 6.4a).

Over the century, the estimated sequestration that could be achieved using lignocellulosic crops alone (250 and 1008 Gtonne for S2 and S5) is 61-84% of total projected sequestration (408 and 1207 Gtonne for S2 and S5; Figure 6.4b). This is in line with an earlier, crop yield-based exploration of BECCS' global sequestration potential, which found that 59% of the sequestration required in a limited global warming scenario (RCP2.6) may be achieved (Kato and Yamagata, 2014). When also including biomass residues, we find that projected sequestration is approached to 88-94%, but not fully achieved (360 and 1132 Gtonne for S2 and S5; Figure 6.4b). In this estimate 0.8 to 2.4 Gha of land is required by 2100 to grow crops for BECCS, for S2 and S5 respectively, which equals 5.1% and 16% of the total land surface area on Earth and of which 53% and 72% are currently natural forests and grasslands. It is important to note that these extreme levels of land demand partly arise due to the time profile of in particular the S2 pathway, and from our assumption to use residues before crops. The cumulative sequestration these pathways demand by 2100 could, biophysically, be achieved with lower land requirements if deployment of crop-based BECCS starts even earlier on, as indicated by the importance of evaluation periods in our analysis (see Figure S8). In any scenario, sequestration potential is drastically increased when deploying BECCS earlier, as also suggested in earlier work (Obersteiner et al., 2018).

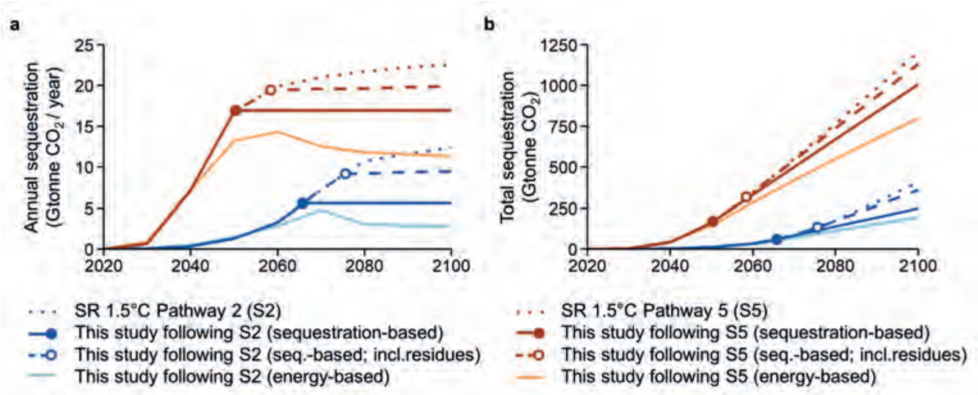

Figure 6.4 | Carbon sequestration potential of BECCS electricity in climate change mitigation pathways. Carbon sequestration refers to negative emissions. **a**. Annual sequestration through BECCS electricity. **b**. Total (cumulative) sequestration through BECCS electricity. Dots indicate the point at which pathway-prescribed sequestration can no longer be kept up with, as additional land conversion no longer results negative emissions over the remaining (evaluation) time until 2100.

6.6 Sensitivities & limitations

Figure 6.5 shows how emission-supply curves of BECCS electricity are influenced by three key parameters. First, keeping bioenergy crop yields constant at their 2020 values decreases BECCS electricity supply potential at negative EFs by 25-32%, while enhanced yield improvement (i.e., global improvement of agricultural management to current best practice, representing SSP1) increases it by 6-11% (Figure 6.5a,e). Second, in line with previous studies (Fajardy and Mac Dowell, 2017; Harper et al., 2018), BECCS electricity supply potential is sensitive to electricity conversion efficiency: a literature-based 5-7% change in conversion efficiency (Table S1) changes supply potential with negative emissions by 6-8% (Figure 6.5b,f). Carbon sequestration potential is, however, unaffected as the carbon capture rate is not influenced by conversion efficiency. Third, more arable lands become available for bioenergy if less land is required for conventional agriculture. Following the SSP1 scenario (with a smaller population and low-meat diet, see Methods), BECCS electricity potential at EFs below zero increases by 21-93% (Figure 6.5c,g). When all three parameters are combined into a 'best case' and 'worst-case' scenario, BECCS energy potential at negative EFs approximately doubles or halves from the default (Figure 6.5d,h). These patterns are similar for lignocellulosic FT-diesel (Figure S13). Our results are less sensitive to variation in other parameters. Doubling supply chain emissions, for instance, only resulted in a 1-5% reduction of BECCS electricity supply potential at negative EFs (Figure S14), though liquid biofuel EFs are more strongly affected (Figure S15).

There are several possible limitations to the biophysical climate change mitigation potential of BECCS. First, our analysis focuses on high-yielding lignocellulosic bioenergy crops and sugarcane. In the boreal forest region, however, yields would typically be low and natural carbon stock losses high, meaning that lower EFs may be achieved by sourcing biomass from sustainably managed forests, if their carbon stocks are maintained (Lundmark et al., 2016; Peura et al., 2018). Under such boreal continuous cover forestry (CCF) we find that electricity supply potential with negative emissions increases by 2.5 EJ/year over a 30 year evaluation period, but *decreases* over longer evaluation periods, as yields are lower than for lignocellulosic crops (Figures S16,17). CCF would, on the other hand, have key benefits in terms of biodiversity conservation and ecosystem services (Peura et al., 2018; Lundmark et al., 2016; Kuuluvainen and Gauthier, 2018). Second, we excluded projected agricultural areas (cropland and pastures) to avoid iLUC effects, but conversion of managed forests could also lead to iLUC emissions, as forestry products like timber and paper are partly sourced from such forests. Third, biomass yields in the LPJml model are not explicitly influenced by soil quality parameters. However, yields are calibrated (see Methods) and we found that over 99% of the BECCS electricity potential with negative emissions is derived from areas with soils that are classified as moderately or highly suitable for rainfed crop cultivation over the continuous period 2011-2100 (Zabel et al., 2014). Lastly, albedo reduction could lower mitigation, which is not accounted for in our calculations. Changes in albedo are typically limited though for grasses and coppiced trees (approximately 5% maximum reduction) (Smith et al., 2016).

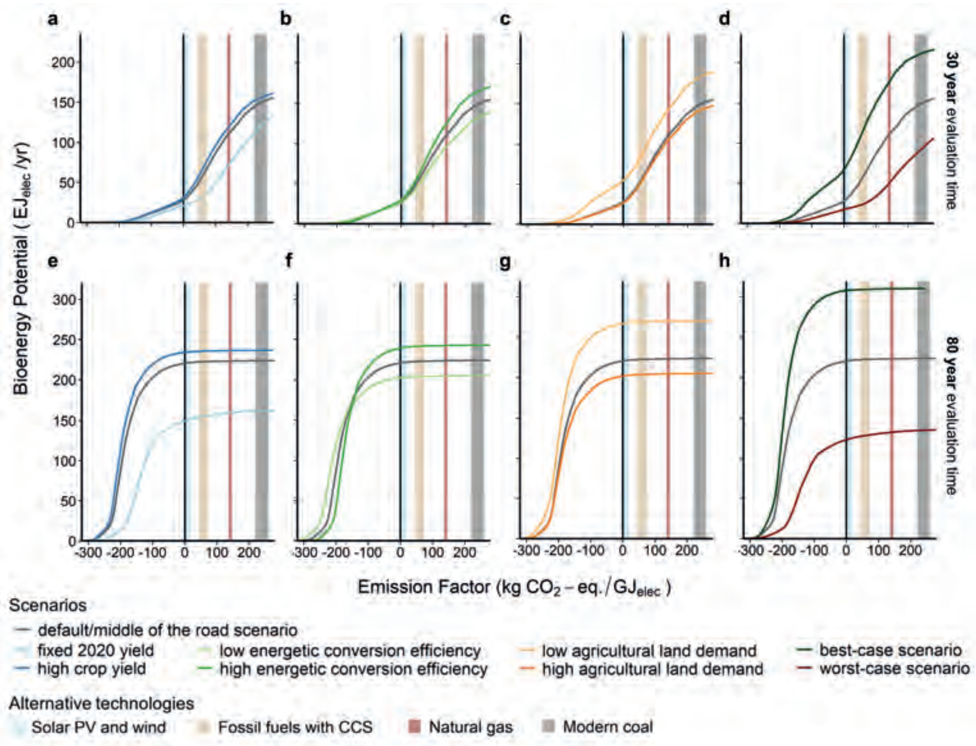

Figure 6.5 | Sensitivity of BECCS electricity emission-supply curves to parameterisation. The default emission-supply curve is plotted in grey in all panels. **a**. Emission-supply curves at constant 2020 crop yields (light blue) and high SSP1 crop yields (dark blue) **b**. Emission-supply curves for low (light green) and high (dark green) biomass to energy carrier conversion efficiencies (based on literature, Table S1). **c**. Emission-supply curves for scenarios with low (yellow) and high (orange) agricultural land requirements (based on SSP1 and SSP3 in IMAGE; default is SSP2). **d**. Emission-supply curves for a best-case (green) and worst-case (red) scenario. **e-h**. these same emission-supply curves for an evaluation time of 80 years, rather than 30 years. Shaded columns indicate EF ranges for alternative electricity generation technologies (Hertwich et al., 2015; Bruckner et al., 2014).

6.7 Implications

We conclude that the climate change mitigation potential of lignocellulosic crop-based BECCS is largest when producing electricity on locations with high biomass yields and relatively low carbon stocks (i.e., abandoned lands and typically warmer temperate and sub-tropical areas), while utilising the original vegetation for bioenergy or materials. We found that the EFs derived for BECCS are crucially dependent on the evaluation time considered, as they account for LUC emissions and foregone sequestration. Our global emission-supply curves and EF maps show that biophysically, many cultivation locations could supply electricity with negative EFs, leading to a large global electricity supply and carbon sequestration potential of 28 EJ_{elec} and 2.5 Gtonne per year over 30 years, 220 EJ_{elec} and 40 Gtonne CO_2-eq. per year over 80 years, and 129 EJ_{elec} and 11 Gtonne CO_2-eq per year over 30 years when utilising initial biomass. The sequestration potential of liquid biofuels with CCS is limited, though BECCS FT-diesel can lead to negative emissions over an 80 year evaluation period and replacing GHG-intensive fossil transport fuels strongly reduces emissions.

Using our global emission-supply curves, we showed that the projected trajectory of BECCS-based sequestration in mitigation pathways S2 and S5 (Rogelj et al., 2018) can biophysically be approached (88-94%), but not fully achieved, as residues and arable land with negative emissions become depleted. The reason for this is partly that especially S2 deploys BECCS later in the century, and that biomass residues are used first, which leads to shorter evaluation periods up to 2100 for crop-based BECCS, and therefore larger land requirements. This highlights that crop-based BECCS should be deployed early on to most effectively contribute to climate change mitigation. Still, the land requirements for BECCS to achieve the cumulative amount of carbon sequestration projected in these pathways are likely to be large to the point of being infeasible, as also suggested in bottom-up assessments of BECCS' sequestration potential (De Coninck et al., 2018).

Depending on the exact scenario around 50-90% of the land area required, carbon sequestered and energy supplied would come from natural forests and grasslands. Since land conversion to BECCS strongly reduces biodiversity (Chaudhary et al., 2015), trade-offs clearly exist between BECCS' climate change mitigating effect and biodiversity conservation (Heck et al., 2018; Stoy et al., 2018; Hof et al., 2018). The mitigation potential of BECCS is further reduced by other environmental (Smith et al., 2016; Kemper, 2015; Bonsch et al., 2016; Fajardy et al., 2018) and socio-political constraints (Fridahl and Lehtveer, 2018; Torvanger, 2019; Bednar et al., 2019; Daggash, 2019), limitations to the amount of developed geologic storage sites (Scott et al., 2015; Baik et al., 2018; Haszeldine et al., 2018; Turner et al., 2018a), and the challenge of upscaling BECCS orders of magnitude from its current demonstration phase (Haszeldine et al., 2018; van Vuuren et al., 2017; Sanchez et al., 2018; Turner et al., 2018b).

Yet, BECCS may play an important role in mitigating climate change and the energy transition, alongside renewables, other NETs (Fuss et al., 2018) and deep-emission reduction (van Vuuren et al., 2018; Grubler et al., 2018). Residues (Hanssen et al., 2019) and waste flows (Pour et al., 2018) form low-impact feedstocks for BECCS with little effect on land-use. Lignocellulosic crop-based BECCS could also be deployed on abandoned agricultural lands (Turner et al., 2018a). Biodiversity and other environmental impacts of BECCS could be reduced using locally optimal crops (Robertson et al., 2017) and supply chain configurations (Fajardy et al., 2018). In all cases, our results indicate that earlier deployment of BECCS greatly increases its climate change mitigation potential, and suggest that policymakers ought to complement BECCS with other options for GHG emission reduction and carbon dioxide removal.

6.8 Methods

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Calculations

GHG emission factors (EF) for feedstock *i* (fast-growing grasses / short-rotation coppicing / sugarcane), carrier *j* (electricity / FT-diesel / ethanol), evaluation time *t* (20-80 years), and location *x* (66,663 land cells; 30 x 30 arcminute raster) were calculated as the sum of GHG emissions minus sequestration per unit energy carrier produced (in tonne CO_2-eq./$GJ_{carrier}$; Equation 6.1).

$$\mathrm{EF}_{i,j,t,x} = \mathrm{Em}_{\mathrm{LUC},i,j,t,x} + \mathrm{Em}_{\mathrm{Fertiliser},i,j,x} + \mathrm{Em}_{\mathrm{Supply\ Chain},i,j} - \mathrm{Seq}_{\mathrm{CCS},i,j} \qquad \textbf{Equation 6.1}$$

LUC emissions (Em_{LUC}) were calculated as the difference in carbon stocks between the bioenergy plantation and a natural vegetation regrowth benchmark at the end of the considered evaluation time (i.e., including foregone sequestration), divided by energy carrier production over the evaluation period (Equation 6.2). Fertiliser N_2O emissions ($Em_{Fertiliser}$) were obtained by converting crop-specific fertiliser emissions to emissions per carrier produced (Equation 6.3). Life-cycle supply chain emissions for the production of the energy carrier, including CH_4 ($Em_{Supply\ Chain}$) were based on literature (Table S1). Net CO_2 sequestration from CCS (Seq_{CCS}) was calculated as the captured amount of carbon per carrier produced minus additional supply chain emissions of CCS per carrier produced (Equation 6.4).

$$Em_{LUC,i,j,t,x} = \frac{\Delta C_{i,t,x} \cdot r}{Y_{i,t,x} \cdot t \cdot f_{loss} \cdot (\eta_{i,j} - \pi_{i,j})}$$ **Equation 6.2**

$$Em_{Fertiliser,i,j,x} = \frac{em_{Fertiliser,x}}{(\eta_{i,j} - \pi_{i,j})}$$ **Equation 6.3**

$$Seq_{CCS,i,j} = \frac{f_{loss} \cdot cc_i \cdot r \cdot \kappa_{i,j}}{(\eta_{i,j} - \pi_{i,j})} - Em_{Supply\ Chain\ CCS,j}$$ **Equation 6.4**

Where: ΔC is the difference in above and belowground carbon stocks (tonne C/ha) between the bioenergy plantation and a natural regrowth benchmark at the end of the considered evaluation time; r is the molar ratio between CO_2 and C (i.e., 3.66); Y is the annual bioenergy crop yield over the considered evaluation time (tonne dry biomass/[ha x year]); t is the evaluation period (in years); f_{loss} is the biomass loss correction factor; η is the biomass to final carrier conversion efficiency ($GJ_{carrier}$/tonne dry biomass); π the penalty in conversion efficiency due to CCS ($GJ_{carrier}$/tonne dry biomass); em are GHG emissions per biomass produced (kg CO_2-eq./tonne dry biomass); cc is the carbon content of the feedstock (tonne C/tonne dry biomass); κ is the carbon capture efficiency of CCS (tonne CO_2 captured/tonne CO_2 emitted) at the power plant or fuel production facility. $Em_{Supply\ Chain\ CCS}$ are the (additional) life-cycle supply chain emissions of using CCS (tonne CO_2-eq./$GJ_{carrier}$). Note that EFs are expressed in kg CO_2-eq. /$GJ_{carrier}$ throughout the main text.

Energy potentials (EP; in $GJ_{carrier}$/year) per grid cell were calculated as production area times net bioenergy yields (Equation 6.5)

Equation 6.5

$$EP_{i,j,t,x} = A_{x,t} \cdot Y_{i,t,x} \cdot f_{loss} \cdot (\eta_{i,j} - \pi_{i,j})$$

Where: A is the land area of each grid cell (in ha).

Global emission-supply curves were determined by sorting all grid cells available for BECCS by ascending emission factor and summing energy potential across these cells. Lignocellulosic bioenergy crop results in the main text were combined from the results for grasses and short-rotation coppicing, by selecting the crop type for each grid cell that results in the lowest EF (details and alternative selection methods are provided in the supplementary information, Figures S3-4). Carbon stocks and bioenergy crop yields were modelled in the (IMAGE-)LPJml global vegetation model and land availability was determined using the IMAGE integrated assessment model, as detailed below. All other parameter values and their ranges are literature-based (Tables S1 and S2). Of these parameters, carbon capture efficiency (κ) stands out, as its value differs strongly among the different energy carriers: 90% for lignocellulosic electricity, 52% for lignocellulosic FT-diesel, 12% for lignocellulosic ethanol, and 24% for sugarcane ethanol. The reason for this difference being that we assume CO_2 emissions from liquid fuel *combustion* are not captured and stored, as these fuels are almost entirely used in transport and other decentralised applications without feasible CCS capability. Furthermore, we assume that only CO_2 from the FT-process or fermentation step itself is captured in the FT-plant or biorefinery. The more disparate flows of CO_2 that for instance arise from the combustion of biomass or fossil fuels for process heat or auxiliary power (modelled as part of supply chain emissions) are relatively small in volume and low in CO_2 concentration and are assumed not be captured, in line with previous work, as explained in detail in Section 14 of the supplementary materials. Emission factors of alternative energy technologies were derived from literature (Table S3). Non-CO_2 GHGs were accounted for using global warming potentials over a 100 year time period based on the IPPC fifth assessment report (Myhre et al., 2013).

Carbon stocks and bioenergy crop yields in IMAGE-LPJml

We used the IMAGE integrated assessment model (Stehfest et al., 2014) coupled to the LPJml global vegetation and hydrological model (Beringer et al., 2011; Müller et al., 2016) to determine carbon stocks and yields per location over time. By default, we used a forced climate scenario via a representative concentration pathway (RCP) leading to 2.6 $W \cdot m^{-2}$ radiative forcing by 2100 (van Vuuren et al., 2011), reflecting substantial climate change mitigation. A warmer climate scenario is explored in the supplementary materials (Figure S14).

Carbon dynamics modelled in LPJml cover aboveground biomass, belowground biomass and soil carbon. We determined carbon stock changes by comparing the difference in carbon stocks at the end of the evaluation time between two scenarios: i) the bioenergy scenario, where land in each available cell is used to

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grow a bioenergy crop (excluding aboveground biomass, which is harvested), and ii) the natural vegetation "benchmark" scenario, where vegetation grows naturally without management. By looking at this difference in carbon stocks, we thus explicitly account for the lost sequestration capacity of natural vegetation that is foregone by using the land for bioenergy crop plantations instead. Three bioenergy crop types were considered: i) grassy bioenergy crops, i.e., fast-growing grasses parameterised based on both *Miscanthus* and switchgrass cultivars, ii) woody bioenergy crops, i.e., short-rotation coppiced trees parameterised based on *Eucalyptus* spp. in the tropics and both willow and poplar in colder areas, and iii) sugarcane. Non-CO_2 GHG emissions of land conversion were not explicitly included here, but based on Whitaker et al. (2018) would typically be below 2% of total GHG emissions per energy carrier in this study.

Yield is determined as the crop-specific rainfed potential biophysical yield in the LPJml model multiplied by a calibration factor that expresses how much of that potential yield is realised. Globally, the average yield potential in LPJml increases by approximately 25-30% from 2020 towards 2100, due to climate feedbacks. The calibration factors were determined by Daioglou et al. (2017, 2019) based on empirical data of historic, current and best-practice yields (Gerssen-Gondelach et al., 2014; Boehmel et al., 2008) and are projected into the future as part of the IMAGE model. They represent agricultural management, including fertilisation, improved crop strains and pest control (Daioglou et al., 2017). In line with historic trends, the calibration factors result in a global average increase in yields of 0.72-1.0% per year for grasses and woody bioenergy crops, and 0.76% per year for sugarcane, from 2020 towards 2100. Energy potentials and EFs were always determined using yields and carbon stock changes from 2020 onwards (e.g., 2020-2060 for a 40 year evaluation time).

Land availability

Which locations are available for bioenergy production was determined using the IMAGE model. It was assumed that areas used for agriculture (cropland and pastures) over the considered evaluation period, are not available for bioenergy production. Default results were based on a median land-use scenario following shared socio-economic pathway 2 (SSP2) (Moss et al., 2010). Scenarios with lower and higher agricultural land demand in the sensitivity analysis were based on SSP1 and 3, respectively. SSP1 includes assumptions on a shift towards less meat-intensive diets and a low population size. SSP3 on the other hand, is characterised by high population growth and low technological development and therefore higher agricultural land requirements. Beside agricultural land, built-up areas were also excluded. The amount of land available for bioenergy was further constrained by a minimum yield threshold. That is, lands yielding less than 2.5 tonne wet biomass per hectare per year (or 10 tonne for sugarcane) as determined in LPJml, were excluded in our analysis. For all crop types these thresholds are about 5% of the global maximum yields per hectare per year.

Land cover types

The original land cover types presented in this analysis were based on IMAGE classification (Stehfest et al., 2014; Figure S2). Specifically, abandoned lands are based on what agricultural lands are abandoned towards 2100, depending on the projected supply and demand of agricultural products as determined in IMAGE. The managed and degraded forests land cover type is defined here as forestland that is in a re-growing state after recent human interventions. It encompasses: i) *managed forests* for wood production, which predominantly occur in temperate and boreal zones, and ii) re-growing *degraded forests* that remain after logging for the most valuable trees or slash-and-burn practices, predominantly in tropical areas. For degraded forests specifically, default LPJml carbon dynamics were re-calibrated based on literature (de Andrade et al., 2017; Rappaport et al., 2018; Bonner et al., 2013; Poorter et al., 2016). We estimated that aboveground carbon stocks in forests that have been degraded within the last 20 years are approximately two-thirds of unharvested carbon stocks, as detailed in section 3 of the SI. In the natural vegetation benchmark scenario, we therefore modelled carbon stocks of degraded forests following the default growth curves for natural forests in LPJml, but starting where aboveground carbon stocks are at two-thirds of their maximum.

Alternative uses initial biomass

When "initial biomass" from the original vegetation is utilised in other sectors, EFs and EPs were calculated by subtracting 80% (Hanssen et al., 2017) of the carbon present in initial stem biomass from the original (pre-conversion) carbon stocks. When initial biomass is used to produce bioenergy, 80% of initial stem biomass is instead added to the overall yield over the evaluation period. It is assumed that initial biomass is used to produce the same energy carrier, including CCS.

BECCS in mitigation pathways

As a starting point of this analysis we took two illustrative climate change mitigation pathways from the IPCC special report on 1.5°C (Rogelj et al., 2018): the S2 middle-of-the-road pathway (MESSAGE-GLOBIOM 1.0 SSP2) and the S5 fossil-fuel and BECCS-intensive pathway (REMIND-MagPIE 1.5 SSP5). The IPCC SR1.5°C online database (Huppman et al., 2019) provides total global carbon sequestered by BECCS electricity (Carbon Sequestration|CCS|Biomass) and primary energy used in BECCS electricity (Primary Energy|Biomass|Modern|w/CCS). We converted global primary energy used to global electricity produced with BECCS, assuming an energetic conversion efficiency of 0.31 $GJ_{electric}/GJ_{biomass}$, following the IPCC AR5 median dedicated biomass electricity plant efficiency (Schlömer et al., 2014). We used 10 year intervals in our calculations, as provided in the IPCC database, with linear interpolation. In the analysis, we deploy land starting with best locations (i.e., with the lowest EFs) and follow the global energy and sequestration-based BECCS deployment rates. We use an evaluation time up until 2100 (e.g., 50 years for capacity installed in 2050, 40 years for 2060, etc.). From 2070 onwards we use the default evaluation time of 30 years to avoid underestimating BECCS potential.

When including biomass residues, we deployed *all* residues available for bioenergy to BECCS, before allocating any land to bioenergy crop production for BECCS. In all cases, residue availability for bioenergy was based on the IMAGE SSP2 baseline scenario and included both agricultural and forestry residues (Table S4). The GHG balance of residues-based BECCS included CO_2 sequestered via CCS (assuming a 50% carbon content; Table S1) and supply chain emissions (based on parameterisation for grassy lignocellulosic biomass, excluding fertiliser emissions; Table S1). Residues were assumed not to cause land-use change emissions or result in foregone sequestration of a natural vegetation reference scenario.

6.9 Data availability

Data supporting the findings of this study are available within the paper and its supplementary information files. Source data for figures and datasets generated during the current study are available online [easy.dans.knaw.nl/ui/datasets/id/easy-dataset:178418].

6.10 Acknowledgements

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Global biodiversity implications of negative emissions from lignocellulosic crop-based bioenergy with carbon capture and storage

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ABSTRACT

Bioenergy with carbon capture and storage (BECCS) based on lignocellulosic crops could provide *negative* greenhouse gas emissions to mitigate climate change, but its land requirements present a threat to biodiversity. Here, we analyse the implications of BECCS for global terrestrial vertebrate species richness, considering both land-use change (LUC) *for* BECCS and mitigated climate change *by* BECCS. LUC impacts are determined using global-equivalent, species-area relationship-based loss factors. We find that global vertebrate species extinctions from LUC per unit of negative emissions are uncertain (0.055-7.7 species lost/Gtonne CO_2 sequestered, over 30 to 80 year evaluation periods), but decrease with: i) longer lifetimes of BECCS systems, ii) less overall deployment of crop-based BECCS, and iii) optimal land allocation (i.e., prioritise locations with lowest species loss per negative emission potential). The positive effect of mitigated climate change on biodiversity is based on earlier meta-analysis and climate response modelling. Tentative comparison shows that LUC effects most likely outweigh climate mitigation effects over a 30 year period, but this trade-off is less clear over 80 years, with a potential small positive effect under optimal land allocation. Both effects and their interaction are, however, highly uncertain and require additional understanding, along with analysis of additional species groups and biodiversity metrics. We conclude that factoring in biodiversity means crop-based BECCS should be used as early as possible to achieve required mitigation over a longer time period, on optimal biomass cultivation locations to minimise biodiversity loss, and most importantly, as little as possible where conversion of natural land is involved, looking instead to sustainably grown or residual biomass-based feedstocks and alternative negative emission technologies.

7.1 Introduction

Most climate change mitigation pathways require negative greenhouse gas (GHG) emissions to limit global warming to 1.5-2 °C (Rogelj et al., 2018). Several options are considered to achieve negative emissions (Fuss et al., 2016; 2018; Smith et al., 2016), of which bioenergy with carbon capture and storage (BECCS) is among the most prominent in mitigation pathways (Rogelj et al., 2018). In the BECCS production chain, atmospheric CO_2 is taken up by growing biomass, which is then combusted to generate energy, while the released CO_2 is largely captured and geologically stored, resulting in negative emissions (Obersteiner et al., 2001; Azar et al., 2010; Gough & Upham, 2011; Kemper, 2015). The reason BECCS is often considered attractive (for instance in energy models) is that it is based on a combination of existing technologies, is scalable, yields useful energy, and may have lower costs than other negative emission technologies (Fuss et al., 2016; 2018; Smith et al., 2016; Hepburn et al., 2019). While BECCS can contribute to climate change mitigation (Harper et al., 2018, Hanssen et al., 2020; Muratori et al., 2020), it also has large potential impacts on the environment through its water, land and nutrient use (Kemper et al., 2015; Smith et al. 2016; Bonsch et al., 2016; Fajardy et al., 2018; Heck et al., 2018; Stoy et al., 2018; Hanssen et al., 2020) and could compete for land with food production (Doelman et al., 2018; Hasegawa et al., 2018, 2020; Fujimori et al., 2019).

The large amount of land required for upscaling crop-based BECCS (Hanssen et al., 2020) could also present a large additional threat to biodiversity (Heck et al., 2018; Nunez-regueiro & Fletcher, 2019), which is already in sharp decline due to (among others) land-use change, over-exploitation and climate change (Hoffman et al., 2010, Barnosky et al., 2011; Dirzo et al., 2014, Ceballos et al., 2017; IPBES, 2019). While land conversion forms an additional strain, mitigated climate change could also prevent biodiversity loss (Thomas et al., 2004; Urban, 2015). Hof et al. (2018) suggested based on climate-based species distribution models that the impacts of bioenergy cropland expansion on global terrestrial vertebrate species richness may be much larger than positive effects on biodiversity of bioenergy-mitigated climate change, but did not include carbon capture and storage. Heck et al. (2018) examined the effect of BECCS expansion on the biodiversity intactness index (a local biodiversity indicator that represents the relative abundance of native species in an area under anthropogenic use). Their results indicate that it is impossible to convert additional natural land for BECCS without further transgressing the planetary boundary of biosphere integrity. These authors did not focus on global species richness or the simultaneous impact of (mitigated) climate change and land conversion on biodiversity. The fact that negative emission options, like BECCS, might be critical to limit climate change to 1.5 - 2°C demands a closer look at the global relation between BECCS' negative emissions and global species extinctions. This includes in particular the global biodiversity loss from land conversion for negative emissions, as well as the trade-off with biodiversity conservation of mitigated climate change.

Here, we aim to provide insight into this relation between negative emissions from lignocellulosic crop-based BECCS and global biodiversity loss, i.e., terrestrial vertebrate species committed to global extinction. We combine full life-cycle, spatially-explicit negative emission potentials for BECCS electricity (Hanssen et al., 2020) with global-equivalent biodiversity loss factors for land-use change (Chaudhary & Brooks, 2018), which are derived per ecoregion based on species-area relationships, for the four classes of terrestrial vertebrates (mammals, birds, amphibians and reptiles). This approach allows us to explicitly estimate: i) regional variation in the contribution to global-equivalent species loss from LUC for BECCS-based negative emissions, and ii) the potential global biodiversity loss from LUC to achieve a certain amount of negative emissions under different land allocation strategies. Moreover, we compare the biodiversity impact of LUC to the potential biodiversity conservation effect of mitigated climate change by BECCS. The effect of mitigation is based on results of a meta-analysis on global species loss from global temperature increase (Urban, 2015) and modelling of the global temperature response to negative GHG emissions (Van Vuuren et al., 2020). Importantly, we also include the temporal scope of these analyses by looking at 30 and 80 year evaluation periods for BECCS.

7.2 Methods

Negative GHG emissions from BECCS

Negative GHG emissions from BECCS refer to the *net* amount of CO_2 that can be taken out of the atmosphere and geologically stored, while considering LUC and supply chain GHG emissions, crop yields, bioenergy conversion efficiencies and carbon capture rates. We derived annual negative emission potentials (tonne CO_2-eq./ha/year; Figure S1) for each 0.5° x 0.5° grid cell from Hanssen et al. (2020). These potentials were based on: i) spatially-explicit (changes in) carbon stock and crop yield estimates obtained from the LPJml global vegetation and hydrological model (Beringer et al., 2011; Müller et al., 2016) coupled to the IMAGE integrated assessment model (Stehfest et al., 2014), and ii) literature-based supply chain emissions, conversion efficiencies and carbon capture rates (for a detailed description, see Hanssen et al., 2020). We specifically used values for electricity with carbon capture and storage (90% capture rate) produced from lignocellulosic bioenergy crops: either fast-growing grasses like Miscanthus and switchgrass, or short-rotation coppicing of Eucalyptus, willow or poplar, depending on which results in the lowest emissions for each cultivation location. We assumed that 80% of *stem* biomass present before bioenergy crop plantation establishment is used to produce BECCS electricity and all remaining initial biomass is burned on-site.

We determined cumulative negative emissions from crop-based BECCS per grid cell (in tonne CO_2-eq.) by multiplying the cell's negative emission potential (tonne CO_2-eq./ha/year) with the area in the cell available for BECCS (hectares) and the time period considered (years). This evaluation time period strongly affects the amount negative emissions achieved. Firstly, because initial emissions from LUC have to be compensated by subsequent BECCS-based carbon sequestration to achieve negative emissions, and the evaluation period effectively sets the amortisation period for these initial LUC emissions. Secondly, a longer evaluation period simply leads to more crop rotations and more carbon sequestration. Thirdly, over longer evaluation periods, the longer amortisation period means that more locations can yield negative emissions. We investigated the influence of this evaluation period, by considering both a 30 and an 80 year period.

The land area available for BECCS was defined from the perspective of the *biophysical* potential to achieve negative emissions. Furthermore, cells and areas within cells with very high biodiversity conservation value (Figure S2) were on forehand excluded from our analysis. Excluded areas thus comprise:

- Grid cells classified as current or future urban area, cropland or pasture in the 21st century (based on the SSP2 baseline scenario in the IMAGE model; Stehfest et al., 2014).

- Grid cells with low bioenergy crop yields (below 5% of the global maximum yield, based on LPJml).

- Grid cells in which no net negative emissions can be achieved, which differs over the specified evaluation periods and was based on Hanssen et al. (2020).

- Water bodies within grid cells.

- (part of) Grid cells of currently protected areas (UN WCMC, 2019).

- (part of) Grid cells of intact forests, i.e., natural areas (including non-forest ecosystems) without human activities that are large enough to maintain all native biodiversity (Potapov et al., 2017).

Biodiversity loss from LUC

As global metric of biodiversity loss from LUC, we used the global-equivalent potential vertebrate species loss factors that have been derived by Chaudhary & Brooks (2018) to determine the influence of LUC on biodiversity loss. Based on species-area relationships (SARs), these species loss factors (number of species that become committed to global extinction per ha of land used) have been derived for four classes of terrestrial vertebrates (based on 6,251 amphibian, 3,384 reptile, 5,386 mammal and 10,104 bird species) and for 804 terrestrial ecoregions across the globe (Figure S3). The species-area relationships that these loss factors are based on take into account: the number of original species present in the ecoregion, the loss of natural habitat, and the average preference of species for new artificial habitat types. A vulnerability score (based on range sizes and IUCN red list status) is assigned to each species group-ecoregion combination to reflect the vulnerability to extinction on a global scale of the both endemic and non-endemic species living within that ecoregion.

The species loss factors have been determined for different land-use types and land-use intensity levels; we selected intensive plantation forestry to represent the bioenergy crop plantations. In 30 out of 804 ecoregions no factors for intensive plantation forestry had been derived. In these instances, we used the factors for intensive agriculture (22 ecoregions) or, when these were not unavailable either, clear-cut forestry (6 ecoregions). For two remaining small pacific island ecoregions no relevant factors were available; these were excluded from the assessment. Ultimately, biodiversity loss from LUC (i.e., species committed to global extinction) was determined for each vertebrate class and each 0.5° x 0.5° grid cell, by multiplying each cell's species group-ecoregion loss factor by the area of the cell that is available for BECCS (see previous section). Uncertainty ranges for ecoregion-specific species richness loss factors, specified as 95%-confidence intervals, were included in our calculations (fully correlated across all eco-regions) to show the statistical uncertainties in our results.

Relating biodiversity loss from LUC to negative emissions at the global scale
After both biodiversity loss from LUC and cumulative negative emissions from BECCS over the considered evaluation periods were quantified per grid cell, their relation at the global scale was derived as a biodiversity response curve to cumulative carbon sequestration. This was done separately for the 30 and 80 year evaluation periods. In both cases the relation between biodiversity loss from LUC and negative emissions can have multiple shapes, depending on what locations are converted for BECCS first. We analysed three criteria to prioritise biomass cultivation locations for BECCS:

i. Prioritise land with the largest carbon negative emissions potential. Grid cells with the largest cumulative negative emission potential (tonne CO_2-eq./ha) are selected first, until all grid cells with net negative emissions had been selected. This minimises land-use requirements.

ii. Prioritise land with lowest biodiversity loss. Grid cells with the lowest biodiversity loss due to land conversion (species/ha; across all four studied taxa) are selected first. This minimises biodiversity loss per amount of land cultivated.

iii. Grid cells with lowest biodiversity loss per negative emission potential (species/tonne CO_2-eq.) are selected first. This minimises biodiversity loss per negative emissions achieved.

Prevented biodiversity loss from mitigating climate change

Mitigating climate change can help conserve biodiversity. We therefore contrasted biodiversity loss (i.e., species committed to extinction) due to land-use change for BECCS with an estimate of the prevented biodiversity loss of limiting climate change through BECCS. This prevented biodiversity loss was estimated using Equation 7.1.

$$\mathrm{PBL} = \frac{\mathrm{PBL}}{\Delta\mathrm{T}} \cdot \frac{\Delta\mathrm{T}}{\mathrm{NE}} \cdot \mathrm{NE}$$ **Equation 7.1**

Where: PBL = prevented biodiversity loss (in % of species saved); ΔT = difference in temperature (in ºC); NE = negative emissions (in teratonne CO_2 [10^{15} kg]).

The percentage of species saved per °C of warming prevented (PBL/ΔT) was estimated based on a meta-regression by Urban (2015) that includes various terrestrial species groups such as vertebrates, plants and insects. We included uncertainty ranges based on the reported 95% confidence interval (Table S1), while looking at (pre-industrial) mean global temperature increases of 2.8ºC and 4.3ºC (in line with RCP 6 and 8.5; Clarke et al., 2014). This (asymmetric) uncertainty was modelled here using a lognormal distribution, with percentiles converted to a standard deviation of the log-values following Slob et al. (1994).

The effect of negative emissions on global temperature (ΔT/NE) was based on the transient climate response to cumulative carbon emissions (TCRE) values reported by Van Vuuren et al. (2020) as a normal distribution (0.62 ±0.12 SD °C/Ttonne CO_2). Overall uncertainty in prevented biodiversity loss per negative emissions (expressed as 2.5-97.5% percentile ranges) was determined using the products of 100,000 random samples of both distributions, for each of the four evaluation-period and temperature-scenario combinations.

7.3 Results

Spatial variability in global biodiversity loss from LUC for BECCS-based negative emissions

Figure 7.1 shows the potential global biodiversity loss from LUC (i.e., vertebrate species committed to global extinction) for the production of BECCS-based negative emissions at a given location. Over a 30 year evaluation period (Figure 7.1a), cumulative negative emissions are relatively limited and the biodiversity losses per unit of negative emissions achieved are therefore relatively high. In almost all locations, sequestering one tonne of CO_2 could contribute the equivalent of 10^{-9} species becoming committed to extinction at the global scale, which over larger areas translates to one species per Gtonne CO_2 sequestered. In many tropical regions, however, potential species loss is more than ten times higher.

An 80 year evaluation period results in much larger cumulative negative emissions per area converted and thus less biodiversity loss from LUC per tonne of CO_2 sequestered (Figure 7.1b). Over this 80 year period, BECCS can also generate net negative emissions in more locations. Potential global-equivalent species loss can still be high (1-10+ species/Gtonne) in areas with high (endemic) biodiversity, typically tropical areas, coastal areas and islands, such as in Southeast Asia and Central America.

Geographical patterns of biodiversity loss are similar across the different terrestrial vertebrate classes, except that conversion of cooler areas results in fewer global extinctions of reptile and amphibian species, as fewer of these species are home to these areas (Figure S4). Furthermore, global patterns of potential LUC-related biodiversity loss for negative emissions are more strongly influenced by species loss factors, which vary by five orders of magnitude (Figure S3), than by negative emissions potential (Figure S1).

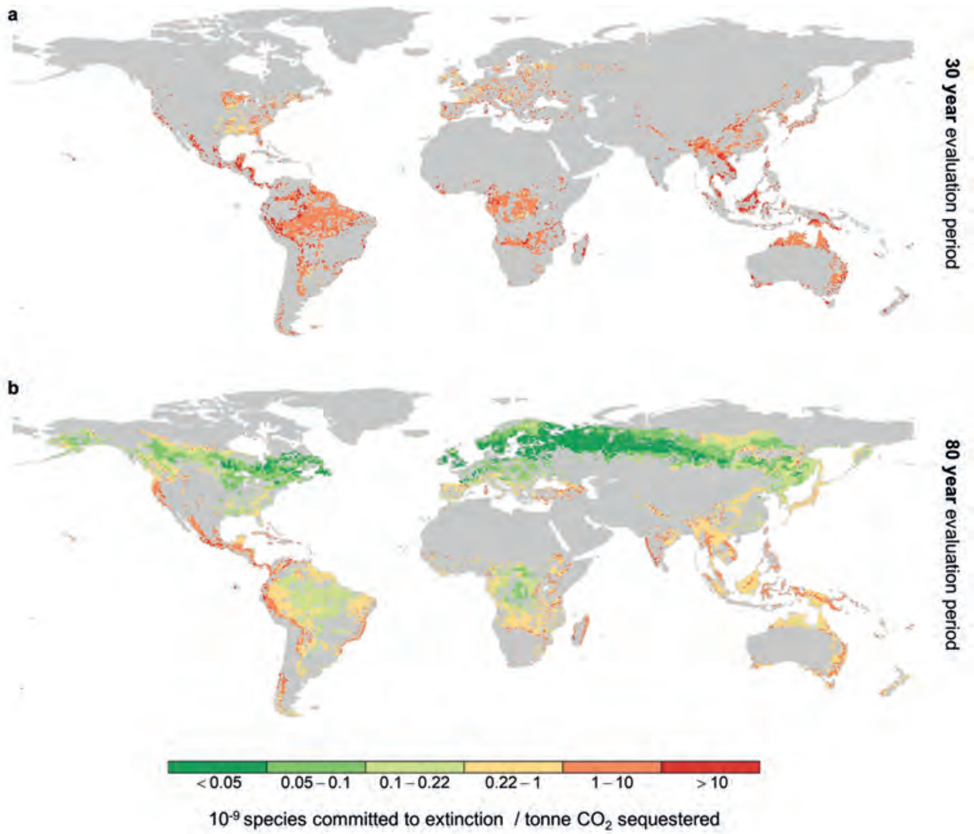

Figure 7.1 | Global-equivalent biodiversity loss associated with land-use change for BECCS-based negative emissions. Indicated are the potential number of terrestrial vertebrate species committed to extinction due to land-use for lignocellulosic crop-based BECCS, expressed in 10^{-9} species per tonne of CO_2 sequestered with BECCS, over **a**. a 30 year evaluation period, and **b**. an 80 year evaluation period. Biodiversity loss and negative emissions per hectare are also separately presented in Figures S1 and S3, respectively. Grey areas were excluded from our analysis and comprise: agricultural land (cropland and pasture), urban areas, inland waters, protected areas, intact forests, areas with low bioenergy crop yields (<5% of global maximum yields) and areas that do not achieve net CO_2 sequestration over the time period considered. This means 389 and 241 ecoregions excluded for the 30 and 80 year evaluation periods, respectively. Note that all protected areas and intact forests (Figure S2) are excluded from our analysis, but that values for grid cells that are partly protected areas or intact forests are plotted on these maps.

The global relation between biodiversity loss from LUC and negative emissions from lignocellulosic crop-based BECCS

BECCS electricity based on lignocellulosic bioenergy crops could sequester around 115 gigatonne of CO_2 over a 30 year period, assuming part of initial vegetation is also used for BECCS electricity. However, achieving this global potential requires all available land that can result in negative emissions through BECCS over the 30 year period (approximately 1.2 Gha). Figure 7.2a shows that conversion of this area into bioenergy crop plantations is (based on our method) expected to result in 760 species becoming committed to extinction (648-889, 95% confidence interval, Figure S5a). This is around 3% of total terrestrial vertebrate species richness and equates to 6.6 (5.6-7.7) species per Gtonne of CO_2 sequestered. Specifically, median species loss includes 321 amphibian species (5% of amphibians), as well as 192 bird (2%), 168 mammal (3%), and 79 reptile (2%) species, which is more than all marine and terrestrial vertebrate extinctions since the 16th century (IPBES, 2019).

At lower amounts of negative emissions, less land is required and different criteria can be employed to select locations for conversion into bioenergy crop plantations. We find that minimising the amount of land used for negative emissions (criterion i), or avoiding the most biodiverse and vulnerable areas (criterion ii), could lead to more global extinctions than prioritising locations with lowest species loss per negative emissions potential (criterion iii; Figure 7.2b). For example, when sequestering 80 Gtonne of CO_2 over 30 years, minimising biodiversity loss per negative emissions results in 137 species committed to extinction (116-164, 95% confidence interval), while minimising land use would double that to 272 species (232-321 95% CI). Geographical patterns of these different land allocation criteria are detailed in Figure S6.

When considering an 80 year evaluation period, BECCS systems can *net* sequester more CO_2, in more areas and over a longer time, resulting in a much larger global sequestration potential. Global species loss per unit of negative emissions achieved is therefore *10-15 times* lower compared to a 30 year evaluation period (compare Figures 7.2a-c) at an average of 0.70 species/Gtonne (0.60-0.82, 95% CI). At lower levels of sequestration, the amount of species committed to extinction per negative emissions achieved is further reduced. For example, sequestering 500 Gtonne of CO_2 over 80 years commits 33 (28-39) species to extinction when prioritising locations with lowest biodiversity loss per negative emissions, that is 0.065 (0.055-0.078) species/Gtonne.

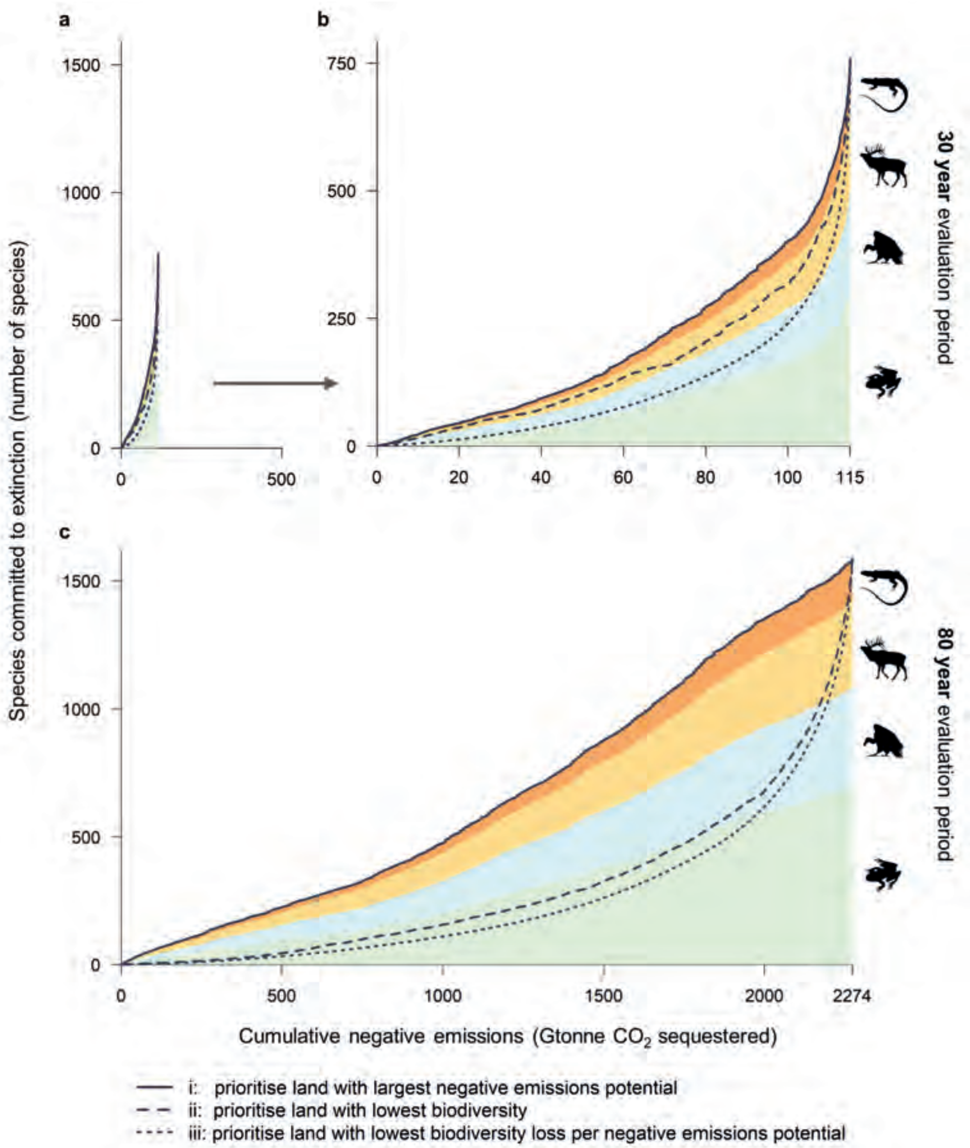

Figure 7.2 | Potential global loss of terrestrial vertebrate biodiversity as a result of land-use change for BECCS. The amount of species that become committed to extinction are shown as a function of cumulative negative emissions from crop-based BECCS. Results are shown for **a**. a 30 year evaluation period, **b**. a scaled version of the 30 year results (note the different axes), and **c**. an 80 year evaluation period. The relation between biodiversity loss and negative emissions differs depending on which land allocation criterion (i-iii) is used. Results are displayed here in different colours for the four classes of terrestrial vertebrates: reptiles, mammals, bird and amphibians, shown as based on criterion i.

The global trade-off between biodiversity loss from LUC and biodiversity conservation through BECCS-mitigated climate change

While land-use change towards bioenergy crop plantations results in biodiversity loss, the climate change mitigation that can be achieved with this crop-BECCS could also *prevent* biodiversity loss. Figure 7.3 shows that the effect of climate change (without any BECCS deployment) could lead to 8-16% loss of global terrestrial (vertebrate) species in 2.8 and 4.3 ^{o}C global warming scenarios, respectively (based on Urban, 2015). More negative emissions from BECCS means increasing effects of land-use change on biodiversity (green solid line), but also decreasing effects of climate change (grey line). Their combined effect can be explored by addition (dotted line), though the true interaction is much more complex, as discussed below.

Over a 30 year evaluation period, biodiversity loss from LUC outweighs prevented biodiversity loss from BECCS-mitigated climate change at all levels of cumulative negative emissions, though both effects are uncertain (Figure 7.3a-b). Biodiversity loss from LUC is exacerbated under other, less optimal land allocation criteria (e.g., criterion i: minimising overall land-use; Figure S7). At higher cumulative negative emissions, more biodiverse land is required explaining the increase in biodiversity impacts from LUC towards the right side of the graphs. The positive influence of climate change mitigation on biodiversity is small, as negative emissions that can be achieved over 30 years are limited.

Over an 80 year evaluation period, more negative emissions can be achieved with BECCS per amount of land used (notice the different scaling in Figure 7.3d-f). This means the climate change mitigation effect on biodiversity is larger and more warming-related species extinctions could possibly be averted. Note that these biodiversity impacts are based on the assumption that negative emissions contribute to 2100 climate targets, which for an 80 year evaluation period implies that all BECCS capacity is in place in 2020. For the 2.8^{o}C baseline scenario, the climate mitigation effect could be larger than the LUC effect on biodiversity under optimal land allocation (criterion iii), though both effects are uncertain (Figure 7.3c). Under other, less optimal land-allocation criteria (e.g., criterion i: minimising overall land-use), LUC effects likely outweigh climate mitigation effects (Figure S7), demonstrating the influence of land allocation. The effect of climate change mitigation on biodiversity is non-linear and strongest when preventing very high temperatures. Therefore, in a scenario of 4.3 ºC warming without the influence of BECCS, the long-term deployment of crop-based BECCS may prevent more species loss from climate change than would be lost due to LUC (Figure 7.3d; dotted line), though these effects and their interaction are uncertain.

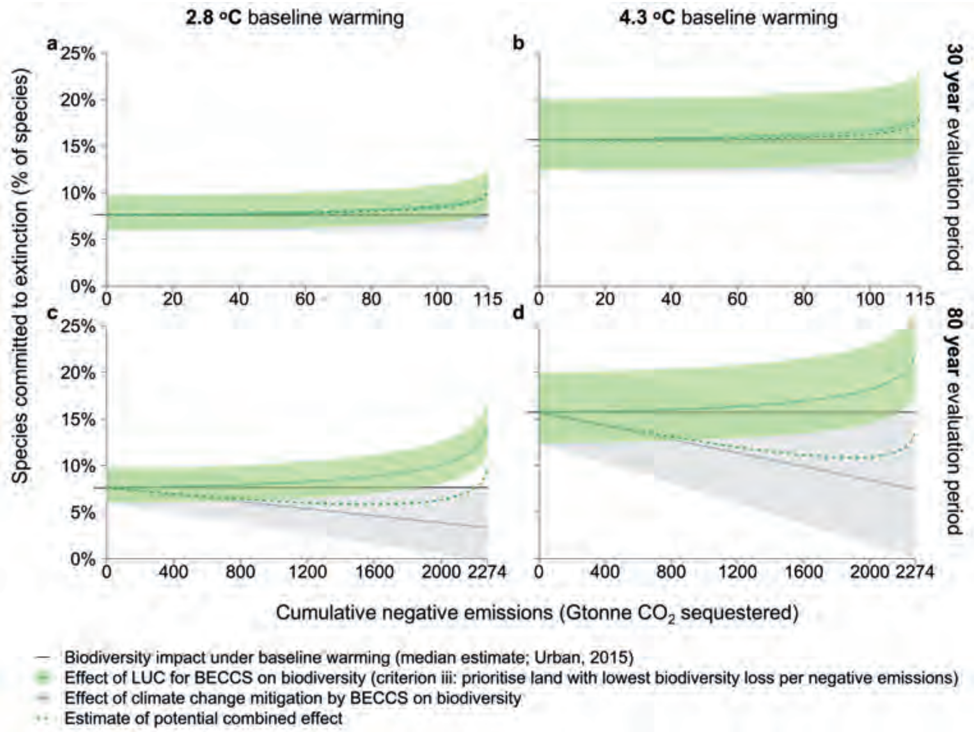

Figure 7.3 | Exploration of the combined effect of land-use change *for* BECCS and climate change mitigation *by* BECCS on global terrestrial vertebrate biodiversity. The amount of species that become committed to extinction is shown as a function of cumulative negative emissions from crop-based BECCS that take place over the specified evaluation period. Results are presented for the use of BECCS over 30 and 80 years (panels **a-b** and **c-d**, respectively; note the different x-axis scaling), and for two baseline warming scenarios: 2.8 ºC and 4.3 ºC warming by 2100, as compared to pre-industrial levels (in line with RCP 6 and 8.5; Clarke et al., 2014). The y-axis intercept shows the assumed biodiversity impact of climate change under baseline warming, without BECCS (based on median estimates by Urban [2015]). With increasing negative emissions from BECCS come increasing effects of land-use change (green line; based on criterion iii: prioritising land with lowest biodiversity loss per negative emissions), but also effects of mitigated climate (grey line). An estimation of their combined (added) effect is shown in the green dotted line, but this excludes any interaction effects. Shading represents the 2.5 to 97.5th percentile uncertainty range for the impacts of land-use change on biodiversity (based on Chaudhary & Brooks [2018]; starting from the uncertainty in the biodiversity impact of baseline warming) and the effect of mitigated climate change on biodiversity (based on Van Vuuren et al. [2020] and Urban[2015]).

7.4 Discussion

The biodiversity impact of LUC for crop-based BECCS

We find that the land conversion required for lignocellulosic crop production for BECCS negatively impacts biodiversity, with on average 6.5 vertebrate species committed to extinction per Gtonne of CO_2 sequestered over a 30 year lifetime, and 0.66 species per Gtonne over an 80 year period. Over both evaluation periods, potential biodiversity loss per unit sequestration is lower at low levels of BECCS deployment, and further reduced when selecting optimal locations (low biodiversity loss per unit sequestration), rather than, for instance, minimising overall land use. On the other hand, biodiversity loss can increase to well over 10 species per Gtonne committed to extinction on highly biodiverse locations, such as tropical islands and coastal areas. When deployed at very large scale, lignocellulosic crop-based BECCS could thus commit hundreds of terrestrial vertebrate species to extinction, representing up to 6% of their overall species richness.

We quantified uncertainty in LUC impacts using the 95% confidence intervals for the global-equivalent species loss factors by Chaudhary & Brooks (2018), representing statistical uncertainty in the underlying species-area relationships. However, impacts of large-scale LUC on *global* vertebrate species richness are inherently difficult to quantify, as empirical data is typically lacking, and not all uncertainties were quantified here. Our use of species loss factors for plantation forestry may, for instance, have led to an underestimation of the biodiversity impacts bioenergy plantations, as these are typically short rotation coppiced trees or Miscanthus with more frequent harvests and disturbances than plantation forestry. On the other hand, alternative biomass production systems could also have lower biodiversity impacts than plantation forestry, as discussed below. Additional uncertainty derives from the scaling factor 'z' that underlies the species-area relationships and species loss factors used in this study, which is differentiated for islands, forests, and non-forests (Chaudhary & Brooks, 2018). SARs may not always be best described by such a power law (Storch et al., 2012), but if described this way, z-values could be also be further distinguished per biome (Kehoe et al., 2017), resulting in a potentially systematic difference with the present analysis.

Beside uncertainty, a further aspect to consider is that biodiversity is multi-facetted (Pereira et al., 2013). The current study focused on global species richness and potential extinctions. This puts emphasis on ecoregions with large amounts of endemic species. Including multiple biodiversity indicators has proven relevant in land-based assessments (Marquardt et al., 2019) and other dimensions of biodiversity that should be included are species abundance and local species richness (Newbold et al., 2015; Ceballos et al., 2017). Their vulnerability to LUC from BECCS could be quantified using recently developed impact factors for land-use and climate change on local mean species abundance (Schipper et al., 2020), allowing a more overarching view of overall biodiversity impacts.

The combined biodiversity effects of LUC and climate change mitigation of BECCS

We tentatively explored the trade-off between species committed to extinction due to LUC for BECCS and the potential species preserved due to BECCS-mitigated climate change. Over a 30 evaluation period, LUC effects most likely outweigh mitigated climate effects for all warming scenarios and land allocation criteria. This suggests that over shorter evaluation periods BECCS has a net negative effect on global vertebrate species richness. Over a longer, 80 year evaluation period, the combined effect of LUC and climate change mitigation is unlikely to be univocal, as it strongly depends on the climate scenario and land allocation criteria assumed. Under optimal land allocation, there could be a small positive effect of BECCS, though there are large uncertainties in the effect of LUC, climate change mitigation and their interaction. For the 80 year results in particular, there is the additional consideration that our biodiversity results assume that *all* negative emissions contribute to 2100 climate targets, i.e. that all BECCS capacity is in place in 2020. When mitigation is achieved later, however, the positive effects of climate change mitigation on biodiversity will be lower.

Both the effects of LUC and climate change mitigation on biodiversity are uncertain. For climate change mitigation, uncertainties were quantified based on the (combined) 95% confidence interval of the species loss meta-analysis (Urban, 2015) and climate response modelling (Van Vuuren et al., 2020) used in this study. The meta-analysis concerns all terrestrial species, including insects and plants. Directly applying these aggregated climate sensitivities to the four terrestrial vertebrate classes adds additional uncertainty, which we could not quantify with the data available. The climate sensitivity of global species richness might also be larger, as suggested by an earlier meta-analysis (Thomas et al., 2004), meaning BECCS' mitigating effect could preserve more species. A promising alternative to meta-analysis is the use process-based approaches to predict the impacts of climate change on biodiversity (Evans et al., 2016; Yates et al., 2018; Bouchet et al., 2019; Briscoe et al., 2019). The climate change mitigating effect of negative emissions from BECCS includes uncertainty ranges (Van Vuuren et al., 2020), but excludes various factors such as carbon cycle feedbacks and non-temperature climate effects. Moreover, the effect of *negative* emissions is inherently more uncertain, as no empirical data exists on large-scale negative emissions.

The combined effect on biodiversity of LUC and climate change mitigation from BECCS was explored by comparing two independently modelled effects. This tentative approach ignores the interaction effects between reduced climate change and enhanced habitat loss. A more accurate estimate of the effect of BECCS or other land-based climate change mitigation measures on biodiversity could be achieved by modelling both land-use change and (mitigated) climate change in conjunction, for instance by modelling how they simultaneously affect species distributions (Visconti et al., 2016; Hof et al., 2018). Using such an approach, Hof et al. (2018) showed for bioenergy *without CCS* that LUC impacts outweigh the climate change mitigation effects on global vertebrate species richness. For BECCS, this integrated species-distribution based trade-off may have a different outcome, owing to BECCS' (much) larger climate change mitigation potential.

Regardless of how the biodiversity trade-off between LUC and mitigated climate change from crop-based BECCS would unfold, mitigating climate change *without* large-scale conversion of natural land would have lower impacts on biodiversity. Agricultural and forestry biomass residues, as well as wastes can provide BECCS feedstocks without additional land requirements, providing up to 30% of total bioenergy feedstock towards 2100 in mitigation pathways (Pour et al., 2018; Hanssen et al. 2019, 2020). Furthermore, sustainable forest management, including selective logging and continuous cover forestry, could provide biomass for BECCS (e.g. Hanssen et al., 2020) with potentially lower biodiversity impacts. In addition, marginal or abandoned agricultural land could be used for crop-based BECCS (Campbell et al., 2008; Gelfand et al., 2013) and cropping could be based here on biodiverse, local and high yielding mixtures of species (Tilman et

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al., 2006; Robertson et al., 2017). Alongside BECCS, other carbon dioxide removal technologies (Smith et al., 2016; Fuss et al., 2018) and renewable energy sources (e.g., Van Vuuren et al., 2018) could provide climate change mitigation, although they too have an impact on biodiversity (e.g., Holland et al., 2019; Popescu et al., 2020).

Conclusions

Based on this study we come to following conclusions:

- Land use-change for lignocellulosic crop-based BECCS can lead to global extinctions of vertebrate species. Considering the quantified uncertainty and the studied variability in BECCS evaluation periods, scale of deployment, and land allocation criteria, the amount of species committed to extinction due to LUC ranges from a minimum of 0.055 to a maximum of 7.7 species per Gtonne of CO_2 sequestered.

- The evaluation period of a BECCS system is a key factor in determining its biodiversity impact. Per negative emissions achieved, fewer species are committed to extinction due to LUC when BECCS systems are operated longer. Short-term operation of BECCS should thus be avoided.

- Which land allocation criterion is used strongly influences biodiversity impacts of crop-based BECCS. It is preferable to select the location with the lowest biodiversity loss per amount of carbon sequestered instead of selecting the locations with the highest amount of carbon sequestered as such.

- Tentative comparison shows that LUC impacts on global terrestrial species richness most likely outweigh the positive effects of climate change mitigation over 30 a year period. This trade-off is less clear over 80 years, though under perfect land allocation there could potentially be a small net positive effect. Both effects *and* their interaction are, however, (highly) uncertain and require additional understanding, along with analysis of additional species groups and biodiversity metrics.

- Factoring in biodiversity means crop-based BECCS should be used as early as possible to achieve required mitigation over a longer time period, on optimal biomass cultivation locations to minimise biodiversity loss, and most importantly, as little as possible where conversion of natural land is involved, looking instead to sustainably grown or residual biomass-based feedstocks and alternative negative emission technologies.

7.5 Acknowledgements

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8 Synthesis

8.1 Introduction

In this thesis I have assessed the potential and trade-offs of using second-generation bioenergy for climate change mitigation. I specifically looked at two research questions:

1. *What are the climate impacts or benefits of current regional bioenergy production in view of other uses of biomass?*

2. *What can bioenergy globally contribute to climate change mitigation over the 21^st^ century, considering biomass supply, negative emission potential and biodiversity trade-offs?*

The first research question was addressed in two regional case studies: Chapter 2 looked at Dutch electricity produced with biomass from pine plantations in the US south-east, and compared this to pulp and panel production or leaving biomass on site. Chapter 3 compared various energy and material applications of residual biomass from Dutch river floodplain management. The methodology used in these case studies was formalised for all forms of (biomass) residues in chapter 4.

The second research question was addressed from a variety of perspectives. Chapter 5 assessed the global amount of biomass residues that is supplied for bioenergy in the 21st century in various (bio)energy demand scenarios across eight integrated assessment models, and compared this to literature-based, bottom-up estimates of residue availability. Chapter 6 analysed spatially-explicitly what global amount of negative emissions can biophysically be achieved with crop-based BECCS for different crops, energy carriers and evaluation periods. This potential was compared to negative emission targets in 1.5 ^{o}C mitigation pathways. Chapter 7 looked in detail at the trade-off between BECCS-based negative emissions and biodiversity loss from land conversion, while also including the potential biodiversity conservation effect of mitigated climate change.

In this chapter, the outcomes of these previous chapters are synthesised. Cross-cutting findings on the climate impacts and benefits of current, regional bioenergy in view of other biomass uses (research question 1) are presented in section

8.2. The potential global contribution of bioenergy to climate change mitigation over the 21st century (research question 2) is discussed in section 8.3. The overall conclusions of this thesis are presented in section 8.4. An outlook on future research and final remarks are given in section 8.5.

8.2 Bioenergy in view of other biomass uses

Bioenergy often leads to GHG emission reductions and therefore climate benefits when the following conditions are met: i) the bioenergy feedstock is residual biomass from forestry, agriculture or landscape management, or is derived from existing biomass plantations, for which it can be assumed that no additional LUC emissions occur due to use of biomass for energy, and ii) a fossil-fuel based, counterfactual energy carrier is replaced. This principle applies to the bioenergy options investigated in this thesis' two regional case studies (chapter 2 and 3). It is also consistent with previous studies on agricultural residues, forestry residues and plantation forestry (Lamers & Junginger, 2013; Gerssen-Gondelach et al., 2014; Creutzig et al., 2015). The size of climate benefits vary, but strongly depend feedstock type, cultivation/harvest location, and the counterfactual energy use that is replaced, as shown in this work and other literature (Lamers & Junginger, 2013; Creutzig et al., 2015). Counterfactuals should therefore always be considered in the assessment of the climate impacts of bioenergy.

The selection of counterfactual is critically important, as it strongly influences the climate impact of the considered biomass use. Accurate counterfactuals are case-specific and depend on: physical properties (what can be replaced), market circumstances (is the replacement economically viable and reasonable), and policy (are measures in place to stimulate or inhibit replacement). What makes a relevant counterfactual for bioenergy or other biomass uses also *changes* over time. If fossil fuels are for instance (locally) phased out, replacing fossil fuels is no longer a realistic counterfactual for bioenergy. Counterfactuals could thus be defined in a dynamic way, as further discussed in section 8.5. A final consideration is that counterfactuals are not always immediately obvious. For instance, leaving biomass on the field may intuitively not have a direct counterfactual, but could enhance soil quality and fertility and therefore reduce artificial fertiliser use.

In assessing bioenergy from residual or existing biomass flows, considering alternative uses or fates of biomass makes for a more comprehensive analysis of the relative climate benefits of all investigated options. In fact, climate benefits of all options can *only* be determined relative to each other, because there is no absolute zero, i.e., no option of not producing these biomass flows. At best, a relative 'zero' option can be defined against which all other uses are compared, for instance based on the previous or most probable use or fate of the biomass flow. Applying this comparative climate impact assessment method in a consistent way requires identifying and quantifying counterfactuals for all alternative uses of biomass. These counterfactuals strongly influence the climate benefits of the alternative biomass uses, as was shown in the case study on Dutch floodplains (chapter 3). Furthermore, different biomass uses also have to be compared per unit of biomass utilised, as the functional units of the different biomass uses vary per type of application. This approach was generalised to determine the environmental impacts of utilising any "residual" flow in chapter 4, and can encompass both by-products (traditional residues), as well as main products that have become redundant due to system inertia. To illustrate this approach of accounting for alternative biomass uses and their counterfactuals, examples from the regional case studies of chapters 2 and 3 are shown in figure 8.1. Beyond the work in this thesis, there are now several examples where this (type of) approach is applied, including for waste gases (e.g., Thonemann & Pizzol, 2019; De Kleijne et al., 2020).

Previous work has shown that climate benefits of bioenergy and alternative uses of biomass strongly vary depending on exact biomass source, conversion pathway and energy carrier or material (e.g., Hoefnagels et al., 2010; Gerssen-Gondelach et al., 2014; Creutzig et al., 2015). The approach derived in this thesis allows direct comparison between different types of biomass use. One generalisation that can be made in this way is that energy applications often have similar or larger climate benefits than material applications of biomass. For example, biomass from pine plantations in the US south-east used for electricity generation in the Netherlands can lead to lower GHG emissions than (a combination of) alternative uses and fates of this biomass, such as paper and panel board production or leaving biomass in the field (chapter 2). This assessment includes both the counterfactual electricity production (the Dutch electricity mix that would be replaced by wood-pellet electricity) and counterfactuals for the alternative uses of biomass (e.g., gypsum board that could be replaced by panel boards). As a second example, residual biomass from Dutch river floodplain management typically leads to larger climate benefits when used for energy (e.g. heat, biogas or combined heat and power) than for materials (chapter 3). The main reason being that bioenergy replaces highly GHG-intensive fossil fuel-based energy. This pattern suggests that in terms of climate, the common paradigm of 'material before energy' uses of biomass (e.g., Ellen-MacArthur Foundation, 2013; Vis et al., 2016; SER, 2020) may be an inaccurate guideline in the current economy and that policy could be based on more comprehensive comparison of material and energy options.

While the approach described above always allows for comprehensive comparison of different biomass uses, it only covers the full climate impact of bioenergy for cases where it can be assumed that biomass is produced anyway. When this assumption does not hold, the climate impacts of bioenergy should be compared to a benchmark of using the land required for biomass growth in a different way (Figure 8.1). With a benchmark of natural vegetation (re)growth, this means the climate impacts of bioenergy should also account for: i) LUC emissions, and ii) the lost capacity of natural vegetation to sequester CO_2 (i.e., 'foregone sequestration').

Whether or not it can be assumed that biomass would already be produced, depends on the decision context and time scale. Biomass from floodplain management is a by-product of a public safety measure and can safely be assumed to be produced anyway. For residual biomass from agriculture and forestry this assumption is typically also made. However, when residues are valorised to a commercial co-product, it can be argued that *part* of any LUC emissions or foregone sequestration associated with the main product should be allocated to the residue-based co-product. It is less clear whether LUC emissions should be included for biomass flows from plantations that have (partially) lost their traditional markets, but continue to produce biomass through economic or physical inertia. This biomass may in the short run be considered as redundant and may be accounted for without a land burden, but evaluation of its prolonged use demands accounting for alternative land-uses.

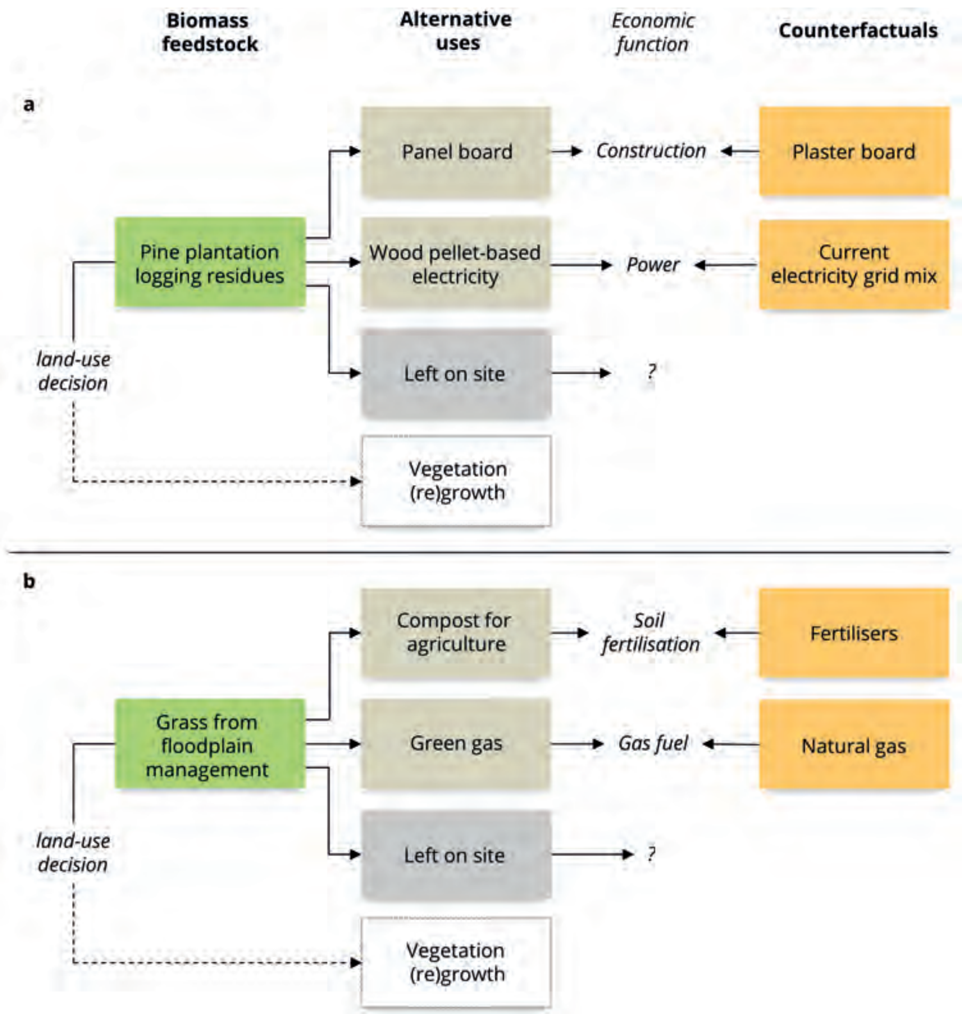

Figure 8.1 | Assessing the climate benefits of bioenergy in light of alternative biomass uses. The approach of assessing bioenergy while taking alternative biomass uses and their counterfactuals into account (chapter 4) is illustrated with examples from: **a**. the regional case study on US south-east pine plantations (chapter 2), and **b**. the regional case study of biomass from Dutch river floodplain management (chapter 3). Note that leaving biomass on the field can have economic functions (indicated with a question mark). Furthermore, the alternative *land* use of natural vegetation (re)growth (indicated here with a dashed line and white box) is important to consider in cases where it cannot be assumed that biomass would be produced anyway.

Generally, the outlined approach is most suitable for bioenergy in local/regional assessments and for the short to medium term (0-30 years). The reasons for this are that the relevant alternative biomass uses, case-specific (dynamic) counterfactuals, and assumptions on degree of redundancy of a biomass flow can more accurately be determined for a specific regional biomass source, in the short to medium term. At a larger spatial and temporal scale, accurately including different options to use biomass, different counterfactuals and different land-uses becomes increasingly difficult. In essence, this is part of what integrated assessment models (IAMs) are designed for, though their spatial, temporal and sectoral aggregation inevitably reduces their level of detail. The simplified, 'counterfactual' approach discussed here may thus provide a valuable additional tool in the assessment of the climate impacts of bioenergy.

8.3 The global climate change mitigation potential of bioenergy

The contribution of bioenergy to the global energy supply and climate change mitigation effort over the 21st century will depend on the availability of biomass. Residual biomass, most importantly from agriculture and forestry, is a good starting point, as it typically leads to lower GHG emission and other environmental impacts and has lower costs than purpose-grown crops (e.g., Whittaker et al., 2011; Creutzig et al., 2015; Boschiero et al., 2016; de Azevedo et al., 2017). Estimates of residue availability for bioenergy in the year 2050 range from 12 to 76 $EJ_{primary}$/year (Fischer & Schrattenholzer, 2001; Hoogwijk et al., 2003; Smeets & Faaij, 2007; Smeets et al., 2007; Hakala et al., 2009; Gregg & Smith, 2010; Haberl et al., 2010, 2011; Cornelissen et al., 2012; Rogner et al., 2012; Lauri et al., 2014; Searle & Malins, 2015; Daioglou et al., 2015a), with a mean of 55 $EJ_{primary}$/year, which equals around 10% of the current global primary energy supply (IEA, 2017b). These so-called bottom-up estimates of residue availability are based on expected trends in population size, diet, consumption, and ultimately agricultural and forestry production. The quantity of residues that can cost-competitively be *supplied* for bioenergy in scenarios for the 21st century varies strongly across IAMs, but for 2050 typically stays within this 55 $EJ_{primary}$/year availability estimate (chapter 5). These model runs also show that residues are typically the first biomass feedstock to be deployed for bioenergy. Residues may thus have a large role in the supply of bioenergy. However, potential logistic and sustainability constraints (Lal, 2005; Janowiak & Webster, 2010; Lemke et al., 2010; Bouget et al., 2012; Lamers et al.

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2013; Liska et al., 2014; Raffa et al., 2015; Poeplau et al., 2015; Repo et al., 2015) warrant additional research. Combining residue use for bioenergy with CCS would further reduce GHG emissions, and would likely result in negative emissions, as shown in chapter 6.

Large-scale deployment of bioenergy, specifically BECCS, is a common feature of many climate change mitigation pathways (Rogelj et al., 2018) and would require purpose-grown bioenergy crops alongside residual biomass. In chapter 6 we showed that lignocellulosic bioenergy crop-based BECCS could biophysically provide large amounts of energy and negative GHG emissions, but requires large amounts of land too. Crop-based BECCS clearly results in more energy supplied and more negative emissions when deployed earlier on and over a longer time period. This is also true for annual amount of negative emissions, as LUC emissions are amortised over longer evaluation periods. Furthermore, more negative emissions are typically achieved: i) when a high carbon capture rate is achieved, for instance during electricity production, ii) when cultivation locations are used with relatively high crop yields and low initial vegetation carbon stocks (often in warmer temperate and sub-tropical areas), as also reported by Elshout et al. (2015), Daioglou et al. (2017) and Fajardy & MacDowell (2017), and iii) when initial vegetation is not burned, but rather used for bioenergy, as also noted by Harper et al. (2018). Land requirements are likely a strong limiting factor on BECCS deployment. For example, when bioenergy crops are deployed alongside residues, 800 Mha of bioenergy cropland (or 5.1% of the global land surface) would be required to reach 88% of projected negative emissions from BECCS in the S2 1.5 ^{o}C mitigation pathway, which would decrease to 61% without residues. Reaching 100%, however, is impossible when following the BECCS deployment rate of this mitigation pathway while using our full life-cycle emission factors and crop yields, as later land conversions would not yield negative emissions on time.

Beyond the biophysical potential of BECCS to mitigate climate change, BECCS' mitigation potential is reduced by: i) environmental concerns, due to its intensive water, land and nutrient use (Kemper et al., 2015; Bonsch et al., 2016; Fajardy & Mac Dowell, 2017; Heck et al., 2018; Stoy et al., 2018; Kato & Yamagata, 2014), ii) socio-political constraints (Fridahl & Lehtveer, 2018), among others due to the challenge of accounting and rewarding negative emissions (Torvanger, 2019; Bednar et al., 2019; Daggash & Mac Dowell, 2019), iii) limitations to the amount of developed geologic storage sites (Scott et al., 2015; Baik et al., 2018; Haszeldine et al., 2018; Turner et al., 2018a), and iv) the challenge of upscaling BECCS orders of magnitude from its current demonstration phase (Haszeldine et al., 2018; van Vuuren et al., 2017; Sanchez et al., 2018; Turner et al., 2018b).

One particular environmental concern related to the large land requirements of crop-based BECCS is the potential biodiversity loss from land conversion (Heck et al., 2018; Núñez-regueiro et al., 2019). Chapter 7 has shown that for terrestrial vertebrate biodiversity these impacts of land-use change should not be neglected. For instance 1-1.6% of terrestrial vertebrate species could become committed to extinction to achieve around 100 Gtonne of negative CO_2 emissions over a 30 year period, and 0.4-2% could become committed to extinction to achieve 1000 Gtonne of negative emissions over 80 years (Figure 7.2). Longer lifetimes of crop-based BECCS systems thus reduce the biodiversity impact per negative emissions achieved. What lands are allocated for bioenergy crop production is also influential. Prioritising locations with lowest biodiversity loss per negative emissions can halve potential biodiversity impacts compared to prioritising land with the highest negative emission potential. Beside these impacts, the contribution to climate change mitigation of BECCS-based negative emissions may also have a *positive* effect on biodiversity conservation. Over a 30 year evaluation period, this effect would be outweighed by the impacts of LUC. Over 80 years, this trade-off is less clear, potentially yielding a small positive effect on biodiversity under optimal land allocation. The exact magnitude of these effects and their interaction is still uncertain, however. In any case, biodiversity impacts could substantially reduce desirable crop-based BECCS potential, while some (other) form of climate change mitigation remains required to prevent the adverse effects of climate change on biodiversity.

Based on the work in this thesis, it becomes clear that bioenergy could contribute to climate change mitigation on a large scale. First, residual biomass from agriculture, forestry, landscape management and various waste streams will likely be a cost-competitive bioenergy feedstock that should be used for bioenergy, where availability (including competition with other uses), logistical and sustainability constraints permit it. Second, bioenergy should be combined with CCS where possible, to enable negative emissions. Maximum negative emissions are achieved via a biomass application with high a carbon capture rate, such as electricity production. A (dynamic) counterfactual should be considered for any final energy carrier though. For example, using biofuels to replace (fossil) kerosene and heavy fuel oil for aviation and shipping could bring larger climate benefits than producing bio-based electricity where it competes with wind and solar power. Third, because of the large land requirements and high potential biodiversity loss associated with land conversion for *crop-based* BECCS, it should be deployed as early as possible, as long as possible, and as limited as possible – alongside other carbon dioxide removal technologies (Smith et al., 2016; Fuss et al., 2016, 2018). Fourth, initial vegetation present before conversion to bioenergy crop plantations should be utilised rather than burned, for instance for additional energy generation. Finally, bioenergy crop cultivation for BECCS can lead to large biodiversity loss. Bioenergy crop cultivation should therefore only be considered on lands with relatively low biodiversity value and high potential for *net* negative emissions, including abandoned agricultural land and marginal lands (Campbell et al., 2008; Gelfand et al., 2013). Cropping could be based here on biodiverse, locally-optimal and high yielding mixtures of species that require less water and fertilisation (Tilman et al., 2006; Robertson et al., 2017). In addition, BECCS could be based on sustainable forestry.

8.4 Conclusions

Based on the findings in this thesis, the following conclusions are drawn on the potential and trade-offs of using second-generation bioenergy for climate change mitigation:

- The use of an accurate counterfactual is of utmost importance in the near-term environmental evaluation of bioenergy and other biomass uses. From these evaluations it can be concluded that bioenergy is a climate-effective way to use residual biomass and often has larger climate benefits than the production of biomaterials.

- Simultaneously considering different applications and their counterfactuals provides a useful framework to comprehensively evaluate the environmental impacts of utilising residual materials, energy or waste flows.

- Residual biomass from agriculture, forestry, landscape management and various waste streams will likely be a cost-competitive bioenergy feedstock in the 21st century that should be prioritised over bioenergy crops, where availability, logistical and sustainability constraints permit it.

- The *biophysical* global negative emissions potential of bioenergy with carbon capture and storage (BECCS) is large, but when based on bioenergy crops strongly depends on the system's lifetime and requires very large amounts of land.

- Large-scale land conversion for crop-based BECCS would have a large impact on global biodiversity. This impact would in most scenarios likely outweigh the potential biodiversity conservation effect of BECCS-mitigated climate change, with a possible exception for BECCS systems with a very long lifetime and under optimal land allocation.

- Because of the land requirements and potential biodiversity loss associated with crop-based BECCS, it should be deployed as early as possible to maximise climate benefits over a long time period, and as limited as possible, alongside other carbon dioxide removal technologies and climate change mitigation strategies.

8.5 Outlook

Future Research

The process of shedding light on older questions inevitably raises new ones. Below I outline several directions for research that could follow up on the work presented in this thesis.

When using the 'counterfactual' approach described in this thesis, the selection of an accurate counterfactual for any energy or material product is crucially important to evaluate its environmental performance. This selection would benefit from a formal standardised procedure. To determine a counterfactual for a product is to answer what a product *would* replace. As discussed in 8.2, standardised counterfactual selection for any product could at least include a product's physical properties (what *could* it replace), market circumstances (is this replacement economically viable and reasonable), and policy context (are measures in place to stimulate or inhibit this replacement). Within these categories various candidate-counterfactuals could be ranked. Dynamic counterfactuals that change over time could also be defined, to reflect (projected) economic changes, or even to form prescriptive “moving targets” with GHG-intensities that decrease over time. Moving-target technology baselines, and the explicit use of counterfactuals in general, could also become part of global climate change mitigation scenarios, as hinted at in recent work by Grant et al. (2020).

Time is a crucial factor in the evaluation of bioenergy. The GHG balance of a bioenergy system changes strongly over time: most of the emissions take place upfront, benefits follow but are spread out in time. This means the GHG balance can be assessed over certain evaluation period (resulting in emission factors), or the break-even time itself can be determined (payback times). These metrics have been used in this thesis, and it has (implicitly) been assumed that emissions and benefits (avoided or negative emissions) can be added up, regardless of when they occur. However, as CO_2 is relatively persistent in the atmosphere, an earlier emission will contribute disproportionally to climate change at a fixed future moment in time. This effect has been well described (e.g., Levasseur et al., 2010; Brandao et al., 2019) and is included in the climate models use to evaluate different mitigation pathways (Rogelj et al., 2018). Its implications for the climate change mitigation potential of BECCS systems, however, have not been comprehensively explored yet, and could provide relevant new insights.

While we compared various biomaterial and bioenergy options on a local scale in the first part of this thesis, a comparison at the global scale has not been made. This would require global assessments of the climate change mitigation potential of various biomaterials. Such assessments have been made for traditional forestry products (Johnston & Radeloff, 2019), building materials (Churkina et al., 2020) and bioplastics (Zheng & Su, 2019), but are lacking for other products like biochemicals. They should also consider possibilities for recycling of materials (Stegmann et al., 2020) and the eventual fate of the biogenic carbon they store. In terms of bioenergy, such a comparison could include work on electricity and fuels presented in this thesis, but also other energy applications including heat or hydrogen production. Systematic comparison of the mitigation potential of these various bioenergy applications and biomaterials could have valuable policy-relevant outcomes. The comparison could be made in terms of potentials, or in a more integrated way, where different applications compete, for instance within an IAM.

In chapter 7, we estimated the amount of terrestrial vertebrate species that could become committed to extinction due to land conversion for crop-based BECCS deployment. We also estimated the potential biodiversity conservation effect of mitigated climate change caused by BECCS. Though the order of magnitude of these effects can be compared, it is inaccurate to simply subtract one from the other, as this would ignore interaction effects between (reduced) climate change and habitat loss, as well as the uncertainties of how climate change and land-use change affect biodiversity (see chapter 7). A better estimate of the effect of BECCS or other land-based climate change mitigation measures on biodiversity could be achieved by modelling both land-use change and (mitigated) climate change in conjunction, for instance by modelling how they affect different species habitat ranges and species distributions (Visconti et al., 2016; Hof et al., 2018).

Considering the clear environmental, socio-political and possibly geological constraints to BECCS, it would be relevant to simultaneously assess the potential of BECCS, other carbon dioxide removal technologies (Smith et al., 2016; Fuss et al., 2016; 2018), possible combinations of technologies, such as enhanced rock weathering combined with (bioenergy) crop cultivation (Beerling et al., 2020), as well as nature-based mitigations strategies. Though presenting a Herculean task, it might provide further insight in what future paths to pursue.

Final remarks

The wide variety of biomass sources and conversion pathways, the resulting differences in GHG balances, and the array of methodological decisions required to integrate these balances into policy-relevant metrics, make the *climate impact* of bioenergy a particularly complex topic. This complexity has led to a heated scientific and societal debate, with some participants arguing to altogether reject bioenergy (e.g., SER, 2020). With regards to this debate, the research presented in this thesis shows: i) the use of bioenergy from various types of currently available residual biomass can result in GHG emission reductions, compared to fossil fuels, and can often outperform biomaterial uses, ii) by 2050, residual biomass could cost-competitively supply around 10% of the current global primary energy supply, and iii) bioenergy with carbon capture and storage from residues and bioenergy crops could biophysically result in negative emissions and contribute to the global climate change mitigation effort in the 21st century, though biodiversity impacts of additional land conversion can be very large. Based on this work, I thus believe that bioenergy should be very critically evaluated with regard to all its constraints and environmental impacts, that additional land conversion for bioenergy should be avoided or very carefully assessed on sustainability, but that bioenergy does have a role to play in the 21st century alongside other technologies and strategies. Altogether rejecting bioenergy could lead to avoidable errors, missed opportunities, and suboptimal management of resources.

This thesis started with a historical perspective on energy. What became clear is that our current fossil fuel addiction is both intense and recent. If a transition to renewable and cleaner forms of energy has indeed gained irreversible momentum (Obama, 2017), fossil fuels may become an historical anomaly, and we will go back to how things have always been: relying on biomass, wind, water and sunlight.

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APPENDICES

APPENDIX I **Detailed documentation of Figure 1.1**

The oldest secure evidence of the use of fire by the genus Homo (specifically H. erectus) is one million years old and described by Berna et al. (2012). The species Homo sapiens is estimated to have arisen sometime between 350,000 and 260,000 years ago (Schlebusch et al., 2017; Hublin et al., 2017; Figure 1.1) and the first anatomically modern humans likely arose around 200,000 years ago. Agriculture is believed to have been independently invented on several locations. The earliest agriculture probably occurred in the Levant, i.e., the eastern Mediterranean and northern Middle East, around 12,000 to 11,500 years ago (Colledge et al. 2004; in figure 1.1 the origin of agriculture is plotted at 11,500 years ago). Agriculture fundamentally changed the way people lived and ultimately enabled complex societies to emerge. It both supplied and increased demand for energy, and it continues to do so to this day. Taking another leap through time, the industrial revolution is generally considered to have started around 1760 AD, powered by the increasingly improved steam engines of the 18th century. Industrialisation drastically increased energy demand, which was ultimately met with fossil fuels, coal at first, followed by oil and natural gas.

The global primary energy supply statistics presented in figure 1.1 were based on several sources. Values up to 1960 were based on Smil (2017) and extracted via Our World in Data (2019). Values for coal, oil, natural gas and hydropower between 1960 and 1990 were based on the BP Statistical Review of World Energy (2018), extracted from Our World in Data (2019). They agree with IEA data (2019) for this period. Energy supplied by renewables and nuclear before 1990 was based on IEA data (2019). From 1990 onwards all values are based on IEA data (2019), with the presented distinction between traditional and modern bioenergy based on several IEA reports (IEA, 2017ab, 2018a). The IEA data and estimates by Smil (2017) for the amount of bioenergy supplied in the period 1960-1990 do not (exactly) match. Therefore, figure 1.1 shows a linear interpolation for this period, i.e., from the 1960 estimate by Smil (2017) to the IEA 1990 value. Modern bioenergy statistics for the period before 1990 are scarce. Figure 1.1 provides a tentative estimate of modern bioenergy supply before 1990 that is based on linear growth from zero at the start of modern bioenergy development in the early 1970s (which saw oil crises and the development of the Brazilian bio-ethanol sector), up to the reported supply of modern bioenergy in 1990 (IEA, 2019).

APPENDIX II Supplementary Information to Chapters II-VII

To save (bio)materials, the Supplementary Information to chapters 2-7 is provided as a single PDF file that is available from the Radboud Repository at **https://hdl.handle.net/2066/230217**.

APPENDIX III Research Data Management

Data used in this thesis can be accessed in the following way:

Chapter 1 No data were produced.

Chapter 2 Hanssen, S.V., Duden, A.S., Junginger, H.M., Dale, V.H. & van der Hilst, F. (2017) Wood pellets, what else? Greenhouse gas parity times of European electricity from wood pellets produced in the south-eastern United States using different softwood feedstocks. *GCB Bioenergy* 9, 1406-1422. doi.org/10.1111/gcbb.12426. All data were published in the article and supplementary information.

Chapter 3 Pfau, S.F., Hanssen, S.V., Straatsma, M.W., Koopman, K.R., Leuven, R.S.E.W. & Huijbregts, M.A.J. (2019) Life cycle greenhouse gas benefits or burdens of residual biomass from landscape management. *Journal of Cleaner Production* 220, 698-706. doi.org/10.1016/j.jclepro.2019.02.001. All data were published in the article and supplementary information.

Chapter 4 Hanssen, S.V. & Huijbregts, M.A.J. (2019) Assessing the environmental benefits of utilising residual flows. *Resources, Conservation & Recycling* 150, 104433. doi.org/10.1016/j.resconrec.2019.104433. All data were published in the article.

Chapter 5 Hanssen, S.V., Daioglou, V., Steinmann, Z.J.N, Frank, S., Popp, A., Brunelle, T., Lauri, P., Hasegawa, T., Huijbregts, M.A.J. & van Vuuren, D.P. (2019) Biomass residues as twenty-first century bioenergy feedstock—a comparison of eight integrated assessment models. *Climatic Change.* doi.org/10.1007/s10584-019-02539-x. All data were published in the article and supplementary information, except for the source data of the article's figures, which are not publicly available. These data are managed by the EMF-33 modelling consortium and hosted by IIASA. For a description of the EMF-33 see doi.org/10.1007/s10584-020-02945-6.

Chapter 6 Hanssen, S.V., Daioglou, V., Steinmann, Z.J.N., Doelman, J.C., van Vuuren, D.P. & Huijbregts, M.A.J. (2020) The climate change mitigation potential of bioenergy with carbon capture and storage. *Nature Climate Change* (10), 1023-1029. doi.org/10.1038/s41558-020-0885-y. All data can be accessed in the DANS EASY archive: [Dataset] doi.org/10.17026/dans-x73-tqeg.

Chapter 7 Hanssen, S.V., Steinmann, Z.J.N., Daioglou, V., Čengić, M., Van Vuuren, D.P. & Huijbregts, M.A.J. Global biodiversity implications of negative emissions from lignocellulosic crop-based bioenergy with carbon capture and storage. *Submitted*. Data will be published in the article and its supplementary information, except data from two key datasets, which are referred to in the article. They are: Chaudhary & Brooks, 2018 (doi.org/10.1021/acs.est.7b05570) and Hanssen et al., 2020 (doi.org/10.1038/s41558-020-0885-y; all data can be accessed in the DANS EASY archive: [Dataset] doi.org/10.17026/dans-x73-tqeg).

Chapter 8 No data were produced.

SUMMARY | SAMENVATTING

Summary

Modern bioenergy is defined as any form of energy outside our bodies that is derived from 'biomass' (i.e., recently living biological materials) and that is subsequently processed into modern energy carriers like electricity and transport fuels. Two main generations of bioenergy are distinguished. First-generation bioenergy is based on food crops that are suitable for bioethanol and biodiesel production. Second-generation feedstocks include: i) wood or herbaceous ('lignocellulosic') biomass that is specifically grown for bioenergy, ii) lignocellulosic biomass that is a by-product ('residue') of agriculture, forestry or landscape management, or iii) organic waste streams, for example from households. These feedstocks are processed into fuels, electricity and heat.

Bioenergy can be renewable and be used in many applications within our existing energy infrastructure. It is also way to make good use of waste and residues. There are sustainability concerns too though, mostly related to the water, land and nutrient requirements of specifically growing crops, trees or grasses to produce bioenergy. In terms of climate change, the uptake of atmospheric CO_2 in growing biomass means that the net greenhouse gas (GHG) emissions of bioenergy can be lower than those of, for instance, fossil fuels. However, GHG emissions from land-use change (LUC), biomass cultivation, processing and transport should also be accounted for in such comparisons. When bioenergy is combined with carbon capture and storage (BECCS), CO_2 produced at the power plant or biorefinery is largely captured and geologically stored rather than emitted back to atmosphere. This can result in so-called net 'negative emissions'.

The main aim of this thesis is to assess the potential and trade-offs of using energy from residual and purpose-grown lignocellulosic biomass to mitigate climate change. I specifically looked at two research questions:

1. *What are the climate impacts or benefits of current regional bioenergy production in view of other uses of biomass?*

2. *What can bioenergy globally contribute to climate change mitigation over the 21st century, considering biomass supply, negative emission potential and biodiversity trade-offs?*

The first line of research (chapters 2-4) assesses the climate impacts of bioenergy by also considering alternative uses of biomass and the counterfactual products they would replace in the conventional economy. This line of research concerns bioenergy from residues and existing tree plantations considered in the short to medium term (0-30 years) and with a regional focus. The second line of research (chapters 5-7) assesses the potential global contribution that bioenergy and BECCS could have towards mitigating climate change in the 21st century, while also exploring its potential biodiversity impacts. It has a longer-term perspective of 30 to 80 years.

Chapter 2 looks at a regional case study of biomass from pine plantations in the US south-east that is converted to wood pellets for electricity generation in the Netherlands. Based on life-cycle assessment, carbon modelling and forestry data, it is shown that the use of this biomass for electricity production can lead to lower GHG emissions than (a combination of) alternative uses and fates, such as paper and panel board production or leaving biomass in the field. The findings account for both the counterfactual electricity production (the Dutch electricity mix that would be replaced by wood-pellet electricity) and counterfactuals for the alternative uses of biomass (e.g., gypsum board that could be replaced by panel boards). Specifically, it is found that the use of thinned trees, logging residues and saw mill residues for electricity has lower emissions within 0-6 years compared to alternative uses. The reason for this delay in climate benefits is that the alternative biomass fates perform better in the short run, due to for instance temporary carbon storage in alternative products or gradually decomposing biomass in the field. For roundwood, on the other hand, achieving net benefits for electricity may take up to 28 years. Overall, the study shows that wood pellet electricity sourced from the right feedstocks can lead to climate benefits in the short run. It also demonstrates that considering alternative biomass uses and their counterfactuals can be critically important in assessing the climate benefits of bioenergy.

Chapter 3 compares the potential climate impacts or benefits of using residual wood and grassy biomass from Dutch river floodplain management for various bioenergy and biomaterial options, accounting for their counterfactuals. The comparison is based on life-cycle assessment and (energy) systems analysis, but also on stakeholder interviews to identify biomass uses and their counterfactuals. Energy applications of biomass, such as biogas from grass or combined heat and power from wood, typically result in larger climate benefits (0-132 kg CO_2-eq. saved per tonne of biomass) than material applications (ranging from 43 kg CO_2-eq. saved to 62 kg of additional emissions per tonne of biomass), or leaving biomass on site (32-176 kg CO_2-eq. emissions per tonne of biomass). Furthermore, woody biomass usually outperforms grassy biomass on a dry tonne basis. Climate benefits are defined here as net avoidance of GHG emissions by replacing a counterfactual. The reason energy applications usually perform best is precisely because their counterfactuals are currently still fossil fuel-based and GHG-intensive. One exception to these findings is the use of grassy biomass to replace peat as a growing medium, which has large climate benefits, due to large impacts of using natural peat. All in all, the analysis shows that including counterfactuals allows for comprehensive comparison of alternative biomass uses and that bioenergy applications from residual biomass can have the larger climate benefits than other uses.

Chapter 4 builds upon chapters 2 and 3 and presents a formalised four-step methodology to assess the environmental benefits of utilising any residual energy or material flow. The approach consists of: i) identifying the residual flow, ii) identifying different uses of a residual flow and quantifying their environmental impacts, iii) selecting an accurate counterfactual for each of these uses, and determining its environmental impacts, and iv) comparing the different uses, including their counterfactuals, per amount of residual flow utilised, and optionally defining one of these uses as a benchmark 'zero' option. This approach allows comprehensive comparison of different applications and optimal use of a residual flow.

Chapter 5 focuses on the global amount of agricultural and forestry residues that can be supplied for bioenergy over the 21st century, based on eight integrated assessments models (IAMs). These IAMs model the economy, biosphere and atmosphere in conjunction and allow exploring what the global energy system and economy would have to look like to achieve climate targets. The comparison reveals large inter-model differences, but generally shows that residues could cost-competitively play a large role in the twenty-first century bioenergy supply. The quantities of residues *supplied* in the models largely fit within previous estimates residue *availability* for bioenergy, which are based on trends in population size and consumption, and average at 55 EJ of primary energy in the year 2050.

Important observed dynamics include that a higher bioenergy demand results in more residues being diverted from competing biomass uses, and that carbon pricing and land protection efforts increase the relative affordability and use of residual biomass. Logistical and wider sustainability constraints of large-scale residue use for bioenergy, however, require additional research.

Chapter 6 provides a global and spatially-explicit analysis of the (negative) GHG emissions of lignocellulosic crop-based BECCS. The analysis is based on the global vegetation model LPJml combined with life-cycle GHG emission data. It shows that crop-based BECCS could biophysically provide large amounts of energy with negative GHG emissions, but requires large amounts of land too. Furthermore, it is demonstrated that (more) negative emissions are typically achieved when: i) cultivation locations are selected with relatively high crop yields and low initial vegetation carbon stocks (often in warmer temperate and sub-tropical areas), ii) initial vegetation is not burned, but instead used for additional bioenergy production, iii) electricity is produced rather than liquid fuels, as it has a relatively high carbon capture rate, and iv) the BECCS system is operated and evaluated over a longer time period. As a final step, it is shown that the projected negative emissions from BECCS in two climate change mitigation pathways for 1.5 ^{o}C (S2 and S5) could biophysically be *approached*. However, considering the potentially very large associated land requirements, substantially less and earlier deployment of BECCS is recommended.

Chapter 7 looks at the global biodiversity implications of negative emissions from crop-based BECCS. The chapter considers the trade-off between biodiversity loss from land conversion to bioenergy plantations, and biodiversity conservation via BECCS-mitigated climate change. The analysis specifically looks at global terrestrial vertebrate species richness, and is based on negative emission data from chapter 6, combined with literature-based global biodiversity loss factors for LUC and for climate change. The results show that land conversion for BECCS has large and uncertain impacts on terrestrial vertebrate biodiversity (0.055-7.7 species lost/ Gtonne CO_2 sequestered, over 30 to 80 year evaluation periods). Biodiversity impacts per negative emissions decrease with: i) longer lifetimes of BECCS systems, ii) less overall deployment of crop-based BECCS, and iii) optimal land allocation (i.e., prioritising locations with lowest species loss per negative emission potential). The effect of biodiversity conservation via mitigated climate change would over a 30 year period be outweighed by the impacts of land conversion for BECCS. Over 80 years, this trade-off is less clear, potentially yielding a small positive effect on biodiversity under optimal land allocation. However, both the LUC and climate effects on biodiversity are (highly) uncertain and hard to compare directly. Overall, crop-based BECCS should thus be used early to maximise mitigation, on optimal cultivation locations to minimise biodiversity loss, and to a limited extent, looking instead to residual biomass-based feedstocks and alternative negative emission technologies.

Chapter 8 discusses and synthesises the main findings of this thesis. The resulting six main conclusions are listed here:

I. The use of an accurate counterfactual is of utmost importance in the near-term environmental evaluation of bioenergy and other biomass uses. From these evaluations it can be concluded that bioenergy is a climate-effective way to use residual biomass and often has larger climate benefits than the production of biomaterials.

II. Simultaneously considering different applications and their counterfactuals provides a useful framework to comprehensively evaluate the environmental impacts of utilising residual materials, energy or waste flows.

III. Residual biomass from agriculture, forestry, landscape management and various waste streams will likely be a cost-competitive bioenergy feedstock in the 21st century that should be prioritised over bioenergy crops, where availability, logistical and sustainability constraints permit it.

IV. The *biophysical* global negative emissions potential of bioenergy with carbon capture and storage (BECCS) is large, but when based on bioenergy crops strongly depends on the system's lifetime and requires very large amounts of land.

V. Large-scale land conversion for crop-based BECCS would have a large impact on global biodiversity. This impact would in most scenarios likely outweigh the potential biodiversity conservation effect of BECCS-mitigated climate change, with a possible exception for BECCS systems with a very long lifetime and under optimal land allocation.

VI. Because of the land requirements and potential biodiversity loss associated with crop-based BECCS, it should be deployed as early as possible to maximise climate benefits over a long time period, and as limited as possible, alongside other carbon dioxide removal technologies and climate change mitigation strategies.

Samenvatting

Moderne bio-energie is elke vorm van energie die verkregen is uit biomassa en verwerkt is tot moderne energiedragers zoals brandstoffen of elektriciteit. Biomassa is een verzamelnaam voor alle organische materialen die geproduceerd zijn door (recent) levende organismen. De zogenaamde eerste generatie van bio-energie is gebaseerd op voedselgewassen die tot bio-ethanol of bio-diesel zijn verwerkt. Voor de tweede generatie worden verschillende typen biomassa gebruikt: i) hout- of kruidachtige ('lignocellulosische') biomassa die specifiek voor bio-energie wordt geteeld (bijvoorbeeld grassen of bomen), ii) lignocellulosische biomassa die als bij-product ('residu') afkomstig is uit landbouw, bosbouw of landschapsbeheer, of iii) organisch afval, zoals mest of GFT. Deze tweede generatie van grondstoffen kan worden gebruikt voor het maken van elektriciteit, brandstoffen of warmte.

Bio-energie heeft een aantal voordelen. Het kan hernieuwbaar zijn, omdat biomassa kan hergroeien. Verder kunnen de geproduceerde energiedragers in onze bestaande energie-infrastructuur gebruikt worden. Bovendien vormt bio-energie ook een manier om afval en residuen nuttig te maken. Er zijn echter ook zorgen over de duurzaamheid van bio-energie. Die gaan met name over het land, het water en de nutriënten die nodig zijn om biomassa te telen voor energieproductie, over de mogelijke destructie van het natuurlijk leefgebied van soorten, en de mogelijke competitie met voedsel.

Wat betreft de impact van bio-energie op het klimaat is het zo dat groeiende planten CO_2 opnemen uit de atmosfeer. Daardoor kunnen de totale broeikasgasemissies van bio-energie lager zijn dan die van fossiele brandstoffen. Echter, leidt bio-energie ook tot emissies als gevolg van het landgebruik en de teelt van gewassen, de verwerking tot energiedragers en hun transport. Die emissies moeten dus ook worden meegewogen in de klimaatimpact van bio-energie. Een speciale variant is de combinatie van bio-energie met de afvang en opslag van CO_2 in de bio-energieketen, bijvoorbeeld het afvangen van de CO_2-uitstoot van biomassacentrales. Door die opslag van CO_2 is mogelijk dat er netto *minder* CO_2 wordt uitgestoten naar de atmosfeer dan dat er is opgenomen door biomassa. Dit noemen we 'negatieve emissies'. Deze combi-variant van bio-energie en CO_2-opslag heet in het jargon 'BECCS' (naar het Engelse: BioEnergy with Carbon Capture and Storage).

Het hoofddoel van dit proefschrift is om beter in kaart te brengen wat het potentieel is van bio-energie om klimaatverandering tegen te gaan (zogenaamde 'klimaatmitigatie'), en welke afwegingen daarbij een rol spelen. De focus ligt op lignocellulosiche biomassa, zowel afkomstig van residuen als ook specfiek voor bio-energie gekweekt. De bijbehorende onderzoeksvragen zijn:

1. *Wat zijn de klimaatimpacts of klimaatvoordelen van regionale bio-energie productie, ten opzichte van andere manieren om biomassa te gebruiken?*

2. *Wat kan bio-energie wereldwijd bijdragen aan klimaatmitigatie in de 21^{e} eeuw, kijkend naar zowel het aanbod van biomassa, de mogelijkheid tot negatieve emissies en de impacts op biodiversiteit?*

In de eerste onderzoekslijn (hoofdstukken 2-4) wordt bio-energie onderzocht naast alternatieve toepassingen van biomassa, inclusief welke producten zij in de reguliere economie kunnen vervangen. In deze onderzoekslijn wordt gekeken naar biomassa uit residuen en bestaande plantagebossen, met een regionale focus en een korte tot middellange tijdshorizon van 0 tot 30 jaar. De tweede onderzoekslijn (hoofdstukken 5-7) onderzoekt de mogelijke bijdrage van bio-energie en BECCS aan klimaatmitigatie in de 21^{e} eeuw en verkent ook de mogelijke invloed op de biodiversiteit. Deze lijn heeft een wereldwijd perspectief en een langere tijdshorizon van 30 tot 80 jaar. Hieronder zijn de verschillende hoofdstukken kort samengevat, inclusief de overkoepelende conclusies die geformuleerd zijn in de synthese van dit proefschrift (hoofdstuk 8).

Hoofdstuk 2 gaat over de klimaatimpact van verschillende typen laagwaardige biomassa uit dennenplantages in het zuidoosten van de Verenigde Staten, waarvan houtpellets (korrels) worden gemaakt voor de stook in Nederlandse elektriciteitscentrales. De analyse is gebaseerd op verschillende technieken, waaronder levenscyclusanalyse (LCA), massabalansen van de koolstofstromen en bosbouwdata. De studie laat zien dat het gebruik van de biomassa uit dennenplantages voor bio-energie tot lagere broeikasgasemissies leidt dan zowel alternatieve gebruiken van deze biomassa (zoals papierproductie of OSB houtpanelen) als het laten wegrotten van biomassa. Voor elektriciteit en andere toepassingen van biomassa is ook rekening gehouden met wat zij zouden vervangen, bijvoorbeeld: fossiele elektriciteit in het geval van elektriciteit uit biomassa, of gipsplaten in het geval van houtpanelen. De precieze resultaten van deze studie verschillen per type biomassa. Voor drie typen biomassa, namelijk dunningshout, resten van het rooien, en residuen uit zagerijen, geldt dat de productie van elektriciteit binnen zes jaar tot klimaatvoordelen leidt ten opzichte van wat er anders met deze biomassa zou gebeuren (materiaaltoepassingen of wegrotten). De reden dat dit even duurt is dat het hout in andere toepassingen vaak tijdelijk is opgeslagen en emissies worden uitgesteld. De vierde en laatste categorie van biomassa is klein rondhout (hele stammen, die ongeschikt zijn als zaaghout). Het gebruik van deze biomassa levert pas later klimaatvoordelen op (in sommige gevallen pas na 28 jaar); in de praktijk blijkt het alternatief voor een groot deel van deze biomassa namelijk te zijn dat de stammen vrij langzaam wegrotten, waarbij de uitstoot dus lang wordt uitgesteld. De uiteindelijke conclusie is dat Nederlandse elektriciteit gebaseerd op deze laagwaardige biomassastromen uit Amerikaanse dennenplantages in veel gevallen op korte termijn tot klimaatvoordelen leidt. Daarnaast is een belangrijke methodologische bevinding dat om bio-energie überhaupt goed te evalueren, alternatieve toepassingen van de biomassa goed in kaart moeten zijn gebracht, én duidelijk moet zijn wat de bio-energie en eventuele alternatieve toepassingen vervangen in de reguliere economie.

Hoofdstuk 3 vergelijkt de mogelijke impacts of voordelen voor het klimaat van verschillende toepassingen van houtige en kruidachtige biomassa uit de uiterwaarden van de Nederlande Rijndelta (Waal, Nederrijn-Lek en IJssel). Wederom wordt in de berekeningen meegenomen wat deze biomassa-toepassingen zouden vervangen in de huidige economie. De analyse is gebaseerd op levenscyclusanalyse en systeemanalyse, maar ook op interviews met verscheidene betrokken partijen (o.a. waterschappen, beheerders en producenten) om duidelijk te krijgen wat de de verschillende toepassingsmogelijkheden van de biomassa zijn en welke producten hiermee in de reguliere economie worden vervangen. Energetische toepassingen, zoals de productie van groen gas of een houtgestookte warmte-krachtkoppeling resulteerden in klimaatvoordelen van 0-132 kg CO_2-equivalent *bespaarde* uitstoot per ingezette ton biomassa. Materiaaltoepassingen, zoals vezel- of compostproductie, bleven daarbij wat achter, variërend van 43 kg CO_2-equivalent bespaard per ton biomassa tot 62 kg CO_2-equivalent aan *extra* emissies per ton ingezette biomassa. Ook het laten liggen van biomassa in de uiterwaarden levert extra emissies op (32-176 kg CO_2-equivalent per ton biomassa; in al deze berekeningen is het feit dat biomassa tijdens de groei CO_2 vastlegt al meegenomen). Een tweede observatie is dat het gebruik van houtige biomassa gemiddeld tot meer klimaatwinst leidt dan het gebruik van kruidachtige biomassa (voornamelijk gras), beiden bekeken per ton droge stof. Zoals gezegd, wordt klimaatwinst van een biomassa-toepassing hier altijd bepaald ten opzichte van het reguliere product dat het zou vervangen. Dat is meteen ook de belangrijkste reden dat energietoepassingen goed scoren: er wordt namelijk fossiele energie vervangen en die stoot veel CO_2 uit. Een materiaaltoepassing die tot uitzonderlijk hoge klimaatwinst leidt, is het gebruik van kruidachtige biomassa om veen te vervangen bij de productie van compost. De winning van natuurlijk veen stoot namelijk veel broeikasgassen uit. Een belangrijke overkoepelende conclusie is dat bio-energie uit deze biomassastromen leidt tot klimaatwinst, ten opzichte van fossiel, maar ook ten opzichte van sommige materiaaltoepassingen. Daarnaast werd in dit hoofdstuk wederom duidelijk dat het voor de analyse van bio-energie belangrijk is om verschillende toepassingen van biomassa systematisch te vergelijken, én daarbij de producten die vervangen zouden worden in de huidige economie mee te wegen.

Hoofdstuk 4 bouwt voort op het werk van hoofdstukken 2 en 3 en formaliseert een nieuwe methode om in bredere zin te bepalen wat de milieu-voordelen zijn van het gebruik van elke vorm van reststromen ('residuen'). Kortgezegd bestaat deze methode uit vier stappen: i) het identificeren van de residuen, ii) het identificeren van de verschillende toepassingen van deze residuen en het kwantificeren van de milieu-impacts van elke toepassing, iii) het bepalen wat elke toepassing zou vervangen in de huidige of toekomstige economie (het zogenoemde 'counterfactual' product), en het kwantificeren van de milieu-impact van elke counterfactual, en iv) het vergelijken van de verschillende toepassingen, inclusief hun counterfactuals, uitgedrukt per hoeveelheid gebruikt residu; eventueel kan hierbij een van de toepassingen aangewezen worden als een 'benchmark' nul-optie. Deze methode maakt het mogelijk om de inzet van reststromen systematisch te analyseren en daarmee eventuele voordelen voor het milieu te maximaliseren.

Hoofdstuk 5 analyseert de hoeveelheid biomassaresiduen uit landbouw en bosbouw die wereldwijd geleverd zou kunnen worden voor bio-energie in de loop van de 21^{e} eeuw. De analyse is gedaan met acht verschillende zogenaamde 'integrated assessment models'. Deze computermodellen modelleren tegelijkertijd de wereldwijde economie, biosfeer en atmosfeer. Ze maken het onder andere mogelijk om te bestuderen hoe de globale energiesystemen en economie er uit zouden kunnen zien om bepaalde klimaatdoelstellingen te behalen. De precieze uitkomsten verschillen sterk per model, maar in de uitkomsten van alle modellen spelen residuen een belangrijke rol in de biomassavoorziening voor bio-energie in de 21e eeuw. Dit komt doordat residuen relatief goedkoop zijn. De gemodelleerde hoeveelheid residuen die geleverd zou worden voor bio-energie passen grotendeels ook binnen eerdere schattingen van de toekomstige *beschikbaarheid* van residuen. Die beschikbaarheid zelf is weer gebaseerd op trends in bevolkingsgroei en consumptie, en bedraagt gemiddeld 55 EJ aan primaire energie in het jaar 2050 (dat is zo'n 10% van het huidige wereldwijde primaire energiegebruik). Naast deze bevindingen zijn er enkele overkoepelende mechanismen waargenomen in de modellen: i) een hogere vraag naar bio-energie zorgt ervoor dat meer residuen worden gebruikt voor bio-energie en minder in andere biomassatoepassingen, ii) Het beprijzen van CO_2-uitstoot zorgt dat residuen relatief nog aantrekkelijker worden dan specifiek voor bio-energie geteelde biomassa, en iii) hogere landprijzen verhogen op eenzelfde manier de relatieve voordelen van residuen. Biomassaresiduen kunnen dus kosteneffectief en waarschijnlijk klimaatvriendelijk een belangrijke rol kunnen spelen in onze energievoorziening. Er is echter meer onderzoek nodig naar de (bredere) duurzaamheid van grootschalig residuengebruik (bijvoorbeeld op het gebied van bodemkwaliteit en biodiversiteit) en naar eventuele logistieke beperkingen van deze decentrale grondstof.

Hoofdstuk 6 analyseert wereldwijd het lokale potentieel van bio-energie met CO_2 afvang en opslag ('BECCS') om tot netto *negatieve* emissies te komen. Specifiek wordt gekeken naar bio-energie op basis van lignocellulosische gewassen (snelgroeiende geknotte bomen en grassen). De analyse is gebaseerd op i) het wereldwijde vegetatiemodel LPJml, dat de gewasopbrengsten voor bio-energie en de mogelijke koolstofverliezen door veranderingen in landgebruik bepaalt, en ii) levencyclusanalyse-data over de emissies en efficiëntie van de bio-energie productieketen. De resultaten laten zien dat deze vorm van BECCS theoretisch (biofysisch) gezien grote hoeveelheden negatieve emissies kan bereiken, maar dat er ook zeer veel land voor nodig is. Daarnaast wordt inzichtelijk gemaakt dat meer negatieve emissies worden bewerkstelligd wanneer: i) biomassa wordt gecultiveerd op locaties waar opbrengsten hoog zijn en er voor ingebruikname voor bio-energie een lage hoeveelheid koolstof lag opgeslagen (vaak zijn dat warme gematigde of subtropische gebieden), ii) de initieel aanwezige vegetatie niet ter plekke wordt verbrand (zonder energiewinning), maar wordt gebruikt, bijvoorbeeld voor bio-energie, iii) elektriciteit wordt geproduceerd in plaats van brandstoffen, aangezien bij elektriciteitsproductie meer CO_2 kan worden afgevangen, en iv) het BECCS-systeem over een langere periode wordt gebruikt (80 jaar i.p.v. 30 jaar), daar de baten dan eerder opwegen tegen de initiële verliezen van koolstofopslag. Als laatste stap in deze studie is het biofysisch potentieel van BECCS, zoals hier bepaald, vergeleken met de hoeveelheid BECCS die wordt gebruikt in twee standaard klimaatmitigatie 'paden' die de werelwijde opwarming beperken tot 1.5 ^{o}C. De hoeveelheid BECCS die benodigd is voor deze paden kan, biofysisch gezien, *benaderd* worden, maar er is enorm veel (deels natuurlijk) land voor nodig. Het valt dus aan te raden om substantieel minder BECCS te gebruiken, maar wel zo snel mogelijk te beginnen om daarmee de baten voor het klimaat te maximaliseren.

Hoofdstuk 7 is gericht op de gevolgen voor de biodiversiteit van het creëren van negatieve emissies met bio-energie en CO_2-afvang en opslag ('BECCS'). Specifiek wordt gekeken naar wereldwijde soortenrijkdom van op land levende gewervelden. Hierbij wordt ook geanalyseerd hoe biodiversiteitsverlies door het aanleggen van bio-energieplantages zich mogelijk verhoudt tot de *beperking* van de biodiversiteitsverlies als gevolg van het voorkómen van klimaatverandering door BECCS. De studie is gebaseerd op de negatieve emissie data uit hoofdstuk 6, gecombineerd met bestaande biodiversiteitsverliesfactoren voor verandering in landgebruik en voor klimaatverandering. De resultaten laten zien dat het in gebruik nemen van land om (lignocellulische) gewassen te telen voor BECCS een grote maar onzekere impact heeft op de biodiversiteit (het mogelijke wereldwijde soortenverlies als gevolg van deze vorm van BECCS ligt in de ordegrootte van 0.55 tot 7.7 soorten per Gigaton vastgelegde CO_2, voor periodes van 30-80 jaar). Per behaalde hoeveelheid negatieve emissies gezien, zijn de gevolgen voor de biodiversiteit minder groot wanneer: i) BECCS-systemen een langere levensduur hebben, ii) er in totaal minder op gewassen gebaseerde BECCS wordt gebruikt, en iii) land op een optimale manier wordt ingezet (oftewel: het prioriteren van locaties met minimaal soortenverlies per behaalde negatieve emissies). Het tegengaan van klimaatverandering met BECCS kan een positief effect hebben op de biodiversiteit, maar over een periode van 30 jaar wordt dit teniet gedaan door de negatieve effecten van de benodigde landgebruiksverandering. Over een periode

SUM

van 80 jaar bezien, is deze afweging minder eenduidig. Mogelijk heeft BECCS dan onder optimale omstandigheden zelfs een klein netto positief (beschermend) effect op de biodiversiteit. De onzekerheden van beide effecten blijven echter groot, en het is moeilijk ze één op één te vergelijken. De conclusie luidt daarom dat BECCS op basis van gewassen het beste snel kan worden ingezet om mitigatie te maximaliseren, maar enkel op optimale locaties om biodiversiteitsverlies te minimaliseren. Bovenal moet BECCS op basis van gewassen beperkt worden ingezet, en in aanvulling op BECCS uit residuen en andere technologieën voor negatieve emissies.

Hoofdstuk 8 vormt de synthese waarin de belangrijkste bevindingen uit de eerdere hoofdstukken worden samengevoegd en besproken. Hieruit volgen de zes hoofdconclusies van dit proefschrift:

I. Om bio-energie en andere toepassingen van biomassa milieukundig te beoordelen is het essentieel om mee te wegen wat deze toepassingen zouden vervangen in de reguliere economie (de zogenaamde 'counterfactual products'). Uit dit type evaluatie blijkt dat bio-energie vaak een effectieve manier is om residuele biomassa in te zetten voor klimaatmitigatie, ook ten opzichte van de productie van bio-materialen.

II. De inzet van reststromen kan op een milieukundig consistente en systematische manier worden beoordeeld door meerdere toepassingen en hun counterfactuals tegelijk te evalueren.

III. Biomassa residuen en organisch afval uit landbouw, bosbouw en landschapsbeheer zullen waarschijnlijk een kostenefficiënte grondstof zijn voor bio-energie in de 21^{e} eeuw. Waar duurzaam beschikbaar zijn deze residuen te verkiezen boven geteelde biomassa voor energie.

IV. De combinatie van bio-energie met CO_2 afvang en opslag ('BECCS') heeft een groot wereldwijd potentieel om negatieve emissies te genereren. Wanneer BECCS gebaseerd is op geteelde gewassen is hier veel land nodig en zijn negatieve emissies sterk afhankelijk van de levensduur van BECCS.

V. Grootschalige veranderingen in landgebruik voor het telen van gewassen voor BECCS zou een grote invloed hebben op de wereldwijde biodiversiteit. Dit effect is dusdanig groot dat bescherming van biodiversiteit door voorkomen klimaatverandering hier waarschijnlijk niet of nauwelijks tegen op weegt, met als mogelijke uitzondering BECCS-systemen met een zeer lange levensduur en een optimale inzet van land.

VI. Vanwege de benodigde hoeveelheid land en het daarmee gepaard gaande biodiversiteitsverlies, moet BECCS zo vroeg mogelijk worden ingezet om klimaatvoordelen over een langere periode te maximaliseren, en zo beperkt mogelijk worden ingezet, naast andere technologieën voor de invang van CO_2 en als onderdeel van een bredere klimaatmitigatiestrategie.

Korte publiekssamenvatting

Biomassa verbranden om de opwarming van de Aarde tegen te gaan?
Biobrandstoffen en elektriciteit uit biomassacentrales zijn niet onomstreden. Wanneer voor de benodigde biomassa landbouwgewassen of bossen gebruikt worden, kan dat nadelig uitpakken voor voedselproductie en de biodiversiteit. Echter, groeiende biomassa neemt ook CO_2 op. Daardoor *kan* de totale klimaatimpact van bio-energie laag zijn, mits het landgebruik en de verdere productieketen van bio-energie weinig broeikasgassen uitstoten.

Dit proefschrift laat zien dat het gebruik van biomassa uit reststromen (bijvoorbeeld van land- en bosbouw) leidt tot een lage klimaatimpact van bio-energie, ten opzichte van fossiel, maar ook ten opzichte van sommige bio-materialen (zoals papier). Hiertoe is een nieuwe rekenmethode ontwikkeld om de toepassingen van reststromen systematisch te beoordelen, inclusief welke reguliere producten zij vervangen. Daarnaast is inzichtelijk gemaakt dat reststromen waarschijnlijk de meest kostenefficiënte vorm van biomassa zijn.

Een tweede onderzoekslijn evalueert de klimaatimpact van bio-energie uit geteelde grassen en houtige biomassa, gecombineerd met de afvang en opslag van CO_2. Deze combinatie-variant (in jargon: 'BECCS') heeft bij een lange levensduur een groot wereldwijd potentieel om netto CO_2 uit de atmosfeer te halen ('negatieve emissies'). Hier is echter veel land voor nodig, met mogelijk grote gevolgen voor de biodiversiteit - zelfs onder optimale inzet van land. De conclusie is daarom dat BECCS uit geteelde biomassa langdurig, maar beperkt zou moeten worden ingezet, naast andere klimaatmitigatiestrategieën.

ABOUT THE AUTHOR

Curriculum vitae academiae

I was born in the beautiful city of Nijmegen on December 11th 1990. At age twelve, I went to the Stedelijk Gymnasium Nijmegen, where I had a thoroughly enjoyable time, made friends for life and learned a lot. After graduating in 2009, I moved to Amsterdam where I earned a BSc in biology (University of Amsterdam, 2013, *cum laude*) and had good times with my fellow biologists. In the spring and summer of 2013, I studied ruins and enjoyed college life at the University of Oxford, the United Kingdom. A country and city I hold dear. Moving back to the Netherlands, I obtained an MSc in Energy Science (Utrecht University, 2015, *cum laude*), which included trips to Oak Ridge National Laboratory and Wageningen University. My PhD project in Nijmegen started in 2016. Four and half years later, I look back at a great and challenging journey that, academically, has culminated with the book in your hands.

Anno 2021, I remain at Radboud University's Department of Environmental Science, starting a new chapter as tenure track assistant professor.

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