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LETTER

Greenhouse gas implications of mobilizing agricultural biomass for energy: a reassessment of global potentials in 2050 under different food-system pathways

Gerald Kalt[1], Christian Lauk[1], Andreas Mayer[1], Michaela C Theurl[1], Katrin Kaltenegger[2], Wilfried Winiwarter[2,3], Karl-Heinz Erb[1], Sarah Matej[1] and Helmut Haberl[1]

[1] Institute of Social Ecology (SEC), Department of Economics and Social Sciences, University of Natural Resources & Life Sciences, Vienna (BOKU), A-1070, Vienna, Schottenfeldgasse 29, Austria

[2] International Institute for Applied Systems Analysis (IIASA) Schlossplatz 1, A-2361, Laxenburg, Austria

[3] Institute of Environmental Engineering, University of Zielona Góra, Licealna 9, PL 65-417 Zielona Góra, Poland

E-mail: gerald.kalt@boku.ac.at

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Supplementary material for this article is available online

Abstract

Global bioenergy potentials have been the subject of extensive research and continued controversy. Due to vast uncertainties regarding future yields, diets and other influencing parameters, estimates of future agricultural biomass potentials vary widely. Most scenarios compatible with ambitious climate targets foresee a large expansion of bioenergy, mainly from energy crops that needs to be kept consistent with projections of agriculture and food production. Using the global biomass balance model BioBaM, we here present an assessment of agricultural bioenergy potentials compatible with the Food and Agriculture Organization's (2018) 'Alternative pathways to 2050' projections. Mobilizing biomass at larger scales may be associated with systemic feedbacks causing greenhouse gas (GHG) emissions, e.g. crop residue removal resulting in loss of soil carbon stocks and increased emissions from fertilization. To assess these effects, we derive 'GHG cost supply-curves', i.e. integrated representations of biomass potentials and their systemic GHG costs. Livestock manure is most favourable in terms of GHG costs, as anaerobic digestion yields reductions of GHG emissions from manure management. Global potentials from intensive livestock systems are about 5 EJ/yr. Crop residues can provide up to 20 EJ/yr at moderate GHG costs. For energy crops, we find that the medium range of literature estimates (~40 to 90 EJ/yr) is only compatible with FAO yield and human diet projections if energy plantations expand into grazing areas (~4–5 million km^2) and grazing land is intensified globally. Direct carbon stock changes associated with perennial energy crops are beneficial for climate mitigation, yet there are—sometimes considerable—'opportunity GHG costs' if one accounts the foregone opportunity of afforestation. Our results indicate that the large potentials of energy crops foreseen in many energy scenarios are not freely and unconditionally available. Disregarding systemic effects in agriculture can result in misjudgement of GHG saving potentials and flawed climate mitigation strategies.

1. Introduction

Substantiated knowledge of renewable energy potentials is pivotal for developing realistic energy and climate-change mitigation scenarios, and for planning energy futures on regional and global scale. The global sustainable potentials of biomass, currently the most important source of renewable energy (IEA 2019, REN21 2019), have been the subject of extensive research (e.g. Fischer and Schrattenholzer 2001, Berndes *et al* 2003, Hoogwijk *et al* 2003, Smeets *et al* 2007, Campbell *et al* 2008, Dornburg *et al* 2010, Haberl *et al* 2010, Haberl *et al* 2011, Deng *et al* 2015, Searle and Malins 2015, Fricko *et al* 2017, Strapasson *et al* 2017) as well as continued controversy (WBGU 2009, Smith *et al* 2014, Robledo-Abad *et al* 2017). Especially,

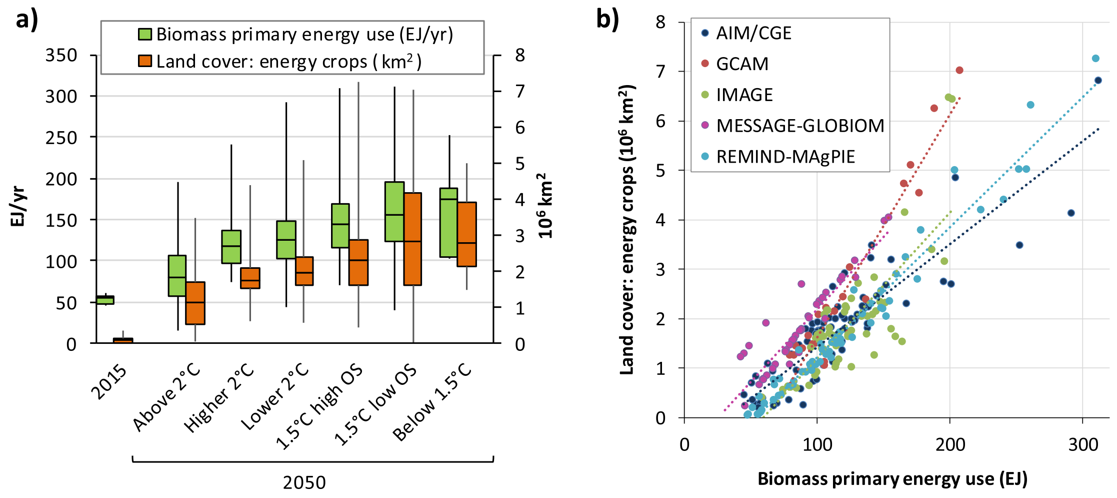

Figure 1. Bioenergy and land dedicated to energy crops in 2050 according to the literature. (a) Biomass primary energy use (primary *y*-axis) and area (secondary *y*-axis) used for energy crop production (2050) in scenarios with different levels of temperature rise. (b) Relationship between biomass primary energy use and energy crop areas in the same scenarios (where available), developed with 5 different integrated assessment models (data from Huppmann *et al* 2018; evaluation and illustration by the authors).

the potential contribution from agricultural biomass, often considered as holding the largest fraction of currently unused biomass resources (see Berndes *et al* 2003, IEA Bioenergy 2007, WBGU 2009, IPCC 2014) seems debatable. While some studies indicate that agricultural intensification will free up large areas for carbon-neutral biomass production while not encroaching into forests, others have shown that meeting the world's future food demand without deforestation could become a major challenge even without dedicating large tracts of land to energy crop production (Tilman *et al* 2011, Erb *et al* 2016b).

Most long-term scenarios towards ambitious climate targets strongly rely on the large-scale implementation of energy crop potentials. Thus determining their magnitude is essential. An analysis of the scenario ensemble (Huppmann *et al* 2018) developed for the IPCC 1.5 °C Special Report (IPCC 2018) and the IPCC Special Report on Climate Change and Land (IPCC 2019b) reveals a clear relationship between global temperature stabilization levels and bioenergy use as well as land being used for energy crop production (figure 1(a)). Moreover, high biomass supply is mainly derived from dedicated energy crops (figure 1(b)) rather than increased forest harvest.

The decisive role of biomass in transformation pathways was also highlighted in the 5th Assessment Report of the IPCC (2014): In most '450 ppm CO_2e-scenarios', substantially more than 100 EJ of biomass are used for energy in 2050 (as compared to 56 EJ in 2017, most of which is traditional fuelwood use (IEA 2019)), and not even one of these scenarios shows biomass use below 200 EJ in 2100. According to IPCC (2019a), the land demand for bioenergy crops in pathways that limit warming to 1.5 °C or 2 °C ranges from 3.2 to 6.6 million km^2 in 2100, representing 20%–42% of current global cropland. Hence, these assessments seem to suggest that vast deployment of bioenergy is indispensable for keeping temperature rise below 2 °C or 1.5 °C (see also Rose *et al* 2014, Rogelj *et al* 2018, Daioglou *et al* 2019), and that considerable land will have to be dedicated to energy crop production to achieve climate policy targets (figure 1, Popp *et al* 2017).

It remains questionable whether large cropland areas can be made available for biomass production without compromising food security. The availability of energy crop potentials depends on various socio-economic developments and underlying narratives; previous studies have identified projections for crop yields and future diets to be among the main reasons for diverging estimates of energy crop potentials (Hoogwijk *et al* 2003, WBGU 2009, Slade *et al* 2014).

The study 'Alternative pathways to 2050', recently published by FAO (2018b), provides an authoritative reference for these parameters that has so far not been utilized for the purpose of assessing agricultural bioenergy potentials. We here present an assessment of sustainable biomass potentials (i.e. respecting environmental limitations and food security) that draws from this source, thereby reviewing results from previous assessments and providing benchmarks for future scenario development. Moreover, the aim of this study is to highlight requirements for and implications of making agricultural biomass available for energy. As Edenhofer *et al* (2011) note, the effects of land use change associated with bioenergy expansion can considerably influence the climate benefit of bioenergy. In other words, mobilizing bioenergy potentials is often associated with greenhouse gas emissions or 'costs' (in the following termed GHG costs) that have been claimed to depend on the magnitude of the bioenergy potential to be mobilized (Haberl 2013).

Reasons for this so far not empirically substantiated claim include carbon stock losses from vegetation or soils that rise with the amount of land dedicated to energy crops, or the need for additional synthetic fertilizer if crop residues are removed from the field in order to use them for bioenergy (Lal 2005, Blanco-Canqui and Lal 2009, Lal and Pimentel 2009, Delgado 2010, Bentsen *et al* 2014).

Moreover, it is insufficient to only appraise GHG emissions directly attributable to biomass supply. In the absence of any land use, ecosystems tend to grow back to their potential carbon stocks (Erb *et al* 2018), or land can be used for other purposes with effects on the overall C budget. These opportunities also need to be considered in a consistent and meaningful account. We here elaborate this notion and demonstrate that a broader perspective that takes into account systemic feedbacks and 'opportunity GHG costs' (i.e. GHG reduction from alternative options, foregone by dedicating land or biomass to bioenergy production (Haberl 2013)) is necessary for understanding climate implications of bioenergy in their entirety.

More specifically, this study aims at substantiating knowledge on the global bioenergy potentials of energy crops, manure and crop residues by taking into account limitations that have so far been disregarded. By deriving 'GHG-cost supply curves' (Haberl 2013), we present a novel concept that holds great value for energy models and integrated assessment models. Economic costs of biomass resources, opportunity costs for diverting agricultural land from production, and other equally important questions related to biomass potentials, such as costs and benefits related to energy and ecosystem services, are not within the scope of this work.

2. Methods

The central tool within the methodological approach followed in this paper is the global biomass balance model 'BioBaM' (Erb *et al* 2016b). BioBaM calculates the balance between biomass supply and biomass demand at the level of 11 world regions, for 14 biomass demand categories and corresponding primary commodities. The model is based on the version applied by Erb *et al* (2016b). For this study, it was extended by modules calculating GHG emissions from the livestock sector, agricultural activities including their upstream GHG emissions (e.g. fertilizer production) as well as changes in carbon (C) stock in biomass and soils resulting from (1) land-cover change (e.g. grassland being converted to cropland), (2) altered grazing intensities and (3) increased removal of crop residues (see below).

In a first step, the base year data underlying BioBaM, originally all referring to the year 2000, are updated to 2012, the base year of the FAO (2018b) scenarios. Our representation of global land use is based on Erb *et al* (2007), updated with cropland data according to FAO (2019b) and new data on intact, unmanaged forest areas (Potapov *et al* 2017). The main reason for not directly using FAO land use data is that grazing land (i.e. land subject to grazing by livestock) is not well represented by the FAO category 'permanent pasture and meadows' because it omits temporary or sporadic grazing. However, non-permanent grazing land plays an important role for livestock nutrition in many world regions (Erb *et al* 2016a). Furthermore, the FAO dataset is characterized by considerable inconsistencies (see Erb *et al* 2007, 2016b, Ramankutty *et al* 2008, Fetzel *et al* 2017b). In consequence, the chosen approach considers, in line with the 'grassland' category by IPCC (2006b), all used or managed lands that are not used for cropping, forestry or infrastructure as subject to a form of livestock grazing and thus assumes these lands to contribute to livestock sustenance.

We run the model with 4 exogenous scenario settings which differ with regard to diets and yields: three scenarios correspond to the FAO scenarios ('Business as usual', BAU; 'Towards sustainability' TSS; 'Stratified societies', SSS). A fourth scenario uses BAU projections for yields but assumes global convergence to a healthy diet ('BAU with healthy reference diet'; Willett *et al* 2019), in order to facilitate insight into possible synergies between health and climate measures and scrutinize the relevance of global food inequalities for the regional distribution and area of cropland potentially available for bioenergy. The scenario data for yields and diets as well as land-use data are provided in the supplementary information is available online at stacks.iop.org/ERL/15/034066/mmedia (SI) to this article.

We consider crop demand for seed, non-food uses other than energy (e.g. fibres) and wasted food by applying constant ratios according to FAO commodity balances (FAO 2019a). To determine the shares of crops being used for non-food/non-energy purposes in the base year 2012, we use data for 'other uses' (FAO element code 5153) and deduct bottom-up estimates for biofuel crops based on REN21 (2013). For sake of simplicity, the ratios of seed, waste and 'other uses' (excluding biofuels) to total crop production are assumed to remain constant until 2050. It could be argued that waste reduction is likely to occur until 2050 (having a positive effect on energy crop potentials), or that the demand for fibres and other industrial non-energy crops could rise steeply due to bioeconomy aspirations (having a negative effect on energy crop potentials), but we consider these influencing factors to be out of the scope of this assessment.

To calculate crop and roughage demand for animal products, we use feed conversion rates according to Bouwman *et al* (2005). Roughage supply potentials from grassland are based on maximum sustainable grazing intensities according to Erb *et al* (2016b), who differentiate between four productivity classes of grazing land (Erb *et al* 2007).

BioBaM produces snapshot scenarios for agricultural production, consumption and GHG emissions in the year 2050, i.e. consistent representations of land use and biomass flows that facilitate the calculation of biomass quantities available for bioenergy and changes in GHG emissions if these quantities are actually supplied. The specific approaches for calculating potentials and GHG costs of each resource type (energy crops, crop residues and manure) are described in the following sections and summarized in table 1.

Important to note, GHG costs, as understood here, do not correspond to supply chain emissions usually considered in life-cycle assessments, nor do they include GHG emissions from harvesting, processing or transportation. Moreover, we (initially) do not account for GHG savings due to avoided fossil fuel combustion (but see Discussion section). GHG costs, as understood in this work, result from systemic feedbacks that are usually disregarded in life-cycle assessments and therefore deserve particular emphasis. In-depth explanations are provided below.

2.1. Energy crops

Dedicated energy crops, specifically fast-growing trees (short rotation coppice) and energy grasses like Miscanthus (summarized as lignocellulosic or 'second generation' energy crops) appear as a favourable biomass resource for two reasons: (1) They provide relatively high yearly biomass yields per unit area. (2) Establishing such plantations on existing agricultural land typically leads to rising natural C stocks (i.e. C benefits additional to those from fossil fuel displacement) rather than a reduction of (possible long-term) C stocks, as is the case when raising harvests in forests.

However, recent studies have highlighted the C benefits achievable through re- or afforestation (Kreidenweis *et al* 2016, Griscom *et al* 2017, Erb *et al* 2018, Fuss *et al* 2018, Houghton and Nassikas 2018, Zhang *et al* 2018, Bastin *et al* 2019, Braakhekke *et al* 2019). This option of using spare land for climate-change mitigation, be it through targeted afforestation or simply by allowing agricultural land to regrow vegetation ('regrowth case'), has been acknowledged as natural alternative to producing biomass for bioenergy (Haberl *et al* 2012, Kalt *et al* 2019). We here consider vegetation regrowth as default counterfactual scenario to energy crop production, and calculate GHG costs as the difference between C stock changes achieved through regrowth (without harvesting) and C stock changes resulting from establishing energy crop plantations (which are harvested periodically). Since aboveground biomass C stocks in energy plantations are largely depleted with every harvest (to supply biomass for fossil fuel displacement) whereas those in the regrowth case continue to rise until they reach a saturation level (unless disturbed by wildfire or extreme weather events), the GHG costs of energy crops are generally larger than zero[4]. GHG costs generally refer to emissions or savings per year; C stock changes are calculated as annual averages during 2012 to 2050 by calculating cumulative values and division by the number of years of our timeframe.

We consider regionally specific energy crop yields, dependence on soil types, climatic conditions and the type of 'natural' vegetation in the respective area and determine representative values for each world region and type of agricultural land by calculating weighted averages on the basis of spatially explicit data. For a detailed description of the methodological approach, see Kalt *et al* (2019) and SI.

The land area available for energy crops in a specific scenario strongly depends on how much grazing land is managed/utilized (see IRENA 2018). We do not allow agricultural land (cropland and grazing land) to encroach into forests because maintaining forest areas is a core environmental objective for various reasons and freeing up land through deforestation is not compatible with our ambition to determine sustainable potentials. However, additional cropland can be made available by land-use change of suitable grazing areas (referred to as highly productive or 'class 1' grazing land' Erb *et al* 2007). All 'grazing land' is assumed to contribute to feeding livestock, although in some places with very low grazing intensity (stocking density). Grazing intensity is calculated as the fraction of aboveground plant growth fed to livestock (Petz *et al* 2014, Fetzel *et al* 2017a). Due to the consistent biomass and land balance approach, all land areas classified as 'degraded', 'unused' or 'wasteland' in other bioenergy studies are considered.

We calculate energy crop potentials under three different premises (figure 2): (a) no grazing intensification; (b) grazing intensification on highly productive sites; and (c) universal grazing intensification. The topmost panel shows a simplified representation of the initial situation, where agricultural land consists of cropland, highly productive and low-productivity grazing land[5]. This initial situation corresponds to the base year 2012 and is calibrated to actual supply and consumption data. Under (a), cropland only becomes available for energy crops if the cropland area required to satisfy food demand decreases[6]. In the

[4] C benefits from fossil fuel displacement are not considered at this stage, so this does not imply that vegetation regrowth leads to higher C benefits than energy crop production; it merely means that the aggregated C stock changes resulting from regrowth are larger than those from energy plantations.

[5] The actual representation of grazing land in the BioBaM model comprises all four classes according to Erb *et al* (2007).

[6] On a global scale, this is not the case for any FAO scenario; on a regional scale, however, this is the case for Eastern & Southeastern Europe and Northern America, for example. We here assume 'regional spare land' as being available for energy crop production, although one could argue that considering a global land balance would be more appropriate for a global assessment. (Under that strict assumption, the energy crop potentials under the 'no grazing intensification' would be zero for all FAO scenarios and only about 3 EJ/yr in 'BAU with healthy reference diet').

Table 1. Main influencing parameters on the availability and GHG costs of the considered biomass resources; modelling approaches and data sources.

Biomass resource	Main influencing parameters on potential	Modelling approach and data for calculating potentials	GHG implications	Methods and data for calculating GHG costs
Energy crops	• Land demand for food, feed and other industrial/non-energy uses • Availability of agricultural land suitable for cultivating energy crops • Energy crop yields • Extent of grazing intensification • Fertilization	• Projections for diets and yields according to FAO (2018); one additional scenario based on a hypothetical global convergence to the healthy reference diet according to (Willett *et al* 2019) • Regionally specific energy crop yields (short rotation coppice) according to Kalt *et al* (2019) • Average calorific value of biomass: 18.5 GJ/ton dry mass (Klass 1998, Haberl and Erb 2006) • Different assumptions regarding grazing intensification (3 cases; see text and figure 2)	• Energy plantations (SRC) usually have larger C stocks per unit area than annual cropland and grassland. Hence, establishing energy plantations leads to C sequestration. • The 'counterfactual scenario', i.e. the missed opportunity of freeing up land for vegetation regrowth is factored in; compared to this alternative, energy plantations have lower C stocks (see Kalt *et al* 2019). • Loss of soil organic carbon (SOC) in grazing areas if land is made available for energy crop production through grazing intensification • Fertilization of energy crops	• Carbon stock changes in soil, litter, above- and belowground biomass (methods and parameters based on (IPCC 2006b), see SI): (i) 'Bioenergy case': conversion of cropland/grassland to energy plantations (short rotation coppice) (ii) 'Regrowth case' (counterfactual scenario): natural succession[a] • Loss of SOC resulting from increased grazing intensity is calculated based on (IPCC 2006b) methods and data; direct correlation between grazing intensity[b] and degradation levels (see table 6.3 in IPCC 2006b) • Difference between (ii) and (i) corresponds to GHG costs of energy plantations • GHG emissions from fertilization of energy crops is disregarded as SRC is usually not fertilized (Dimitriou and Rutz 2015)
Crop residues	• Crop production • Residue-to-product ratios (RPR) • Maximum sustainable removal rates • Types of residues: (dry straw, stover etc for burning; leaves etc for anaerobic fermentation) • Other competing uses (e.g. feed, bedding)	• Projections for crop yields, diets etc (see above) • RPR and residue demand for non-energy uses (feed and animal bedding) based on Krausmann *et al* (2008); future RPR for cereals in presently low-yielding regions calibrated to historical data • Two maximum removal rates are assumed: 40% (low estimate) and 60% (high estimate) • Assumed heating values: Dry residues: 18.5 GJ/t_{dry} Wet residues: 264 m^3 methane/t_{dry} corresponding to 9.5 GJ/t_{dry} (based on KTBL 2019)	• Reduced residue input to cropland results in loss of SOC (i.e. C stock reduction; depends on climate zone, soil type etc) • Nutrient loss (to be compensated by synthetic fertilizers) • Reduced N_2O emissions from residue application to soils • Increased N_2O emissions due to additional N fertilizer application • Methane leakage in anaerobic digestion plants using wet residues	• Loss of SOC due to residue removal is calculated based on (IPCC 2006b) methods and data (see SI) • Methane leakage in anaerobic digestion plants: assumed 1% of methane yield; see Liebetrau *et al* (2017) • Emissions from residue/fertilizer input determined according to (IPCC 2006b) Tier 1 approaches • Upstream emissions from fertilizer production are based on Frischknecht *et al* (2005) and Wernet *et al* (2016)

Table 1. (Continued.)

Biomass resource	Main influencing parameters on potential	Modelling approach and data for calculating potentials	GHG implications	Methods and data for calculating GHG costs
Livestock manure	• Livestock, determined by animal product demand (number and types of animals) • Housing conditions and intensity determine feasibility/likelihood of utilization	• Assumed methane yields (m^3 per tonne of volatile solids in manure): pig manure: 300 m^3/t_{VS}; poultry manure: 320 m^3/t_{VS}; ruminants: 209 m^3/t_{VS}; Calorific value of methane: 36 MJ m^{-3} (based on Scarlat *et al* 2018, KTBL 2019) • Starting with the total (theoretical) manure potential, we apply limitations reflecting feasibility: manure from indoor housing only; and manure from intensive systems only	• Reduced N_2O and CH_4 emissions from manure management (depend on default livestock/manure management systems) • Methane leakage in anaerobic digestion plants • Possibly reduced N_2O emissions if digestates are applied as fertilizers instead of manure	• GHG savings due to anaerobic treatment displacing conventional manure management systems are calculated according to (IPCC 2006b) methods based on the situation in 2012 (housing and manure management systems) • Methane leakage in anaerobic digestion plants: assumed 1% of methane yield; see Liebetrau *et al* (2017) • We assume that digestates are generally applied as fertilizer (instead of the processed manure); in want of conclusive data, N_2O emissions associated with digestate and manure application are assumed to be identical

[a] 'Natural succession' here means that vegetation is assumed to be allowed to regrow without intervention, as opposed to planting of specific tree species. The notion of 'natural succession' is widely used in the bioenergy literature and does not imply that the emerging vegetation were 'natural vegetation' in the sense of vegetation-ecological concepts.

[b] Grazing intensity is defined as ratio of grazed or mowed biomass to net primary production (Erb *et al* 2016b).

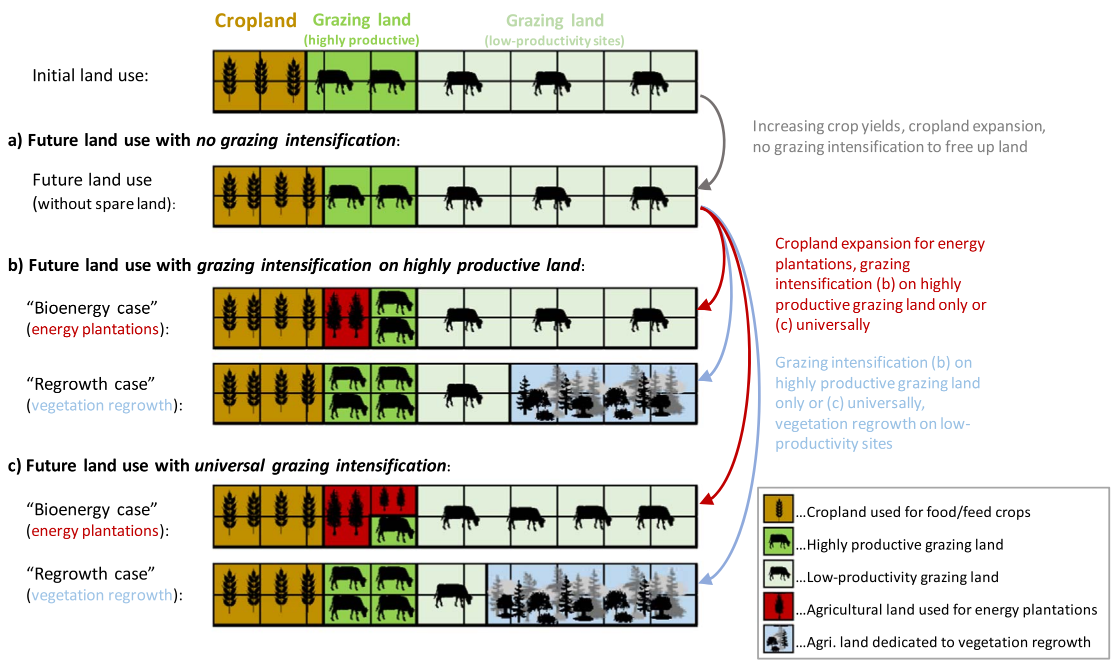

Figure 2. Illustration of the core assumptions related to land use. Note that area sizes and grazing densities are purely hypothetical.

counterfactual case ('regrowth case'), the same area that is available for energy crops is assumed to regrow vegetation that is not harvested. Under the assumptions (b) and (c), the counterfactual cases are characterized by more complex considerations, because the assumption that the same (highly productive) areas as in the bioenergy case would be set free in the regrowth case under intensification pressure would be against economic rationale: in order to free up land for energy crop production, grazing can be intensified up to a maximum sustainable level. This could happen either on highly productive grazing land only (figure 2(b)), or universally (figure 2(c)). In the latter case, grazing is shifted from highly productive land to less productive, in order to maximize the area for energy crops. The counterfactual (regrowth) cases are shown below the respective 'bioenergy cases': here, increased grazing intensities on highly productive sites set free less productive areas (which are least profitable), facilitating vegetation regrowth. Regrowth areas are larger than energy crop areas, but naturally also less productive, i.e. accumulate less C in biomass than more productive land planted with energy crops in the counterfactual scenario would absorb.

With regard to energy crop yields, we assume the regionally specific values according to Kalt *et al* (2019). With scenario-specific global weighted averages around 10 tonnes dry mass per hectare and year, these yields are in the medium range of literature estimates (see IEA Bioenergy 2007, Berndes *et al* 2003, Slade *et al* 2011, 2014, and the SI to Kalt *et al* 2019).

2.2. Crop residues

Assessments of crop residue potentials are usually based on crop production, crop-specific residue-to-product ratios and maximum sustainable removal rates. The assumed sustainable removal rates vary widely: a literature review in Scarlat *et al* (2010) shows values ranging from 15% to 82% (most values being in the range of 30 to 60%), depending on crop type and tillage practices. Bentsen *et al* (2014) cite rates 'between nothing to everything, with a trend towards recovery rates [...] between 25% and 60%'. The literature is, however, inconclusive in terms of crop-specific sustainable removal rates, and IPCC (2006b) methods suggest that the amount of residue input to (or removal from) fields is just one factor affecting SOC stocks but generally has an influence on SOC (Monforti *et al* 2015).

We here consider sustainable removal rates of 40% ('low estimate') and 60% ('high estimate'). We use IPCC (2006b) Tier 1 default methods and parameters to determine the loss of SOC resulting from residue removal for bioenergy (tillage practices are assumed to remain unchanged). This SOC loss is part of the GHG costs of crop residue potentials and depends on site-specific parameters like soil type and climate (see SI for detailed methods). GHG emissions from additional fertilizer demand required for a balanced nitrogen cycle are also taken into account (i.e. upstream emissions of synthetic fertilizer production, as N_2O emissions from fertilizer application are offset by reduced N_2O emissions from residues left on the field). For wet residues (leaves etc from pulses, roots and vegetables), which are assumedly always converted to biogas in case of energy use, we further consider methane

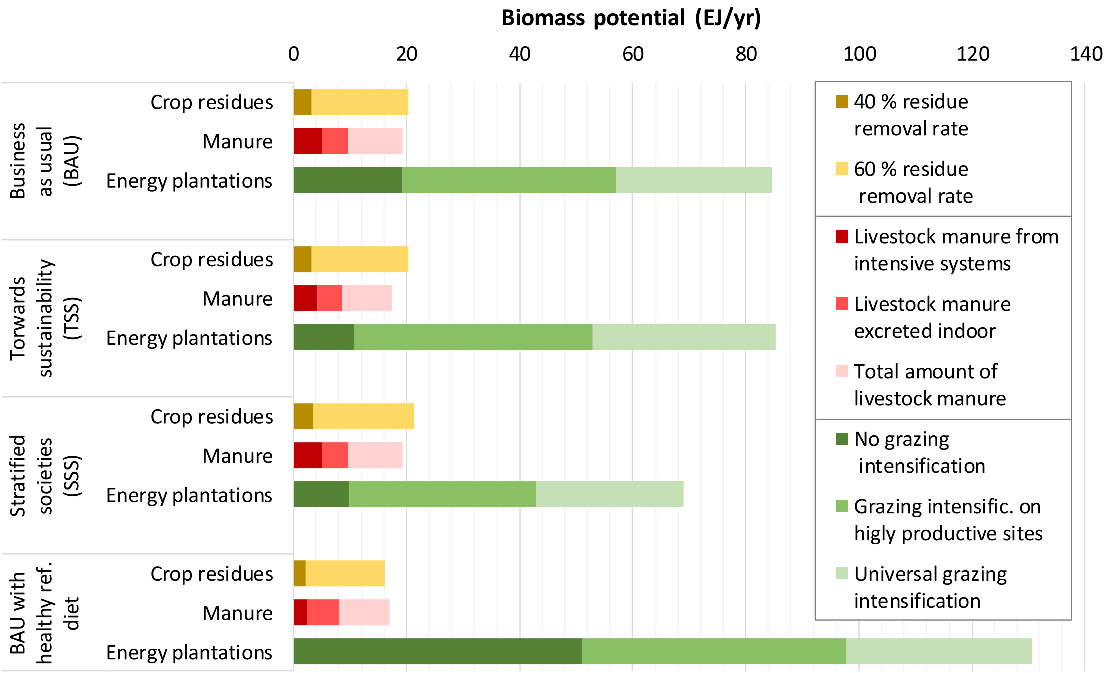

Figure 3. Agricultural biomass potentials in the scenarios based on FAO (2018a) yields and diets, and a BAU scenario assuming global convergence to a healthy reference diet. BAU, TSS and SSS differ with regard to diets and yield projections. The categories are ordered according to their degree of restrictiveness, with most likely or most easily accessible potentials ranked first and represented by bars in dark colours; less likely or less easily accessible categories are shown as additional amounts in brighter colours. The energy crop potentials in the 'regrowth case' (see below) are zero by definition.

leakage in anaerobic digestion plants as GHG costs. Dry residues (straw from cereals, oilcrops etc) are assumed to be used in combustion plants, causing CO_2 emissions that correspond to the CO_2 sequestration during plant growth, i.e. are not assumed to bear additional GHG costs.

In the literature it is sometimes suggested that since SOC stocks are more sensitive to tillage practices than residue removal (Daioglou *et al* 2016), C stock impacts can be offset by adopting SOC-enhancing management practices. This, however, bespeaks a flawed counterfactual assumption: changing tillage practices is an option for raising C stocks in soils regardless of whether residues are removed for energy or not. For assessing the GHG impact of removing residues for energy, we must assume the same tillage management in the 'bioenergy case' as in the counterfactual case. Otherwise, we would disregard opportunity GHG costs.

Following common practice (e.g. Scarlat *et al* 2010, Searle and Malins 2015), non-energy uses of residues (mainly feed and animal bedding) are prioritised over bioenergy. Hence, only the difference between the amounts determined from maximum removal rates and demand for non-energy uses is considered as bioenergy potential. Assuming that current practices in residue use (specific to each world region, based on Krausmann *et al* 2008) are maintained, this considerably limits the quantities available for bioenergy.

2.3. Livestock manure

Bioenergy potentials from livestock manure are calculated as biogas potentials using specific biogas yields for poultry, pig and ruminant manure (see table 1). We estimate theoretical potentials (all manure excreted by livestock) as well as technical potentials, assuming two different limiting factors: first, we assume that only manure from indoor housing is available for anaerobic digestion, and second, we assume that this is only the case for manure from intensive livestock systems. The respective world region specific shares are adopted from the data basis of the GAINS model (Amann *et al* 2011, Winiwarter *et al* 2018) and Robinson *et al* (2011) and Lowder *et al* (2016), respectively, and are provided in the SI.

With regard to GHG costs or implications, we consider CH_4 and N_2O emission reductions resulting from default manure management being replaced by anaerobic digestion, and methane leakage in anaerobic digestion plants. With regard to management systems, we differentiate between pasture, liquid and solid systems and apply emission factors according to Winiwarter *et al* (2018) (N_2O) and the GLEAM model (FAO 2018a) (CH_4). Data on the distribution among systems are adopted from the GAINS model and are specific for each world region.

Two particularities about the GHG costs of manure are noteworthy: First, due to the fact that avoided emissions clearly exceed methane leakage, the GHG costs are negative (i.e. GHG savings). Second, in contrast to the GHG costs of energy crops and residues,

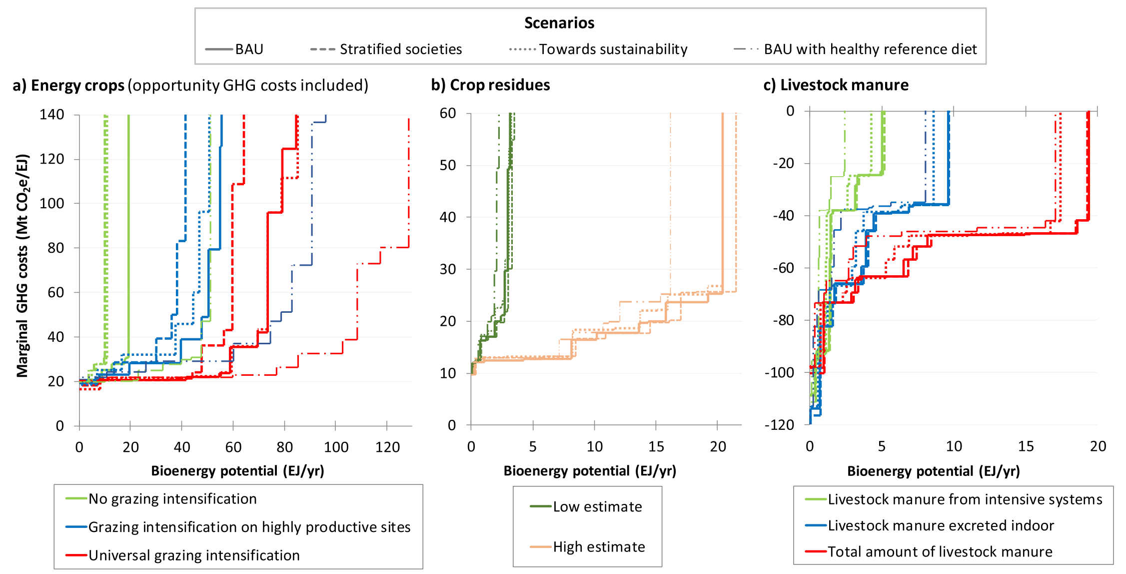

Figure 4. Global GHG-cost supply curves for energy crops, crop residues and livestock manure in four scenarios. Note the different ranges of the *y*-axes, and especially the negative values in panel c. GHG costs of energy crops (panel a) are due to 'opportunity GHG costs', resulting from the foregone opportunity of vegetation regrowth on agricultural land. GHG cost curves without opportunity costs are provided in the SI.

these GHG savings are usually taken into account in life-cycle assessments.

The following table summarizes the main influencing parameters on the availability and GHG implications of mobilizing each resource. Moreover, our modelling approaches and data sources for calculating availability and GHG impacts are specified.

3. Results

3.1. Potentials

For the three scenarios based on FAO projections, the agricultural bioenergy potentials do not differ significantly; especially not with regard to crop residue and manure potentials (figure 3). In the BAU scenario, which is based on the narrative of the 'SSP2 scenario' (O'Neill *et al* 2017) and assumes failure in addressing challenges for food access and stability (FAO 2018b), the *energy crop potentials* without grazing intensification are close to 20 EJ/yr and about twice as high as in the 'Towards sustainability' (TSS) and the 'Stratified societies' (SSS) scenario. Grazing intensification potentials are similar in BAU and TSS (up to 85 EJ/yr), but significantly lower in SSS (up to 69 EJ/yr). The reasons are higher food crop yields in BAU as compared to TSS and less meat-rich diets (as compared to SSS) in highly developed world regions (see SI), resulting in more spare cropland in 2050. Under the assumption of global convergence to a healthy diet, energy crop potentials are significantly larger than in the other scenarios.

Energy production potentials from *manure* reflect the structure of livestock systems and differences between the scenarios in animal product consumption: the total amount of manure corresponds for close to 20 EJ/yr in BAU and SSS. About half of that is attributable to indoor housing and one fourth to intensive systems. In TSS, which assumes 'virtuous social, environmental and economic dynamics' and 'fairly generalized [food] equity' (FAO 2018b) and even more the healthy diet scenario, animal product consumption declines in highly developed and increases in least developed regions. In total, the manure potential in these scenarios is lower than in BAU and SSS, and shifted regionally and in terms of livestock systems (lower proportion of intensive systems).

Crop residue potentials are about 20 EJ/yr in all FAO-based scenarios if 60% sustainable removal rate are assumed and less than 4 EJ/yr for a removal rate of 40%. Due to generally lower crop production in the 'healthy diet scenario', this scenario shows a lower residue potential despite a reduced residue demand for feed and animal bedding. All resources combined, agricultural biomass potentials are lowest in the 'Stratified societies' scenario, based on a narrative of self-protected elites, failure in conserving natural resources and mitigating climate change, increased poverty, food insecurity and poor nutrition.

Results on the level of world regions are provided in the supplementary information.

3.2. GHG-cost supply curves

By associating the different types of biomass potentials with their respective GHG costs (see table 1), ranking them in ascending order and drawing them as stepped curves, we arrive at the GHG-cost supply curves shown in figure 4. The contribution of each of the 11 world region is shown individually, so the curves are composed of up to 11 'steps' (each step represents one world region; potentials may be zero in individual regions). GHG costs are determined as average values per energy unit, i.e. as total difference in GHG emissions to the counterfactual case divided by the total potential.

The GHG costs of energy crops are typically low if no grazing intensification is assumed (figure 4(a)). Through intensification, larger potentials are made available, but usually also at higher GHG costs: apart from intensification itself (which leads to C stock loss on grazing land), claiming highly productive areas for energy crops shifts land-use patterns in a way that is often (albeit not necessarily) detrimental to C balances in the 'bioenergy case', as compared to the 'regrowth case'. The numerical results depend on numerous parameters, e.g. initial and maximum grazing intensities on the region's grazing areas, the area distribution among grazing classes (see SI) and influencing factors on C stock changes resulting from land-use change at a specific site (e.g. soil type, climate zone, speed of C sequestration in vegetation). This explains the wide range of GHG costs among scenarios and the fact that changes in GHG costs from one scenario to another are not uniform. Nevertheless, it can be concluded that intensification, while making area available for energy crops, also leads to increasing GHG costs. Full exploitation of energy crop potentials therefore appears as inefficient in terms of climate effects, as GHG costs are partly within the range of combustion emissions of the most relevant fossil fuels (natural gas: 56 Mt CO_2e/EJ; lignite: 101 Mt CO_2e/EJ (IPCC 2006a)). The curves illustrate that if the missed opportunity of vegetation regrowth (i.e. opportunity GHG costs) is factored in, energy crop potentials start at GHG-costs of 20 Mt CO2e/EJ and may easily rise to much higher values.

Crop residues are usually more favourable in terms of GHG costs (figure 4(b); note the different scaling of axes). Differences in GHG costs between world regions and scenarios are mainly due to regionally diverse shares of crops[7] and different soil and climate types affecting the extent of soil C stock loss when residues are removed for bioenergy. Typical GHG costs of crop residues are estimated between 10 and 25 Mt CO_2e/EJ. Relatively high GHG costs of about 50 Mt CO_2e/EJ in the 'low estimate' (i.e. 40% removal rate) occur for low-yielding crops with wet residues in developing countries and are likely worst-case estimates.

Livestock manure is clearly most beneficial in terms of GHG implications (figure 4(c)), because considerable quantities of CH_4 and N_2O emissions are avoided when anaerobic treatment is substituted for conventional manure management systems. The size of the potential is, however, relatively small when limitations to actual feasibility are considered; the total amount of livestock manure certainly has only illustrative character. Even the 'indoor potential' is not considered to be fully exploitable, given the large contribution of small-scale farms with indoor housing in developing countries. The limitation to manure from intensive farming is considered a reasonable approximation of a techno-economically feasible potential of biogas from manure if appropriate support schemes are in place; economics are a significant barrier to utilizing the energy and climate mitigation potential of livestock manure (see Einarsson and Persson 2017, Scarlat *et al* 2018 for Europe).

In the case of manure, differences in GHG costs between world regions and scenarios originate from different distributions among animal species, intensities, housing and manure management systems. The highest average GHG savings are possible in industrialized world regions where liquid manure storage systems are common (e.g. Western Europe, North America).

Figure 5 aggregates GHG-cost supply curves from each of the three biomass sources. We differentiate between a 'high' (using the maximum potential estimates in each case) and a 'low mobilization' case (see figure caption). This representation emphasizes the differences in sizes and GHG costs of biomass potentials, especially the dominance of energy crops in the high mobilization case. Moreover, the figure illustrates that without intensification (associated with GHG costs), the global agricultural biomass potentials are quite moderate and cannot cover the additional biomass consumption projected in ambitious climate mitigation scenarios (see figure 1).

3.3. Land use

Figure 6 illustrates the global structure of agricultural land use in 2012 and 2050. The figure includes the 'bioenergy case' as well as the counterfactual 'regrowth case', and illustrates how the mechanisms described in figure 2 manifest in each scenario. Without grazing intensification, cropland areas of 580 000–3130 000 km^2 are available for bioenergy.[8] Under universal grazing intensification, the available area extends to 4010 000–7770 000 km^2.

By comparison, up to 15 240 000 km^2 of agricultural land (mainly grazing areas) could revert to wilderness until 2050 under the land sparing rationale of the 'regrowth case'. This could contribute to an unprecedented trend reversal in human appropriation of natural resources and regeneration of biodiversity and ecosystem services. This highly relevant trade-off inherent to large-scale bioenergy deployment is often overlooked.

[7] Note the different treatment of dry and wet residues (see table 1).

[8] Actual energy crop areas in 2012 are estimated at 500 000 km^2.

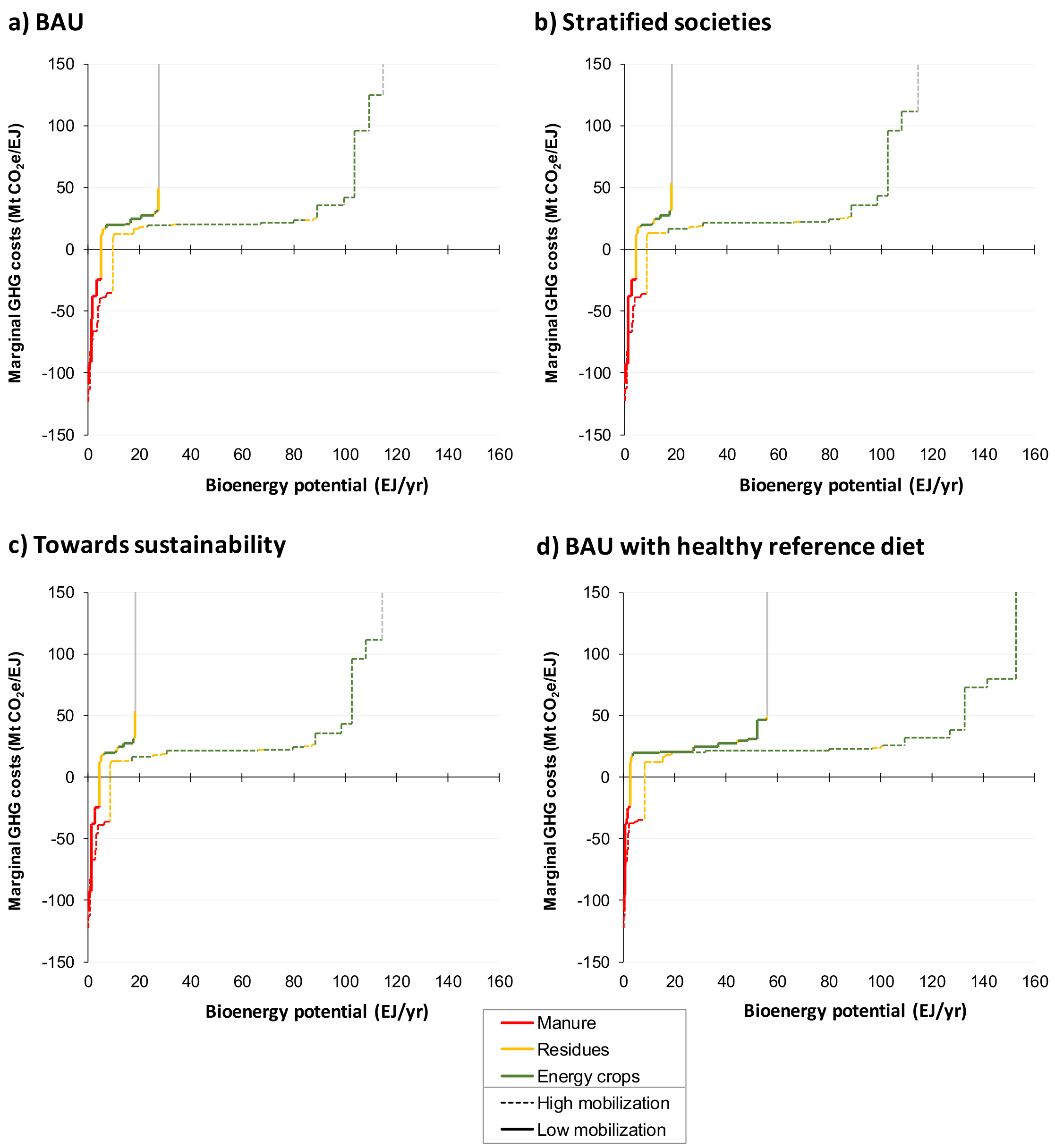

Figure 5. Aggregated global GHG-cost supply curves for agricultural biomass in four scenarios. ('Low mobilization' (solid lines) refers to 'no grazing intensification', 'low estimate' for crop residues and manure from intensive livestock systems only, 'High mobilization' (dashed lines) to 'universal grazing intensification', 'high estimate' for crop residues and livestock manure excreted indoor). GHG costs of energy crops are due to 'opportunity GHG costs' representing the foregone opportunity of vegetation regrowth on agricultural land.

4. Discussion

4.1. GHG costs of energy crops

Whether opportunity GHG costs need to be factored in or not depends on the scope and system boundaries of the respective analysis or modelling approach. We observe that agricultural intensification freeing up agricultural land for energy crops is often an ex-ante assumption, but there are no considerations of alternative uses of freed-up land. In this case, it is correct and important to factor in opportunity GHG costs of vegetation regrowth, as energy crop production eliminates GHG savings that would otherwise occur. This is why opportunity GHG costs are by default included here. In other instances, it is correct to disregard opportunity GHG costs; for example within an integrated assessment model that adequately reflects GHG effects of intensification, implications of changes in grazing pressures etc, and further includes vegetation regrowth as a land-based mitigation option. In this case, the (direct) GHG costs of energy crops are usually negative because establishing perennial energy plantations on grassland or cropland with annual crops leads to C stock gains in soil, biomass and litter (see SI).

If vegetation regrowth is considered as counterfactual scenario to energy crop production, it could be assumed that the same area should be assumed for energy crop production and vegetation regrowth. However, counterfactual scenarios should represent

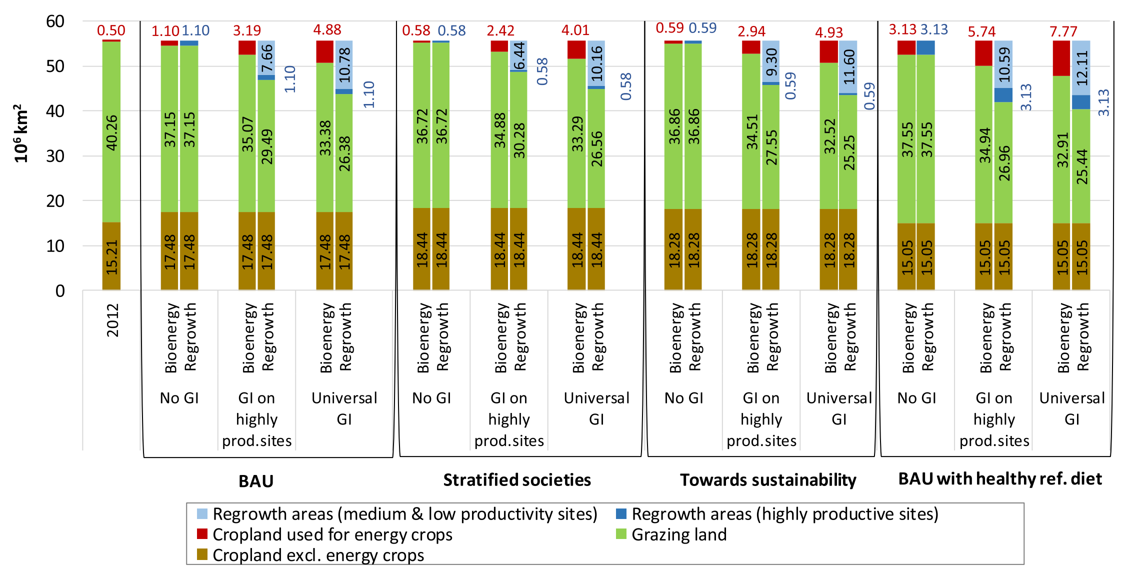

Figure 6. Agricultural land use patterns in 2012 and scenarios to 2050 (GI: grazing intensification).

the *most likely* situation that would occur in the absence of the considered events (the factual case; see Parish *et al* 2017); it is highly unlikely that vegetation regrowth would occur on the highly productive land where profitable energy crops cultivation is possible. It is much more likely that low-productivity areas would be abandoned, while highly productive areas are used for food or feed production. This is the reason for different land-use patterns in the factual ('bioenergy') and counterfactual ('regrowth') cases illustrated in figure 1. It implies that regrowth areas are usually larger than energy crop areas (see figure 6), but at the same time less efficient in accumulating C. This is adequately reflected in our calculations, because each productivity class has their specific C accumulation dynamics, based on their distributions across climate zones, soil types and ecological zones (see SI).

However, for illustrative purposes we also calculated the GHG costs of energy crops under the assumption that regrowth occurs on the same areas where energy crops are cultivated. Under this assumption, the global weighted average GHG costs are 40%–65% lower than under default assumptions (see SI). Ranges on world regional level are much larger, reflecting the different patterns in land distribution, C accumulation dynamics and livestock densities.

4.2. Comparison with bioenergy potentials in previous studies

Table 2 shows that our results regarding biomass primary energy potentials are largely in agreement with the low to medium estimates in literature. Due to the large number of original assessments of global biomass potentials as well as review articles, this comparison is limited to reviews and recent studies that provide valuable comparable values on global scale or for Europe/EU. For more comprehensive reviews of global biomass potential assessments, we recommend IPCC (2014) and Slade *et al* (2014).

With regard to cropland availability and energy crop potentials, our approach of assuming different levels of grazing intensification helps explaining the exceedingly wide ranges in the literature. Our energy crop potentials *without grazing intensification* are partly lower than the ranges according to review studies, indicating that intensification is a basic assumption in most potential assessments. However, some previous studies have even reported zero potential for dedicated energy crops (see Edenhofer *et al* 2011, Roth *et al* 2018). With regard to the highest estimates in literature, we conclude that they are not compatible with current FAO projections regarding food and feed crop yields and diets. Expectations on future energy crop yields are intrinsically uncertain and appear as a main contributing factor to vast differences in biomass potential assessments (see Searle and Malins 2014, 2015, Slade *et al* 2014). With weighted global averages of about 10 tonnes dry matter per hectare, our assumptions are in the medium range of literature assumptions (cf IEA Bioenergy 2007).

Our results for crop residues are also well comparable to previous assessments on global and European/EU scale: For the EU, they are in good agreement with Scarlat *et al* (2010). Compared to Scarlat *et al* (2019) on European scale and Daioglou *et al* (2016) on global scale, our low estimates appear quite conservative. Our conclusion is that BioBaM assumes higher shares of residues being used in the livestock sector, because our sustainable removal rate in the low estimate (40%) is not exceptionally conservative.

The result of a previous assessment for 2050 with a precursor version of BioBaM (Haberl *et al* 2011) was 64–161 EJ of agricultural biomass per year (excluding manure). The upper value refers to a universal 'fair

Table 2. Comparison of the results from this study with potentials in selected literature. With regard to our results, we here only include potentials in FAO-based scenarios.

Topic and geographic coverage	Reference and data	Results from this study (FAO-based scenarios, 2050)
Land available for energy crops (global)	Searle and Malins (2015) (review): 10^6–37×10^6 km^2 (2050) WBGU (2009): 2×10^6–5×10^6 km^2 (2050) Slade *et al* (2014) (review): 790 000–6.1×10^6 km^2 (including scenarios to 2100)	No grazing intensification: 580 000–1.1×10^6 km^2 Grazing intensification on highly productive sites: 2.42×10^6–3.1×10^6 km^2 Universal grazing intensification: 4.01×10^6–4.93×10^6 km^2
Energy crops (global)	Berndes *et al* (2003) (review): 47–238 EJ/yr (2050) Roth *et al* (2018); Searle and Malins (2015) (reviews): 0 to > 1000 EJ/yr (2050) IPCC (2014) (review): < 50 to > 500 EJ/yr (technical potential in 2050); 'high agreement in literature': 25 to ~40 EJ/yr; 'medium agreement': ~40 to ~90 EJ/yr Edenhofer *et al* (2011): 0–700 EJ/yr (technical potential of energy crops on surplus agricultural land 2050) Revised potential[a] according to Searle and Malins (2015): 45–111 EJ/yr (2050)	No grazing intensification: 9.9–19.4 EJ/yr Grazing intensification on highly productive sites: 43–57 EJ/yr Universal grazing intensification: 69–84.7 EJ/yr
Energy crops (EU)	Bentsen and Felby (2012) (review): 3–56 EJ/yr (only 2 studies > 20 EJ/yr)	No grazing intensification: 3.7–5.3 EJ/yr; Grazing intensification on highly productive sites: 6.7–8.6 EJ/yr; universal grazing intensification: 10.9–13.1 EJ/yr
Crop residues (global)	Daioglou *et al* (2016): 'Available potential' (ecological potential minus amount used as animal feed) in scenarios to 2100: 20–28 EJ/yr Slade *et al* (2014) (review): 10–66 EJ/yr	High estimates: 20.5–21.5 EJ/yr Low estimates: 3.2–3.5 EJ/yr
Crop residues (EU/Europe)	Scarlat *et al* (2010): 1.09–1.90 EJ/yr (EU-27; based on crop production 1998–2007) Scarlat *et al* (2019): 2.1 EJ/yr (EU-28), 2.6 EJ/yr (Europe) (sustainable potential considering technical and environmental constraints; based on production in 2000–2015) Bentsen and Felby (2012) (review): ~0.5–5 EJ/yr	High estimate: 2.2–2.5 EJ/yr Low estimate: ~0.8 EJ/yr (total of Eastern and Western Europe)
Livestock manure (global)	Edenhofer *et al* (2011): 5–50 EJ/yr	Intensive livestock systems only:4.3–5.2 EJ/yr, Indoor housing only: 8.6–9.7 EJ/yr All livestock manure: 17.4–19.4 EJ/yr
Livestock manure (Europe)	Scarlat *et al* (2018) for period 2009–2013: 0.64–0.92 EJ/yr	Intensive livestock systems only:0.37 to 0.47 EJ/yr, Indoor housing only: 0.72 to 0.93 EJ/yr All livestock manure: 1–1.3 EJ/yr

[a] Searle and Malins (2015) perform a reassessment of global bioenergy potentials with harmonized parameters and assumptions.

and frugal' diet, i.e. a global convergence scenario similar to 'BAU with healthy reference diet'. Despite entirely revised input data and slightly different assumptions regarding grazing intensification, the results for energy crops are surprisingly consistent (about 130 EJ/yr, assuming universal grazing intensification).

Potential assessments must be explicit on underlying assumptions and narratives (regarding intensification, socio-economic developments etc), in order to facilitate realistic visions about the future role of bioenergy. Requirements for biomass mobilization, especially the need for grazing intensification, are often insufficiently addressed in the literature, contributing to confusion regarding large ranges of biomass potentials and their causes. Furthermore, notions about currently unused land are questionable. Many studies found that these lands may in fact be under more or less intensive use, in particular in developing countries with widespread pastoral livestock systems (Young 1999, p 199, Lambin and Meyfroidt 2011, Nalepa and Bauer 2012, Baka and Bailis 2014, Exner *et al* 2015, Erb *et al* 2016a, Bartels *et al* 2017). The assumptions that energy crops would be cultivated in low-productivity areas (marginal land) are also dubious. Economic rationale and empirical evidence (see Novo *et al* 2010, Rathmann *et al* 2010, Harvey and Pilgrim 2011, Piroli *et al* 2012, Lapola *et al* 2014,

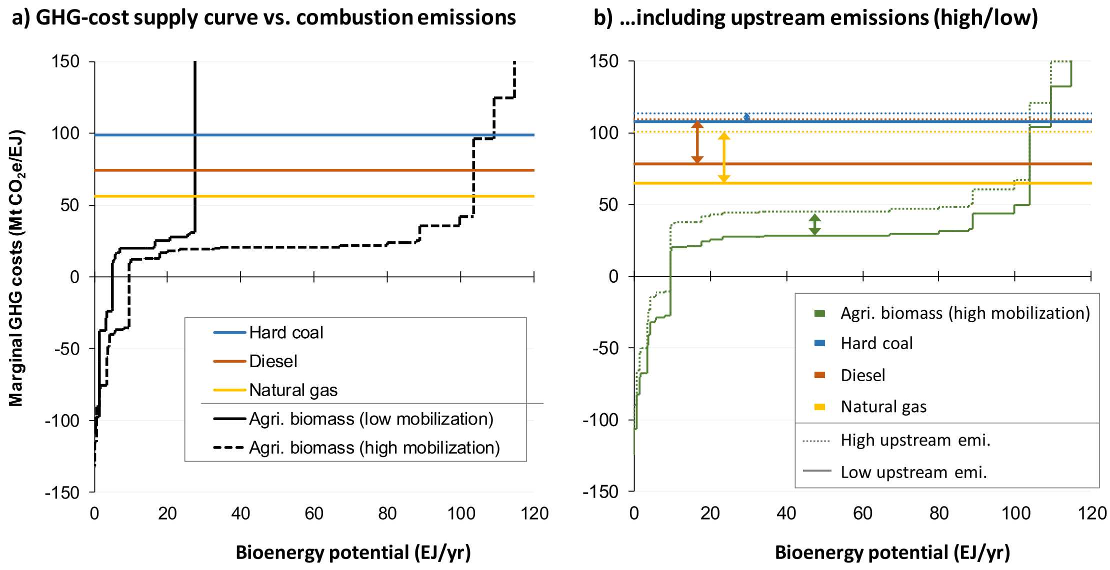

Figure 7. Comparison of GHG costs of agricultural biomass (BAU scenario) with combustion emissions of fossil fuels. (a) Upstream emissions are disregarded. (b) Illustrative ranges for upstream emissions resulting from different transport distances, fossil fuel deposits etc (Kalt *et al* 2019); only high mobilization case is shown in (b).

Rajcaniova *et al* 2014) suggest that energy crop production will (i) enhance competition for existing cropland and (ii) foster intensification and cropland expansion into highly productive grassland.

4.3. Considerations on applications of GHG-costs curves

GHG-cost curves convey additional information that, in the context of climate mitigation, is just as crucial as primary bioenergy potentials. Information on the GHG implications of mobilizing biomass potentials is of paramount importance for developing prudent climate and energy strategies and avoiding GHG leakage effects. Therefore, we suggest implementing GHG-cost curves in energy models and integrated assessment models that do not (or insufficiently) cover GHG emissions from agriculture and land-use.

To quantify net GHG savings achievable with different biomass resources, we can compare GHG-cost curves with combustion emissions from fossil fuels. Combustion emissions from biomass may be disregarded in this context because C cycles of agricultural biomass are brief; and lasting effects (C stock changes) are considered in the GHG costs. Figure 7(a) illustrates that the GHG costs of agricultural biomass in the BAU scenario are mostly lower than combustion emissions of fossil fuels. However, differences among biomass fractions are considerable, and systemic effects can lead to GHG costs that even exceed fossil fuel combustion emissions[9]. Marginal net GHG costs from agricultural bioenergy may be notably lower or higher than the avoided fossil fuel emissions depending on the amount of biomass to be used.

To determine actual net GHG savings from bioenergy (as compared to different fossil fuels), two more aspects must be considered: first, upstream emissions from fossil fuels as well as biomass production and supply that are not considered in our GHG costs (e.g. transport, storage). Depending on the respective supply chains, fossil fuel deposits etc, upstream emissions can be considerable, as figure 7(b) illustrates. And second, this comparison referring to primary energy disregards differences in subsequent conversion processes to final or useful energy. Bioenergy technologies usually have lower conversion efficiencies than fossil fuel-based counterparts. Thus, a GHG comparison on the level of primary energy, as shown in figure 7, is of limited significance. A thorough analysis of the net GHG balance of agricultural bioenergy with consideration of the various biomass conversion paths and their fossil-based counterparts is, however, beyond the scope of this work (see Kalt *et al* 2019).

Since average GHG emissions/sinks from C stock changes are sensitive to the considered timeframe, the time reference of GHG-cost supply curves is of particular importance. The counterfactual scenario of energy crop cultivation, vegetation regrowth, is characterized by declining annual C benefits, as biomass accumulation eventually becomes saturated and net C uptake converges towards zero (see Kalt *et al* 2019). Hence, the GHG costs of energy crops depend on the considered timeframe and typically decrease for longer timeframes.

[9] In the BAU scenario, this is true for energy crops in 'Central Asia & Russia' and 'East Asia'; because in these regions grazing intensification can set free large areas for vegetation regrowth, while energy crop yields in suitable areas are relatively low.

5. Conclusions

Although fairly consistent with previous assessments of residue and manure potentials, this study does not confirm the large energy crop potentials often reported in literature. Assuming the latest FAO projections for yields and diets, and based on a consistent and comprehensive biophysical model we find that energy crop potentials strongly depend on intensification on grazing land. Without grazing intensification, potentials are situated at the lower range of literature estimates. Depending on changes in diets and crop yields, between 10 and 85 EJ of energy crops can be supplied in 2050 without expanding agricultural land. Large energy crop potentials are not readily and unconditionally available. They are conditional on intensification and considerable changes in global land-use patterns: In addition to cropland expansion for food and feed production (about 2–3 million km^2 in scenarios based on FAO projections), about 4.9 million km^2 of agricultural land need to be diverted to energy crop production to mobilize the maximum potential of 85 EJ/yr.

Assuming perennial energy crop plantations, the direct C stock changes associated with this land-use change are beneficial for climate mitigation. Yet if 'opportunity GHG costs', representing the foregone opportunity of vegetation regrowth, and C stock changes from grazing intensification are factored in, the beneficial direct C stock changes are clearly outweighed. Consequently, the net GHG benefits of energy crops displacing fossil fuels diminish and may even become net costs, depending on regional parameters, supply chain emissions and characteristics of the bioenergy pathway as well as the (fossil-based) reference system (see Kalt *et al* 2019).

Systemic GHG effects are inherent to all types of agricultural biomass. Removal of crop residues for bioenergy reduces soil carbon stocks. Although the resulting GHG costs are expected to be moderate in relation to GHG benefits achievable from fossil fuel displacement, this must be considered in technology-specific assessments of GHG saving potentials as well as scenario analyses. Agricultural practices to offset adverse effects of residue removal (e.g. cover crops or compost application; see Blanco-Canqui and Lal 2009, Blanco-Canqui 2013, Mouratiadou *et al* 2019) should be an integral part of bioenergy strategies envisaging the use of crop residues. Yet, it is important to note that such practices could be implemented anyway, regardless of whether residues are removed for bioenergy or not. So even if direct GHG costs are offset through targeted agricultural practices, opportunity GHG costs are likely to remain (depending on how efficient these practices are at different residue removal rates).

In case of livestock manure being used for biogas production, highly beneficial direct GHG effects occur because methane emissions from conventional manure management are avoided. Harnessing the potential of livestock manure could therefore yield a double dividend in terms of GHG mitigation and—with a realistic global potential (intensive livestock systems only) of 4.3–5.2 EJ/yr—a noteworthy contribution to future energy supply. This energy and GHG saving potential is, however, small in relation to the burden the livestock sector inflicts in terms of land requirements and GHG emissions. Our scenario assuming global convergence to healthy (low-meat) diets features significantly higher energy crop potentials (+32 to +46 EJ/yr compared to BAU) and considerably lower GHG emissions from manure management and enteric fermentation (−470 Mt CO_2e/yr).

Our study shows that systemic GHG effects are relevant and should not be easily dismissed in long-term decarbonisation scenarios and integrated modelling approaches. The concepts of 'opportunity GHG costs' and 'GHG costs curves' appear as a promising way of factoring in indirect, systemic GHG effects, particularly when LULUCF[10] and agricultural emissions are beyond the scope of the respective assessment or modelling approach. Generally, by associating bioenergy potentials with their GHG costs, profounder insights are provided than by merely quantifying potentials.

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Data availability statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

ORCID iDs

Gerald Kalt https://orcid.org/0000-0003-4087-8902
Christian Lauk https://orcid.org/0000-0002-4173-1753
Andreas Mayer https://orcid.org/0000-0002-6975-7082
Michaela C Theurl https://orcid.org/0000-0002-2617-9091
Katrin Kaltenegger https://orcid.org/0000-0001-7751-7794
Wilfried Winiwarter https://orcid.org/0000-0001-7131-1496

[10] Land use, land use change and forestry (see Penman *et al* 2003).

Karl-Heinz Erb https://orcid.org/0000-0002-8335-4159
Sarah Matej https://orcid.org/0000-0003-4576-4189
Helmut Haberl https://orcid.org/0000-0003-2104-5446

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