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Q1

Evaluating the Potential for Harmonized Prediction and Comparison of Disposal-Stage Greenhouse Gas Emissions for Biomaterial Products

Q2

David Glew, Lindsay C. Stringer, Adolf Acquaye, and Simon McQueen-Mason

Keywords:

biorefineries
greenhouse gas (GHG)
hemp
industrial ecology
life cycle assessment (LCA)
waste

Q3

Summary

The carbon footprint (CF) of biofuels and biomaterials is a barrier to their acceptance, yet the greenhouse gas emissions associated with disposing of biomaterials are frequently omitted from analyses. This article investigates whether harmonization is appropriate for calculating the importance of biomaterials' disposal. This research shows that disposal stages could double a biomaterial's CF, or reduce it to the point that it could claim to be zero carbon. Incineration with combined heat and power coupled with on-site energy production in the biorefinery are identified as prerequisites to being zero carbon. The article assesses the current UK waste infrastructure's ability to support a low-carbon bio-based future economy, and finds that presently it only achieves marginal net reductions when compared to landfill and so cannot be said to support low-carbon biomaterials, though the article challenges the polluter pays principle where low-carbon disposal infrastructure are not available. Reuse and recycling are shown to have the potential to offset all the emissions caused by landfill of biomaterials. However, the savings are not so great as to offset the biomaterial's upstream emissions. The study explores the ability to overcome the barriers to incorporating disposal into life cycle assessment while identifying limitations of using harmonization as an assessment method. Specifically, data availability and industry consensus are flagged as major barriers. The study also uses sensitivity analysis to investigate the influence of methodological choices, such as allowing additional reuse and recycling stages, classifying biomaterials into different types, and choosing between opposing allocation methods.

Introduction

Businesses are often held responsible for their environmental and social performance (Seuring et al. 2008), through, for example, the polluter pays principle (Ambec and Ehlers 2014; Huber and Wirl 2015), among other laws and regulations. Industry is therefore required to comply with many different legislative targets requiring greenhouse gas (GHG) emissions to be reduced, including the European Union's (EU) Emissions Trading Scheme and 20-20-20 Targets as well as the UK's Renewable Energy and Energy Efficiency Directives (Committee on Climate Change 2015). Targets set in legislation are reinforced by a rise in consumer demand for softer measures, such as proof of low environmental impacts through certification schemes and preference for socially responsible products (Golden et al. 2010; Lynes and Andrachuk 2008; Nawrocka et al. 2009; de Boer 2003). Thinking differently about the use of petrochemicals and exploring the development of

Q4

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DOI: 10.1111/jiec.12421

Editor managing review: Miguel Brandão

Volume 00, Number 0

biomaterial alternatives represents one possible direction businesses could take (Ragauskas et al. 2006).

Biomaterial alternatives to petrochemicals are perceived to offer the potential for industry to both deliver "greener" products and meet growing consumer demands for sustainability. In the UK alone, the market for biomaterials products (including natural fibers, bioplastics, and biofuels) was expected to triple over the period 2012–2015 (NNFCC 2012). Even after the recession, these scales of growth are anticipated to be mirrored in the United States where estimates suggest an 80% growth in biobased materials production between 2005 and 2025 (OECD 2014).

Understanding the environmental impacts of biomaterials requires comprehensive assessment approaches that provide insights into the potentially substantial and sometimes indirect negative externalities relating to production, use, and disposal. Already there is growing awareness of biomaterials' impacts from land-use practices, and indirect land-use changes (LUCs), as well as energy-intensive supply chains through the use of machinery, fertilizers, and pesticides (Overmars et al. 2011). In many cases, GHG emissions are used as an indicator of environmental impacts associated with the production of biomaterial feedstock, usually using life cycle assessment (LCA) approaches (Fahd et al. 2011; Cherubini and Ulgiati 2010; Harding et al. 2007; Falloon and Betts 2010; Seguin et al. 2007). LCA is useful in providing holistic supply-chain evaluations (Roy et al. 2009; Acquaye et al. 2012b; Guinée et al. 2001). However, LCA can be undertaken in rather a partial way. Assessments may be justifiably curtailed for a variety of reasons, often attributed to data unavailability, to reduce complexity and cost, or simply because the system boundaries that the investigator is interested in require only partial assessment. This phenomenon is one of the justifications for the development of input-output LCA (Rowley 2009). When LCA is undertaken in a partial way, disposal and end-of-life (EoL) impacts are often excluded. In this article, it is argued that the disposal emissions have an important influence on the overall carbon footprint (CF) of biomaterials and should not be readily omitted.

Considering disposal emissions is not essential in all LCA; however, initial studies show that omitting it can significantly limit the completeness of biomaterial LCA (UNEP 2010; Wang et al. 2012; Stichnothe and Azapagic 2009; Shen et al. 2010; Glew et al. 2012; Ross and Evans 2003). Although not a widely studied area, some research suggests that despite the current lack of exposure, there may be demand for industry guidelines on how to achieve and articulate low-carbon disposal options for biomaterials (Glew et al. 2013). Biomaterial feedstock can be manufactured into myriad products. Thus, to consider what the disposal options of an unknown product may be at this stage would be difficult. However, even when a biomaterial product has been produced, there may still be several reasons why disposal presently remains relatively under-represented in their LCA:

1. It is not clear who "owns" the benefit of reducing GHGs of disposal; the manufacturer, the waste management company, or the consumer (Häkkinen and Vares 2010). This technical externality explains that if a company cannot claim benefits (of GHG reductions), they have little motive to consider it through, for example, design for deconstruction (Nemoto and Goto 2004).
2. Consumers may not choose to dispose of products in preferred, low-carbon ways and so the calculations must remain hypothetical.
3. LCA can be a costly and complicated undertaking with many uncertainties, so the results may be difficult to interpret (Golden et al. 2010).
4. Several methodological options for measuring disposal emissions of products exist (Ekvall and Tillman 1997), meaning that different scope and goals, functional units, and system boundaries can each be used. This limits the ability of individual LCAs to be compared to one another, therefore limiting any clear ranking of options from being provided.

This article's aim is to develop a harmonized LCA tool and evaluate whether such a tool can help to address these four barriers. In doing so, it provides useful insight into whether the UK waste infrastructure is able to support a low-carbon bio-based economy and the role disposal plays in biomaterials' ability to be low or zero carbon.

Life Cycle Assessment Harmonization

The principle of harmonization has been accepted by some as useful (Schlegel and Kaphengst 2007), though there are specific benefits and limitations with its use over conventional LCA. These are summarized in figure 1.

Harmonized LCA tools have already been introduced, for example, the Standard Assessment Procedure (SAP) used to estimate the energy efficiency of UK buildings (DECC 2013), and perhaps more pertinently for our study, in the EU, the Biofuels calculator BioGRACE[1] project. The latter was designed specifically to compare emissions of different biofuels and set emissions thresholds, ensuring that comparisons can be made between biofuels. Harmonization means that smaller organizations, perhaps incapable or unable to afford to undertake their own LCA, can compete alongside larger biofuels producers, given that everyone must use the freely available associated generic biofuel LCA tool. This means that they are required to use the same functional unit, system boundary, and allocation procedures. They can also all use the same default values for controversial or difficult to calculate inputs and processes, such as LUC or improved agricultural practices.

Given the progress made using harmonized LCA tools in the bio*fuels* arena, it is possible that bio*material* markets may also adopt this evaluation methodology for their products. However, if a BioGRACE-like tool were to be adopted by the biomaterials industry, it is important to note that unlike biofuels, biomaterials ultimately require disposal. This article focuses on the specific complexities around *disposal* and how these may be addressed by harmonization in LCA. To do this, it develops a tool

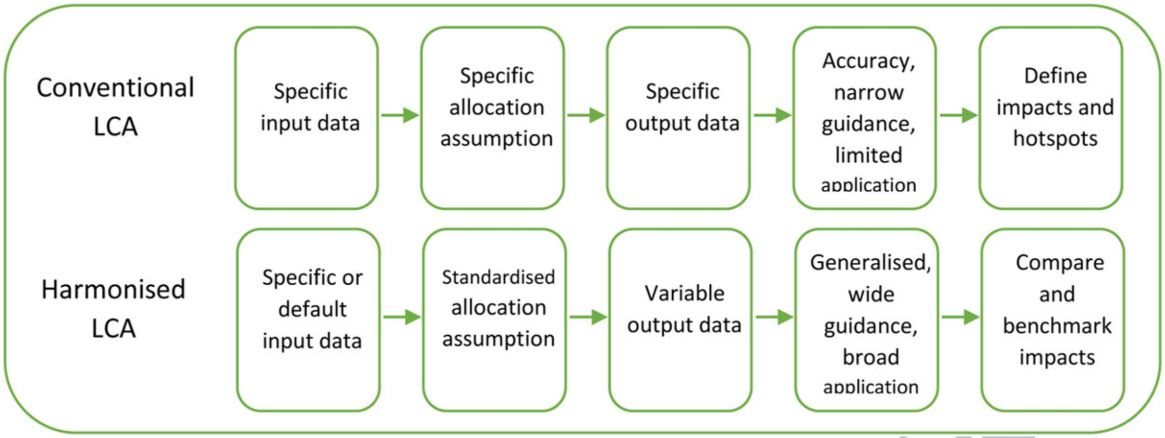

Figure 1 Comparison of conventional and harmonized approaches to LCA. LCA = life cycle assessment.

taking a **H**armonised approach to consider **E**nd-of-life impacts using **LCA**, hereafter referred to as "HELCA."

Providing harmonized data on GHG emissions of disposal scenarios may be useful for biomaterial companies, given that they may use the information to design products to align with particular disposal options or to educate their customers regarding low-carbon disposal options as part of their corporate social responsibilities. Our study may also be useful for policy makers interested in using harmonization as a means of standardizing quantification of the impacts of biomaterial product disposal.

Biomaterials and Feedstock

The term biomaterials is used in this article to capture a diverse range of products, such as natural fibers, oils, and bioplastics, which are increasingly touted as sustainable alternatives to petrochemicals, and which may require disposal (Vandermeulen et al. 2012). Because HELCA is being used as a model in this article to test the viability of harmonization, it is important to capture a wide range of products. This article therefore investigates European hemp (*Cannabis sativa*) as a biorefinery feedstock because it has multiple product outputs, including paper, plastics, textiles, fuel, and food (Van der Werf and Turunen 2008). This is especially useful as a case study because a product can be disposed of in several different ways, so the harmonization of multiple scenarios can be tested.

Life Cycle Assessment Method

In order to compare a variety of scenarios, HELCA is implemented using a consequential LCA (cf. Georgilakis 2006). It follows the accepted process of first setting the objective, system boundary, and functional unit (1: goal and scope) and then quantifying supply-chain inputs and outputs (2: life cycle inventory). Following these steps, the environmental burdens are applied (3: life cycle impact assessment [LCIA]) before interpreting the results (4: interpretation) (ISO 2006).

Life Cycle Assessment Goal and Scope

Aim and Objectives

The aim of the study is to investigate the feasibility of using a harmonized LCA tool and evaluate whether such a tool can help to address the previously mentioned four barriers to including disposal in LCA. We also have two complementary objectives: to provide insight into whether the UK waste infrastructure is able to support a low-carbon bio-based economy and identify the role disposal plays in biomaterials' ability to be low or zero carbon.

Allocation

This article is concerned with emissions caused by disposal. This section describes how these emissions (and avoided emissions) will be apportioned to all the different products made by the biorefinery.

Two allocation approaches may be explored, which we call *individual* and *collective* approaches. These are similar to approaches already established by Hermann and colleagues (2011). HELCA will start by using the *individual* approach, in which 100% of net emissions associated with sending paper to a landfill site will be allocated to paper, and 100% of textiles' net emissions will be allocated to textiles and so on. If the product is a consumable product, such as animal feed, it is assumed to have no disposal emissions. This approach inevitably locks bias into the calculation, though. For example, those co-products that happen to only have high-carbon EoL options will have to receive all their associated emissions, even though they may have been a necessary part of the biorefienry process. Conversely, those products that can be reused easily and can attract low- or zero-carbon disposal options will benefit, even though their production was necessarily accompanied by other products with more polluting disposal fates.

Conversely, in the *collective* approach, the total net emissions of disposing of all the paper and all the textiles, and so on, are summed and then allocated back to individual products based on their relative importance to the supply chain. To do this, we must select one of the three conventional allocation methods: (1) economic, (2) energetic, and (3) mass as described by Finnveden and colleagues (2009). Although energy is a measure of a fuel's worth, it is not necessarily an indication of a biomaterial's importance. For this reason, energy will not be used in this study. For low-volume, high-value products, such as chemicals, it may be argued that *economic* allocation should be used because these may be the primary justifications for the biorefinery's activities. For example, if 10 grams (g) of chemicals may be as profitable as 1 kilogram (kg) of paper, some would argue that they should each receive an equal share of the emissions associated with sending 1 kg of paper to landfill, even though the chemicals were not directly responsible. Unfortunately, data on costs are difficult to acquire and fluctuate considerably. For this practical reason, economic allocation is therefore also rejected. Mass is the final recognized allocation method. Because biorefineries produce textiles, food and feed, paper, and to a lesser extent plastics, for all of which *mass* can be an indication of some of their relative importance in the overall supply chains, mass allocation is used for this harmonized tool. Bias still exists. However, the more massive the product, the more emissions allocated to it, whether it was responsible for them or not. Accepting a limited amount of bias is necessary in all LCA. However, in harmonized tools, the bias is consistently applied, regardless of who uses it or why; thus, in some way, it may be deemed less influential in the interpretation, though cannot be ignored completely. The influence of allocating emissions according to this collective approach is investigated in a sensitivity analysis.

Functional Unit

Functional units greatly influence an LCA and how it may be interpreted (Cooper 2003). It is not the purpose of this research to investigate how cultivation, transport, and processing affect a biomaterial's impacts, and there are already many examples of upstream biomaterials LCA (Essel 2012; Sevenster 2014; Pawelzik et al. 2013; Weiss et al. 2012). This article focuses on disposal options and their associated emissions only, and it is the responsibility of the functional unit to capture this.

The functional unit also identifies the impacts being considered. In this article, impacts are limited to GHG emissions. This was done for simplicity, because GHGs are a standard unit used in legislation and because they are commonly cited environmental impacts for which data are available (UNEP 2010; EC 2001). Impacts beyond GHGs are not considered, though in principle, other impacts, such as acidification or contribution to eutrophication, could be included if there were sufficient demand and data.

Biomaterials are often, but not always, produced in a biorefinery. This article assumes that biorefineries are the supply chain for the products in the assessment. The approach may be applied to biorefineries making just a single product or a dozen. Biorefineries can produce a spectrum of marketable products with all providing some revenue (Cherubini 2010a); thus, the functional unit needs to address kg of feedstock, not kg of biomaterial product. This acknowledges that no biomaterial product is made in isolation from any other, and so articulations of the benefits of products with low-carbon disposal options should be tempered by the problems of intrinsically linked higher carbon co-products. One of the benefits of a harmonized tool is to provide this degree of control to ensure that systems as a whole are considered and that cherry picking cannot take place. The Q8 functional unit used in this LCA is therefore: *disposal grams carbon dioxide equivalent per kilogram (g CO_2-eq/kg) of biorefinery feedstock*.

System Boundary

HELCA addresses the net emissions associated with disposing of biomaterials. This excludes biomaterial cultivation, production, and transportation to the consumer, which might be referred to as a "cradle to use" LCA. Instead, this LCA could be referred to as a "grave to grave."

Conventionally, there are four methods of allocating upstream emissions to products that result from reuse and recycling in LCA: (1) cutoff; (2) loss of quality; (3) closed loop; and (4) 50:50 (Nicholson et al. 2009). Because we are not considering upstream emissions, these methods cannot be applied here and we must consider a fifth: the *substitution method*.

The substitution method operates under the system expansion approach, whereby data are sought on the emissions that can be claimed to be saved by offsetting consumption of virgin materials through reuse, recycling, or energy recovery; these are then subtracted from the total emissions as credits. This approach is adopted in other harmonized tools, for example, in calculating the impacts of LUC caused by biofuel cultivation in BioGRACE, where credits or penalties are applied.

In the sensitivity analysis, the influence of one additional reuse or recycling scenario in addition to a final disposal is investigated for individual co-products (i.e., two disposal options in total), because one of the benefits of biomaterials is reported to be their ability to be reused. This allows for the benefit of reusing and recycling products to be investigated, but stops short of exploring the influence of multiple reuse stages. The reason for this is to limit the complexity of the tool. Also, because products tend to deteriorate as they are reused or recycled, one reuse stage was considered appropriate. Although curtailing future uses may be seen as a possible limitation, in a harmonized tool, it allows assessments to be comparable, removing the possibility of bias where biomaterial suppliers could otherwise assume unrealistic numbers of reuse LCA to their advantage. HELCA is therefore based on the following equation:

$$\begin{aligned} \textit{Total disposal emissions} &= \{(a_1 \times (b_1 + b_2)) \\ &\quad + (a_2 \times (b_1 + b_2)) + \ldots (a_n \times (b_1 + b_2))\} \end{aligned}$$

where: a = co-product 1, 2, n as a % of total feedstock yield (allocated by mass); b = EoL net GHG emissions of disposal option 1 and 2 (g CO_2-eq/kg)

Life Cycle Inventory

Setting default values for the emissions that should be incurred or credited by particular disposal options is essential to this harmonized tool. Here, too, bias will manifest, for example, avoided emissions from recycling paper must be applied as a standard credit, regardless of any variations specific to quality or type of recycling facilities that might exist between or within countries (Ekvall and Tillman 1997). However, this is a necessary limitation of harmonization to stop users ignoring disadvantageous data sources, even when it may be the most reputable and appropriate. As harmonized tools develop, they can acquire levels of sophistication to address heterogeneity of the real world as seen in the case of the BioGRACE tool, which allows biofuel producers with enhanced (low-carbon) agricultural techniques to claim a standard credit from which others will not benefit. Similarly, future editions of HELCA may look to provide standard credits for companies or countries with, for example, efficient recycling regimes.

Upstream Inputs

Although upstream supply-chain emissions (raw material extraction, transport, and LUC, and so on) of biomaterials are not assessed in this article, they receive much attention elsewhere (Cherubini 2010a, 2010b; Cherubini and Jungmeier 2010; Searchinger 2010; Van der Werf and Turunen 2008). However, in order to add a sense of scale and address the second research question posed around zero-carbon biomaterials, an upstream LCA for hemp is briefly described. One study that assumes good agricultural practice (GAP) states that cultivating 1 kg of hemp may cause 350 g CO_2-eq of emissions (Van der Werf 2004). GAP cannot be guaranteed, and a second
Q9 assessment shows that cultivating 1 kg of hemp fiber feedstock results in 1,600 g CO_2-eq (González-García et al. 2010). Taking the latter study as a worst-case scenario, we can add this to the processing GHG emissions caused by weaving 1 kg of hemp bast fibers for which ecoinvent,[2] a widely used LCA database (Frischknecht and Rebitzer 2005), predicts that 406.7 g CO_2-eq[1] are emitted. Because only one third of the hemp actually ends up as fiber (EIHA 2012), we must divide our current total of 2,006.7 g CO_2-eq by 33, giving approximately 662.2 g CO_2-eq/kg of hemp feedstock. Using multiple data sources to derive this total raises the possibility of using LCA with different system boundaries and assumptions, and increases the chance of double counting or omitting important information. Again, these are some of the problems that harmonization aims to overcome.

Downstream Outputs

This section describes the default data used in HELCA and table 1 describes two hypothetical biorefineries, A[3] and B,[4] used to explore how choices made in the biorefinery can affect the overall disposal emissions of the hemp feedstock.

Data on GHG emissions caused by disposal of biomaterials are relatively scarce and often not provided with much granularity, therefore categories in table 1 have been developed to aggregate different biomaterials under umbrella *types* for which GHG emissions data are available. This represents a limitation requiring us to assume that all bioplastics have the same disposal emissions as one another and all textiles have the same emissions as one another and so on. Clearly, heterogeneity of biomaterials exists and information on the impact of their differing chemistry on their disposal emissions would be required to capture this within a harmonized tool. Should harmonization be adopted, agreement would need to be reached by industry experts over the absolute net emissions for different biomaterials, how many subcategories products could be split into, and how often these were to be revised and updated.

Deciding the category into which a product should be placed is highly influential to the ultimate GHG emissions allocated to it. No systematic way to make these distinctions exists, so the user must make a competent selection. This represents an area of uncertainty, and should a harmonized tool be adopted, it is another issue for which there would need to be a set of definitions that must guide harmonization grouping rules for different products. In this article, the influence of designating different products in different biomaterial classifications to those shown in table 1 is assessed in the sensitivity analysis.

Once the biomaterial grouping conventions are agreed, the disposal options available to each group must be identified. Table 2 shows the GHG values proposed for the 12 disposal options in this research. These were selected to represent current and future biomaterial disposal technology given that a harmonized tool would need to be comprehensive and cover all possible disposal options, however scarce. Any option not listed in a harmonized tool cannot be considered. This will represent a limitation of harmonized tools, though it would be possible to include new disposal options as they become available.

GHG values are sourced predominantly from the European Commission's (EC) report on waste and climate change (EC 2001), with a few exceptions described here. The EC report includes Europe-wide average emissions for transport and processing, and credits energy recovery and recycled products, both of which avoid consumption of virgin petrochemical alternative resources elsewhere (i.e., a system expansion approach). The report assumes that 50% of textiles are organic; we have therefore doubled the textile landfill decomposition emissions because we are interested in 100% biomaterials.

We have allocated lignocellulosic waste the emissions from landfill for municipal solid waste MSW as a proxy because these data were not available. Such limitations exist because this is the first attempt of its kind to provide a harmonized set of default data for biomaterial disposal emissions. Keeping the data from one source as far as possible was important to avoid mixing of LCA methods. The EC's data were used because they are the most complete and represent a transparent reference point. These data are based on EU15 sources and so are relevant, to some extent, to the UK because no nation-specific data were available. As with all harmonized tools, rules for updating these values could be established as more data become available. Indeed, the adaptation of a harmonized tool would likely prompt more research into refining these values, given that data gen- Q10
erated by specific process LCA upon which harmonized LCA builds improve. One benefit of harmonization where uncertainty around data exists is that there would be consensus on which data are to be used so that cherry picking was avoided.

Table 1 Co-products of hemp biorefineries and their HELCA product types

Biorefinery A co-products	*HELCA type*	*Mass (%)*	*Biorefinery B co-products*	*HELCA type*	*Mass (%)*
Bird feed	Food/consume	10	Flower essential oils	Food/consume	8
Dust	Lignocellulosic waste	9	Seed and feed	Food/consume	7
Fish feed	Food/consume	1	Dust	Lignocellulosic waste	2
Plastic	Bioplastic (PET)	1	Animal bedding	Textiles/fibers	29
Paper	Paper	19	Shivs in construction	Textiles/fibers	7
Insulation	Textiles/fibers	7	Composite board	Textiles/fibers	4
Animal bedding	Textiles/fibers	34	Insulation	Textiles/fibers	7
Composite board	Textiles/fibers	15	Mulch	Textiles/fibers	1
Waste	Putrescible/biowaste	4	Paper	Paper	15
		1[illegible]	Waste	Putrescible/biowaste	21
					100

Note: PET = polyethylene terephthalate.

Table 2 GHG emissions from end-of-life scenarios (g CO_2-eq/kg co-product)

End-of-life strategy	*Textiles*	*Putrescible*	*Paper*	*Bioplastic (PLA)*	*Food/consume*	*Lignocellulosic material*
Landfill	31	762	255	386	—	349
Landfill energy recovery	15	730	223	354	—	327
Recycling	−3,169	—	−600	−1,761	—	—
Reuse	−3,193	—	—	−1,785	—	—
Incineration with electricity	−303	−66	−235	—	—	−10
Incineration with CHP	−880	−224	−691	—	—	−348
Ethanol	−47	−81	−72	—	—	−47
AD with electricity	—	−104	−104	—	—	—
AD with CHP	—	−185	−184	—	—	—
Compost	−37	−37	−37	—	—	−37
On-site heat	—	−690	—	—	—	−690
On-site electricity	—	−1,010	—	—	—	−1,010

Note: g CO2-eq/kg = grams carbon dioxide equivalent per kilogram; PLA = polylactic acid; CHP = combined heat and power; AD = anaerobic digestion. Q11

Data on disposal emissions for bioplastics and ethanol conversion were not given in the EC's report. Using different data sources adds uncertainty. However, to complete the tool, data on bioplastics' landfill emissions are taken from Madival and colleagues (2009). These data were also used in a UK government waste management LCA analysis (WRAP 2010). Landfill gas utilization from bioplastics is assumed to save 32 g CO_2-eq as in the case of paper and putrescible according to the EC report. In addition, the EC assumptions on the emissions saved through avoiding petrochemical plastics through recycling and reuse are also used for bioplastics, given that they could replace petrochemical alternatives. Data from Hermann and colleagues (2011) were reviewed with a view to providing data on the GHG emissions associated with polylactic acid's (PLA) incineration, composting, and anaerobic digestion (AD); however, their method of allocating credits for producing energy or, in the case of composting, avoiding the use of alternative products, differed from that used in the EC data, and therefore in this analysis, no values were allocated for PLA for these technologies.

Harmonization's benefits to industry are to remove the cost of undertaking LCA, increase access to LCA capabilities, and to set a level playing field upon which comparisons or regulation and certification can be made. Should industry wish to pursue harmonization, comprehensive data need to be produced, so as to avoid the use of data with disparate inherent assumptions. In this instance, PLA constituted only 1% of biorefinery A's outputs and thus this limitation will not substantially affect this study's findings.

All ethanol conversion data used are based on Schmitt and colleagues (2012), where we have taken lignocellulosic waste and textiles/fibers' ~~to~~ be similar to their woody organic yard waste, and our putrescible/biowaste is assumed to equate to their MSW. Textiles/fibers, lignocellulosic waste and bioplastics are not suitable for compost or AD because they are deemed in UK guidelines too fibrous for successful AD (Environment Agency 2008a, 2008b). Neither waste categories are suitable for recycling or reuse.

Total emissions saved for reuse and recycling are equivalent to the EC default for savings achieved by offsetting the need for an alternative petrochemical product. Emissions associated with product reuse include default transport and mobilization emissions of 10 g CO_2-eq/kg to account for collecting the products and distributing them to new users. This is the same for recycling emissions, which, in addition, includes extra processing energy of 24 g CO_2-eq/kg to convert the used items into new products.

On-site energy production is only viable from lignocellulosic waste and putrescible/biowaste made in the biorefinery, that is, *dust* and *waste*, and their energy yields are calculated using the lower heating value of hemp, 15.9 megajoules per kg (Prade 2011). A conversion efficiency of 85% is set for heat production[5] (EC 2010) and a power to heat ratio of 1:3 is used[6] to calculate the productivity of on-site electricity production (EPA 2000), offsetting 0.2407 kg CO_2-eq per kilowatt-hour (kWh) for gas and 0.5246 kg CO_2-eq/kWh for electricity (Carbon Trust 2011).

Data presented in table 2 are averages and default values based on EU15 technologies set within HELCA in order achieve harmonization. Using these as exact values may therefore be misleading, for example, differences in disposal infrastructure efficiencies between nations will affect actual savings achieved. Another limitation is the broad product classifications, for instance, in the lack of distinction between cardboard and paper and there are no data for biocomposite, insulation, or some other hemp co-products. This required the selection of a relevant biomaterial type as a proxy. These are necessary omissions attributed to data availability. Such limitations can be a feature of harmonization given that not every specific situation can be accommodated. However, it should nevertheless be noted that regular updates are not uncommon in harmonized tools, as illustrated by the technologies and default values used in SAP, which are often updated (DECC 2013).

Having defined the default values, table 3 shows the typical UK disposal fates for each co-product as described by government reports (DEFRA 2011). Typical UK EoL scenarios are used to investigate how the current UK infrastructure supports low-carbon biomaterials. Generally, UK landfill sites operate without gas recovery[7] and energy from waste plants seldom have combined heat and power (CHP). The most common waste sent for incineration is MSW, including other/waste and bioplastics. Paper is usually recycled whereas all non-MSW, such as building materials, are sent to landfill. All agricultural and horticultural products are assumed to be composted given that facilities using these products (amenity parks, farms, and so on) are likely to undertake composting on-site to avoid waste disposal costs.

Results and Discussions

Life Cycle Impact Assessment

Tables 4 and 5 show the LCIA for biorefineries A and B, respectively. The influence of adding an additional reuse or recycle stage is investigated in the sensitivity analysis. The top line in each table describes the emissions anticipated from the typical UK waste infrastructure previously described in table 3; the subsequent lines describe the absolute potential of each disposal technology for each co-product. The results show that hemp's disposal could add an additional 139 g CO_2-eq/kg of hemp feedstock to its supply chain emissions if landfill was the only disposal option pursued or, alternatively, it could have a net reduction up to –790 g CO_2-eq/kg hemp feedstock when a combination of on-site and incineration CHP are used. Clearly, the disposal stage of biomaterials is hugely influential to their overall CF. Any biomaterial LCA that does not include disposal in its scope can only claim to be a partial analysis.

In both biorefineries, landfill and then landfill with electricity recovery provide the greatest net emissions because of the carbon released in decomposition. All the other options produce a net reduction in emissions, that is, they offset energy or products that would have caused even more GHG emissions. Nevertheless, compost only marginally reduces emissions because it replaces an already relatively low-carbon product. Ethanol conversion and AD provide some GHG reductions, but are not ranked particularly high, especially if AD does not incorporate CHP, indicating that the efficiency or yields are relatively poor at present. Also, these are technologies that are only suitable for certain products, neither of which are suitable options for bioplastics. AD, on the other hand, can only be considered for paper waste and putrescible waste.

Recycling and reuse as one-off disposal options show some significant benefits for paper and plastics. The sensitivity analysis in the next section analyzes the cumulative benefits achieved when reuse and recycling are coupled with another final disposal fate. Incineration with CHP is clearly the number one low-carbon option in both biorefineries A and B, though after this the order of preference between the disposal options changes. This is mainly attributed to the classification *types* that each product is given in HELCA and the fact that more on-site heat recovery was possible in biorefinery B, whereas biorefinery A made a greater quantity of products thought to end up in incineration with CHP plants. The definitions of each group greatly influence their ability to yield net disposal GHG reductions. The influence of changing the type of the biomaterials is assessed in the sensitivity analysis.

One feature of biorefinery B that meant it had higher net emissions was that it produced more animal feed, which is not allocated a disposal option at all. This makes animal feed look like a relatively high-carbon product; however, this may be relatively disingenuous because it is a necessary part of the supply chain for those other products that do have potential for offsetting emissions. The impact of using a *cumulative* allocation approach, where all the co-products are allocated a chunk of the overall net disposal emissions based on their mass, is discussed in the sensitivity analysis.

Is the UK Waste Infrastructure Able to Support a Low-Carbon Bio-based Economy?

The typical UK disposal scenario applied to biorefineries A and B was moderately low carbon: –127 and –69 g CO_2-eq/kg hemp, respectively. The main benefits are provided from on-site heat production at the biorefinery sites, from recycling of paper and a small amount from composting animal bedding, thus there is room for improvement. Landfill of biomaterials is the only scenario that increases net GHG emissions (even the inclusion of electricity recovery on landfill sites makes only a

Table 3 Typical UK end-of-life scenarios for hemp products

Biorefinery A co-products	*UK typical disposal*	*Biorefinery B co-products*	*UK typical disposal*
Bird feed	n/a	Flower essential oils	n/a
Dust	On-site heat	Seed and feed	n/a
Fish feed	n/a	Dust	On-site heat
Plastic	Incineration with electricity recovery	Animal bedding	Compost
Paper	Recycling	Shivs in construction	Landfill
Insulation	Landfill	Composite board	Landfill
Animal bedding	Compost	Insulation	Landfill
Composite board	Landfill	Mulch	Compost
Waste	Incineration with electricity recovery	Paper	Recycle
		Waste	Incineration with electricity recovery

Note: n/a = not applicable. Q12

Table 4 End-of-life scenarios for hemp biorefinery A (disposal O_2-eq/kg feedstock)

Co-product description	*Bird feed*	*Dust*	*Fish feed*	*Plastic*	*Paper*	*Insulation*	*Animal bedding*	*Composite board*	*Waste*	
Selected product type	*Food/ consume*	*Ligno-cellulosic waste*	*Food/ consume*	*Bioplastic (PET)*	*Paper*	*Textiles/ fibres*	*Textiles/ fibers*	*Textiles/ fibers*	*Putrescible/ biowaste*	*Total*
Typical UK	—	−62	—	0	45	2	−13	5	−3	−25
Landfill	—	31	—	4	48	2	11	5	30	132
Landfill, Electricity	—	29	—	4	42	1	5	2	29	113
Reuse	—	0	—	−18	—	—	—	—	—	−18
Recycling	—	0	—	−18	−138	—	—	—	—	−156
Incineration, Electricity	—	−1	—	—	−45	−21	−103	−45	−3	−218
Incineration, CHP	—	−31	—	—	−131	−62	−299	−132	−9	−664
Ethanol	—	−4	—	—	−14	−3	−16	−7	−3	−47
AD, Electricity	—	0	—	—	−20	—	—	—	−4	−24
AD, CHP	—	0	—	—	−35	—	—	—	−7	−42
Composting	—	−3	—	—	−7	−3	−13	−6	−1	−33
Onsite Heat	—	−62	—	—	—	—	—	—	−28	−90
Onsite CHP	—	−91	—	—	—	—	—	—	−40	−131

Note: kg CO2-eq/kg = kilograms carbon dioxide equivalent per kilogram; PET = polyethylene terephthalate; CHP = combined heat and power; AD = anaerobic digestion. Q13

marginal improvement), yet currently it is estimated that over 50% of council-collected waste sent to landfill (DEFRA 2015) is biodegradable. Thus, if a low-carbon and bio-based economy is likely in the future, then the UK's waste landscape needs to shift away from landfill as a default. Some of these issues may already be seen to be recognized in legislation, for example, by the EU Waste Directive, which aims to phase down recyclables and biodegradable waste from landfill to 25% by 2025 (EC 2014).

Alternative technologies, such as ethanol and AD, may provide a future solution. However, their net carbon savings are currently lower than incineration can provide. There may be alternative motivations to pursue these technologies, such as public acceptability compared to energy from waste plants, increasing the localization of power generation, or it may prove more cost-effective than mass-scale incineration, an area that is not assessed in this article.

According to these results, to improve the UK waste infrastructure so that it may better support a low-carbon bio-based economy, improvements should target increasing recycling and reuse rates and, critically, expand the number of energy from waste (EFW) sites in the UK that incorporate CHP. Currently, only 13% of MSW goes to EFW plants and these do not usually have CHP (DEFRA 2014), thus it is unrealistic to suggest that the UK waste infrastructure could support a low-carbon bio-based economy without significant change.

What Role does Disposal Play in Biomaterials Being Low or Zero Carbon?

The estimate of downstream GHG emissions for 1 kg of hemp feedstock was 662 kg CO_2-eq. For hemp to be considered a net zero-carbon product, it must have a disposal option that represents a savings of equal magnitude. According to table 4, in biorefineries A and B, the most effective disposal option for end products to reduce emissions was through incineration with CHP. The most effective disposal option for lignocellulosic and putrescible waste material in the biorefinery was on-site CHP (because it avoids the losses and transport penalties associated

Table 5 End-of-life scenarios for hemp biorefinery B (kg CO_2-eq/kg feedstock)

Co-product	*Essential oils*	*Seeds*	*Dust*	*Animal bedding*	*Shivs for construct*	*Composite board*	*Insulation*	*Mulch*	*Paper*	*Waste*	
Selected product type	*Food/consume*	*Food/consume*	*Ligno-cellulosic waste*	*Textiles/fibers*	*Textiles/fibers*	*Textiles/fibers*	*Textiles/fibers*	*Textiles/fibers*	*Paper*	*Putrescible/biowaste*	*Total*
Typical UK	—	—	−14	−11	2	1	2	—	−140	−13	−7
Landfill	—	—	7	9	2	1	2	—	38	152	213
Landfill, Electricity	—	—	7	4	1	1	1	—	33	146	193
Reuse	—	—	—	—	—	—	—	—	—	—	—
Recycling	—	—	—	—	—	—	—	—	−105	—	−105
Incineration, Electricity	—	—	—	−88	−21	−12	−21	−3	−35	−13	−194
Incineration, CHP	—	—	−7	−255	−62	−35	−62	−9	−104	−45	−578
Ethanol	—	—	−1	−14	−3	−2	−3	—	−11	−16	−50
AD, Electricity	—	—	—	—	—	—	—	—	−16	−21	−36
AD, CHP	—	—	—	—	—	—	—	—	−28	−37	−65
Composting	—	—	−1	−11	−3	−1	−3	—	−6	−7	−31
Onsite Heat	—	—	−14	—	—	—	—	—	—	−138	−152
Onsite CHP	—	—	−20	—	—	—	—	—	—	−202	−222

Note: kg CO2-eq/kg = kilograms carbon dioxide equivalent per kilogram; CHP = combined heat and power; AD = anaerobic digestion.

Q15

with connecting to the grid). Thus, if hemp supply chains only adopted these disposal options, the net contribution to its CF would be –790 kg CO_2-eq/kg for biorefinery A and –730 kg CO_2-eq/kg for biorefinery B. This is sufficient to indicate that hemp could theoretically be carbon neutral. However, as stated in the section titled *Is the UK Waste Infrastructure Able to Support a Low-Carbon Bio-based Economy?*, incineration with CHP is not a common waste treatment in the UK and so this scenario may never be achieved in practice.

Q14

An alternative solution to achieving zero carbon may be the use of on-site heat recovery or CHP given that this is shown to provide the greatest GHG savings possible for lignocellulosic waste, which avoids the losses involved with connecting to the national power grid. Manufacturers who do this would have a much stronger basis to carbon neutrality claims, and it should be recommended for inclusion in future carbon-neutral biomaterials certification schemes that may develop.

Savings achieved by the next best disposal options after onsite CHP, on-site heat recovery and incineration with CHP, would be provided by incineration with electricity. However, this would not yield sufficient savings for hemp to claim to be carbon neutral. This means that incineration with CHP and on-site energy recovery is not only the most effective means to achieve carbon neutrality, but also they are integral. Currently, however, they are not widespread technologies and therefore it is unlikely that hemp and therefore other biomaterials in the UK would be zero carbon, unless they have exceptionally low-carbon upstream supply chains.

Sensitivity Analysis

This section investigates the influence on GHG emissions of three methodological decisions: (1) allowing products to take advantage of a reuse or recycle stage before disposal; (2) changing the biomaterial classifications of the co-products; and (3) applying a cumulative allocation approach to distribute net disposal emissions.

Additional Reuse Cycles

Paper and bioplastic co-products are, in many instances, able to be reused or recycled before their ultimate disposal. In doing so, the potential for additional net savings should increase. To illustrate this, we have applied such reuse and recycling stages to the biorefinery A analysis, as shown in figure 2.

When the paper and plastics produced by biorefinery A are recycled, then the cumulative benefits are sufficient to offset almost all the emissions that would occur if the co-products were then finally sent to landfill. Soiled paper and textiles cannot be easily reused, and so the same effect is not observed when considering reuse only. Reductions as a result of reuse and recycling are observed in the typical UK case. Despite these additional net savings, however, no scenario other than incineration with CHP can cause reductions near to the 662 kg CO_2-eq threshold, a level that equals the net upstream supply-chain emissions assumed for cultivating and producing the hemp. Thus, simply reusing or recycling a biomaterial is unlikely to be sufficient if the aim is a zero-carbon product, given that incineration with CHP would still be required. Only one reuse cycle has been investigated here because it is likely that biomaterials will degrade during the reuse or recycling process, and it may not be possible to reuse them multiple times. There is very little difference between reuse and recycling in this assessment because there is very little difference in the processing emissions provided by the EC data. In addition, the EC makes the assumption that the reuse and recycling has occurred to provide a product of equal value, where, in fact, downgrading may take place, in which case the savings calculated would be overstated.

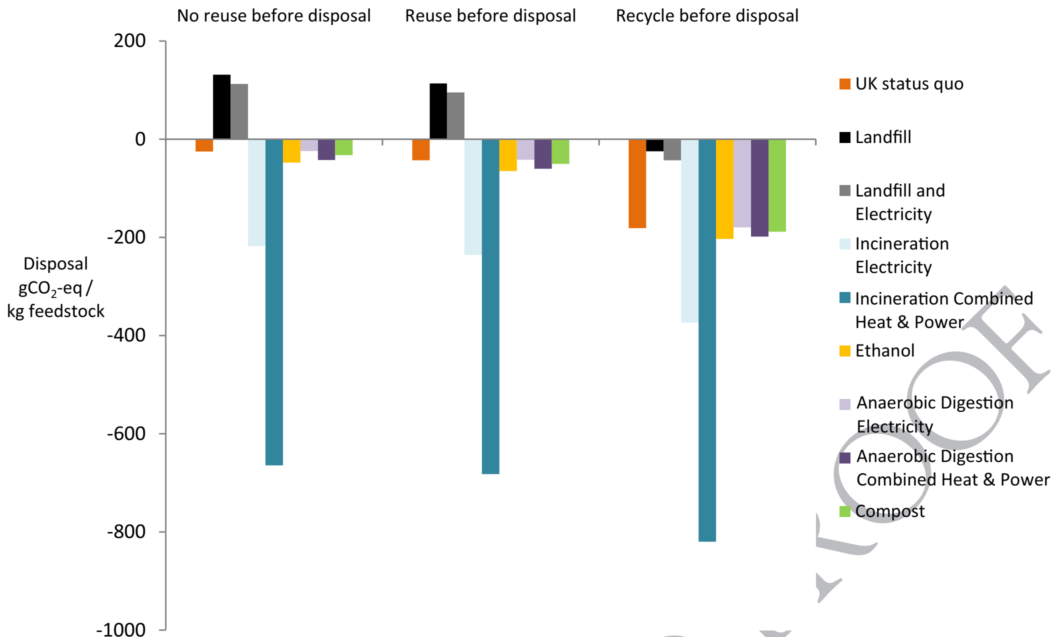

Figure 2 Biorefinery A with reuse and recycling.

Biomaterial Classification

In biorefineries A and B, we made assumptions on the classifications for each biomaterial product. This was done because the data regarding disposal emissions caused by biomaterials are scarce and only available at a low level of disaggregation, for example, no distinction is made between cardboard and paper. In making these selections, there is inevitable ambiguity as to which products should be in which category types. Whereas this may be straightforward for products such as paper, plastics, and animal feed, it is less so for textiles and lignocellulosic waste. A second analysis of biorefinery A was therefore undertaken to explore this. Each of the co-products previously categorized as textiles were, this time, identified as lignocellulosic waste. The results of making this switch are shown in figure 3.

As can be seen, the major influence occurs with the amount of emissions released by landfill and the reduction in the amount of energy that can be recovered by incineration. The reason for this substantial change in GHG emissions (see table 2) is because the data for MSW were used as a proxy for lignocelluosic waste and compared to textiles, MSW is assumed to release more emissions during decomposition and less energy can be recovered from it during incineration. By removing the textiles component, the net carbon emissions of the biorefinery substantially increase. This highlights the implications of using relatively broad data sets. As such, if a harmonized tool were to be more widely used, it would need to produce guidelines for selecting classification types, as well as providing more refined data in order to reduce these biases.

Individual vs. Cumulative Allocation

In LCA, it is common to allocate emissions between co-products according to their relative importance in the supply chain (in our case, according to their mass), irrespective of whether they were directly responsible for a large or small portion of these (see the section titled *Allocation*). We investigate the effect of this approach on our biorefinery by allocating the total net emissions to each co-product according to their mass. Figure 4 compares the results of this for the typical UK scenario on biorefinery A. The total disposal emissions, assuming the typical UK waste management decisions previously described from the biomaterial, do not change, but as can be seen, the distribution of these between the co-products does.

Under the *individual* method, all net disposal emissions were allocated to those co-products that were directly responsible for them and the standout low-carbon product was paper, owing to it being recycled, and the lignocellulosic dust that was used to generate on-site heat. The composite board and insulation were shown to be net emitters because their only available disposal option was to be sent to landfill. This may be seen to be unfair, in some instances, given that all the products were a necessary part of the production and so should all receive some credit for the net emissions saved. This situation is described by the cumulative allocation data.

Using the cumulative method, the net emissions reductions are shared more equitably between co-products, so much so that there are now no co-products that are seen to be net emitters of GHG emissions at their disposal stage. Indeed, animal bedding, as the most massive product in this scenario, takes the greatest share of the emissions saved in the disposal stage, even though it was not directly responsible for the vast bulk of these. If this were deemed to be a more equitable method of allocation, the implications of applying this would need to be considered when setting the rules for a harmonized tool, because it diminishes the savings of some while improving those of others.

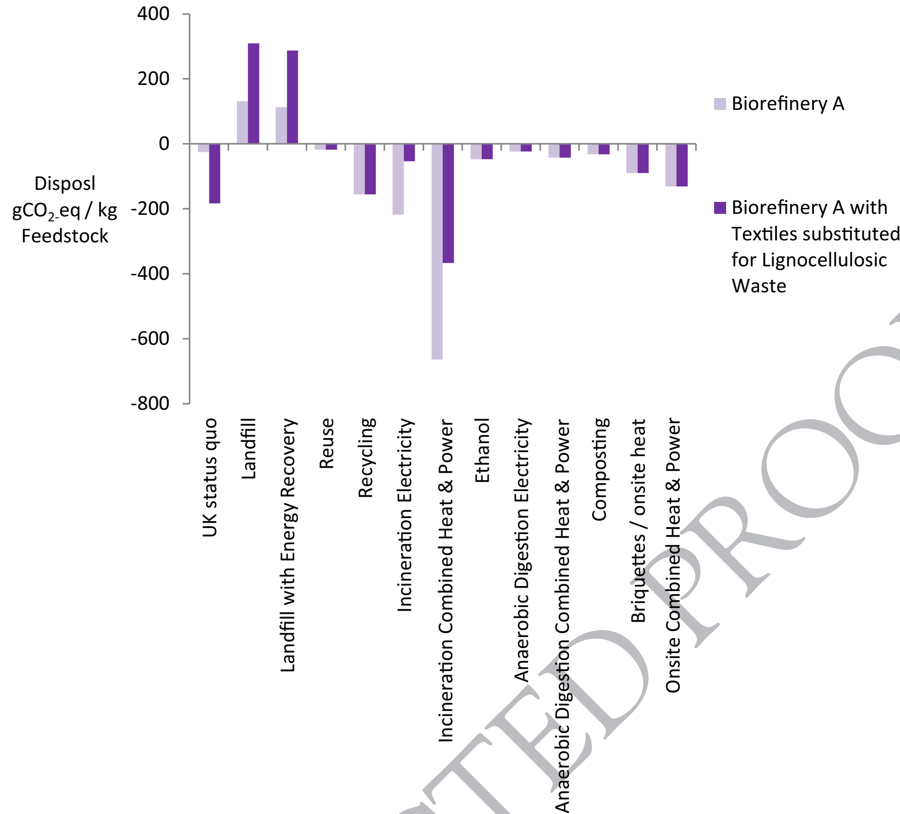

Figure 3 Influence of classifying textiles as lignocelullosic waste.

Implications

Life Cycle Assessment Interpretation

We have provided various discussion points throughout the presentation of the results; however, in the *Introduction*, we also described four barriers to disposal being incorporated into biomaterial LCA. We have used two hypothetical hemp biorefineries to investigate this and now discuss how the barriers to harmonization have been addressed.

Barriers to Harmonization

The first barrier presented in the *Introduction* regards the question of ownership over the disposal emissions. Harmonization could remove this uncertainty. We identified two ways in which the net emission reductions could be related to the individual products (i.e., how the emissions should be "owned"). With the individual allocation method, it was clear which coproducts were responsible for the most savings. However, with the cumulative method, the emissions saved were more equally distributed. A key strength of harmonization is that it may select one of these methods and ensure that all stakeholders are consistent in this; alternatively, results by both methods may be reported.

The second barrier addressed was the problem that the disposal fate of individual products is not known. As with the first barrier, the allocation method will be important and harmonization can provide clarity on this. We have shown in the extremes that if all products are sent to landfill, they will cause a net increase in emissions, but if they are sent to energy from waste plants, they may even be carbon neutral. By identifying the typical UK case, we can come to some kind of compromise, in that there is likely to be a slight reduction in emissions attributed to the disposal of biomaterials. Though this may not articulate the ultimate potential of each biomaterial, it nevertheless represents the most likely scenario. Harmonization would ensure that each company is required to use the same values, whether typical disposal fates are set at an EU or national level.

The third barrier argued that LCA can often be costly and complicated and, in so doing, limit its application by those without required resources or skill sets. On this account, HELCA could also be said to have had some success, given that the data sets are made available and the rules are set (assuming a consensus was achieved), meaning that an intimate knowledge of LCA is not required to complete the assessment. With this being the case, individual producers could use the tool for their own products and each receive a disposal stage net GHG value that, within reason, would be comparable. Harmonization therefore means that producers could cheaply and simply undertake LCA for their products. However, this might come at the cost of not crediting specific cases of more-efficient energy recovery infrastructure. Nation-specific data could alleviate this to some degree or, alternatively, it is not unusual to allow standard credits for good practice in harmonization, as in the case

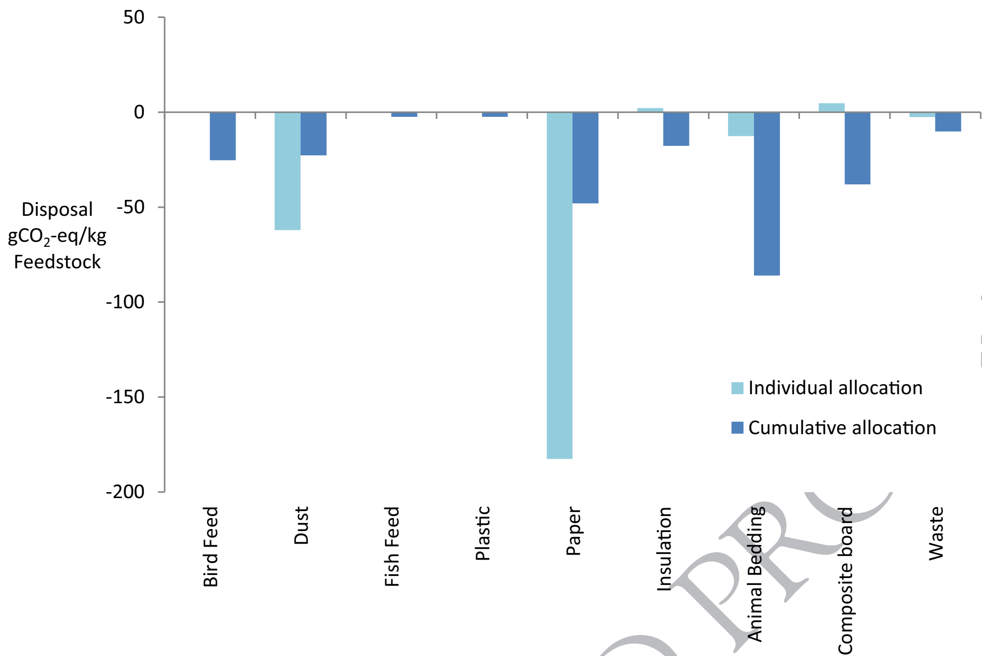

Figure 4 Comparison of individual and cumulative allocation methods on typical UK scenario for biorefinery A.

Q16

of BioGRACE crediting good agricultural practice for biofuels producers.

Harmonization was intended to remove the number of "what ifs" that we identified as the fourth barrier. Whereas a standardized approach will, to an extent, achieve this, and we have justified our particular methodological steps throughout this article, often there were several reasonable alternatives that could have been taken. These include, for example, choosing to base the functional unit around the feedstock rather than specific biomaterial products, whether to use individual or cumulative allocation methods, how many reuse cycles to allow, and which proxy classifications and emissions data to use where there was no perfect match. Whereas harmonization would necessarily reduce these what if barriers, the study has highlighted the sorts of challenging issues that will require scientific and industry consensus for a harmonized tool to be accepted.

Limitations and Future Improvements

Disposing of biomaterials will have many effects on the environment, society, and economy. Yet, in this article, the scope of the study was limited to GHG emissions. Research suggests that biomaterial's main impacts may not be GHG, but be associated with eutrophication and eco-toxicity (Tabone et al. 2010). Whereas the scope of our study has been clearly set out from the beginning, in other future studies, other impacts could be considered. Further refinement in data would improve the accuracy of the results, for example, to allow the distinction between the types of bioplastics or between cardboard and paper, or to describe more fully the carbon balance of ethanol production. Being able to select the "home" nation for the analysis would also make the tool more useful so that nations' own waste infrastructure and efficiencies are accurately represented in the GHG value. Our assessment has centered on the emissions of a hemp biorefinery, and it may not be possible to make conclusions on biomaterials in general from the hemp results. In particular, replicating the analysis on a range of other feedstock would be useful. Investigating the industry and policy appetite for harmonization in this area would certainly make for an interesting study.

Conclusions

The study has highlighted disposal's influential role in LCA, raising the possibility that biomaterials, specifically hemp, could be zero carbon given the right mix of disposal infrastructure. It has also provided evidence to suggest that current UK waste infrastructure may not be able to support a low-carbon biomaterials-based future economy. This perhaps challenges the idea that the producer is wholly responsible for the emissions associated with their products given that the waste infrastructure of the country in which they operate may ultimately control whether or not their biomaterial products may be low carbon.

Disposal emissions are known to be fundamental to the overall CF of biomaterials, but are not equally well represented in biomaterial LCA literature. This study undertook an LCA of hemp to investigate the impact on GHG emissions of the disposal stage of biomaterials. Specifically, the article presented harmonization as an approach to overcoming the four barriers of disposal stages being incorporated into biomaterial LCA studies, these barriers being: ownership of emissions; uncertainty of disposal fates; cost of analysis; and comparability of results.

Although the study suggests that adopting harmonization could be one solution to these barriers to encourage disposal impacts to be included into biomaterial LCA, it also identifies several weaknesses. These weaknesses need to be considered before such a system could be developed to anything like that representing the EU biofuel harmonized LCA approach. For example, accurate data and general data availability and acceptability have been shown to be key limitations. Specifically, issues to consider are that the diversity (even within categories) of biomaterials means that waste databases will need to be large; consideration needs to be given as to how categorization of biomaterials could take place without requiring user-informed decisions; efficiency of waste infrastructure and technology varies based on location and the degree of specificity and options to refine data will need to be established; and linking disposal emissions to upstream biomaterials emissions will be required if complete LCAs are desired.

Some questions around which methodological steps to take have also been shown to be important in this study and consensus on adopting the cumulative or individual allocation approach will be needed, as well as setting limits on the number of times that a product can be claimed to be usefully reused or recycled.

Acknowledgments

The research is funded by the Yorkshire White Rose Consortium of Universities. None of the authors are aware of any conflicts related to the preparation of this research.

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