Validating low carbon bio-renewable alternatives to petrochemicals: exploring end of life impacts

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Abstract

A need to reduce anthropogenic carbon emissions has prompted a trend for industry to switch from fossil to biorenewable feedstock, but it remains unclear if this is always a 'low carbon' decision. The concept that biomaterials can have lower greenhouse gas (GHG) emissions than petrochemicals has been grasped enthusiastically, however one aspect in particular has been somewhat overlooked when considering life time emissions; their disposal. Low carbon waste management is not a new concept, but its application to waste streams with high bio content is not well understood.

This thesis employs mixed methods to investigate the impact of end of life scenarios on the GHG emissions associated with biorenewable materials. A hybrid life cycle assessment (LCA) of a biomaterial and petrochemical product shows that end of life scenarios have a bigger impact on overall GHG emissions for waste biomaterials than those based on petrochemicals and shows that biomaterials can be lower carbon if disposal is taken into account.

In order to understand how such benefits from biomaterials can be realised, fourteen interviews with biomaterial industry stakeholders were then conducted and provided insights from which policy options to promote low carbon disposal are developed. A focus group with nine experts considered these options and made recommendations to raise the profile of disposal via encouraging product purity, stimulating demand and updating collection infrastructure. One other recommendation was to provide more transparency regarding the benefits of particular disposal options on specific biomaterials and in order to help with this the final part of this thesis is devoted to the development of a low carbon decision support tool for biomaterial disposal options based on LCA and tested on two hemp biorefineries.

This tool was used to rank all the disposal options according to GHG emissions as well as cost effectiveness, particularly useful in locations where preferred strategies may not be available for example where there is no district heating infrastructure to support Combined Heat and Power (CHP). Its results confirm the waste hierarchy but also shows novel technologies such as ‘ethanol from waste’ are can be both low carbon and economically competitive. This tool can both help biorefinery operators to design low carbon disposal options into their products, as well as help guide waste and biomaterial policy decisions. The tool suggests that existing UK disposal infrastructure for municipal solid waste streams is not designed with biomaterial waste in mind, and that a rethink in waste disposal and its funding may be required to ensure future bio-based economies achieve better reductions in carbon emissions.

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On accompanying CD:

- HELCA tool (Excel, for full functionality please use version 2010)
- HELCA Guide Book (PDF)

Preface

This thesis consists of work carried out between 2010 and 2013 to attain the qualification Doctor of Philosophy. It comprises three academic papers submitted to the Journal of Cleaner Production (published in 2012), Waste Management (in press in 2013) and the Journal of Industrial Ecology (under review, 2013). The PhD student, David Glew, was the lead author of all three papers, developing the original ideas, undertaking the research and interpreting the results. Co-authors provided guidance on research design and data analysis and provided editorial input to enhance the flow and intelligibility of the written work.

Acknowledgement

This thesis was made possible by the White Rose Consortium of Universities' generosity, Nigel Mortimer's pointers, Adolf's instruction, Lindsay's passionate supervision and Simon's trust and direction.

I would like to especially thank my wife, without her unaccountably warm support, critical eye and kindly delivered rebukes this thesis may exist but it wouldn't be anywhere near as good. Thank you.

Author's declaration

This is to certify that:

- The thesis comprises only my original work towards the PhD except where indicated in the Preface
- Due acknowledgement has been made in the text to all other material used.

David Glew

1. Introduction

This chapter establishes the aim of the study as well as introducing the significant terms, concepts and methods used in the research. Following this a critique of the relevant literature is given in Chapter 2. Three academic papers have been submitted for publication from this PhD around the concept of disposal impacts and biomaterial alternatives to petrochemicals these are presented in Chapters 3, 4 and 5 respectively. Those presented in Chapters 3 and 4 have been published or are currently in press. The paper presented in Chapter 5 is currently under review. The thesis closes with some final remarks in the concluding Chapter 6.

1.1. Research aim

Biomaterials are often presented as low carbon alternatives to petrochemicals. The aim of this research is to identify the importance of disposal when calculating the life cycle GHG emissions of biomaterials in order to develop more ‘realistic’ calculations of the GHG emissions of biomaterials and enable their comparison to petrochemicals over the full life cycle of a product.

1.2. Rationale and significance of the thesis

Petrochemicals are an important yet controversial resource and balancing their costs and benefits is not straightforward. In terms of problems, they contribute to climate change, pollute the air and waterways and their extraction disrupts ecosystems and communities (IPCC, 2007b, Verbruggen and Al Marchohi, 2010). These problems are exacerbated by the fact that they are not always found in the places in which their benefits are enjoyed meaning that different groups can bear the costs compared with those enjoying the benefits (IPCC, 2007a). In terms of benefits, petrochemicals afford decent living standards to those who use them, providing energy, products, transport and support for food production, all of which mean demand for petrochemicals is anticipated to grow over the next 20 years (OPEC, 2010).

Finding alternative renewable technologies and resources that share the benefits of petrochemicals but create fewer problems is an attractive proposition. As the old adage goes ‘the Stone Age did not end because of a stone shortage’, instead new technologies made survival more efficient. Thus, it may not necessarily take a shortage of petrochemicals, a ‘peak oil’ situation, before substitute resources become more commercially viable, technically possible and socially desirable (Verbruggen and Al Marchohi, 2010). Biomaterials and bioenergy are such alternatives available now and although these are used on a relatively small scale and can have large financial and environmental costs, interest in them is growing (Gallagher, 2008). Indeed, some believe the world is heading towards the growth of bio-based economies (Vandermeulen et al., 2012). Thus, this thesis, which contributes to understanding biomaterials’ environmental impacts, provides a useful and important contribution.

Specifically, this thesis addresses the impacts of disposal on the overall environmental impact of biomaterials. Growing feedstock for biomaterials and bioenergy has been shown to various degrees to cause significant GHG emissions, land use change (LUC), and affect food production and prices in addition to other environmental and social impacts (Gallagher, 2008). The disposal of biomaterials is less well studied but is an important part of their supply chain which can influence their GHG emissions. Biomaterials’ end of life options can be very different to that of petrochemicals since they are biodegradable and can have greater potential for recycling and energy recovery (European Commission, 2010a, European Commission, 2009b, Sarasa et al., 2009). Thus, in order to understand if biomaterials can be favourable alternatives to petrochemicals, understanding the complete picture, including the impact of disposal, is important.

1.3. What are biomaterials?

It is useful to start by defining what constitutes a biomaterial. Different words are commonly used in the literature to describe bio-based products though each term can actually refer to something specific. Figure 1.1 defines the more common terms; those highlighted in green are the focus of this research.

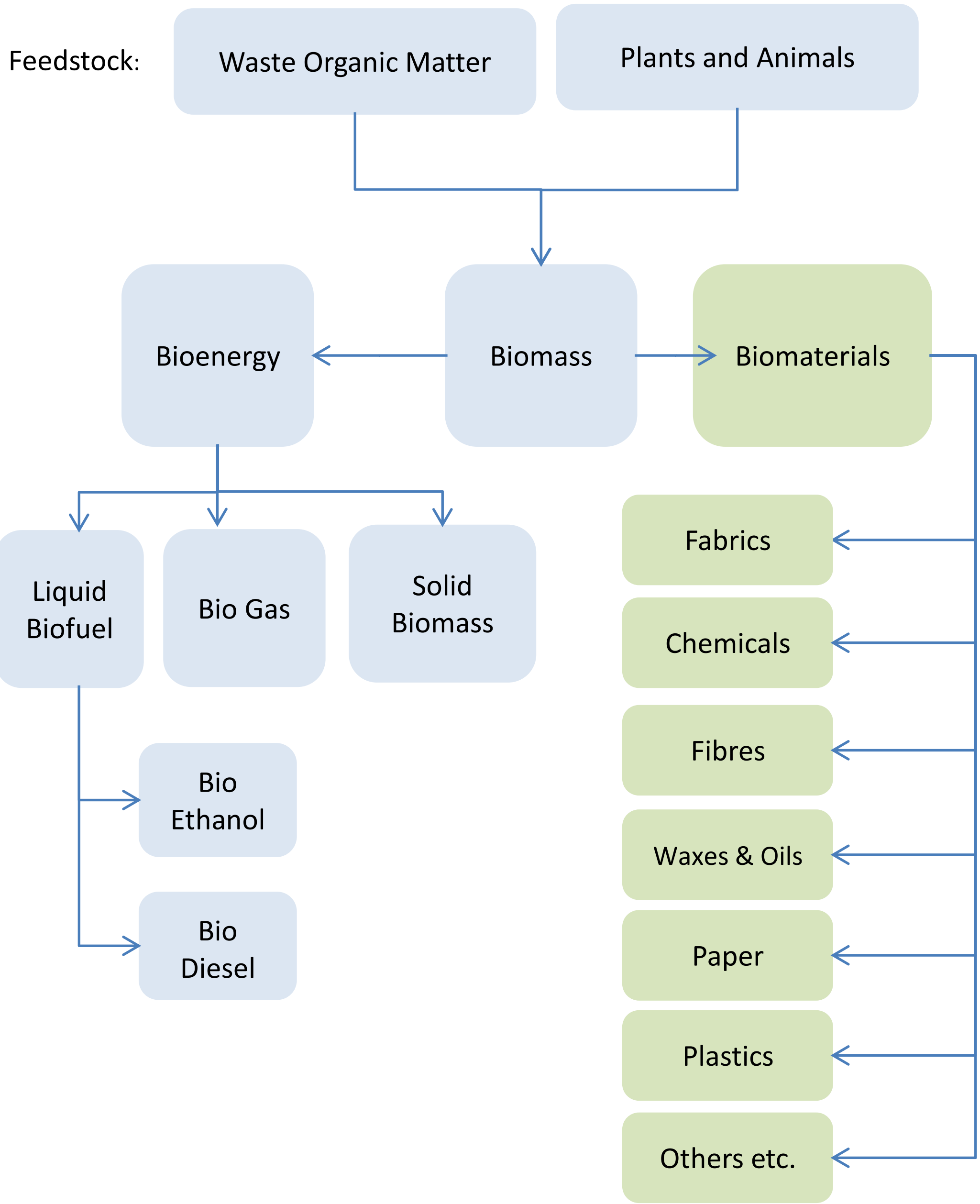

Figure 1.1 'Bio' nomenclature (developed by author)

Biomaterial ‘feedstock’ can come from a wide variety of sources including crops, trees, biomass from marginal land or even residues from processes like sawmills or municipal waste collection (Gallagher, 2008). Biomaterials (which are the focus of this research and shown in green in Figure 1.1) are manufactured in biorefineries.

These are factories that convert biological feedstock into different co-products. Biorefineries supply many industries: the pharmaceutical industry uses their chemicals; and the construction, clothing and automotive industries use their fibres and composites. Bio plastics are often used for packaging or to make products. The food and cosmetics industries use waxes, fats and essential oils made in biorefineries. Biorefineries also produce energy and fuels, though this is not always their central function.

The wider bio-based market is estimated to be worth 22 trillion Euros per annum in the European Union (EU) (Geoghegan-Quinn, 2010) and demand is growing (Salas, 2010). It is difficult to extract from this what the direct biomaterial and biofuel markets may be though there are suggestions that biofuels may only constitute around 14%[1] of the direct biomaterial market, yet they receive the majority of the media and academic attention. Indeed, predictions of future feedstock requirements often only consider that required for biofuels and bioenergy, ignoring other biomaterial based demand completely (Haberl et al., 2010). This means the predictions of future feedstock requirements could be greatly understated.

1.3.1. **Current thinking on biomaterials**

Depending on which side of the debate one sits, biomaterials either promote sustainable development and resource security, mitigate climate change and reduce dependence on oil because they release no fossil carbon, can be owned by local communities and are renewable; or, they emit more net greenhouse gases (GHG)

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[1] Based on 2010 trade estimates of biomass $572.9 billion and biofuels $56.4 billion ENVIRONMENT LEADER. 2010. *Biomass Market to Hit $693 Billion by 2015* [Online]. Available: http://www.environmentalleader.com/2010/09/20/biomass-market-to-hit-693-billion-by-2015/?graph=full&id=1 [Accessed 05/12 2012], CLEAN EDGE INC. 2011. *Global Clean-Energy Predicted Growth* [Online]. Available: http://climatecommercial.files.wordpress.com/2011/03/global-clean-energy-projected-growth-2010-2020-us-billions.jpg [Accessed 05/12 2012].

than burning fossil fuels, pollute ecosystems, exacerbate world hunger and devastate habitats, local communities and biodiversity through increasing deforestation (Tilman et al., 2009, Cherubini, 2010).

Many of these criticisms are laid at the door of ‘first generation’ feedstock that are derived from intensively grown food crops like wheat, soy and maize whose sugars are relatively easy to extract and use (de Vries et al., 2010). Conversely, ‘second generation’ feedstock (which must instead come from *lignocellulosic* crops that have higher lignin and cellulose content) are often considered to be less contentious, especially if they are sourced from woody material, non-food crops (like hemp, flax and miscanthus) and waste. These second generation industries are more embryonic because the technology for producing usable sugars from these materials is not yet competitive with first generation approaches and despite the potential environmental benefits they are currently less profitable (Black et al., Cherubini and Jungmeier, 2010).

Presently, first generation bio-feedstock dominate the market and are predicted to do so over the next 30 years since these are the most commercially viable (Offermann et al., 2011). Interest in second generation biomaterials is by comparison small but increasing (NNFCC, 2012). There is a similar bias in the environmental literature which focuses almost exclusively on the way a biomaterial or biofuel is sourced and produced (rather than disposed of), with sourcing and production even being supported by certification and regulation (Schlegel and Kaphengst, 2007). This research is novel because it considers the sustainability of biomaterials from the opposite direction by determining the effects of disposal on the biomaterial’s life cycle GHG emissions. Disposal has far less of a presence in the biomaterial literature and the legislative arena, even though waste management has a high profile in wider climate change debates (Coggins, 2001, ECOTEC, 2000, UNEP, 2010).

1.4. The difference between ‘low carbon’ and ‘sustainable’

‘Sustainable development’ and ‘sustainability’ are terms used in many different ways to describe a variety of ideas. Their exact definitions are not universally agreed upon but they both focus on three pillars: i) environment, ii) society and iii) economy (Robèrt et al., 2002, Elghali et al., 2007, Kates et al., 2005) and incorporate a future dimension that suggests that sufficient resources need to be maintained for future generations. This means issues as diverse as GHG emission reductions to social responsibility and profitability can all claim to cross-cut sustainability debates in some way, and it remains a challenge to consider each aspect simultaneously (Clift, 2007).

The literature differentiates between sustainability and sustainable development (Espinosa et al., 2008, Gomar and Stringer, 2011). Sustainability is regarded as eco-centric since it prevents activities that are harmful to the natural environment, for example, by diverting agricultural expansion away from pristine forests (Phalan et al., 2011). Sustainable development is more anthropogenic in its focus, since it encourages activities with some impacts on the environment if they replace other more damaging practices, such as selectively harvesting products within forests instead of clear felling for timber (Pearce et al., 2003). The concepts of ‘strong’ and ‘weak’ sustainability are similarly used in ecological economics to distinguish eco-centric approaches where all types of capital (including natural resources) are considered equally within strong sustainability. This contrasts with a more anthropogenic centred approach where social and economic capitals are prioritised in weak sustainability (Ekins et al 2003). The research in this thesis is investigating biomaterials as potentially less environmentally harmful products than petrochemicals. As such, the thesis may be said to be concerned primarily with ‘sustainable development’ or ‘weak’ sustainability. Any reference to ‘sustainability’ or ‘being sustainable’ refers to these concepts and alternate terms are only used to aid the flow of the writing.

Exact definitions of sustainability vary depending on an individual’s priorities which may be broad or narrow. This makes the term sustainability difficult to use without causing confusion (Frazier, 1997). For example, in some reports GHG emissions may be considered more important than the risk of eutrophication and biodiversity loss or vice versa depending on the values of the report’s authors. Although it can be useful to simplify assessments by using only one criterion to evaluate sustainability it nevertheless introduces bias (Ahlroth et al., 2010). To reduce bias, assessments can study a range of indicators across a range of different dimensions, though this adds complexity, requires additional work and inevitably some weighting may still persist in the selection of indictors (Kates et al., 2005).

In this research although the three pillars of sustainability are discussed, the focus is mainly on low carbon biomaterials and GHG emissions. This approach is justified because data sets are more abundant for GHGs than for other impacts; GHGs are a familiar lexicon of policy and political discourse; existing legislation and sustainability assessments on biofuels already target GHGs; GHGs are independent of geography whereas the importance of other impacts can vary with location; companies are often familiar with GHGs as a key performance indicator; consumers are often familiar with GHGs through the popularisation of carbon foot printing labels; and finally, measuring sustainability in a holistic way would require multiple methodologies and a time frame beyond that available for this research.

1.5. Why study biomaterials and not biofuels or other renewables?

Bio-feedstock are particularly important because they offer a range of potential alternative products like plastics, textiles and chemicals compared to other renewables like wind or solar, which are limited to providing only heat and power. Even in terms of energy-only products, bioenergy can be more land efficient than solar power and offers a more stable supply than wind power, while it can uniquely produce liquid fuels that can be used in internal combustion engines and stored and transported easily (Carus, 2010, Salas, 2010). Bio-feedstock is therefore afforded a unique position and attracts considerable attention.

Most studies on environmental impacts and predicted future land use tend to focus on biofuels only and omit the contribution of biomaterials altogether (Haberl et al., 2010), whereas this study only focusses on biomaterials. The lack of consideration given to biomaterials in the literature means the impacts of switching from a petrochemical to a bio-based economy are not fully understood, and redressing this balance is one of the motives for this thesis. For example the European Union (EU) produced biofuels legislation with mandatory sustainability criteria as part of the EU Directive on the Promotion of the Use of Energy from Renewable Sources (RED) (European Commission, 2009a) but omitted other biomaterials, despite these having potentially the same negative impacts on the environment and society. This unequal treatment can distort the market and the scenario may arise where a producer sees their feedstock rejected on sustainability grounds for biofuels but accepted to make other types of biomaterials.

There is a great variety of biomaterials currently on the market, and products can be made from first generation energy crops like palm oil (*Elaeis guineensis*), soy (*Glycine max*), wheat (*Triticum*) and maize (*Zea mays*) as well as second generation lignocellulosic feedstock like hemp (*Cannabis sativa)*, flax (*Linum usitatissimum*) and willow (*Salix*) (Elsayed et al., 2003). Throughout this research hemp is used as an exemplar feedstock since it is a second generation up-and-coming feedstock currently being used in EU markets to provide a wide range of products.

1.5.1. **Hemp**

In the 1800s hemp was traditionally used for making ropes and paper, while today it has myriad uses such as in papers, fabrics, composite waxes, oils, feed, food and also fuel (Cherrett et al., 2005, Johnson, 2010). Hemp is considered to be a lignocellulosic second generation crop. Under certain production conditions it has been shown to require low agricultural inputs. It is increasingly being grown in Canada, Europe and Australia to produce biomaterials but it does require more complex processing (Abass, 2005). Studies have publicised hemp's ability to be used as a biofuel whereas others identify its flexibility in being a material that can be used to make multiple products (Finnan and Styles, González-García et al., 2010, Poiša L. et al., 2009, Prade, 2011, Rice, 2008, DEFRA, 2004, Johnson, 2010, Turunen and Werf, 2006).

1.6. Overview of methods used in this study

The research in this thesis is both quantitative and qualitative and uses methods including Hybrid Life Cycle Assessment (LCA) (Chapter 3), focus groups and interviews (Chapter 4), and Harmonised Process LCA (Chapter 5). These research methods are described in detail in the relevant chapters though a general introduction is provided here in order to set the scene.

1.6.1. Hybrid life cycle assessment

The first research paper presented in Chapter 3 of this thesis uses Hybrid LCA to study the GHG emissions of a product which uses hemp as one of its inputs. Simplistically, LCA collects input data on the whole supply chain from start to finish and everything in between then converts these inputs to impacts- GHG emissions in the case of this research. Hybrid LCA combines conventional Process LCA (where inputs are directly measured) with Input Output (IO) LCA (which uses economic data to calculate emissions rather than taking direct measurements). Thus, hybrid LCA are able to pick up emissions that are missed by Process LCA alone, and provide more specific advice than using IO in isolation. The LCA in this research specifically focusses on how the GHG emissions of a hemp product changes depending on its disposal option compared to a petrochemical alternative.

1.6.2. Interviews and focus groups

A second aspect of the research, having investigated the impact that end of life scenarios have for biomaterials compared to petrochemicals, was to establish the awareness of industry representatives regarding the importance of disposal options when considering the carbon footprint of their products. Interviews and focus groups were used as data collection methods and the sampling and approach are presented in detail in Chapter 4.

Interviews are used as a research tool to extract current thinking from a group of respondents, in this case the stakeholders from the biomaterials industry. Semi structured interviews specifically are established techniques used to gain a snap shot of opinions. Stakeholders from all stages of the biomaterial supply chain were interviewed, including growers, manufacturers and retailers from various biomaterial industries using different biomaterials including hemp.

Focus groups are used as a complementary, more targeted research method (Tashakkori and Teddlie, 1998) and were used in this research to glean the expertise from stakeholders from academia and industry in the waste, biomaterial and sustainability sectors so there could be a more thorough investigation of the important themes arising from the interviews. Focus groups also allow discussion of potential recommendations that may encourage disposal impacts to be incorporated into the operations of the biomaterial industry (Neuman, 2004).

1.6.3. **Harmonised life cycle assessment**

Having discovered how disposal is viewed by the biomaterial industry and what they thought should be done about it, the final stage of the research is to investigate a means to improve the transparency of the impact on carbon emissions of different disposal scenarios. LCA is returned to in Chapter 5 where a process LCA is proposed that ‘harmonises’ or standardises the calculation assumptions and methodology as well as some of the input data to enable benchmarking of the disposal GHG emissions across the biomaterials industry. Theoretical Hemp biorefineries are used as a case study to investigate how hemps life cycle GHG emissions change when different disposal options are pursued.

1.7. Contribution to knowledge

This thesis verifies the view that the disposal of a biomaterial can be more important than the disposal of a petrochemical in terms of GHG emissions. It gives an example of using up and coming hybrid LCA methodology on biomaterial as well as petrochemical supply chains. The research reveals the specific attitudes and organisational barriers experienced by the biomaterials industry which explain why disposal is less well considered in GHG calculations and business psyche. It initiates a policy discussion on a topic where no legislation currently exists and identifies policy options not previously considered before to ensure biomaterial disposal is not at odds with climate change objectives. The first attempt to quantify disposal GHG emissions using harmonisation in LCA via a decision support tool is made. This may pave the way for more widespread consideration of this form of LCA. The research considers the implications of existing UK waste management habits on a future bio-based economy and highlights that infrastructure and attitudes may need changing before such a future may be considered low carbon.

1.8. Chapter summary

The literature provides evidence that biomaterials can represent realistic low carbon alternatives to petrochemicals though there is currently inadequate knowledge of the influence of disposal on their GHG balance. This chapter has introduced some of the key concepts underpinning the research and has provided a broad overview of the quantitative and qualitative techniques employed. The aim of the research has been presented and the key knowledge contribution the thesis will make to the understanding of how a bio-based economy could also be low-carbon has been set out.

2. Literature review

2.1. Introduction

This chapter discusses the consensus, disagreements and gaps in the literature on biomaterials. It highlights their predicted growth and perceived low carbon characteristics and identifies the current ways in which disposal influences their GHG emissions. It also critically evaluates the appropriate research methods used in the thesis and explains why hemp is a useful archetypal biomaterial feedstock for the study.

2.2. The need for bio renewable replacements for petrochemicals

In the last decade there has been a growth in the number of studies and development of policies and legislation aimed at reducing GHG emissions. For example, the European Renewables Directive (RED) in 2003 set an EU target to achieve 22% of electricity from renewable sources by 2020; the Stern Report (Stern, 2007) highlighted the economic case for mitigating climate change; the IPCC's 4th Report made predictions on the scale of climate change and its consequences (IPCC, 2007b); the implementation of emissions trading in the EU was introduced, aimed at curbing GHG emissions from major emitters (Egenhofer, 2007); the UK Climate Change Act (2008) set GHG reduction targets of 34% by 2020; and the Kyoto protocol that set out limits on GHG emissions for participating nations was ratified and entered into force (Feroz et al., 2009). This assortment of activities highlights that there is political consensus that the consumption of petrochemicals should be reduced. The motives for replacing petrochemicals range from mitigating climate change (IPCC, 2007b) and enhancing energy and resource security (Bauen, 2006, Prior et al., 2012) to concern over escalating fuel prices brought about by 'peak oil' (Verbruggen and Al Marchohi, 2010).

Global consumption of petrochemicals is increasing though their market share is diminishing as renewable resources grow faster, as depicted for the energy market in Figure 2.1.

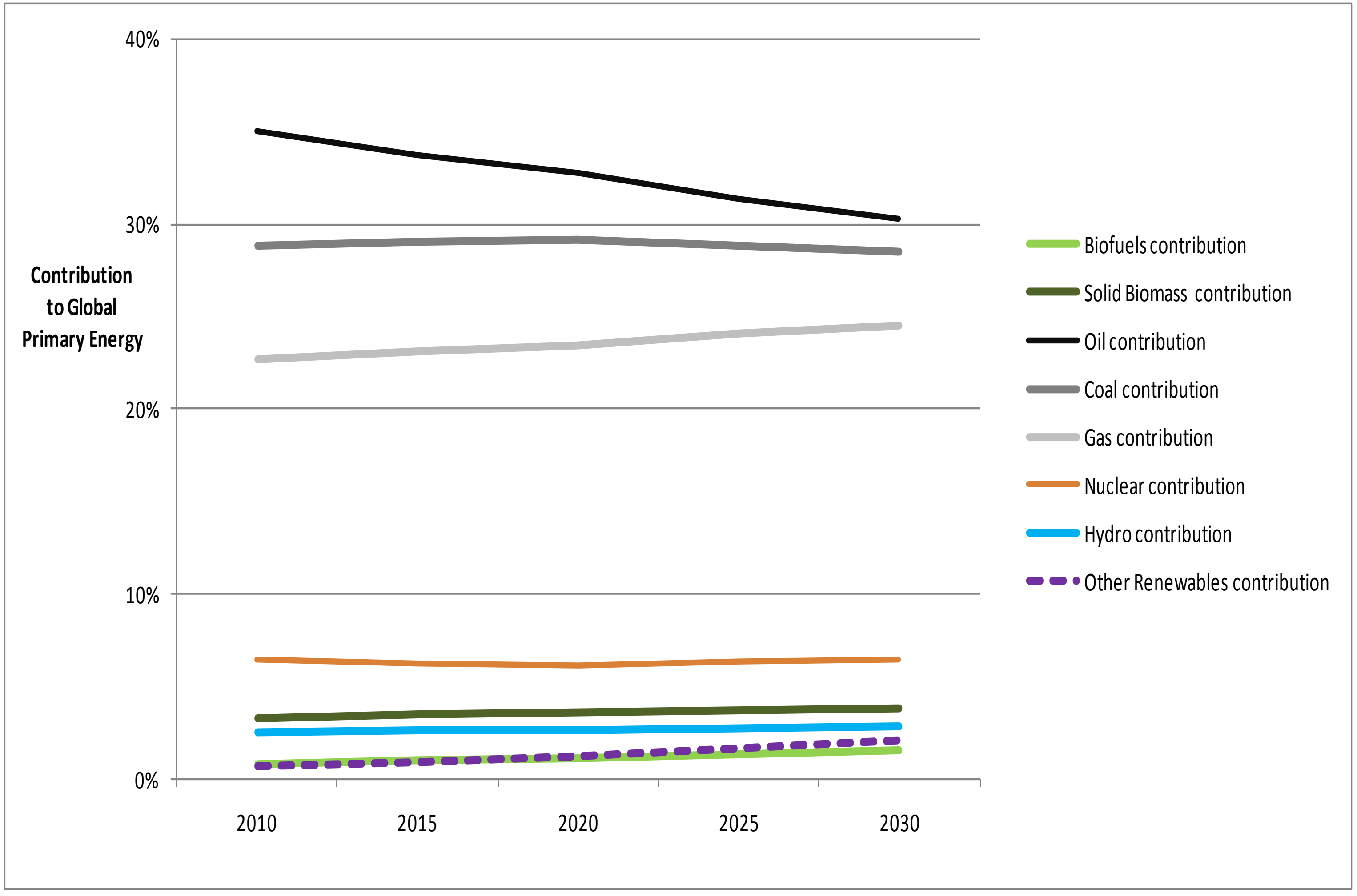


Figure 2.1 Contributions to global primary energy production (OPEC, 2010)

Here, bioenergy is shown to have superior growth rates to petrochemicals and yet it will still make up less than 6% of global primary energy. The transition away from petrochemicals is still therefore at an early stage and if, as predicted, growth in bio-based economies takes place, mass expansion in biomaterials (and their impacts) beyond current levels can be expected.

2.2.1. **Biomaterial products**

Many products can be made from bio-feedstock though bioenergy often steals the limelight despite other biomaterials being more numerous. Timber and textiles are familiar 'everyday' products and equally ubiquitous are waxes, oils and chemicals, though these are less conspicuous. Emerging materials such as bio-plastics and bio-composites are not yet commonplace in consumers' psyche but according to organisations like the National Non Food Crops Centre (NNFCC) they are fast gaining market shares (NNFCC, 2012).

There is some debate as to the most effective use of bio-feedstock. Most comparisons only consider energy-based options; electricity is often deemed the lowest carbon ahead of making liquid fuel and using solid biomass for heat (Ohlrogge et al., 2009, Campbell et al., 2009). Yet, there is a counter argument that the unique ability of bio feedstock to make liquid fuels should be valued most highly, since electricity can be produced using other renewable resources such as photovoltaics, wind, hydro and heat via geothermal and solar thermal technology (Lewis, 2010). No literature could be found that compares the usefulness of using feedstock for energy vs. non-energy uses or compares which may be considered to be lower carbon. This may be an important oversight given that biomaterials are more abundant and their products are highly valuable and useful.

2.2.2. **Global biomaterial yield**

Data on global biomaterial yields are not collated by a single official body since there are so many disparate interested parties and potential biomaterial uses that they do not sit easily under one collective banner. Predictions of feedstock requirements for future bio-based economies often focus only on bioenergy production, omitting other biomaterials entirely (van Vuuren et al., 2009, Offermann et al., 2011, Haberl et al., 2010). The United Nations (UN) estimate that solid biomass and liquid biofuels may contribute 25% of the world's energy needs in the next 20 years (UN Energy, 2006). Currently bioenergy constitutes around 10% of primary global energy production though this is difficult to measure since two thirds may be informally used for cooking and heating in developing countries (Heinimö and Junginger, 2009). More conservative estimates such as those in Figure 2.2 that do not capture informal energy state bioenergy represent less than 5% of global primary energy.

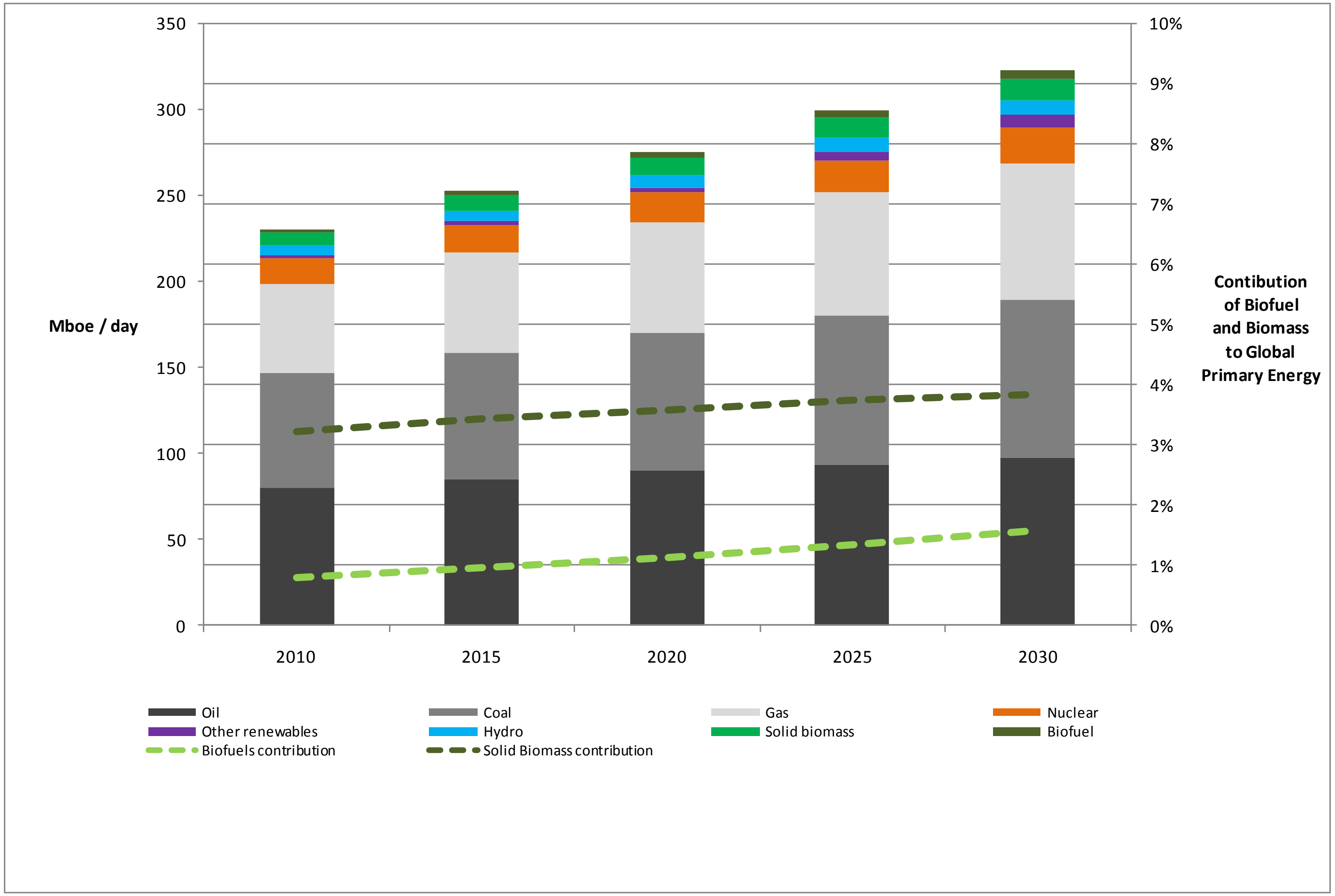


Figure 2.2 Global biofuel and primary energy up to 2030 (OPEC, 2010)

Figure 2.2. shows that all energy types will grow in terms of their contribution to global primary energy over the next 20 years, but liquid biofuels and solid biomass will grow faster than oil and coal. In the context of global energy this remains a small component; liquid biofuel is currently 1% of global primary energy and will increase to 1.5% by 2030 while solid biomass will grow from 3% to 4%. Putting this in context, this projected growth is equivalent to around double the UK's annual energy needs (BERR, 2008). The numbers seem modest but it is important to consider how much land is required to meet this level of demand.

2.2.2.1. Biomaterial land requirements

Biomaterial feedstock is often criticised for being land hungry and land use is a high profile component of sustainability and there are even organisations devoted to measuring the footprint of a product or nation such as the Global Footprint Network. Despite this no universally recognised estimate of land use needed for biomaterials industry has been agreed upon so an estimate is made here.

In terms of bioenergy requirements assuming an average primary productivity for 'global land' (combining cropland, pastures and unproductive land types) of 9.5MJ/m^2 of bioenergy per year (Haberl et al., 2010), around 0.000162 mboe could be produced per hectare and so 6,173 ha would be needed to make 1 mboe. Annual estimates of 3,832 and 6,449 ha of indirect land use change (iLUC) are also predicted by some to be incurred for every mboe of bioenergy produced (Bowyer, 2010). According to OPEC, global annual bioenergy production is around 3,358 mboe based on 9.2 mboe/day (OPEC, 2010). This would therefore mean 20,728,395 ha of land, equivalent to over 80% of the UK's landmass in addition to between 12,867,856 to 21,655,742 ha of iLUC, which combined would mean an area very roughly around the size of Germany would be needed to satisfy current global bioenergy production as shown in Figure 2.3. A report by the Food and Agricultural Organisation (FAO) claimed that the direct land needed to satisfy global bioenergy production in 2004 was 13,800,000 ha and will be 34,500,000 in 2030, which agrees with the order of magnitude of these extrapolations, though also identifies the extraordinary difficulty in predicting LUC and iLUC (FAO, 2010).

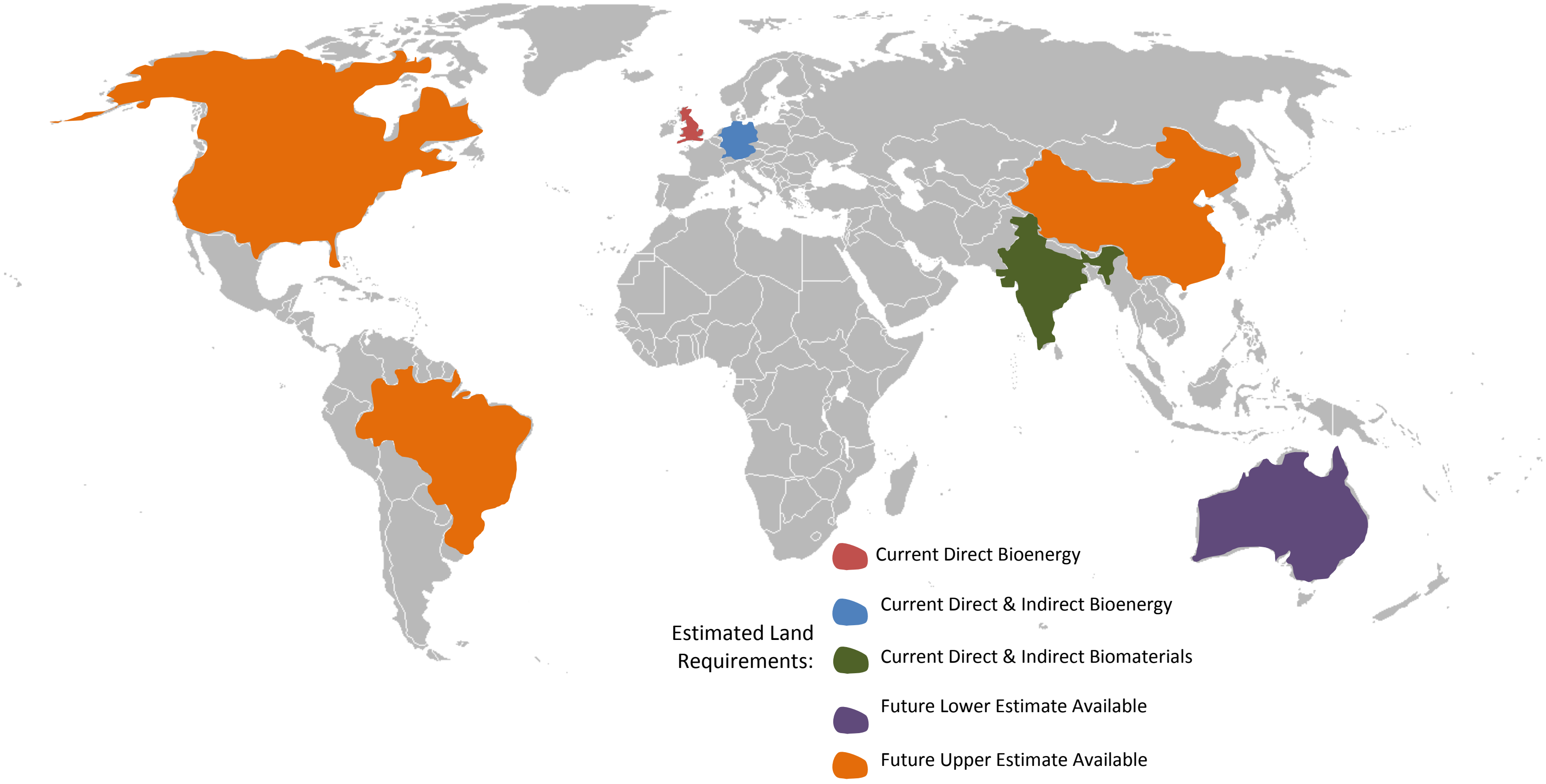


Figure 2.3 Approximations of 2010 global land requirements for bio-feedstock

In order to extrapolate from this to the land requirements for all biomaterials, we may consider the relative size of the biomaterial and biofuel only markets. The *BioEconomy Towards 2020* conference held in Brussels in 2010 claimed that the wider bio-economy including all associated and supporting or indirect industries is worth around 2 trillion Euros (Geoghegan-Quinn, 2010, Cunningham, 2010, Salas, 2010, Lieten, 2010, Bowles, 2010). This figure describes the whole bio economy including agriculture and forestry and any supporting industries, thus it is difficult to extract details on the worth of the direct biomaterials or bioenergy industry.

Other industry trend reports suggest the direct biofuels industry was worth US$ 83 billion[2] in 2010, However, this may only represent around 14% of the entire direct biomaterials market which is reported to be worth US$573 billion[3] in 2010. Scaling up the land requirements for bioenergy to that for all biomaterials according to this 14% ratio would result in land requirement for all biomaterials as 239,977,071 to 302,743,835 ha (which is just short of an area the size of India as shown in Figure 2.3).Nevertheless, this is disingenuous since the land requirements for $1 of biofuel does not necessarily equal that for $1 of biomaterials. What this does show however is that it is reasonable to assume that current land requirement estimates for bioenergy are well short of that actually required to grow feedstock for the entire biomaterials industry.

Haberl et al. (2010) review estimates of available land for bioenergy which they find range wildly, from a conservative 60,000,000 ha (an area around 80% the size of Australia shown in Figure 2.3) to an optimistic 3,700,000,000 ha (an area just less

[2] http://www.cleanedge.com/sites/default/files/CETrends2012_Final_Web.pdf, accessed 07/03/2012

[3] http://www.environmentalleader.com/2010/09/20/biomass-market-to-hit-693-billion-by-2015/?graph=full&id=1, accessed 07/03/2012

than the combined total of Canada, USA, Brazil and China as shown in Figure 2.3). This range is immense because of differences in each report's assumptions on land productivity, the use of residues and competing land uses. Haberl et al. predict the most likely total to be somewhere around the lower estimates.

Deciding whether there may be enough land for both energy and non-energy feedstock is therefore a difficult task and no convincing estimates exist to explicitly state this is the case, nor at what loss to existing land uses. Importantly, areas of existing cropland, mountains, protected areas, deserts, lakes and rivers and urban areas would also need taking out of the equation before concluding with confidence if there is in fact enough room to grow all the feedstock that the biomaterial markets may demand in the future. Of major concern in all the reviews is where exactly land conversion will take place, as well as the impacts LUC and iLUC may have in terms of e.g. habitat destruction, loss of carbon stocks, land grabbing, displacement of food production, threats to protected areas and the marginalisation of traditional land uses (Gallagher, 2008, Rulli et al., 2013).

According to Figure 2.4 total global feedstock is expected to grow over the next 20 years so that it may constitute between 2% and 4% of world agriculture. This could be as much as the land area currently used to grow rice in 2010 (114 million ha) or wheat (225 million ha).

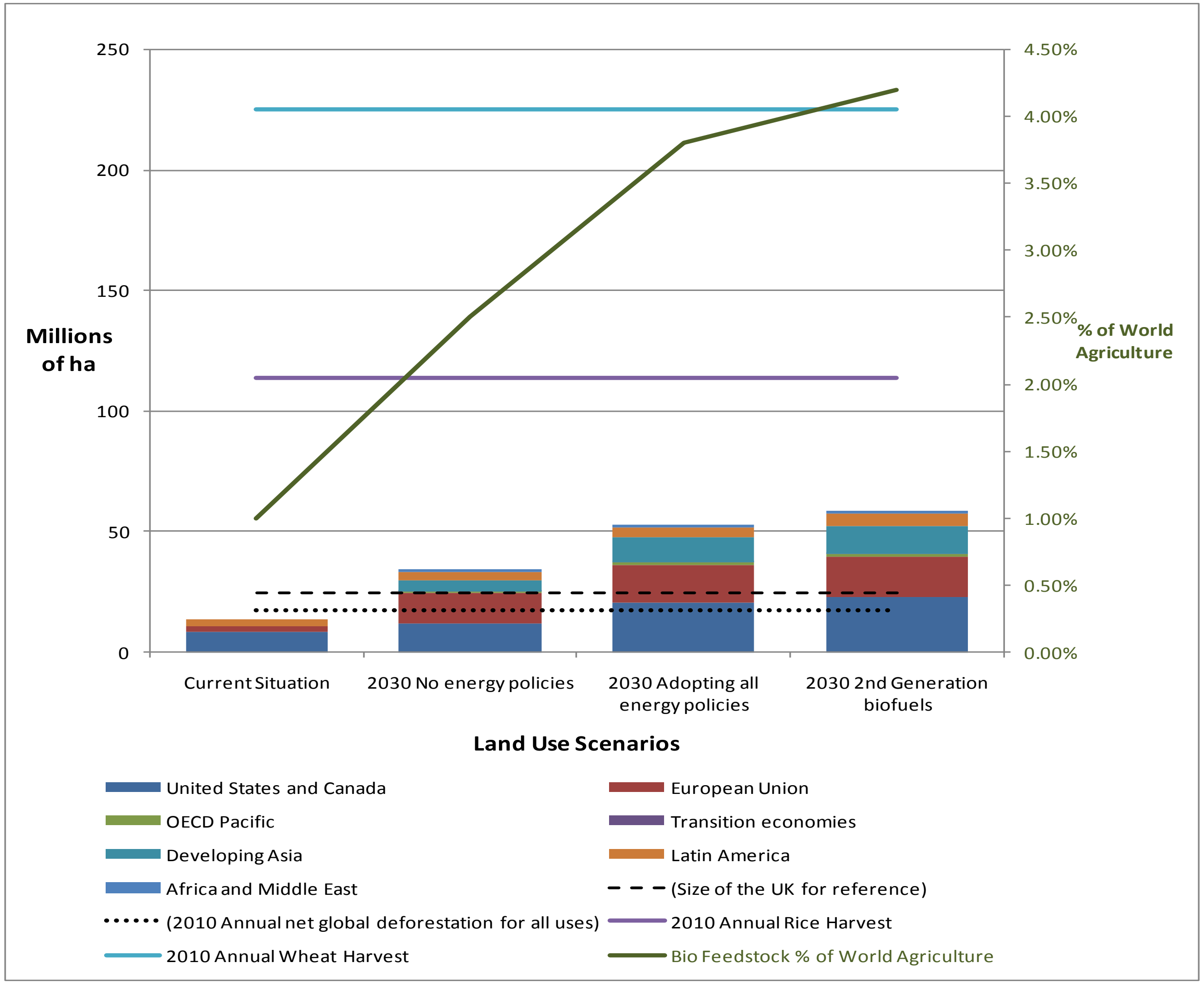


Figure 2.4 Predicted land use change scenarios (FAO, 2010)

The yield predictions made here are estimates and although they are transparent and reasonable they rely on data sets that use different modelling techniques and assumptions. In addition, there is a significant degree of supposition. For example, the average annual yield of bioenergy from a piece of land will vary enormously depending on its fertility and climate and there is also significant uncertainty regarding how to scale up the non-energy biomaterial land requirements given that an energy yield per hectare is irrelevant for non-energy products. These headline figures are nevertheless informative and provide the reader with a means of contextualising the scale of importance of biomaterials globally though they should not be viewed as authoritative.

More concrete concerns centre on the location of the perceived growth in feedstock. According to Figure 2.4 the bulk of the production is predicted to occur in Europe, the United States and Canada, where there are already intensive agricultural systems. This indicates that further increasing land use intensity may be limited and therefore additional production must come from LUC on land previously deemed unsuitable for agriculture for whatever reasons, or replacing food crops, both of which are controversial issues. The contribution of Latin America appears to remain relatively constant where there is already a well-established supply chain for biofuels, especially in Brazil and Argentina. The growth of production in developing Asia, may therefore be more cause for concern, since here there can be less comprehensive environmental protection and policing of regulations, and land rights are more tenuous (Barbier, 2004, Cuffaro, 1997). In addition, much of the land is currently covered by forest so the emissions from LUC here may be greater than in Europe and the US where less forest conversion may take place.

Figure 2.5 does not identify iLUC caused by any crop or other land use displacement yet additional GHGs will be emitted where an existing land use has been displaced somewhere elsewhere (Brown, 2009, Cornelissen and Dehue, 2010, Lapola et al., 2010). Predicting iLUC is very difficult since it is unlikely that the location where activities are displaced can be known (Cornelissen and Dehue, 2010). Activities can be displaced onto vulnerable or valuable land where the impacts may remain unreported making iLUC difficult to regulate and making it challenging to predict its GHG emissions. LUC is captured within some GHG regulatory schemes like the EU's RED and in the IPCC guidelines, though iLUC is less well understood. Areas of high carbon or biodiversity value such as forests or wetlands may still therefore be threatened by iLUC even if they are protected from LUC through sustainability schemes like the EU's RED (Tipper et al., 2009, RFA, 2010).

Similarly to the EU RED and IPCC default factors for the emissions relating to LUC, attempts have been proposed for harmonising the methodology for calculating iLUC factors. For example, the average estimate of 20t CO_2/ha/year as summarised in the International Energy Agency (IES) report (Brown, 2009) and the Round Table on Sustainable Biomaterials state that between 30 and 103 gCO_2eq will be caused per MJ of biofuel. This means iLUC could cause between around an additional third of the original quoted GHG emissions of the fuel or more than double them (Cornelissen and Dehue, 2010).

There is some concern that this 'factor' approach may therefore be less appropriate for iLUC since it will be so difficult to predict and police. Where the true iLUC is greater than the described factor this value may be used instead, thus under-reporting iLUC and increased biomaterial feedstock production. Alternative policies such as applying iLUC as LUC to the displaced product may be equally difficult to manage (Brown, 2009). The EU's approach offers an alternative to this, demanding a report to be written on iLUC before RED certification is awarded, though this is not binding and may be viewed as relatively weak. It is fair to say the jury is out on how to incorporate iLUC in biomaterials GHG assessments. Given that there is all this effort put into quantifying these ethereal impacts it may seem incongruous that so little effort by comparison is placed on more tangible influence over biomaterial's impacts; their disposal emissions.

2.3. Low carbon biomaterials and the influence of waste

The terms 'low carbon' and 'carbon footprint' refer to low GHG emissions not just low CO_2 emissions and is one of the motives often cited for seeking replacements for petrochemicals. The GHG emissions of a biomaterial supply chain are distributed unevenly across geographical areas so their impacts may not necessarily affect those who purchased the product (Peters and Hertwich, 2006). This provides a barrier to their measurement especially if vulnerable people are affected (Klein, 2000).

Simplistically, biomaterials may be seen as low carbon or even carbon neutral alternatives to petrochemicals since they absorb CO_2 (one of the most abundant GHG emissions) prior to emitting it when they are burned or ultimately decompose. It has been shown however that biomaterials and bioenergy can actually emit more GHG than petrochemicals. Many studies show that biofuels have greater GHG emissions than petrochemicals if there is heavy use of fertilisers, significant N_2O soil emissions or if yields are affected by local conditions (Searchinger et al., 2008, Hillier and Murphy, 2010, Cherubini, 2010). In addition, GHG balances are adversely affected when emissions from LUC and iLUC are considered that e.g.

result in the conversion of high carbon stocks land like peat and forests (Gallagher, 2008, Searchinger et al., 2008). Changing land use has been shown to represent the third biggest GHG emission in a biofuel's life cycle after the feedstock's actual yields and the emissions linked to fertilizer use (Bernesson et al., 2006). The exact emissions from LUC depend on the vegetation that is lost as well as the different feedstock management regimes (Cherubini et al., 2009, Webb et al., 2010, Börjesson and Tufvesson, 2010). To overcome this uncertainty, default emissions based on the IPCC reports give values for emissions from soils for NO_2 and CO_2 among others. However, it is likely that the IPCC averages will in most instances be either too high or too low (Hillier and Murphy, 2010).

These problems are taken seriously, and the UK government commissioned the Gallagher Review to investigate the influences on low carbon biofuels. Consequently, sustainability criteria were enshrined in the EU biofuels' legislation (European Commission, 2009a, Gallagher, 2008). Similar scrutiny on a multilateral scale is not yet paid to all biomaterials despite plans to do so for solid biomass fuels. Currently, only individual assessments exist for other specific biomaterials (van Dam and Junginger, 2011) and there is no consensus on how issues that affect the carbon footprint of products should be addressed for non-fuel products (Börjesson and Tufvesson, 2010, Elsayed et al., 2003, Acquaye et al., 2011). Having said this, there are overtures to suggest that the industry is realising that biomaterials may equally have as many negative consequences as biofuels. For example, the Roundtable for Sustainable Biofuels (RSB) had, at the time of writing, recently changed its name to the Roundtable for Sustainable Biomaterials.

There remains however a lack of standardised assessment for biomaterials on a wider scale despite the RSB's advancements. This means there is little consensus within GHG emissions assessments and in particular, little consensus on how we should deal with a feature that is unique to biomaterials which could not be considered for biofuels: disposal (Ekvall et al., 2007, Nicholson et al., 2009, Pawelzik et al., 2013). Disposal GHG emissions are not considered for bioenergy assessments since the feedstock is burned and because the carbon released was fixed by the feedstock initially while it was growing. The same is true for biomaterials to an extent in that the emissions released when a biomaterial decomposes were originally absorbed by the feedstock. However, before this final end of life fate, there are several options available to the biomaterial that can have the effect of reducing net life cycle GHG emissions by displacing additional consumption or extending the useful life of a product. These are reuse, recycling and energy recovery referred to as the waste hierarchy (European Commission, 2010b).

Studies show waste management is responsible for around 3 to 5% of anthropogenic global GHG emissions (UNEP, 2010) and the waste hierarchy is addressed extensively in the waste literature (Kong et al., 2012, Ross and Evans, 2003, UNEP, 2010, Zhao et al., 2009) though studies often address municipal solid waste (MSW). This means assessments seldom link back GHG savings from the waste hierarchy into the carbon footprint of the original product. One reason for this may be a lack of clarity on who 'owns' emissions and emissions savings. For example, complications arise with recycled products where it is not clear if emissions should be allocated to the new product or should be attributed to the original product. These decisions can influence GHG emissions significantly (Nicholson et al., 2009). Energy production from waste products also complicates the calculations since the energy excluded from the assessment could be attributed to the original product as a net emissions saving from avoiding fossil energy or shared between the co-products (Gnansounou et al., 2009, Thamsiriroj and Murphy, 2011).

Beyond theoretical barriers to attributing waste emissions to products there is the practical difficulty that the producers and waste managers of a product are usually different companies. This makes efforts to join up the supply chain emissions difficult. There are further complications in that there is no guarantee that a consumer will dispose of a product in the way it was intended so it may be disingenuous to attribute GHG savings to a product on the presumption of a particular disposal fate. In addition the presumption that biomaterials may be 'green' products could perhaps influence individuals and organisations to be less critical of them and so ignoring their potential waste disposal problems, akin to the idea that putting something on a pedestal makes one less likely to notice its faults.".

Despite these barriers, it is generally agreed there will be savings in GHG emissions when landfill is avoided. However, the exact savings will depend on the waste composition and available technologies (European Commission, 2001, Zhao et al., 2009, Kong et al., 2012)

2.4. Wider impacts of biomaterials

GHG emissions are just one concern for the acceptability of biomaterials as replacements to petrochemicals.Other environmental, social and economic considerations need also to be taken into account. Some approaches to appraise the appropriateness of biomaterials therefore consider multiple issues. Figure 2.5 shows the key issues addressed by current bioenergy sustainability certification schemes from around the world.

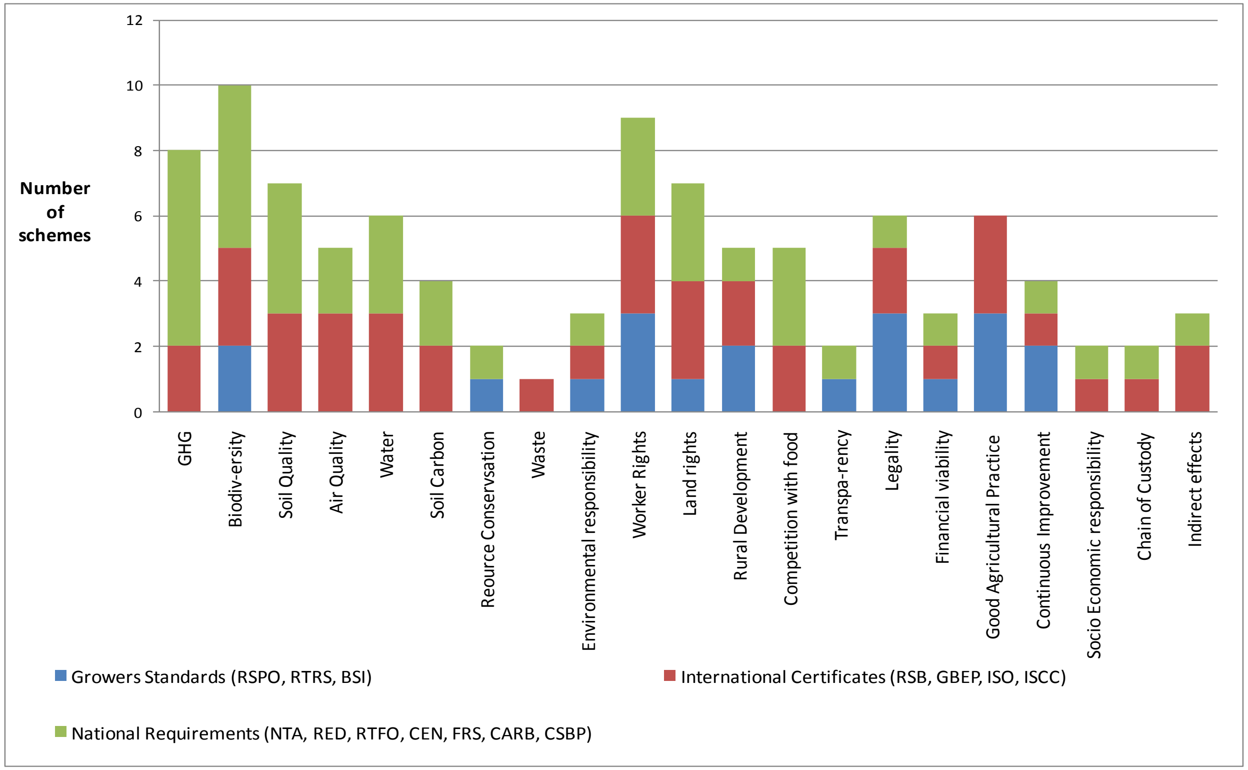


Figure 2.5 Bioenergy certification criteria (adapted from (Scarlat and Dallemand, 2011))[4]

[4] Roundtable for sustainable Palm Oil (RSPO), Round Table on Responsible Soy Association (RTRS), British Standards Institution (BSI), Roundtable for Sustainable Biomaterials (RSB), Global Bioenergy Partnership (GBEP), International Standards Organisation (ISO), International Sustainability and Carbon Certification (ISCC), Dutch technical agreement for sustainably produced biomass (NTA), Renewable transport fuel obligation (RTFO), European Committee for Standardisation (CEN), Financial Reporting Standards (FRS) California Air Resources Board (CARB), Council on Sustainable Biomass Production (CSBP)

The wider issues for bioenergy are the same issues faced by all biomaterials. *Growers' standards* are more biased towards socioeconomic issues, as their members' welfare is of utmost importance. The *national requirem*ents favour environmental issues such as GHG and emissions to soil, air and water which again may not be too surprising since they are influenced by the politics of climate change and often favour tangible environmental impacts that can be measured and reported. The *international certificates* show more equality in their treatment of all three pillars of sustainability (economy, society and environment) which may reflect their broader range of stakeholders. This demonstrates that the bias of those stakeholders undertaking an assessment defines its priorities.

Agriculture has environmental impacts for example on waterways, soil and air due to fertilizers, pesticides, irrigation, drainage or other inputs and activities. There is a consensus that environmental impacts like eutrophication risk and acidification of water ways are likely to be higher for first generation bio-feedstock supply chains than for petrochemicals (von Blottnitz and Curran, 2007, Börjesson and Tufvesson, 2010). The social impacts of growing bio-feedstock are less well documented in the literature, though there are studies on the general issues identified by Figure 2.5 of workers' rights (RFA, 2010), transparency (Gnansounou et al., 2009) and legality (FAO, 2010) as well as the more biomaterial specific issues concerning land rights (Barbier, 2004, Carus, 2010), rural development (Rist et al., 2009, Thamsiriroj and Murphy, 2011), good agricultural practice (Offermann et al., 2011), chain of custody (Black et al., 2011) and competition with food (Fischler, 2010, Tilman et al., 2009, UN Energy, 2007).

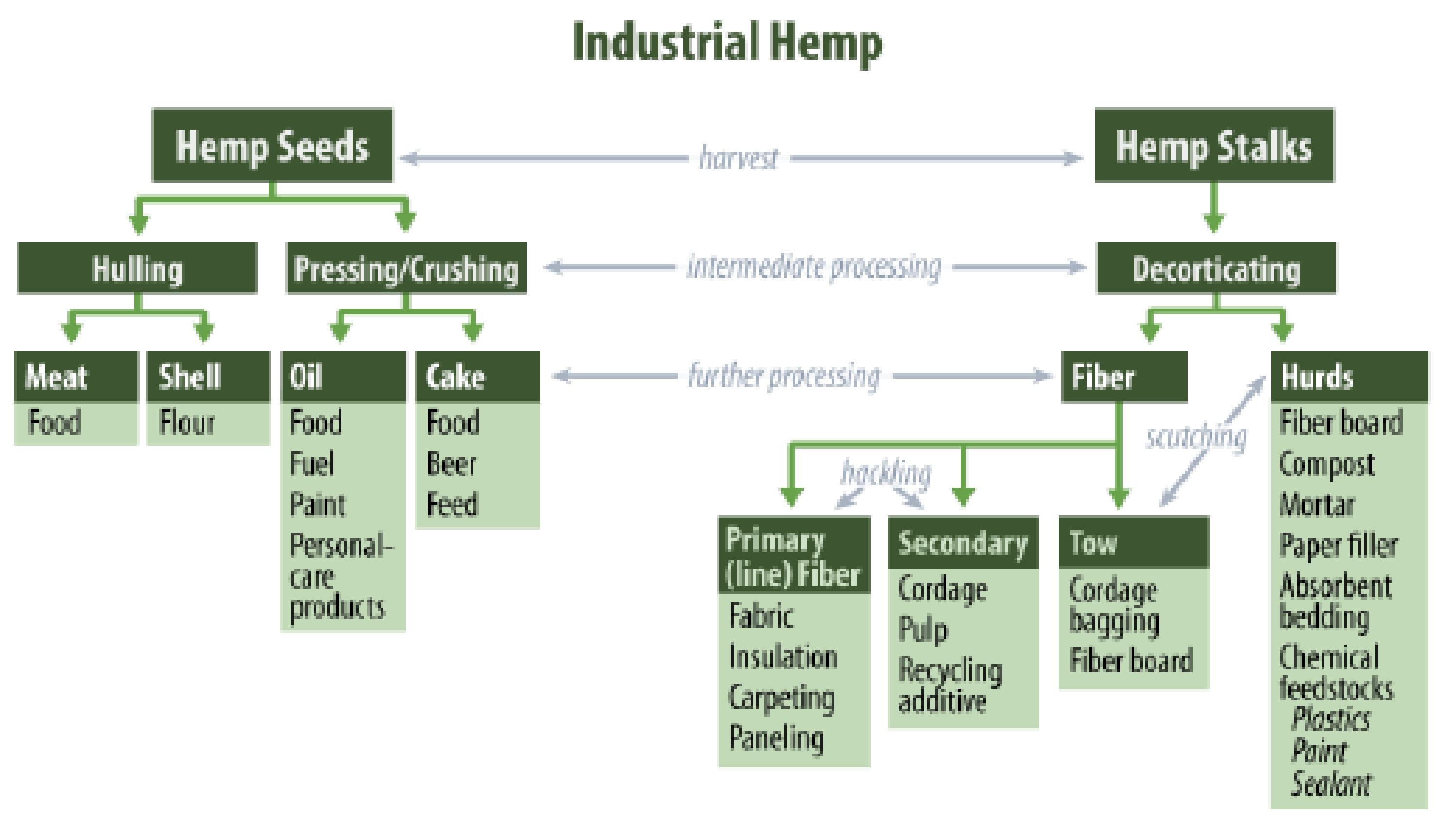
Tilman (2009) highlights the considerations over competition of 'fuel vs. food' in two ways: initially via the social cost of hunger but also through the economic problems caused by influences on food prices that using crops for fuel or other products can have. Crop price fluctuations caused by weather and oil prices fluctuations are already a major concern for farmers worldwide and the introduction of new biomaterial markets could cause more instability (Fischler, 2010, Hill et al., 2006, Cornelissen and Dehue, 2010).

Biomaterials markets are likely to become more important when international trade of feedstock increases and is sourced from locations with potentially less stringent environmental and social protections (Haberl et al., 2010). As stated, these wider problems relate mainly to first generation plants, and so fast-tracking the expansion of second generation feedstock which do not have such negative wider implications could reduce the overall impacts of future bio-based economies (Berndes et al., 2010).

2.5. Research focus

Although these wider socio-economic and other environmental issues are relevant to the bio-based economy debate, GHG emissions remains the most studied impact. A review by Von Blottnitz showed that only 7 out of 47 LCA on biofuels considered issues other than GHGs (2007). One reason for this is the difficulty in measuring the impacts but also perhaps the lack of emphasis placed on these indicators by legislation like RED, which require GHG emission calculations but only encourage comments to be made on other issues. As such, researchers may lack the incentive or investment to measure them (Weale et al., 2011). As explained in the previous chapter this research focuses on GHG emissions which are the most pertinent factor to low carbon disposal options for biomaterials.

Throughout the thesis, hemp (*Cannabis sativa*) is used as an exemplar biomaterial feedstock since it produces many biomaterial products. Hemp is touted as being a potentially low carbon crop that requires no pesticides, since it out-competes weeds, uses little fertilizer or irrigation in temperate areas and potentially causes little LUC and iLUC since it can be grown on marginal land (though it will require more inputs if not grown on cropland) (Johnson, 2010). Hemp was widely used at the beginning of the 20th Century for rope, textiles and paper. However it was virtually abandoned as a crop in most of the UK and USA due to competition from cotton and synthetics but also because of bans associated to its narcotic relatives (Cherrett et al., 2005). It is currently becoming popular again and can be used to produce an array of products as shown in Figure 2.6.



Source: CRS, adapted from D. G. Kraenzel et al., "Industrial Hemp as an Alternative Crop in North Dakota," AER-402, North Dakota State University, July 23, 1998, http://purl.umn.edu/23264.

Figure 2.6 Uses for hemp (Johnson, 2010)

For these reasons Hemp is used as a common thread to tie the different results chapters together. This aids the flow of the research and grounds the study in the context of a realistic emerging biomaterial.

2.6. Chapter summary

The literature sets out strong and quite polarised positions on either side of the biomaterials debate. When the specific claims of each side are analysed, they appear to have significant backing. Both petrochemicals and biomaterials can cause significant harms to the environment, and society but also have the potential to provide benefits. As yet, neither can be said to be preferred in general terms though many individual examples are well presented.

3. Research methodologies

There is a wide array of sustainability assessment tools available in the literature. These include Ecological Footprinting, Risk Assessment, Strategic Environmental Assessment (SEA), Environmental Impact Assessment (EIA), Cost Benefit Analysis (CBA), Material and Substance Flow Analysis, Energy Analysis and Life Cycle Assessment (LCA) (Robèrt et al., 2002, Jeswani et al., 2010, Ahlroth et al., 2010) and Triple Bottom Line assessments (Foran, 2005, Wiedmann et al., 2009). Each assessment has different methodologies and data requirements, and use assessment units which are best suited to their aims. For example, footprinting uses land area, LCAs usually measure GHG emissions and risk assessments produce the probabilities of certain scenarios occurring (Robèrt et al., 2002). This means they each make slightly different sustainability claims and are therefore difficult to compare (Hacking and Guthrie, 2008). These are summarised in Table 3.1.

Table 3.1 Summary of Sustainability Assessments

Assessment	Unit	Strengths	Weaknesses
Ecological Footprinting	Hectares / number of Earths	Visually powerful, simple concept	Specific impacts not identified
Risk Assessment	% risk	Applicable to wide array of situations	Limited description of sustainability impact
SEA	Broad sustainability priorities	Guides decision makers on a wide range of issues and priorities	No quantification of impacts or specific calculations
EIA	Environmental impacts of projects	Identifies specific problems and advises how to minimise impacts	Specific to geographical location and not applicable to supply chains
CBA	Monetary unit	Ability to compare different issues using a common unit	Converting non-monetary impacts causes problems
Material / Substance Flow Analysis	Quantities of inputs	Maps entire supply chains showing where largest inputs exist	Does not relate quantities of inputs to impacts
LCA	GHG or other Environmental Impacts	Identifies hotspots, widely used, available data, quantification of specific impacts	Can't address all issues and methodologies are varied

TBL	Company's social, economic, environmental impact	Only tool to advise on impacts on economy, society or environment	Specific to company activities not product supply chains or projects

The research presented here is concerned with GHG emissions for which many studies conclude that LCA is the most appropriate and widely accepted methodology (von Blottnitz and Curran, 2007, Kaltschmitt et al., 1997, Cherubini and Jungmeier, 2010, Acquaye et al., 2011, Treloar et al., 2001). The majority of 'bio' LCAs have been undertaken on the production of biofuels as these have the most developed markets and legislation, specifically wheat and maize in Europe and USA, palm oil from Asia and USA and sugar cane from Brazil (de Vries et al., 2010). Given this existing precedent, LCA is used as a methodological tool in this research.

As biomaterials and their disposal are less studied it was deemed to undertake qualitative research in order to establish the LCA priorities. Interviews (Brenner et al., 1985, Wilson et al., 1998) and focus groups (Billson, 2006, Tang and Davis, 1995) are robust qualitative techniques that can elicit priorities and extract expert insights and these have therefore also been selected as tools for this research. This mixed methods approach to research should help provide stakeholder validation of the findings (Tashakkori and Teddlie, 1998). The specific methods and techniques that are used are described in detail in the results chapters, where it was important to detail the methodology for the purposes of publication. In addition however, the methods are critiqued in general terms here.

3.1. Process life cycle assessment

One advantage of LCA is that unlike some tools it has an international standard, ISO14040, which sets out guidelines for consistency of method and interpretation.

Because LCA is commonly used its pitfalls are widely known, and awareness of its limitations enables more useful interpretation (Finnveden et al., 2009). Useful outcomes of LCA include discovering hotspots within supply chains where the largest impacts occur and tracking improvement over time (Acquaye et al., 2011). LCA is a flexible and transparent tool and using consequential LCA allows the assessment of various scenarios in order to pinpoint changes in GHG reductions corresponding to changes in disposal fates.

The literature shows that bio renewable LCAs are generally Process LCA that follow the International Standard Organisation's (ISO) 14040 four phases: 1) goal and scope definition; 2) inventory analysis; 3) impact assessment and; 4) interpretation (ISO, 2006). The scope and goal definition phase sets out the background and intention of the study and defines the boundaries and detail of data collection. For example, an assessment will usually consider all inputs to the point at which further data collection has a marginal effect on the overall results (often a 5% difference). Since this point is not defined in the ISO it will usually depend on the resources and time available to the assessor to choose when to stop collecting data and therefore can be an area of inconsistency between assessments (Crawford, 2008).

The Life Cycle Inventory (LCI) is the second phase which involves collection of the background data and which can be time consuming (Minx et al., 2009). This may include collating energy bills from a factory or recording the amount of fertilizer used on a field.

The Life Cycle Impact Assessment (LCIA) is the third phase in which the boundaries and data in the LCI are applied to the specific product or system under review, and the total energy consumption of the factory may be scaled down to the one product being assessed or the "functional unit". Assumptions regarding how to scale the impacts down may affect the results in this stage and so sensitivity analysis can be helpful in assessing each option. This can inform decisions such as whether energy use in a factory should be attributed evenly across all the items that were produced or by the number of hours it took to make each specific product.

Life cycle interpretation is the final phase during which a summary of the results in the LCIA is given in accordance with the goal and scope definition. In this stage, weighting and grouping can provide a further degree of inconsistency between LCAs (Ahlroth et al., 2010). It is the responsibility of the researcher to provide guidance on the use of the LCA relevant to its specific goal but also to report all the data, assumptions and weighting that were used. There is not currently a detailed explanation in the ISO of how this should be formatted or presented to the reader (Finnveden et al., 2009) thus 'unknown unknowns' may remain unchallenged.

LCA are used in this research following these standards to quantify the GHG emissions of biomaterials' disposal and attempts are made to identify any limitations and drawbacks of the methodologies used.

3.1.1. **Uncertainty in process LCA**

Data accuracy is an area of concern for Process LCAs. For example, some LCA may use defaults and industry averages or some may take primary data (Cherubini et al., 2009, Hillier and Murphy, 2010, Wiedmann, 2009). The use of defaults such as the IPCC values for N_2O emissions from agriculture and the use of fertilizers (Acquaye et al., 2011) is controversial; N_2O emissions are related to soil type, orientation, moisture and management as well as weather so using one single value is an unsophisticated approach. One report suggests the use of N_2O defaults rather than actual data can change an LCA by 300% (Cherubini, 2010).

Attempts to deliver greater accuracy often attract complexity. An alternative to the IPCC defaults is the Nomenclature of Territorial Units (NUTS) information developed by the EU to provide regional N_2O soil emissions (Webb et al., 2010). However the variation within NUTS regions can be great. Some argue that N_2O emissions and agricultural LCA in general need to be calculated seasonally on a field-by-field basis for the true value to be known, as the samples from different sides of the same field or the same sample in different years can give greatly different values (Scharlemann and Laurance, 2008). Clearly this is not feasible and therefore default values such as NUTS and the IPPC values are used as a necessary compromise (Hillier and Murphy, 2010).

Where there can be primary data collection of input values, defaults need not be used. In most cases secondary data from LCA databases will then be used to convert inputs into environmental impacts. This means that even when defaults for input data are not used, data quality varies depending on which LCA database is used. For example, data may have varying number of years over which data collections stretch or the number of sample points used or the degree of weighting applied may be different in each database (Eldh and Johansson, 2006). This compromises the ability to compare between assessments without a detailed analysis of all the assumptions (Elghali et al., 2007). Such scrutiny is not always possible however, as reports are often shortened to fit the format of journals and raw data may not always accompany the report (Gnansounou, 2008). In these instances, failure to investigate the assumptions may lead to misleading conclusions (Ekvall et al., 2007). In order to combat these limitations, sensitivity analysis is used to provide some degree of certainty on the robustness of the claims. This may include using different input data sets or altering the methodological assumptions such as the way weighting and allocation takes place or even extending the system boundary (Pesonen et al., 2000).

Although accepting data inaccuracy is often necessary and common practice by those undertaking assessments, very little literature exists on how well these areas of error are understood by a) those outside the assessment process and b) those who may be interested in using the data. As such, this would be a fruitful area for future research and seems to have significant implications for policy makers. For example governments may use the results from an LCA from one nation to justify their policy yet their own nation's situation, soil type, climate, infrastructure, technology, electricity make up etc. may invalidate the former nations' findings in the context to which it is being applied.

In addition to data uncertainties, methodical nuances can be important in LCA. When a bio renewable product is made there are usually co-products or uses for the waste materials in the supply chain. Gnansounou et al. (2009) found that even when comparing similar studies, CO_2 emissions for some co-products could differ by 200% if different allocation methods were used, even though total emissions across all co-products remained constant. Thus, it is possible that two studies can state significantly different results, simply because of the choice of methodologies or data sources. This provides the opportunity for selective reporting to favour a preference, i.e. by using the methodology or data that gives the highest or lowest CO_2 emissions.

This could contribute towards public distrust of LCA and science in general and so must be treated very carefully (West et al., 2010).

When a co-product, such as, waste heat or 'dried distiller's grain with solubles' (DDGS) avoids the use of virgin resources it is common that the equivalent GHG emissions of the avoided product can be deducted from the LCA (Wang et al., 2011). This is often called the displacement, the system expansion or the substitution method of allocation, and can with regards to biofuels change the LCA results by over 50% compared to other allocation methods depending on the importance of co-products in the overall supply chain (Malça and Freire, 2006). Other types of allocation method are called economic, mass or energetic allocation. In these instances the total emissions from the process can be split across the two products (biofuel and animal feed) based on the economic value, mass or energetic potential of the products (Wang et al., 2011).

A slightly different form of allocation can occur where there are two distinct processes for each co-product that can be easily separated; this is called the process purpose allocation method. An example of when this can be used is that the dryer in the wheat ethanol plant is used exclusively for drying out the DDGS, therefore the emissions associated with it can be subtracted from the wheat and put solely on the DDGS co-product (Wang et al., 2011). This may happen anyway as a result of a detailed data collection though some studies refer to it as a type of allocation method.

Thus, in addition to paying attention to where data has come from it is equally important to understand how these data are treated before attempting to compare results between different LCA. Using default data and setting standardised assumptions, so called harmonisation, may reduce the accuracy of the LCA result but it is a useful approach where multiple assessments need comparing and benchmarking, as is the case in, for example, the RED biofuels legislation (European Commission, 2009a).

3.2. Input output and hybrid LCA

Originally there were two main types of LCA 1) Process LCA and 2) Input Output (IO) LCA. These can be combined to create a Hybrid LCA. The boundary for data collection in process LCA is defined by the assessor, usually as the point at which significant differences are no longer made by adding additional individual inputs. This often means the direct energy used by manufacturing equipment would be included but the embodied energy used in making the equipment may be too small and so omitted (Finnveden et al., 2009). The arbitrary selection of boundary definition adds uncertainty and incompleteness into the results. In addition there are certain inputs that may be very obscure and difficult to capture, for example, the inputs that went into an advertising campaign for a product may be too difficult to measure and allocate in an assessment. Some reports suggest the cumulative effect of all the missing inputs can result in a 50% truncation of emissions associated with a product by process LCAs (Crawford, 2008).

To tackle this truncation, IO LCA can be used. The advantage of IO is that it is reckoned to be more 'complete' than Process LCA (Wiedmann et al., 2011) as well as being potentially less time consuming and costly. IO takes an economic accounting approach instead of taking direct measurements of transport distances, energy consumption and quantities of inputs. National economic statistics tables published by governments that record financial transactions between sectors in economies show how much each sector purchased from another (Suh and Huppes, 2005). Since each sector also publishes their total GHG emissions, the economic data can be converted to GHG emissions, and so by knowing how much was spent on each sector to make a product, the LCA can also be known (Acquaye, 2010, Wiedmann et al., 2011).

Using financial transactions means that no resources are missed and the data can be considered a complete picture of the aggregate resource flows so that no truncation can occur. Lenzen (2002) developed an inverse matrix to apportion the relative resource consumption of each sector to another as a ratio as opposed to an actual value. In the UK the Office of National Statistics (ONS) collects and presents economic data in an Input Output matrix which aggregates all the industries into a total of 138 sectors. The GHG emissions of each sector are also known and therefore the GHG per pound sterling spent can be apportioned to the relevant sectors. IO therefore does not require the assessor to artificially draw a line (system boundary) of where to stop counting emissions, so the smaller inputs, ignored by process LCA, can still be accounted for in IO assessments, providing a more complete indication of emissions in IO LCAs (Crawford, 2008).

3.2.1. **Uncertainty in IO and hybrid LCA**

IO is a broad brush approach and assumes for example that all the companies within the construction sector are average, thus the problem of disproportionality exists whereby an otherwise efficient sector may be brought down by some individually poorly performing companies (Freudenburg, 2006). This effect is magnified if the sector boundaries include quite disparate subsectors. For example, road building has very different inputs and emissions to house building. In addition, fluctuations in the price of inputs can seriously change the perceived carbon footprint of a product if the financial transaction statistics and emissions conversion tables are not up to date, though this may be a relatively small limitation as statistics are collated more quickly.

One problem with IO is found in addressing international supply chains. Some countries will have no available sector-based financial or emissions data and so must be grouped in with 'rest of the world' type classifications. Clearly, this is a concern, though multiregional input output tables are becoming more sophisticated and wide reaching and so this may be less concerning in the future (Minx et al., 2009). In summary IO LCA has problems regarding its data resolution and refining products into ever smaller categories may resolve the problem of aggregated data. Thus, IO can boast to provide generalised emissions advice for different types of products, not specific supply chains like process LCA, however its data is provided with greater completeness and speed. In order to achieve both the specific accuracy of process LCA and the general completeness of IO LCA the two may be joined to create a hybrid LCA and a more robust assessment. However, in doing this, errors of both types of assessments may also be combined and the possibility of double counting emissions is introduced (Acquaye, 2010).

Both Hybrid and process LCA are undertaken in this thesis to provide an insight into the GHG emissions of biomaterials' disposal. The use of quantitative assessments is useful in measuring known impacts and LCA is a relevant tool to discover the GHG emissions of biomaterials and unveil the influence of disposal. It is common to employ additional research methods in different phases of a single piece of research, taking a so-called mixed methods approach (Tashakkori and Teddlie, 1998). Qualitative research is therefore also undertaken in this study to complement the quantitative LCA work.

3.3. Interviews and focus groups

Qualitative research can be useful in applying context to quantitative research and in identifying salient issues to complement and direct quantitative studies (Trainor and Graue, 2013). Specifically the research presented in this thesis may be thought of as having a ‘cyclical mixed method design’ since it uses quantitative research (hybrid LCA) to identify the problem of disposal emissions, qualitative research (interviews and focus groups) to understand how this problem manifests in the biomaterials industry and to identify barriers to change and finally returns to quantitative research (process LCA) to propose a solution to these barriers (Trainor and Graue, 2013). Within the qualitative research approach mixed methods are again employed sequentially. Interviews are first undertaken, the results from which inform a focus group in the second phase.

There are various qualitative research methodologies such as case studies, questionnaires and observations. Interviews and focus groups are used in this research. Specifically, semi-structured interviews were selected as they allow open-ended questions to be asked, enabling respondents to describe their attitudes surrounding particular topics, while focus groups allow more thorough and dynamic investigation of key themes (Trainor and Graue, 2013). Semi-structured interviews and focus groups were selected in preference to case study and observation techniques as there was no opportunity to integrate with a biomaterials partner organisation.

Using interviews with a range of stakeholders from the biomaterial industry was essential because waste disposal touches upon many parts of a product's life cycle; the producers may ensure raw materials are pure, the manufacturers can design for easy disassembly of products and the retailers have an interface with consumers. As such a variety of opinions to be collect through the interviews of the whole industry was necessary. Questionnaires were not employed because a large sample size was not paramount and because they are inferior to face to face interviews in eliciting detailed descriptions and lucid thoughts (Thakur, 2005).

For the focus groups it was important to assess the outputs from the interviews with experts who had experience and knowledge of the biomaterials industry, sustainability and the waste sector. It was also useful that they should not be directly employed or funded by the biomaterials industry lest they have any vested interests in a particular waste option. In addition it was important that they were not part of the sample that were interviewed so that an entirely new group could reassess the issues and respond to the interview results without prior bias.

Like quantitative data, qualitative data too can have many uncertainties, especially in the context of grounded theory where instead of testing preconceived hypotheses directly the ideas and approaches show themselves as a result of investigations (Trainor and Graue, 2013). For example theoretical saturation points are reached when pursuing more interviews fails to provide more insights into a developing theory and so the research is aborted at this point (Glaser and Strauss, 1967). Clearly this relies on competent assessment of any developing trends and so raises the possibility that crucial information may be missed.

In addition to sample size uncertainties there is scope for inconsistencies in the interpretation of qualitative assessments. Coding and categorizing of data into relevant and emerging themes is used to explain what has been found, yet this relies on a competent selection and dissection of the data, assuming misrepresentations are infrequent and salient patterns do not go unseen (Trainor and Graue, 2013).

Combining the results from this range of research techniques enables the study to make more insightful conclusions than using any one method alone. The use of mixed methods allows the research to target the priorities in biomaterials research regarding GHG emissions and disposal's impact and its iterative nature allows the research to react to the initial insights found.

3.4. Chapter summary

In general, literature on the GHG emissions of biomaterials is slow to catch up with that on biofuels despite their arguably superior magnitude. Specifically, there is currently a dearth of information on the unique influence of disposal on the GHG emissions of a biomaterial's life cycle. The combined use of qualitative and quantitative research methods has potential to contribute insights to address this knowledge gap.

4. The influence of disposal on biomaterials

4.1. Introduction

Chapter 2 introduced the idea that decisions made during disposal stages can change a product's carbon footprint. However, despite the importance of end of life scenarios, they are by no means commonplace in the sustainability assessments of biomaterials as shown in Chapter 3. The research presented in this thesis attempts to quantify the significance of this omission by measuring the impact that end of life scenarios have for an exemplar biomaterial product: a natural (hemp) fibre mattress compared to its petrochemical alternative.

A paper was published in 2012 in the Journal for Cleaner Production from this research entitled 'How Do End of Life Scenarios Influence the Environmental Impact of Product Supply Chains? Comparing Biomaterial and Petrochemical Products' (Glew et. al., 2012). This can be seen in Appendix I where the full methodology, justifications, data, results and conclusions can be found. In the paper GHG emissions from the production of the two mattresses and their end of life scenarios are compared using a consequential integrated hybrid LCA.

Data were taken from well-known process LCA databases and combined with industry average IO emissions data provided by the Office of National Statistics, and the effects of disposal options on GHG were investigated, the full methodology can be seen in Appendix I. It is shown that natural fibre (biomaterial) pocket spring mattresses emit marginally less greenhouse gasses (GHG) than foam (petrochemical) pocket spring mattresses. However, when end of life scenarios are considered, the results suggest much larger GHG emission reductions for natural fibre than foam mattresses. The paper also identifies that had the LCA considered only process emissions then 25% of the actual supply chain emissions would have been truncated from the assessment. The IO emissions associated with the biomaterial supply chain were not shown to be discernibly different in scale to the petrochemical foam supply chain. Hybrid has been used successfully in many sectors and on many products and this research validates its usefulness extends to biomaterial specific supply chains.

Refurbishing natural fibre mattresses and reusing the springs, coupled with recycling the waste components, can reduce GHG emissions by 90% compared to sending the mattresses to landfill. Incinerating mattresses via combined heat and power plants for electricity production and converting the waste textiles to ethanol are also shown to reduce GHG emissions, though to a lesser extent than refurbishment and recycling. Mattresses are normally disposed of via landfill however designing for reuse and recycling, coupled with supportive policy and legislation, may encourage more natural fibre mattresses and recycling. Such changes could save between 210 and 2,092 thousand tCO_2-eq in the European Union annually (Glew et. al., 2012).

Sensitivity analyses are undertaken in the paper to predict the impact of common variables in the LCA methodology; functional unit, data quality and data selection. Regarding the functional unit against which to measure the emissions from the mattresses the assessment are switched from an area basis (m^2) to a unit of weight (kg), data quality is assessed by assuming only half of the savings identified are actually achievable and the influence of data selection is tested via the use of alternative input GHG values. In each sensitivity analysis there is no remarkable change in the findings of the LCA implying that the conclusions are robust.

This assessment has serious implications for all biomaterial producers, showing that if being low carbon is an ideal then end of life scenarios are an essential part of the story. However this is not the end of the story, in addition to these findings further investigation were made which in the interests of brevity could not be included in the paper but they are presented here.

4.2. Further reflections on methodology, results and limitations

The potential nuances in LCA are vast and including them all in an academic journal article is not possible. The following sections provide additional analyses pertinent to the wider research presented in the thesis. Specifically, the influence of the following eight issues are explored: 1) Statistical significance, 2) Dissimilable biodegradable carbon (DDOC), 3) Local sourcing, 4) Recycled material, 5) Cost benefit analysis (CBA), 6) Multiple products, 7) Issues beyond GHG emissions, and 8) Land use change (LUC). Table 4.1summarises the findings.

Table 4.1 Summary of Sensitivity Analyses

Further Sensitivity Analyses	Influence / Comments
Statistical significance: biomaterials vs. petrochemicals	Biomaterials do not have significantly lower carbon than petrochemicals unless they are reused.
Statistical significance; end of life scenarios	Landfill has significantly high GHG emissions than other disposal options, re-using and recycling however cause significantly less emissions.
Dissimilable bio-degradable organic carbon	Organic carbon is locked into biomaterials marginally reducing their emissions in landfill.
Local sourcing	Locally sourcing materials could reduce GHG emissions by around 10%.
Recycled inuts	Incorporating recycled inputs would constitute double counting since the full savings achieved by recycling the mattresses are allocated tot eh mattress.
Cost benefit analysis	Landfill is one of the most costly disposal options whereas incineration and recycling may provide some revenue.
Multiple products	GHG emissions from pure foam are relatively unchanged regardless of disposal options, like-for-like foam and hemp swaps can reduce emissions but functionality limits this substitution.

Beyond GHG	Certain biomaterials have high impacts on water depletion, eutrophication, acidification and land use, foam has more impacts in resource consumption.
Land use change	Including LUC could double the natural fibre mattress GHG emissions

4.2.1. **Statistical significance: biomaterials vs. petrochemicals**

Undertaking statistical analysis on the *inputs* to the LCA i.e. between the GHG references, is unusual since no 'true' or 'average' value for the GHG of a material can exist across more than one geographical location or specific supply chain. Statistical analysis of the *outputs* of the LCA however can yield meaningful results.

In order to understand which end of life scenarios caused natural fibre mattresses (biomaterials) to have significantly fewer GHG emissions than foam mattresses (petrochemical) a Chi^2 test was performed as shown in Table 4.2. This shows the likelihood that the difference in GHG emissions between the two mattresses is either due to chance or to actual differences between the supply chains. Certainty to the 95% confidence level that a difference is not simply due to chance is denoted by '*'.

Table 4.2 Chi^2 assessment to compare biomaterials and petrochemical products

	Natural Fibre Mattress Observed (O) ($KgCO_2eq/m^2$)	Memory Foam Mattress Observed (O) ($KgCO_2eq/m^2$)	Expected (E) ($KgCO_2eq/m^2$)	Chi^2 $\sum[(O-E)^2/E]$	Probability Based on 1 Degree of Freedom
Landfill	81	88	84.5	0.29	0.40
Reuse - Landfill	76	83	79.5	0.30	0.40
Recycle	39	56	47.5	3.09	0.92
Reuse - Recycle	8	36	22.0	17.53	0.999*
Incineration CHP	64	72	68.0	0.52	0.5
Reuse - Incineration	57	67	62.0	0.90	0.6

Conversion to ethanol and Landfill	62	79	70.5	2.10	0.8
Reuse Conversion to ethanol and Landfill	37	65	51.0	7.29	0.99*
Conversion to ethanol and Recycling	43	58	50.5	2.17	0.85
Reuse Conversion to ethanol and Recycling	17	41	29.0	9.56	0.99*
Conversion to ethanol and Incineration	46	58	52.0	1.50	0.7
Reuse Conversion to ethanol and Incineration	20	50	35.0	12.35	0.999*

Table 4.2 shows that the biomaterial mattress has significantly lower GHG emissions than the equivalent petrochemical mattress under four end of life scenarios: ‘Reuse and Recycle’, ‘Reuse, Conversion to Ethanol and Landfill’, ‘Reuse, Conversion to Ethanol and Recycling’ and ‘Reuse Conversion to Ethanol and Incineration’. Two interesting themes emerge:

- Firstly, biomaterials may not always have fewer GHG emissions than petrochemicals since only four of the ten treatments yielded statistically significant results.
- Secondly, disposal is critical in determining whether biomaterials are less polluting in terms of GHGs than petrochemicals. In order to claim that biomaterials have significantly lower emissions than petrochemicals they must guarantee that the biomaterials will be reused in some way prior to their disposal.

4.2.2. **Statistical significance; end of life scenarios**

A second Chi^2 test was performed and presented in Table 4.3 to calculate how likely it is that different GHG emissions found between different end of life scenarios are caused by the scenarios themselves and are not merely due to chance. Again, certainty that the difference is not caused by chance to the 95% confidence level is indicated by a '*'.

Table 4.3 Chi^2 assessment to compare of end of life scenarios

	Memory Foam Mattress (O) ($KgCO_2eq/m^2$)	Memory Foam Mattress (E) ($KgCO_2eq/m^2$)	Chi ^{2} ∑(O-E)2/E	Probability Based on 1 Degree of Freedom	Natural Fibre Mattress (O) ($KgCO_2eq/m^2$)	Natural Fibre Mattress (E) ($KgCO_2eq/m^2$)	Chi ^{2} ∑(O-E)2/E	Probability Based on 1 Degree of Freedom
Landfill	88	63	10.083	0.995*	81	46	27.51	0.995*
Recycling	56	63	0.882	0.5	39	46	1.13	0.5
Incineration CHP	72	63	1.241	0.75	64	46	6.90	0.99*
Ethanol and Landfill	79	63	4.037	0.95*	62	46	5.65	0.975*
Ethanol and Recycling	58	63	0.460	0.25	43	46	0.18	0.25
Ethanol and Incineration CHP	63	63	0.001	0.025	46	46	0.00	0
Reuse Landfill	83	63	6.254	0.975*	76	46	20.00	0.995*
Reuse Recycling	36	63	11.380	0.995*	8	46	30.61	0.995*
Reuse Incineration CHP	67	63	0.261	0.25	57	46	2.56	0.9
Reuse Ethanol and Landfill	65	63	0.028	0.1	37	46	1.59	0.75
Reuse Ethanol and Recycling	41	63	8.133	0.995*	17	46	18.06	0.995*
Reuse Ethanol and Incineration CHP	50	63	2.766	0.9	20	46	14.03	0.995*

Table 4.3 shows that the GHG emissions of 'Landfill', 'Ethanol and Landfill', and 'Reuse and landfill' were significantly higher than the emissions of the other scenarios. This implies any disposal fate involving landfill will cause higher GHG emissions for both biomaterial and petrochemical products. This is unsurprising, and supports the argument for policies like landfill tax that have long been established to reduce waste going to landfill (Morris et al., 1998).

Table 4.3 also shows 'Incineration with CHP' in the natural fibre mattress caused significantly higher GHG emissions than other options whereas this was not the case for the foam mattress. Thus, although incineration is seen as preferable to landfill, it may not be a low carbon waste option for biomaterials.

Two end of life scenarios that caused significantly fewer GHG emissions in both biomaterial and petrochemical mattresses; 'Reuse and Recycling' and 'Reuse Ethanol and Recycling'. The implication is that reuse and recycling reduces life cycle GHG emissions regardless of the material's origin. The natural fibre mattress under the 'Reuse Ethanol and Incineration CHP' scenario also had significantly lower GHG emissions showing that biomaterials' ability to produce ethanol gives it the potential to be a low carbon alternative to petrochemicals. Currently no policy for diverting organic waste to ethanol production exists, though this may be a reflection of the infancy of the industry. These results suggest policy support for recycling and ethanol conversion should be prioritised above incineration with CHP for biomaterials.

In summary, the statistical assessments of significance strengthen the waste hierarchy of 'Reuse', 'Recycle', 'Ethanol Production', 'Incineration', and 'Landfill' and also support the idea that combinations of options should be sought where possible. It also further supports the view that end of life scenarios hold the key to unlocking biomaterials' potential to offer low carbon alternatives to petrochemicals.

4.2.3. **D*issimilable* bio-degradable organic carbon (DDOC)**

DDOC values represent how much organic carbon in a material is broken down in landfill and emitted to the atmosphere as CO_2 or methane (CH_4), compared to how much stays intact and is stored within the landfill, as measured over a 100 year period (Biswas et al., 2010). Data on DDOC is scarce because of the long time periods involved and the difficulty of assessing the contents of landfills so values for textiles are often aggregated (European Commission, 2001). In the mattress LCA textiles are assumed to be 50% synthetic and 50% organic[5] fibres. DDOC values and GHG emissions from landfill should be applied only to organic textiles since synthetic textiles only contain fossil carbon which is not biodegradable and so is not attributed any GHG emissions.

Table 4.4 shows the impact of these assumptions. Treating the synthetic and organic textiles as separate inputs alters the ‘landfill’ GHG emissions. Since DDOC is now only attributed to organic textiles, this causes their GHG emissions to increase, whereas the GHG emissions from synthetic textiles falls to only include the emissions from transportation and processing. The impact on ‘Recycling’ is zero since organic and synthetic textiles are judged to have equal potential to replace virgin textile production. Emissions arising from ‘incineration with CHP’ increases for synthetic textiles, as these emit fossil carbon during combustion. Conversely, they fall for organic textiles which do not contain fossil carbon. Finally the potential to produce ethanol from synthetic textiles is removed, meaning only organic textiles can be converted to ethanol and achieve GHG reductions.

[5] ‘Organic’ refers to organic carbon, from a biological source i.e. any plant or animal, it does not refer to only organically grown or certified plants and animals.

Table 4.4 Revised GHG emissions for end of life scenarios per kg of material

Material	Landfill (KgCO2eq/Kg)	Recycling (KgCO2eq/Kg)	Incineration CHP (KgCO2eq/Kg)	Conversion to Ethanol (KgCO2eq/Kg)
Previously combined assumption of Textiles	0.015	-3.169	-0.162	-2.587
Organic Textiles	0.030	-3.169	-0.880	-2.587
Synthetic Textiles	0.008	-3.169	0.586	0.000

Table 4.5 uses these revised numbers to re-calculate the end of life emissions for the mattresses. The influence of landfill is interesting in that both mattresses contain more than 50% organic textiles so the DDOC and therefore GHG emissions caused by landfill increases by 71% for the natural fibre and 50% for the foam mattress. Since there are less than 50% synthetic textiles in the mattresses, less fossil carbon is emitted during combustion, meaning the benefit of the CHP is improved dramatically: by 57% for the foam and by 280% in the natural fibre mattress. Conversely, where all the textiles had been assumed previously to be converted to ethanol, when the synthetic fibres are omitted, the ethanol yield is reduced, and therefore the GHG emissions savings are reduced by 45% for the foam and 18% for the natural fibre mattress.

Table 4.5 Revised GHG emissions ($kgCO_2eq/kg$) of end of life scenarios per m^2 of mattress

	Foam Mattress		Natural Fibre Mattress	
	Previously combined assumption Textiles	Separate Synthetic and Organic Textiles	Previously combined assumption Textiles	Separate Synthetic and Organic Textiles
Landfill	0.16	0.22	0.34	0.58
Recycling	-33.88	-33.88	-70.95	-70.95
Incineration CHP	-1.73	-2.50	-3.63	-13.82
Conversion to Ethanol and Landfill	-27.66	-15.43	-57.92	-47.52
Conversion to Ethanol and Recycling	-27.66	-30.40	-57.92	-60.26
Conversion to Ethanol and Incineration	-27.66	-12.71	-57.92	-45.20

The net impact of these changes on the overall mattress GHG emissions is shown in Figure 4.1. The 'landfill' scenario changes by less than 1%. The most significant change takes place for ethanol conversion which shows a GHG increase of 20% in the 'reuse ethanol and incineration scenario' compared to the original assessment. Improvements are seen for the Incineration options though as a general rule these changes are insufficient to make incineration a preferable end of life option to ethanol conversion or recycling.

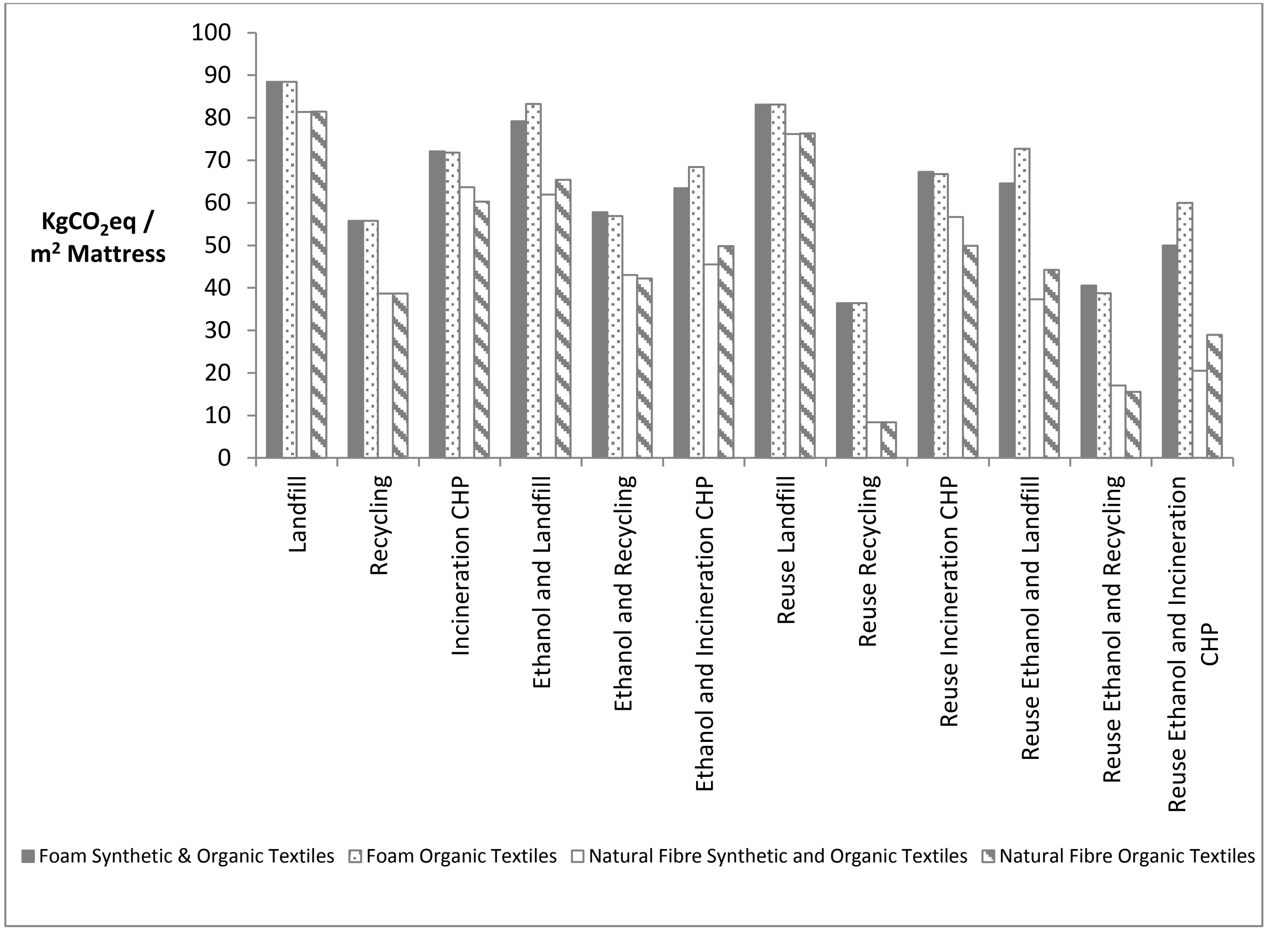


Figure 4.1 Effect of separating organic and synthetic textiles on DDOC and resulting GHG emissions

In summary, the specificity and detail of an LCA's data can be crucial to its accuracy. In this assessment, despite including more realistic assumptions regarding the textile composition in the mattresses, the resulting change in DDOC in landfill and the different incineration and ethanol conversion rates were not sufficient to change the waste disposal hierarchy previously presented.

4.2.3.1. Animal vs. plant fibres

Refining this sensitivity analysis further to define individual DDOC values for each natural fibre was not possible in this desk based study as very little data is published about the DDOC of different textiles. This is perhaps because of the difficulty in gaining access to and controlling the conditions within landfills, not to mention the length of time one must wait in order to measure the decay of organic material. Generally, research into the generation of GHGs from landfill treats all waste streams together as municipal solid waste (MSW), making it difficult to draw out specific conclusions for different textiles or other items. It is therefore unsurprising that organic textiles or natural fibres are often all assumed in the literature to have equal DDOC, though this assumption may not necessarily be true.

The lignin content in plant fibres and the keratin content in animal fibres are the main determinants of a textile's biodegradability. Lignin has been argued by some to not degrade significantly under anaerobic conditions and therefore it can be assumed it does not release any of its carbon in landfills (US EPA, 2012, Barlaz et al., 1989). A report for the Sustainable Landfill Foundation in the Netherlands shows that lignin (assuming a wet landfill) can also protect cellulose from breaking down, inhibiting the release of its carbon too. Thus, textiles and biomaterials with high lignin content may be expected to emit relatively fewer GHG emissions in landfill i.e. have a low level of DDOC (Oonk, 2010). However, if we take the most abundant natural plant fibres in the mattresses hemp and cotton, we can see that hemp has close to 3% lignin content and there is no discernible amount of lignin in cotton at all (Summerscales et al., 2010). Thus natural fibres used in mattresses are not highly lignified materials and one could therefore argue that the impact that lignin has on their DDOC and therefore on their overall GHG emissions is relatively small, i.e. virtually all of the carbon in the plant fibres is likely to be broken down.

Wool and other forms of animal hair can be up to 95% keratin, an insoluble protein that, similar to lignin, is very slow to break down under aerobic conditions (Cardamone et al., 2009). Keratin is said to be made up of just less than 50% carbon (Earland and Knight, 1955) meaning that relatively large amounts of carbon in animal hair may be stored for long periods of time in landfills, especially if anaerobic conditions exist (Bálint et al., 2005). The implication is that wool and other animal fibres may have a much lower DDOC than plant fibres given that the majority of carbon locked in the wool is likely to remain un-degraded in the landfill. Intuitively this may therefore hint at wool and animal hair being a more environmentally friendly fibre than plant fibres. However, viewing this phenomenon in isolation can be misleading since the life cycle GHG emissions for wool production is much higher than that of hemp or cotton fibres, being over 14 kgCO_2eq/kg, compared to 0.84 and 3.07 kgCO_2eq/kg respectively.

In summary, the importance of the specific DDOC on the biomaterial lifecycle GHG emissions has been shown to be very small, so the assumption that all the natural fibre textiles have equal DDOC is deemed to be acceptable for this research. DDOC clearly influences GHG emissions of biomaterials sent to landfill and may be the focus of more specific research in the field of biomaterials' decomposition. This would inform the wider discussion around the best way to dispose of biomaterials and identify which less-lignified biomaterials would be more polluting in landfill than others.

4.2.4. **Local sourcing vs. imports**

LCA commonly use secondary data since primary data collection on large scales is impractical and expensive. Secondary data however are incapable of distinguishing nuances in supply chains. For example, there may be differences in energy efficiencies, production techniques or transport distances for the same product made in different countries. Often these inaccuracies are dismissed as limitations, however in the case of this research it is possible to identify the GHG emissions savings for the mattresses that use local raw materials rather than imported materials. The secondary LCA data used in the calculations includes average transport distances to account for the movement of component parts in the production of each material. However, this does not incorporate the transport distances covered in delivering the materials to the mattress factory. This section attempts to quantify the importance of sourcing locally by including the GHG emissions caused by these additional transport distances.

The wool and hemp were sourced locally from Yorkshire farms and the majority of mattress components were sourced from multiple locations within the EU. However, the steel, brass, cotton and animal hair (except wool) were all sourced from China. Precise locations vary according to business conditions, thus it is difficult to get an accurate reflection of where imports are coming from. For the purposes of this research it has been assumed that all components sourced from the EU come from

Germany, where many of the components are actually sourced. The transport distance from Leeds (the location of the factory in Yorkshire) to Dresden (a major manufacturing area within Germany) is 1,450km by road[6]. This was taken to be the representative distance that all the EU imports travelled. It could not be known if components crossed the English Channel by rail or ferry, thus road was assumed for this section of the route. Crossing the English Channel may occur at a variety of ports and it would be impractical to identify each material's specific point of entry into the UK. In addition, this part of the journey is considered to be a minor contributor to the overall transport GHG emissions and both ferry and rail have lower GHG emissions per km than road haulage. Thus, the assumption of road haulage was taken for simplicity on the understanding this would yield a worse-case but consistent estimate. China has several large ports and no single one is responsible for all the materials sourced to make the mattresses, thus, the components sourced from China were assumed to have been shipped 17,807km[7] from Guangzou which is one of the main industrial areas and shipping ports in China that ships many textiles to Europe. The UK port of Portsmouth was assumed to be the UK recipient port, since this has a trade connection with Guangzou. Therefore, an additional transport of 418km by road to Leeds is assumed.

In the following local sourcing scenarios, all the natural fibre fillings and steel have been assumed to be sourced in Yorkshire, travelling by road from within 30km (from farms in North and West Yorkshire) and 60km (from Sheffield) respectively of the factory. This is a realistic estimate based on knowledge of the mattress factory's supply chain. The Department for the Environment, Food and Rural Affairs'

[6] Maps.google.co.uk

[7] Sea-distances.com

(DEFRA) guidelines on GHG intensity for various types of haulage was used to calculate the GHG emissions per km of transport of the mattress components. These are: 0.132 kgCO_2eq for large lorry road haulage and 0.013 kgCO_2eq for long distance shipping (DEFRA, 2008). The effects of including this additional transport on the lifecycle GHG emissions for the natural fibre mattress are shown in Figure 4.2.

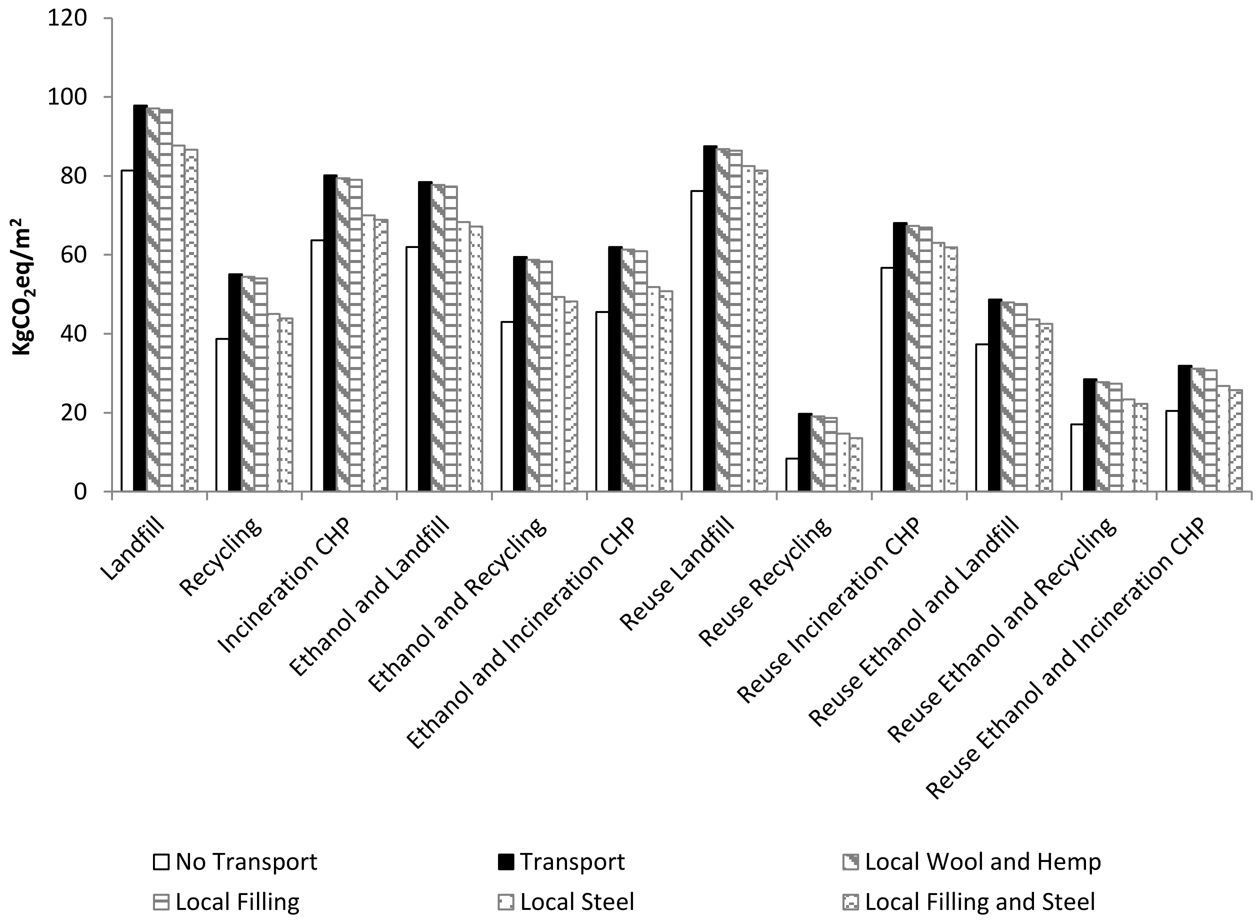


Figure 4.2 Addition of transport emissions to scenarios of natural fibre mattress component locations

The black columns in Figure 4.2 show the original GHG emissions, this excludes transport emissions for delivering components to the mattress factory. The white columns show the change when the additional transport emissions are added, resulting in , a 15% increase. The patterned bars in Figure 4.2 show the potential reduction in emissions achieved by locally sourcing key components from Yorkshire, which results in only small reductions in GHG emissions. By far the greatest reduction comes through sourcing local steel which is the heaviest single component in the mattresses and therefore requires more fuel to transport. All of these savings are relatively modest however compared to those achieved via end of life scenarios.

In summary, omitting transport emissions can affect the accuracy of LCA results, though it has not changed the general findings. This study found that end of life scenarios are more influential than transport in determining overall GHG emissions. Local sourcing can, nevertheless, be an important step in reducing GHG emissions and may also be useful as a marketing tool and business opportunity since it can also improve transparency in the supply chain and resource supply security.

4.2.5. **Recycled vs. virgin material**

Using recycled materials is a popular approach to reduce environmental impacts. In this study, the benefits of replacing virgin materials are already accounted for in the end of life scenarios. Allocating any further benefit to the mattress for using recycled materials may be seen to be double counting the benefit of recycling. However, complications arise where recycled materials are used and these are then recycled again at the end of life, thereby avoiding two lots of virgin material (assuming this additional recycling can be guaranteed). Considerable work has been done on allocating credits for such scenarios within LCA and there is no consensus on which approach should be adopted. For example, it is unclear whether the benefit of recycling should be passed to the product being recycled, the product being made from recycled goods or shared somehow between these. The chosen method tends to reflect the aim of the study which can aid decision making but makes comparisons with other assessments more difficult (Shen et al., 2010, Ekvall and Tillman, 1997).

When a product can be recycled multiple times its quality can suffer. In the case of natural fibres, part of the material may be lost to waste, fibres may become shorter and hence each regeneration has an increasingly limited application and marginally lower quality, which further complicates the assessment. Given that recycling was the most effective end of life scenario, understanding how multiple recycling affects GHG emissions is important but involves a high degree of propositions and 'what ifs' that make the hypothetical calculations relatively meaningless.

In summary, in this research it has been assumed that the future recycling of the mattress components is attributed to the mattress. Credits for any recycled materials used in the mattress are therefore allocated to the previous supply chain from where this recycled material came. Simplistically this avoids double counting, but it is also a practical approach, as it is not known how many times the materials can usefully be recycled, nor does it try to identify or emphasise which phase of the potential recycling stages should be afforded the most credit. Issues around recycling and emissions allocation are already well debated in the literature (Ekvall and Tillman, 1997, Ekvall et al., 2007) and taking this specific line of assessment further is therefore outside the scope of this research.

4.2.6. **Cost benefit analysis (CBA)**

The most effective end of life scenario in terms of reducing life cycle GHG emissions may not necessarily be the most cost effective scenario. Performing a CBA to find the cost per kg of GHG emissions avoided in each mitigation measure would show the most attractive commercial end of life scenario. However, undertaking a full CBA of landfills, incineration and recycling plants is a huge undertaking. Such large projects have already been tackled by governments and other large organisations around the world, though it is important to note that most are concerned with municipal waste schemes. Using this existing data to extract information on individual materials therefore inherently has a high degree of uncertainty associated with it. This section provides a useful discussion on the cost effectiveness of saving 1kg of GHG emissions by each of the disposal options in turn but stops short of attempting a full CBA which is beyond the scope of this thesis.

4.2.6.1. Economics of landfill

Landfill caused additional GHG emissions of 1.02 $kgCO_2eq$ and 1.07$kgCO_2eq$ for the natural fibre and memory foam mattress respectively, thus it is not possible to calculate the cost of saving 1 kg of GHG. This suggests there is a negative “double whammy” to landfill in that it costs money to do in addition to not making any GHG savings. Landfill tax in the UK rose in April 2012 to £64 per ton. In the case of the foam (67.79kg) and natural fibre (61.61kg) mattresses, this would mean a dumping charge of £4.34 and £3.94 respectively. Some landfill sites generate electricity through burning landfill gas which can be sold to the national grid, which further complicates the economics of landfills. Furthermore, it is not clear who should take responsibility for the costs of disposal. Consumers may not be subject to Landfill

Tax at municipal waste sites, however, these often do not admit bulky wastes like mattresses, in which case, consumers may have to pay for special collections that exceed the cost of the tax. A search of local councils in the UK shows charges for bulky items ranging from £10.50 per item in Northumberland[8] to £20 in Mid Sussex[9]. Landfill therefore does not represent a good option from the perspective of GHG savings or cost.

4.2.6.2. Economics of incineration and CHP

A common alternative to landfill for mass mixed waste is incineration. Similarly to landfill sites, incineration plants produce electricity which can be sold to the national grid. In some instances heat is exported as well. A report by the World Bank showed an that an income of between £12.55 to £21.33 per tonne of municipal waste could be generated using incineration and CHP (Rand et al., 1999). These data are nevertheless more than 10 years old, and in reality, given that the income is closely tied to the price of electricity which has increased significantly over the last decade, they are out of date. Assuming they were still valid today as indications, and given that the natural fibre mattress was shown to avoid 48.92kgCO_2eq and weigh 61.61kg, the income from avoiding 1 kg of GHG emissions via incineration could be claimed to be between £0.26 and £0.44. This obviously incorporates very high uncertainty. For example it assumes that 1 tonne of mattresses yields the same electricity as 1 tonne of municipal waste and it doesn't allow for technological advancements or efficiency savings over the last ten years. However, it is useful as an order of magnitude estimate to the cost of avoiding 1kg of GHG emissions via incineration.

[8] http://www.northumberland.gov.uk/default.aspx?page=874, accessed 12/04/2012

[9] http://www.midsussex.gov.uk/8215.htm, accessed 12/04/2012

4.2.6.3. Economics of refurbishing

Refurbishing the foam and natural fibre mattresses saved 16.08kgCO_2eq and 15.69kgCO_2eq respectively in the scenarios presented here. The cost to the manufacturers of refurbishing a mattress is incorporated into the sale price of the mattress which makes calculating the exact cost of these avoided GHG emissions difficult. However, considering that the refurbishment is a replacement for a new mattress, it could be argued that the production costs of a new mattress, which may also factor in a degree of profit, are offset against this, so that the saving is achieved at no additional cost. In this instance the GHG savings are essentially 'free'.

4.2.6.4. Economics of saving 1kg of GHG via recycling

There are several recycling companies in the UK which collect mattresses and will strip and sort the waste streams and sell these into new supply chains. The gate fee for a consumer to have their mattress recycled is between £2 and £4 depending on their proximity to collection hubs and the processing sites, assuming no collection is required (Personal Communication 2011 REF). This is on a par with landfill tax. Therefore, where the costs of collection are greater than this, recycling may not be deemed viable and from a rational economic view, mattresses will be sent to landfill instead.

The cost to the consumer is only a small part of the overall economics of recycling. Commercial sensitivity means little data is available on the profit achieved through recycling mattresses and hence the economic benefit of saving 1 kg of GHG, however the mattress recycling industry is relatively healthy, supporting 28 mattress companies in the UK (up from just 4 companies worldwide in 2008) and collecting well over 1 million mattresses every year, indicating that profits may be achieved (Bagnall, 2012).

A report published by Friends of the Earth (FOE) summarises the economics of recycling various materials, showing that the savings achieved can be highly variable especially when the external benefits of recycling are monetised and internalised into the overall economics. Ferrous metal recycling for instance, can save between £49 and £3,239 per ton and although some plastics can make £460 per ton, some may not provide a profit at all (FOE, 2003). Price fluctuations linked to market supply and demand clearly make recycling, like all commodity markets, unstable. Using values from the FOE report of £297 per ton for steel recycling, £48 per ton for plastics and £66 per ton for textiles could provide an income of £12.16 per mattress. Recycling the mattress achieved GHG savings of 97.11kgCO2eq which puts the income generated by avoiding 1 kg of GHG emissions through recycling at around £0.13. This is around half that of the incineration option and may seem quite a small incentive, however the most up to date price for recycled materials that could be found and used here was over 10 years out of date and so this may be more profitable today. Despite the significant uncertainty, it is reasonable to assume that the cost of achieving GHG savings is negative (i.e. profitable) and therefore may be seen as an attractive disposal option from both cost and GHG emissions reduction perspectives.

4.2.6.5. Economics ethanol conversion

There are currently no ethanol-from-waste biorefineries in the UK. This lack of infrastructure is a sign that the start-up investment is high and that the technology is not mature. In addition there are different processing methods which can greatly influence yields and therefore GHG emission savings per ton of waste feedstock (Schmitt et al., 2012). Given the lack of commercial-scale plants, it is not possible to accurately calculate the cost of each kg of GHG emission saved (which would require experimental data). It may be reasonable to assume that the costs per kgCO_2eq avoided are therefore likely to be higher than for recycling or refurbishment since no industry yet exists for ethanol production from waste. However, this may become a more important disposal option as incentives for producing biofuels and reducing waste increase. In the face of escalating petroleum costs, ethanol conversion from waste may soon compete financially with recycling and incineration (Faraco and Hadar, 2011).

4.2.6.6. CBA Summary

Landfill is not an attractive disposal option for GHG savings or from a financial perspective. Conversely, incineration with CHP and recycling can provide GHG savings in addition to economic returns and may therefore be preferable options; indeed these are the most advanced industries. The CBA of refurbishing biomaterial products to avoid GHG emissions is difficult to quantify since the costs are often internalised in the initial transactions. Similarly, cost data is not yet available for ethanol conversion because the industry is in its infancy. In general, the problems of data accuracy are significant and high levels of uncertainty make it difficult to place faith in quoted figures. However, the discussion around the relative cost effectiveness of reducing GHG emissions of each scheme is useful because it can often be profit, not the desire to 'do good', that determines whether a low carbon technology or practice will prosper.

4.2.7. Multiple products

Mattresses are made from several materials making straightforward comparisons between biomaterials and petrochemicals problematic. Simple foam slab mattresses are relatively common and represent perhaps a more appropriate product against which to compare biomaterial pocket spring mattress to the foam and spring memory foam mattress in the paper. One complication is that the simple mattresses are not of equal price and so therefore may not be considered to be equal in quality or performance. The relative performance of luxury versus standard products is a debate that is beyond the scope of this study and so is accepted in the paper as a limitation.

4.2.7.1. Simplified foam slab mattress

This section presents results for the LCA of a simple foam slab mattress following the same methodology used in the journal paper for the natural fibre mattress. It uses additional data supplied by the mattress manufacturers (Table 3.5) which show that foam makes up around 85% of the foam slab mattress by weight.

Table 4.6 Foam slab mattress components

Component Name	Equivalent Material	Quantity (kg unless stated)
Memory Foam	PUR Foam	20.400
Contura Foam	PUR Foam	11.600
Natural Weave	Woven Cotton 39% & Viscose 61%	2.800
Labels / Cards	Paper	0.043
Poly Bag	Extrusion Film	1.035
Corner Protector	Cardboard	1.850
Bubble Wrap	Polyethylene terephthalate	0.141
Direct Electricity	Electricity (KWh)	2.411
Indirect Electricity	Electricity (KWh)	1.311
Direct Heating	Gas (KWh)	0.200
Indirect Heating	Gas (KWh)	0.115
Transport	Diesel (litres)	3.109

The life cycle GHG emissions for the foam slab mattress using the same end of life scenarios that were applied to the pocket spring foam and natural fibre mattresses are shown in Figure 4.3. A remarkable trend can be seen that under every end of life scenario the foam slab mattress shows no discernible reduction in GHG emissions. This infers that foam does not have any useful options available to it at the end of its life. As a result, foam generally has higher life cycle GHG emissions than alternative non-petrochemicals, regardless of the end of life scenario adopted. This may provide a more definitive comparison between biomaterials and petrochemicals since it implies that the benefits accrued by the foam pocket spring mattress in its end of life stages were provided by the natural fibres and springs, not the foam. This seems to further strengthen the findings that only cradle to grave assessments are able to articulate the advantages, in terms of GHG emissions, of using biomaterials over petrochemicals.

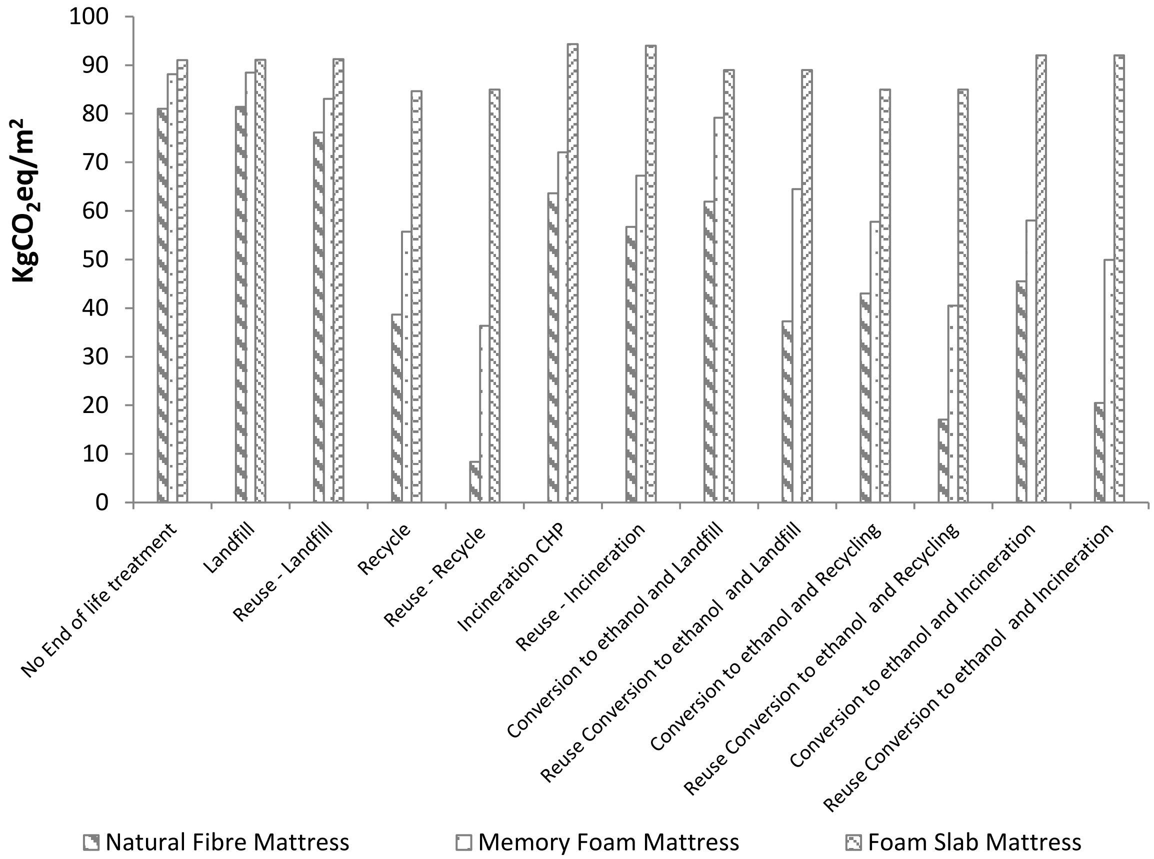


Figure 4.3 Comparison of foam slab mattress GHG emissions

In summary, quantifying the benefits of using biomaterials on the overall life cycle GHG emissions of products that are inherently made from an array of components may be a significant challenge for the biomaterials industry. In this study under the landfill option there is a negligible difference in GHG emissions between using biomaterials or petrochemicals. However, if end of life scenarios are considered, biomaterials show an overwhelming reduction in GHG emissions compared to the simple foam mattress.

4.2.7.2. Substituting in hemp 'like for like'

Another interesting contribution to the debate is that natural fibre alternatives to petrochemicals are not always fully transferable and may require complementary products. In the case of replacing foam in a mattress, natural fibre fillings alone cannot provide sufficient support and so require springs. Including the complete list of components in each mattress in the GHG calculations averts any problems regarding bias being placed on biomaterial products that may otherwise have inferior functional performance if, for example, the springs were ignored. However, this is a complicating factor when making comparisons between petrochemical and biomaterial products. Within the mattress there are petrochemical products that theoretically could be swapped like-for-like with biomaterials with no reduction in function or additional inputs being required. Table 4.7 presents an assessment of the impact of replacing all the synthetic 'Flexbond' material that is used as one of the fillings in the foam mattress with hemp.

Table 4.7 Like-for-like comparison of biomaterial and petrochemical mattress fillings

Material	GHG density of Material ($KgCO_2eq/Kg$)	Current Fabric Quantity (Kg / m^2 Mattress)	Current Emissions ($KgCO_2eq$ / m^2 Mattress)	100% Hemp Fabric Emissions ($KgCO_2eq$ / m^2 Mattress)	Avoided Emissions ($KgCO_2eq$ / m^2 Mattress)
Hemp	0.57	0.63	0.36	0.62	
Flexbond	2.7	0.45	1.21	0.00	
		Total	**1.57**	**0.62**	**0.95**

Replacing the synthetic material yields a small saving in GHG emissions. However, the overall emissions of the foam mattress are over 88 $KgCO_2eq/m^2$ so switching to hemp saves just over 1%. This is a small saving compared to those achieved via end of life options like recycling. Despite this, it is a relatively unobtrusive change as it requires little alteration to existing production lines and may be complementary to other approaches. Consequently, this change is more likely to be undertaken and if this saving were replicated in every one of the 35 million mattresses sold every year in the EU, it could be seen as a useful step in reducing the GHG emissions of foam mattresses.

4.2.8. **Beyond GHG emissions**

Assessing all three dimensions of sustainability (environment, society and economy) within one single study can be problematic because of issues of subjectivity and data collection (Robèrt et al., 2002). This chapter has therefore focused only on quantifying GHG emissions. This decision was made because economy-wide data produced by the Office of National Statistics (ONS) needed for the hybrid LCA methodology currently only exists for GHG emissions, though other environmental impact categories are anticipated soon. Simple process LCA databases already include information on other environmental impacts, though it must be noted that these will also share the same limitations regarding data quality as those discussed for the GHG values quoted in LCA databases. Presented here are the process LCA impacts for both the natural fibre and foam mattresses using data supplied by ecoinvent v2.2 for the following five commonly reported areas in agricultural based LCA: 1) water consumption, 2) resource consumption, 3) land use, 4) eutrophication risk and 5) acidification risk.

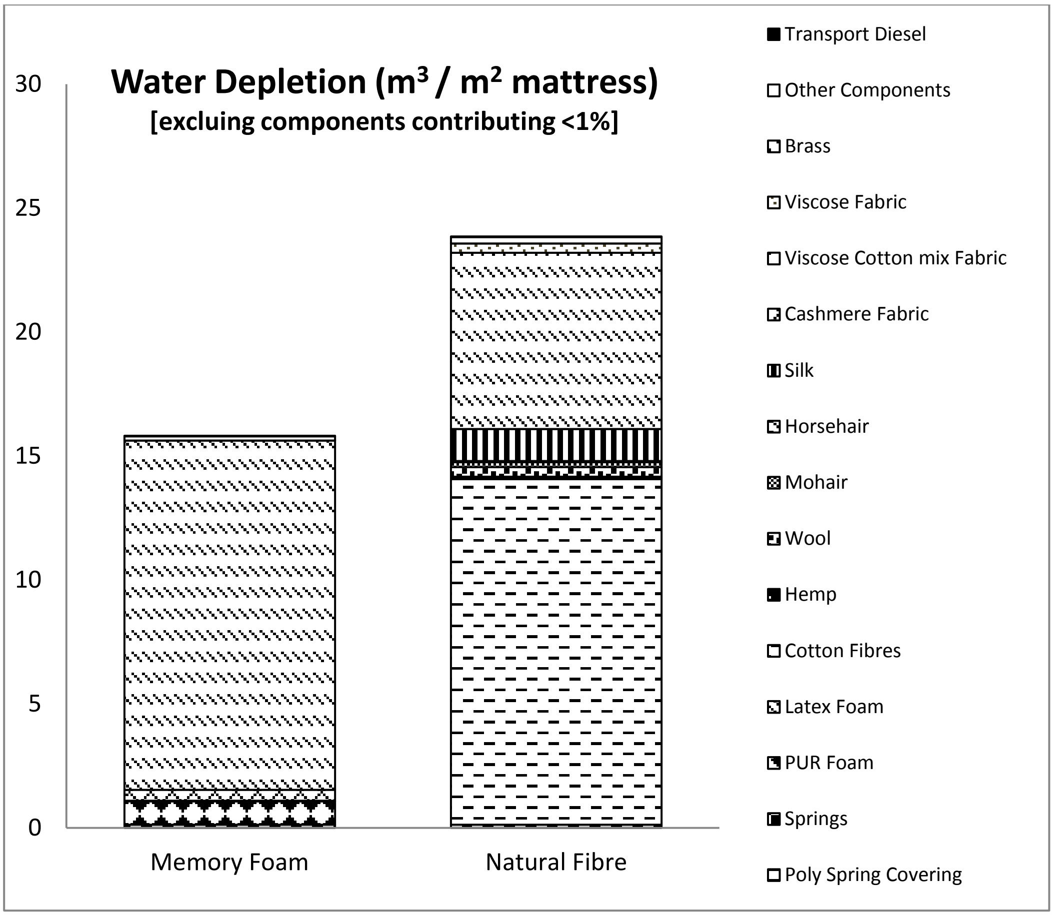


Figure 4.4 LCA of mattresses: water depletion

As is perhaps understandable given the cotton content of the natural fibre mattress (cotton has very high irrigation requirements), the foam mattress is shown in Figure 4.4 to cause much less water depletion. The viscose and cotton mix makes by far the greatest contribution to the foam and is second only to cotton in the natural fibre mattress. Silk is the third largest contributor despite its relatively small contribution by weight to the natural fibre mattress. PUR foam is shown to have a higher water usage than the combined depletion caused by the natural fibre fillings of all the animal hair and hemp combined, thus if the cotton, viscose and silk were targeted or replaced, the natural fibre mattress may not have such a great water dependency.

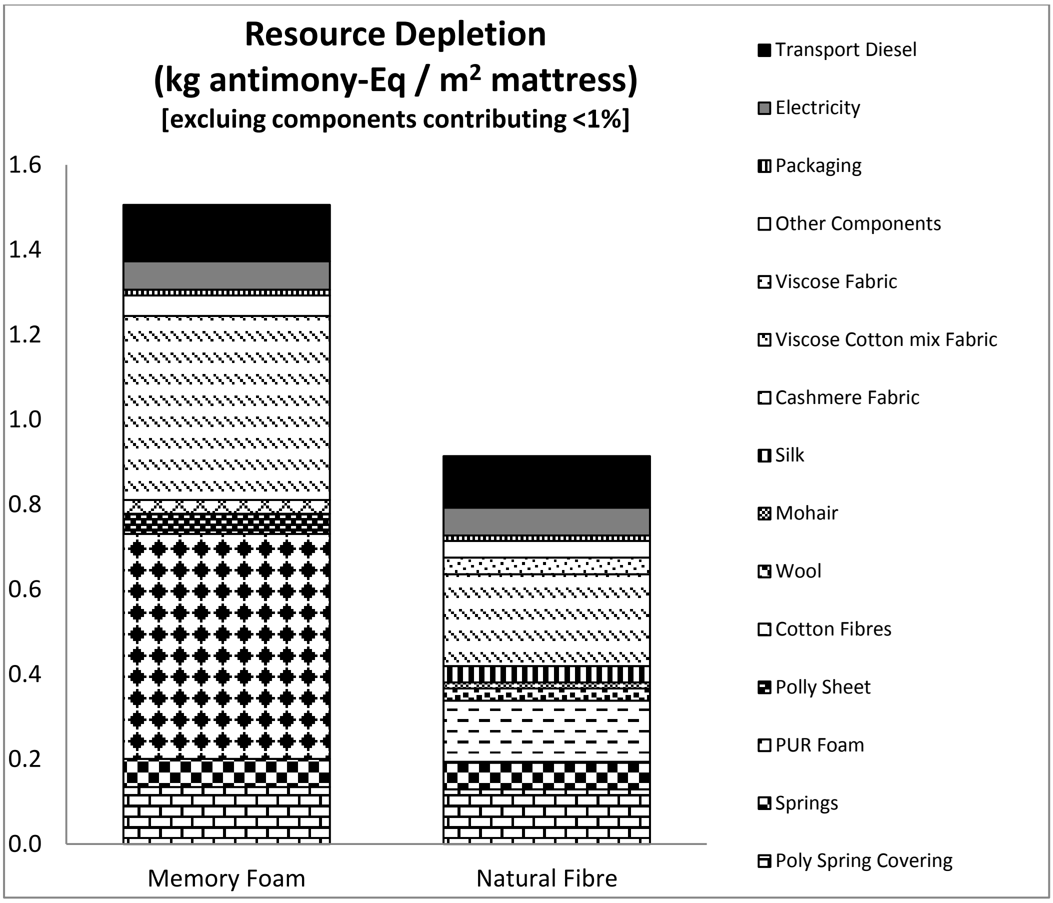


Figure 4.5 LCA of mattresses: resource depletion

Figure 4.5 shows that PUR foam is the biggest resource depleting component, followed by the viscose cotton mix and the cotton fibres, which require large quantities of oil based fertilizers. Apart from these impacts, the mattresses perform relatively similarly, having for example, near identical resource consumption from poly spring covering, steel springs, diesel and the electricity consumption used in the processing of the mattress and its delivery to the shops from the factory. Similar to water depletion, targeting the major problem components could significantly reduce the overall resource depletion properties for both mattresses.

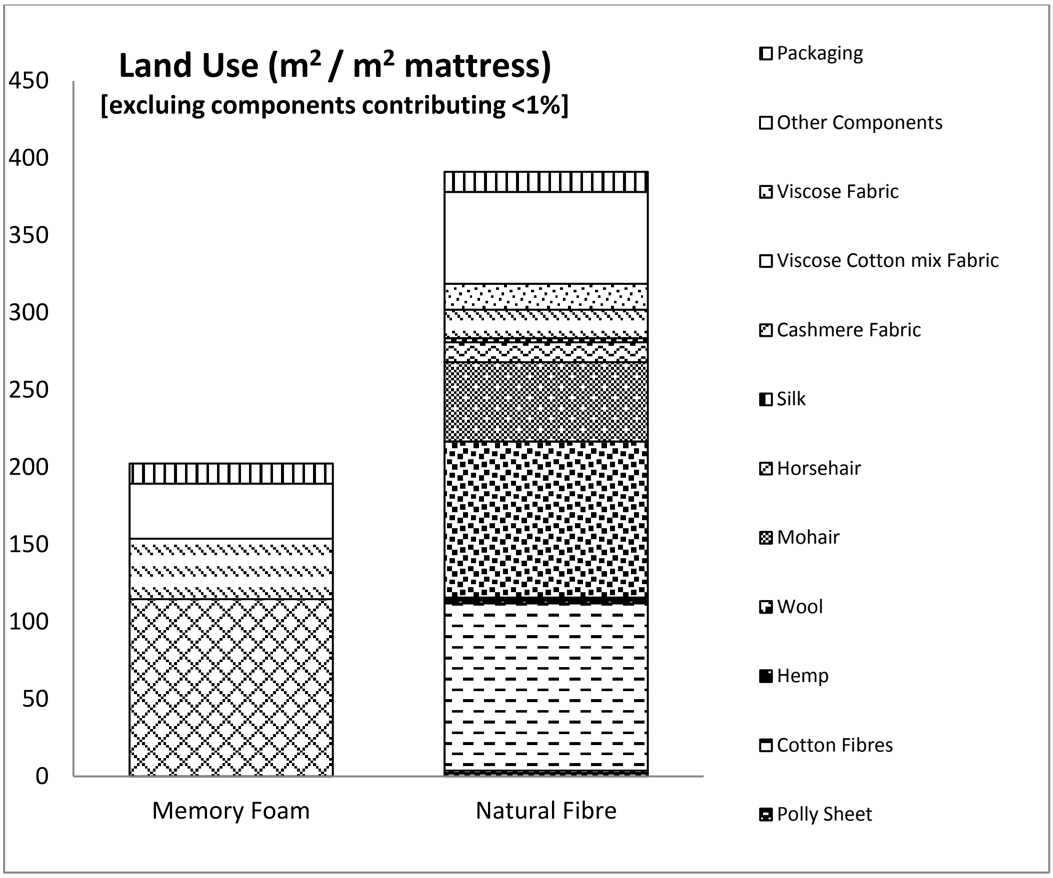


Figure 4.6 LCA of mattresses: land use

The land use required to grow feedstock for the natural fibres is greater than that needed to make foam; roughly double according to Figure 4.6. Cashmere and the viscose cotton mix present the biggest land use demands for the foam mattresses. The wool and mohair are the stand out components for the natural fibre mattress, despite their relatively small quantities. This is significant, showing that although animal hair has been shown to have lower water depletion and resource requirements than plant fibres, animal production is land hungry. Exact comparisons should be treated with caution however, as the land often used for animal husbandry can be marginal or otherwise unproductive land, whereas crops require land that must have good productivity and therefore may be in demand for other products such as food.

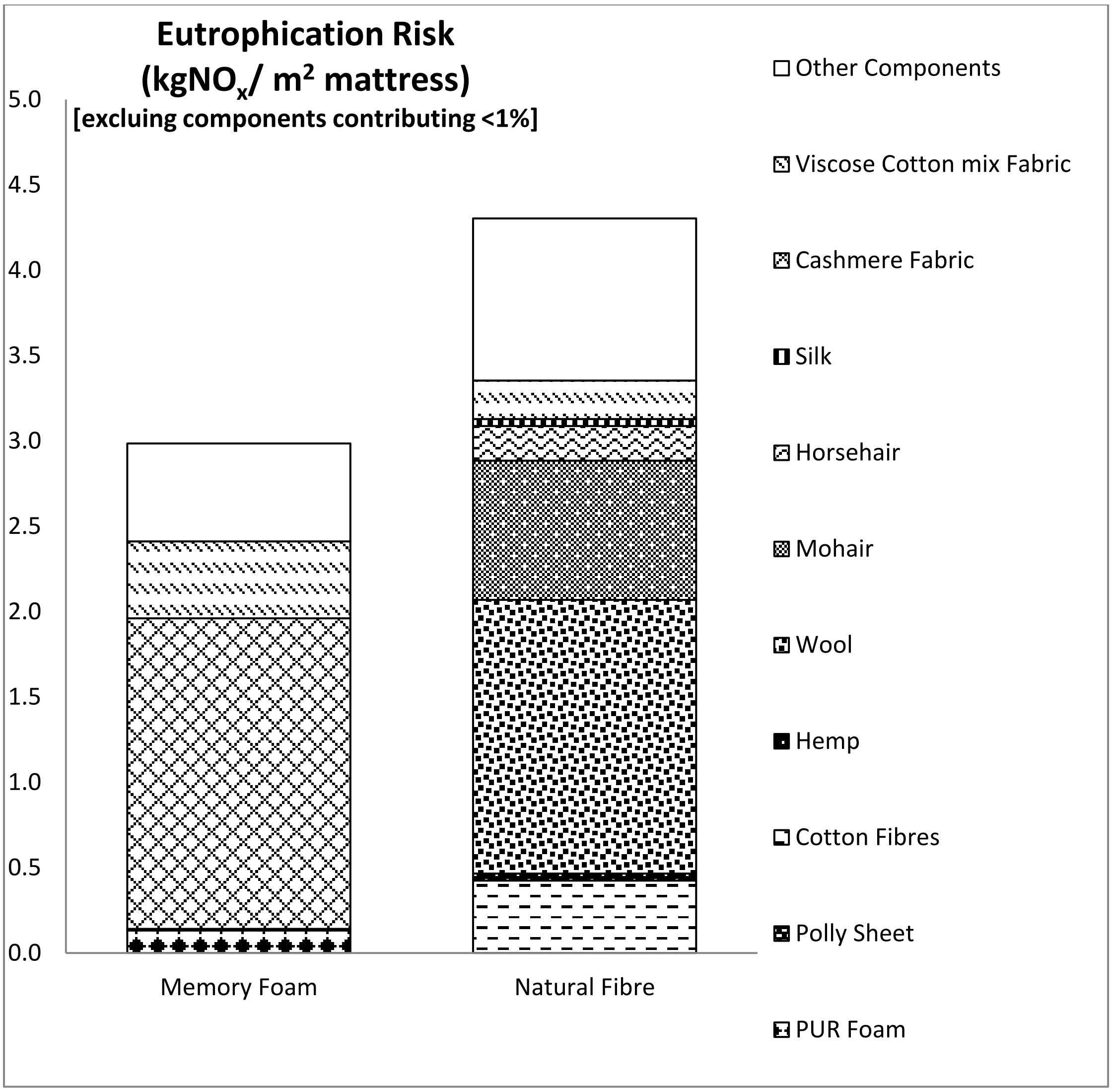


Figure 4.7 LCA of mattresses: eutrophication risk

Similar to the land use and water depletion trends, the agricultural components contribute the major risk factors in eutrophication pollution (Figure 4.7). Specifically, these come from wool, mohair and cotton for the natural fibre mattress, and cashmere and the viscose and cotton mix for the foam mattress. This is largely due to fertilizer applied to the ground but also in the case of animal hair, the nitrogen-rich faeces of sheep and goats contribute to the eutrophication risk.

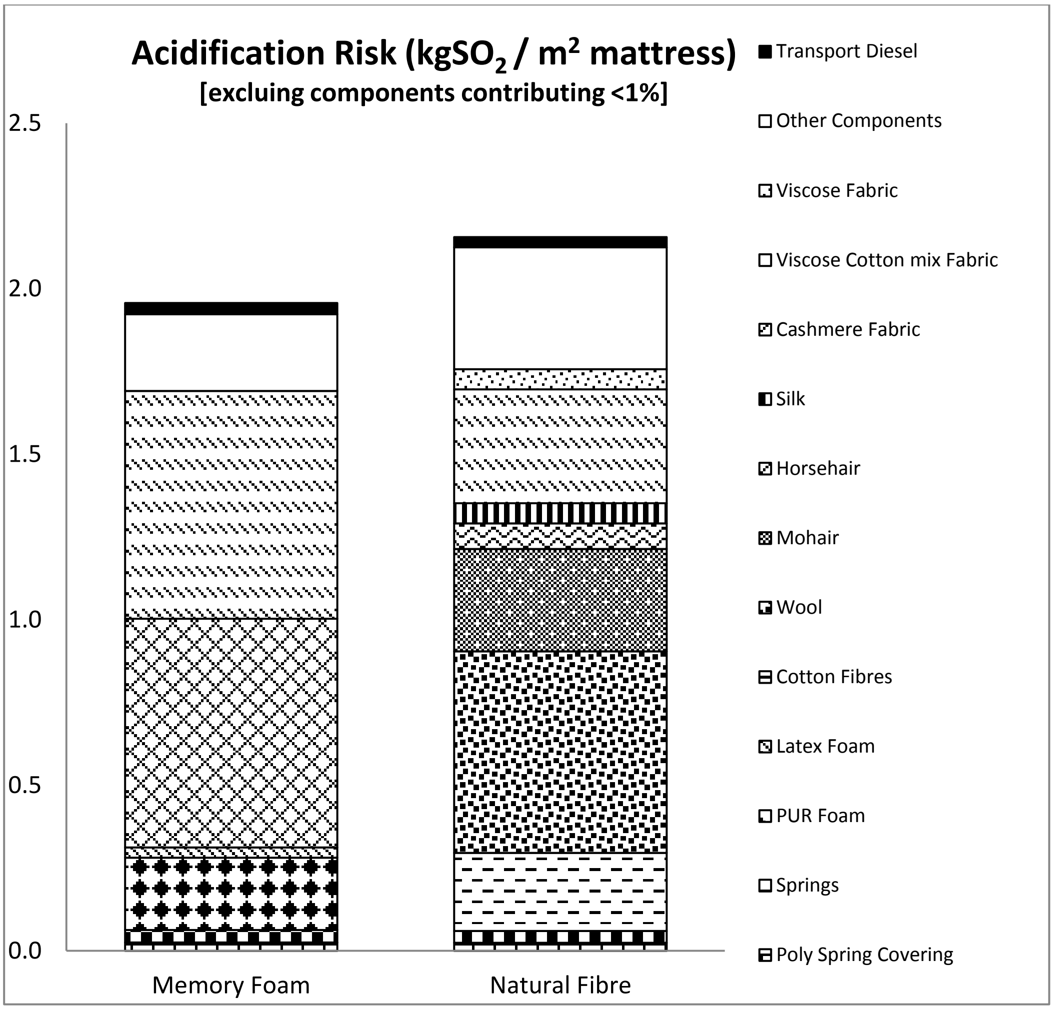


Figure 4.8 LCA of mattresses: acidification risk

As can be seen in Figure 4.8 the comparison of acidification shows the mattresses to both present similar risks. The cashmere and viscose and cotton mix are the largest contributors again due to fertilizer application but also perhaps as a result of gases emitted by viscose production which has a particularly chemically intensive process. These are closely followed by the wool and mohair, which may again be responsible for gaseous emissions related to digestion and excretion. The chemical engineering required for PUR foam production is also potentially important in acidification, being much more prominent than in the other environmental impacts.

In summary, the biomaterials in both the natural fibre and foam mattresses were responsible for the majority of the environmental harms described in the figures above. Targeting problem materials, namely cotton, viscose, wool and mohair and replacing these with less intensively produced alternatives, like hemp, or using the same products but using less intensive production methods, could significantly reduce the environmental impact of the mattresses. It is important to view the results in the proper context, such as in the case of land use, where it may not be appropriate to equate $1m^2$ of marginal or unproductive land needed for wool production with $1m^2$ of high quality arable farmland. This analysis has shown that in order to define the relative environmental merits of petrochemicals versus biomaterials, one must define which environmental characteristics are most important to the aim of the project. It is also important to understand the limitations and context of the data.

4.2.9. **Land use change (LUC)**

The final extrapolation undertaken was to examine more closely the impact of LUC on the GHG emissions of the mattresses. The reason this is important is that GHG emissions can be released from existing carbon stocks in the soil and vegetation by cultivating feedstock (Searchinger et al., 2008). This is a hot topic in the industry and much research is directed towards this and indirect land use change (iLUC), where growing feedstock on a piece of land that is used for forestry for example pushes this forestry to another area, which in turn causes more land use potentially emitting more GHG emissions (Cornelissen and Dehue, 2010, Brown, 2009).

According to the Centre for Ecology and Hydrology, land converted to cropland in the UK will, very broadly, emit up to $20tCO_2/km^2$. However, there are many outlying soil types in the UK where this could be closer to zero or even up to

40tCO_2/ km^2 (Hallsworth and Thomson, 2011). Despite this uncertainty, this is useful as an indication of LUC in this study. Furthermore these estimates are in line with the 19.16tCO_2/km^2 that the BioGrace[10] calculator shows as defaults assuming the conversion of 'Native Forest' into 'Cropland in Europe'. This calculator was developed by Intelligent Energy Europe to provide harmonised calculations of biofuels GHG emissions in Europe and follow the EU RED calculation rules.

Data taken from Figure 4.6 (which shows the foam and natural fibre mattresses were responsible for 0.0203km^2 and 0.0389km^2 respectively), is combined with LUC of 20 tCO_2 per km^2. This results in the foam and natural fibre mattresses emitting through LUC potentially an additional 406kgCO_2eq and 778kgCO_2eq respectively. Given that the mattresses themselves were estimated to emit only 243 kgCO_2eq and 264 kgCO_2eq respectively, omitting LUC from the LCA could underestimate GHG emissions by over 100%. If the feedstock has come from land the previous use of which was already for crops, no additional LUC emission will be emitted, though there may be some iLUC.

The uncertainty around such broad brush calculations makes the results relatively unreliable and they could conceivably be an order of magnitude out. The location of the LUC is a key factor. The calculation presented here assumes LUC in the UK. If LUC was to take place in locations where primary forests exist which store vast quantities of soil and biomass carbon, this value could be expected to increase significantly. Conversely, if existing cropland was used very little, then additional GHG emissions may be expected. Complications surrounding chains of custody and origin, previous land use, specific climate conditions and differing soil types make

[10] www.biograce.net

calculating LUC a serious cause for concern especially given its potentially large influence on the overall LCA. However, the degree of specific experimental data collection that would be needed to ensure robust estimates of GHG emissions caused by LUC often mean that default values carrying large errors and averages are cautiously used.

In summary, the biomaterial mattress, although shown to have potentially lower GHG emissions than the petrochemical mattress, could actually be seen as the most polluting option in terms of GHG if LUC is included and if forest land has been converted into cropland to grow the feedstock. However, the degree of uncertainty in such claims is high. It is nevertheless useful to know the relative impact of any LUC that may take place from biomaterials and petrochemicals prior to stating which is the least polluting.

4.2.10. **Implications**

The GHG emissions of a natural fibre and foam mattress have been compared using a consequential integrated hybrid LCA. Data was taken from well-known process LCA databases and combined with industry average IO emissions data provided by the Office of National Statistics. The findings show that the way in which the mattresses are disposed of is a very influential part of their life cycle emissions. The least polluting hierarchy was found to be reuse and recycle, then energy recovery from ethanol conversion, and incineration with CHP, and finally, the most polluting and still the most common option, landfill. The biomaterial product caused much lower emissions than the petrochemical alternative the further up this hierarchy that disposal occurred, with the greatest improvement compared to petrochemicals being in the recycling and reusing scenarios. There was only a marginal difference when the mattresses were sent to landfill.

There are three major implications of this research. The first relates to claims made that biomaterials are low carbon alternatives to petrochemicals; the second focusses on the way waste is treated as the dynamics of product providence shift to become more bio-centred; and the third relates to methodologies for measuring GHG emissions in biomaterials.

First, this research shows that it is disingenuous to assume that biomaterials are necessarily low carbon alternatives to petrochemicals. It is more accurate to state that biomaterials are only significantly lower carbon alternatives to petrochemicals either if their disposal first incorporates recycling or if they incorporate reuse and conversion to ethanol. If biomaterials have conventional disposal fates of landfill and incineration they may not be considered low carbon alternatives to petrochemicals.

Second, in moving to a bio-based economy nations should rethink their mass waste options to account for the additional organic waste that will make up a greater proportion of their mass waste streams. The disposal of all waste, including mattresses, has developed over decades based on a waste stream with high petroleum content in addition to high contents of metals, paper and glass. This has meant that landfill has prospered despite its higher GHG emissions, though more recently, growth in recycling and incineration has been seen and further diversification of waste’s fate is needed to provide low carbon biomaterials. There has been little value in pursuing alternative disposal options such as ethanol conversion and reuse in petroleum based world but in a bio-based economy the dynamics of waste disposal will change and alternative scenarios may be capable of generating more value, profit and resource security.

Finally, the complexities in measuring disposal impacts are a major challenge for the biomaterial industry since this can make or break their sustainability argument, but it is nevertheless a route worth pursuing. The commonly used cradle-to-gate LCA is not a suitable assessment for biomaterial products and may be providing a bias towards petrochemical products. Ensuring sourcing is sustainable is already high on the agenda of conventional sustainability assessments which have boomed in recent years, yet end of life scenarios have often been an notable omission. In a bio-based economy, the way sustainability assessments are approached must change to incorporate disposal if the most favourable outcomes are to be understood. End of life scenarios should be raised in the agenda as there will be increasingly profitable and efficient disposal options that were not previously conceivable or appropriate in a petroleum based economy or in the biofuels market. In a bio-based economy the importance of where a product ends up should be seen to be as important as where it has come from.

4.2.11. **Unanticipated results**

Local sourcing of goods was not found to cause particularly significant GHG emissions reductions to the mattresses, nor was using materials with low biodegradability, compared to the reductions achieved by the end of life scenarios. The most cost effective method of reducing GHG emissions was not clear, though a good market already exists for recycling and incineration with CHP. In addition, profit may be achievable through refurbishment and ethanol conversion when these industries mature. There is great difficulty in comparing biomaterials within a composite product to foam alternatives. However, when the 'noise' of supplementary components are removed, it becomes much more apparent that biomaterials have much lower GHG emissions than petrochemicals and that the waste hierarchy is much less effective at reducing emissions from petrochemicals.

Looking into more issues affecting sustainability than simply GHG emissions muddies the comparison further, showing that petrochemicals have virtues in using less water and land and causing less risk of eutrophication. The difference in their potential to cause acidification is less distinct and biomaterials cause less abiotic resource depletion. The higher pollution rates and land use change linked to biomaterials becomes even less apparent if key polluting feedstock like cotton, wool and viscose are avoided. Thus, the type of biomaterial may also be key to its environmental sustainability in addition to its disposal scenario.

4.2.12. **Limitations**

Data quality has been explored extensively in this research and uncertainty in the calculations has been shown to be high for a range of reasons though these have not been found to jeopardise the conclusions in the research. Sensitivity analyses throughout have attempted to place the uncertainty into context, though it is impractical for any LCA to completely remove uncertainty.

4.2.13. **Concluding comments and future research**

Areas that may be explored further in terms of LCA methodology include the identification of appropriate methods of allocating credit for recycling and ways to adequately capture the additional impacts of LUC. The paper also shows the importance of selecting an appropriate functional unit, .

The economics of disposal is an area where further research may be required. This may also help prioritise investment into new disposal infrastructure more suited to bio-based economies. This research notes the lack of current infrastructure for ethanol conversion which may be a problem for bio-based economies. Identifying opportunities and developing technology for demonstration scale biorefineries for ethanol conversion will be a useful step.

Finally, this research suggests that the mind-set of industry and policy makers may need to change so that they require end of life scenarios to be captured within their sustainability assessments and internal operations. A first step to achieving this may be to understand the readiness and appetite of the current biomaterial industry for such changes. This forms the basis of the subsequent piece of research in this thesis, presented in Chapter 4.

5. Industry views on sustainable biomaterials and end of life scenarios

5.1. Introduction

As discussed in Chapter 4, recovering biomaterials reduces their life cycle GHG emissions, but the rate of biomaterial recovery is below that of other materials. Currently, the focus of industrial sustainability assessments and the claims made by companies striving to be ‘green’ is often on the sourcing of biomaterials, for example, addressing how to minimise land use change impacts and regulate GHG emissions caused in agriculture. Omitting the importance of disposal emissions is a large oversight considering that the policy context in the EU promotes the bio economy, and growth in bio-based products is predicted to be significant (Vandermeulen et al., 2012).

Despite the benefits that a potential shift towards a bio-based economy may bring little evidence exists to suggest that the issue of sustainable disposal of biomaterial products is being taken seriously by the industry, government or academia. The next piece of research presented in this thesis investigates this hypothesis by investigating the opinions of industry stakeholders.

A paper was produced from this research which was published in 2013 in the journal Waste Management entitled Achieving sustainable biomaterials by maximising waste recovery; this is presented in Appendix II of this thesis where the full methodology, results and conclusions can be viewed. In summary, initially fourteen semi structured industry interviews were undertaken with biomaterial industry stakeholders (mainly managers) to gauge the status quo attitudes on how waste is viewed alongside other priorities. Interview findings were analysed using coding of interview responses and semiotic clustering to develop themes, patterns and key issues and look at how the characteristics of stakeholders influenced opinions. There was a high degree of uncertainty on how best to deal with waste, some consensus on needing more information and more demand for waste biomaterials before anything could be done and a lot of disagreement on the importance, scale and responsibilities associated with biomaterial waste. From the discussions however three clear policy options emerged to describe how intervention in the market could assist greater recovery of biomaterials with a view to reducing GHG emissions, these were (1) do nothing; (2) develop legislation; and (3) develop certification standards.

An expert focus group made up of senior members from waste, biomaterial and sustainability organisations then discussed these policy options. The experts considered that action was required, rejecting the first scenario. No preference was apparent for scenarios (2) and (3). Experts agreed that there should be collaboration on collection logistics, promotion of demand through choice editing, product ‘purity’ could be championed though certification and there should be significant investment and research into recovery technologies and infrastructure. These considerations were finally incorporated into the development of a model for policy makers and industry to help increase biomaterial waste recovery. This model concluded that maximum waste recovery of biomaterials will require a multi-pronged approach. Conventional waste legislation such as bans and taxes were deemed inappropriate due to the diversity and complexity of biomaterial products. Instead a combination of increasing demand for product purity (via government procurement), and potential purity certification standards coupled with greater investment in research and infrastructure were preferred. Significantly the readiness of consumers for bio-specific strategies was questioned and it was thought that producers needed greater clarity on what options were available for different biomaterial products. Some final thoughts on the research methods, results and implications of this research which were not included in the paper owing to brevity are presented here.

5.2.Further reflections on methodology, results and limitations

The journal article refers to five documents that were sent to stakeholders; 1) a concept note for prospective interviewees, 2) a post analysis summary of the interviews, 3) a 1 page summary of the interview results for the experts, 4) a concept note for the experts, and 5) a summary of the focus group outputs. These are presented in the Appendices III to VII.

This study was conducted as a follow up to Glew et al. (2012) and intended to obtain a snap-shot of how the biomaterial industry viewed and addressed disposal emissions. Contributors came from a range of backgrounds though of course, as is common in survey and focus group based research more stakeholders could have been contacted. The response rate was lower for the focus group than it had been for the interviews: 26% compared to the 34%. Limited time, the fact that some companies have a policy not to participate in research, and because snowball sampling eventually draws dead ends, means there is a possibility that the sample did not capture all the issues relating to the research. This lower response rate is perhaps to be expected due to the necessity to travel to York to participate in the focus group. In total, nine experts attended the focus group which is a useful size for data collection in exploratory research aimed only at expert stakeholders rather than larger confirmatory social studies (Billson, 2006, Tang and Davis, 1995). Steps were made in an attempt to encourage high calibre experts and a maximum of £100 contribution towards travel expenses and an invitation to tour the Biorenewable Development Centre at the University of York after the event were used as incentives. Limited resources meant a second focus group was not organised in London (where more experts were located), so it was therefore not possible to give those who could not make the trip to York a chance to air their views. This may have resulted in the unique views of some experts being missed.

Consumers were not consulted in this research yet it is known that consumer behaviour influences the quantity of material that is recycled and the success of schemes to reduce waste (Boer, 2003, Coggins, 2001, TLC, Last Accessed 2012). The reason for this was that the research identifies the structural impediments and opportunities within the industry such as transparency and infrastructure problems. It did not set out to design or compare consumer friendly ways of encouraging more biomaterial recovery. In addition the choice to omit consumers from the research sample was also justified to an extent by the responses gathered from industry and from experts, who suggested the issues are currently not palatable for consumers and that it would be better to consult consumers only when industry themselves had greater knowledge of the problems and were better able to articulate the issues to consumers.

In summary, steps were taken to maximise the results' robustness and the response rates and sample selection techniques used in this study are not likely to have adversely influenced the validity of the interview findings or focus group conclusions.

5.2.1. **Data collection**

Remote forms of data collection (e.g. questions via email) were not used in this research despite their potential to reach a wider audience. One reason for this is that they do not enable physical gesticulations and vocal intonations to be noted and so in depth analysis is not so easily undertaken (Gillham, 2005). The interviews were audio-recorded where permission was granted and so there was the possibility to use discourse analysis software to interrogate the data according to key words and themes. This software was ultimately not selected as a tool partly because not all the interviews were recorded and also because the backgrounds of the respondents were very diverse meaning the sample did not share a common terminology or 'norms'. This limited the ability to compare the frequency with which key phrases are mentioned. Content analysis was used instead, which gives the researcher more flexibility in developing themes and determining how relationships and patterns are drawn (Collier and Scott, 2010).

5.2.2. **Coding**

In the research, coding was the basis upon which patterns and conclusions were drawn. However there are some dangers that should be considered when using codes in this way. The use of codes is illustrative and useful as a tool but codes themselves are not explanations (Manor-Binyamini, 2011). The process of coding is subjective and so there is the chance that there may be some misinterpretation of the comments. This may especially be the case where certain trends emerge at a later stage and analysis must be redone for the earlier interviews and may not be as fresh in the researcher's mind. This was overcome to an extent by relaying summaries of the results to the interviewees and experts for comment, though only two participants chose to respond. In addition, note taking is not fool proof and certain emotions, feelings or meaning behind some comments may be missed or some comments may be omitted entirely. Coding was used in this research with these limitations in mind and so it is hoped that any errors were minimised.

5.2.3. **Unanticipated results**

The policy scenario ‘do nothing’ was a relatively well represented view in the interviews with industry stakeholders which confirms there may be a general reluctance to taking on-board environmental advice or improving performance regardless of a company’s size or financial performance (Hitchens et al., 2005). Yet ‘do nothing’ was hardly discussed by the experts who were all keen that some form of intervention would be beneficial. There may be some form of ‘NIMBYism’ taking place here where it is easier for those not directly affected to favour some form of interference.

The unanimous calls for more research showed that companies and to some extent, experts, are confused by the concept of life cycle impacts and instinctively wary that they could get things wrong. This indicates that fear is potentially stopping them from doing anything at all.

An overarching policy on biomaterials like that for biofuels was ruled out because the diversity of products made by biomaterials would make legislation too complex. Product-specific policy or certification could increase confusion since it will result in multiple simultaneous schemes. This was nevertheless still deemed preferable to an overarching policy because the practicalities of capturing all biomaterials within one set of criteria were thought to be unworkable.

5.2.4. **Implications**

The study has highlighted the lack of attention paid to disposal in the biomaterial industry and that there is a diverse range of suggestions for preferred interventions. At the same time, it has identified that there is a desire for ‘something to be done’. This may act as a spring board to give confidence to policy makers to engage in discussions with industry or encourage existing voluntary recovery schemes to expand their operations.

The challenges posed by efforts to increase waste recovery often focussed around ‘unknowns’. This implies that greater research on the practicalities and logistics of disposal options would be a good starting point from which to shift attitudes and practices to ensure more biomaterials are recovered. This is an area towards which investment should be directed in the first instance. This study has shown that attempts to expand existing waste legislation to encompass biomaterials may not prove successful and that policy makers should look towards specific policy or incentives that address supply and demand by improving purity of products, ease of recovery and developing nationwide collection infrastructure.

5.2.5. **Limitations**

Key limitations were discussed surrounding the response rates, sample coverage and the inherent problems of using coding. There is the potential that these will impact the robustness of the results, however, steps have been taken to minimise the influence of these limitations such that they are unlikely to significantly reduce the integrity of the findings.

5.2.6. Concluding comments and future research

Industry interviews revealed that high start-up costs and unknown risks, a lack of knowledge of potential opportunities, insufficient sorting and reprocessing technology, embryonic collection infrastructure, immature public understanding, and competing priorities, cause recovery rates to be low. Three possible policy scenarios to address this emerged from the data: 1) do nothing, 2) develop legislation and 3) develop certification. Nine experts from the biomaterial, sustainability and waste fields analysed these scenarios in a focus group. They suggested that intervention was needed which should target the supply and demand of recovery in addition to bringing about an industry consensus on collection logistics and infrastructure. They surmised that strict legislation or burdensome requirements should not be used and instead suggested promoting the purity and recycled content of biomaterial products through government procurement as a priority over targets or taxes.

The research raises several potential avenues for further research. The policy context for most UK companies often includes the rest of the EU, and in the case of those multinational companies who took part in this research, policy in many other parts of the world can affect their operations. This research is based in the UK and it is therefore unclear if the study were conducted in other countries whether it would yield the same results; there are studies which suggest contrasting business views on environmental issues between EU nations (Keil et al., 2002). It may be an interesting extension to this study to replicate the research in other countries to draw parallels and differences across nations and see if the conclusions are universal or specific to UK conditions.

One of the main suggestions from the research is that each biomaterial should have its own form of regulation or certification that addresses its disposal options separately to other products. There may therefore be some benefit in arranging additional product specific focus groups for each biomaterial. Work has already been started in this regard with the textiles industry and UK government (Morley, 2009), though no similar schemes exist for bio plastics or construction materials for example.

One of the main concerns from both industry and experts was that before any decisions on intervention could be made there needed to be greater transparency over which end of life scenarios were best suited to particular biomaterials. This lack of knowledge was a barrier for companies who in other respects were keen to invest in making their products ‘green’. Developing a tool to empower companies and policy makers to make informed decisions and recommendations on the end of life scenarios of their biomaterial products forms the basis of the next chapter.

6. Promoting low carbon disposal decision making in the biomaterial industry

6.1. Introduction

We have thus far established that disposal significantly affects life cycle GHG emissions of biomaterials more so than for petrochemicals, and yet industry and experts do not have sufficient knowledge or decision making tools. Consequently, disposal remains unaccountably low on the biomaterial agenda. This is a problem if a future bio-based economy also has ambitions to be low-carbon. This next piece of research in this thesis addresses this conundrum through the development of a decision support tool.

This tool ‘HELCA’ uses default values for the GHG emissions caused by various disposal scenarios and was tested on two hypothetical hemp biorefineries. HELCA was developed as part of a paper submitted to the Journal of Industrial Ecology in April 2013 entitled Biomaterials and environmental sustainability: Predicting disposal-stage GHG emissions via a harmonised life cycle assessment tool (HELCA), which is presented in full in Appendix VIII. The full methodology, justifications for decisions and data source selections as well as the final results and conclusions and implications are provided I the paper however these are summarised here.

HELCA is a 'process' based LCA tool rather than an 'IO' or 'hybrid' based assessment since it is intended for use by both biorefinery operators and policy makers. These stakeholders are not likely to have access to the costs of various disposal options which is necessary for allocating emissions in IO LCA, thus this extension was not included in HELCA. In addition IO data on GHG emissions can be several years out of date and are not yet available in refined form for each distinct disposal options for biomaterials such as anaerobic digestion, composting or ethanol production; therefore a process approach was preferred.

There is a precedent for harmonised LCA industry tools in European biofuels legislation since it allows disparate users the means to compare products and benchmark themselves in addition to undertake assessments to meet compliance requirements without needing to employ specialists, something not possible when using specific individual assessments that cater to specific situations using specific data sources and assumptions etc. A widespread tool can also raise the profile of the issue addressed. In this case harmonisation is used to quantify and rank disposal options according to their GHG emissions and cost effectiveness.

HELCA shows that Hemp biorefineries could operate carbon neutrally depending on their end of life scenarios. In general HELCA finds that the greatest GHG emission benefits are obtained through reuse and recycling in combination with incineration and CHP generation, a conclusion mirrored by earlier research. However it also identifies however that using energy onsite also reduces GHG emissions and anaerobic digestion with CHP, ethanol conversion and anaerobic digestion with electricity generation have the next largest GHG reductions above composting. Despite its emissions savings, incineration is among the most costly ways of reducing GHG emissions, along with composting. The net costs of anaerobic digestion are negligible and onsite energy production and ethanol conversion may provide net revenue.

The importance of having a tool like HELCA instead of general guidance that favours Incineration with CHP is that it has a ranking feature for the disposal scenarios. This allows policy makers to understand the consequences of different waste disposal options and thereby help ensure waste policy is GHG and cost efficient for example where preferred disposal options such as incineration with CHP are not available. This is often the case in the UK and other nations where district heating systems are rare, HELCA can identify the next best alternatives and quantify the likely difference in GHG emissions and cost.

HELCA addresses an essential gap in the current carbon accounting of biomaterials and may be a useful tool to promote the concept of complete life cycle or cradle to cradle GHG assessments. In general terms this may encourage firms to bridge the gap between design and disposal where possible. Often companies do not have any control over the waste disposal of their products, in this instance HELCA could be used by those companies who are conscious of their environmental impacts to inform their customers and promote particular low carbon disposal options to gain good will and provide evidence of their wider concerns.

Additionally those companies may also be able to adapt the design of their products to suit particular geographical regions where only certain disposal options are available. For example HELCA shows there is generally always a benefit to encouraging design for deconstruction so that products can be reused, but in areas where incineration with CHP dominates the waste infrastructure there may be little benefit in companies trying to make their products more suitable to anaerobic digestion by removing any impurities. The opposite may be true if companies were operating in regions where bioethanol from waste hubs were being established in the future, where there could be a particular benefit in ensuring products had few impurities and could more easily be suited to ethanol production.

The concepts of waste disposal, GHG emissions and cost effectiveness are pertinent themes for biomaterials which will become yet more important in future bio-based economies. The following section pulls out some of the discussion points of HELCA that in the interests of brevity were not included in the paper.

6.2. Further reflections on methodology, results and limitations

Although transparent and reputable sources have been used where possible (European Commission 1997) it may be argued HELCA was not the purpose for which these data have been produced and so do not share common assumptions and limitations, and that combining them in one study could be considered problematic. This is a feature of the embryonic stage of HELCA and should the tool be adopted by a wider policy or industry audience, new default data could be sourced specifically for this purpose.

Specifically the use of static data would need to be addressed. Should the tool be adopted as an online resource then regular updates could be issued, as is currently the case for other online assessment software including BioGrace for calculating life cycle biofuels or the UK government's Standard Assessment Procedure for producing energy performance certificates (EPCs) for houses. This would ensure all users were using the most up to date data but also crucially that they were using the same data."

6.2.1. Unanticipated results

Perhaps the most surprising results from HELCA were the cost effectiveness data. These are also perhaps the least robust answers that HELCA provides due to the variable nature of price information but also because of the use of defaults in generating the yield data. The upper and lower cost and yield estimates to some extent provide more clarity but also identify the imprecision in using cost estimates.

Conventionally the embryonic technology surrounding ethanol production from lignocellulosic material has meant that facilities are small scale and therefore economies of scale are not realised. The majority of infrastructure has been directed towards first generation biofuels which are now produced on semi mass scale and are deemed more profitable. However the question perhaps should not be whether first generation biofuels are more profitable than second generation biofuels (which they generally are) but rather, whether biofuel made from waste is more profitable than electricity made from incineration (which HELCA suggests it is). This indicates that investment in waste to ethanol conversion may be money better spent than the simplistic comparisons to first generation biofuels may suggest.

Composting is a relatively popular ‘green’ activity that councils are more or less applauded for undertaking, and any environmentally conscientious consumer will pursue at home. HELCA suggests that composting may be not be such a beneficial activity since although it saves some GHG emissions and is a low cost technology, it generates so little revenue that it is rarely cost effective. It can however offer other benefits beyond GHG savings if used domestically in e.g. allotment gardening. The implication of the poor cost effectiveness of GHG savings attributable to composting is that perhaps investment may be better directed to diverting compostables towards other end uses. This advice is of course determined by the default cost and GHG data input into HELCA and may not take into account more locally or informally incurred costs or benefits of composting. In addition, there are many social considerations that HELCA does not take into account which may ensure composting remains a core feature of the disposal landscape. This is a useful example of the importance of viewing the outcomes of HELCA in the context of the concept of harmonisation and the uncertainty that this brings.

6.2.2. **Implications**

HELCA confirms some commonly held beliefs surrounding the efficacy of end of life scenarios. For example, the well-known waste hierarchy defines recycling to be more favourable to energy recovery. Beyond this, more insightful interpretations can be drawn from HELCA. For example, where the national infrastructure is ill-designed to support mass scale CHP, as in the UK, HELCA can quantify the benefits of pursuing the next best alternatives. Countries without the means to employ mass CHP may decide to invest instead in ethanol conversion research and facilities or in encouraging producers to design products with ethanol conversion in mind rather than pursue incineration for power generation only. Thus, even though the most effective means of reducing GHG emissions may not be undertaken, a more practical or cost effective scenario may be prioritised using HELCA. HELCA is useful in that it can be interpreted in context to the specific circumstances of those using it.

The cost effective dimension of HELCA is a double edged sword. In one respect it makes a useful contribution to knowledge, highlighting that the most effective method of reducing GHG emissions may not always be the most cost effective. However, introducing cost estimates highlights the fallible nature of harmonised tools and the need to keep them up to date and relevant to the users. One further criticism with such a tool is it omits other sustainability criteria. Being low carbon is arguably important yet it leaves many other sustainability questions unanswered. Despite this, there are many other tools which focus on biomaterials' chain of custody (FSC), sourcing GHG emissions (RED and Roundtable on Sustainable Biomaterials) and even their equitable treatment of local communities and workers (Fair trade). These may be used in conjunction with HELCA to provide holistic sustainability guidance for biomaterials.

HELCA was developed in response to issues raised in the academic papers presented in chapters 3 and 4. Industry stakeholders highlighted concerns that they were not able to understand which end of life options were most suited to their biomaterials, and they craved an independent means of verification. HELCA attempts to provide this. In broad terms, HELCA confirms the existing wisdom that reuse and recycle are still deemed to be priorities and so strengthens the case for companies designing for deconstruction. This supports the conclusions drawn from the expert focus groups in the previous chapter, that improving purity and designing for deconstruction should be prioritised in biomaterials policy and research.

6.2.3. **Limitations**

Much has been discussed on the data limitations, largely tackling challenges of availability, accuracy and lack of specificity. These are a feature of the existing collective knowledge of research on biomaterials' disposal options and are not a direct criticism of HELCA. However, if the uncertainties are severe enough they may reduce the validity of any conclusions drawn through the use of HELCA.

Some methodological limitations of HELCA nevertheless remain. For example, the reuse and recycling options are limited to one life cycle, where in reality a product may be reused or recycled several times before its disposal. In addition, the recycling and reuse options do not include an option for downgrading the usefulness of the new product which is what happens in most cases. It is less common that recycling a product yields exactly the same utility as its previous use, which is the scenario assumed by HELCA.

The version of HELCA presented is limited in its ability to cope with more than ten co-products. This limit was used because the hemp biorefineries under investigation had nine and ten co-products respectively. However, it is feasible that more co-products may be produced. In this instance HELCA would need to have further co-product options added. This is not a limitation of the theory of HLCA which can cope with unlimited co-products, only a limitation associated with the version of HELCA presented in this thesis.

Perhaps one of the limitations with the most profound consequences on HELCA is the tool's inability to further refine the categories of biomaterials due to data limitations. The consequences of having broad categories like 'paper' is that the nuances between for example cardboard and newspaper cannot be assessed. Differences between two seemingly similar products may prove to be significant and when further refinements in the data are possible, this could make the claims of early HELCA versions like that presented in this thesis less valid. Until more nuanced categorisation is possible, the existing claims, albeit generic and broad brush, are the best that are available without breaking the mould of harmonisation and embarking on specific and costly investigations for each individual product and potential disposal scenario. Independent, non-harmonised LCA of biomaterials has not yet brought about significant action on disposal impacts in the biomaterials industry and has not been able to provide a common reference point from which to act.

6.2.4. **Concluding comments and future research**

HELCA would benefit from further refining of GHG data for different co-product types. This may include, for example, making distinctions between the energy recovery rates or recyclability of highly lignified textiles that are tough to break down versus those that are relatively easy, or between simple bio plastics that are designed to readily biodegrade and more complex ones that are not. Unfortunately data are not currently available to make such refined distinctions.

HELCA is based on EU data for GHG emissions of disposal options and cost. An interesting comparison may be to update the HELCA defaults relevant to other nations to see if the conclusions drawn are specific to Europe or if they are universally held. Similarly, future predictions on disposal emissions and market costs could be input into HELCA to make it a predictive tool, informing the industry not just on the status quo but of future trends. This may inform where investment in infrastructure and research should be placed to maximise future GHG emission reductions.

Undertaking trial runs of HELCA using real world industry and policy stakeholders may make an interesting study. Receiving feedback from potential users would help refine future revisions of HELCA, ensuring it was meeting the requirements of industry and policy makers and that it was using the most relevant data. The user interface could also be field tested and revisions may be made to make it more user-friendly. This research was intended as a proof of concept study only. Professionals from information technology and experts on disposal technologies who were authorities on the GHG emissions of different scenarios would be needed to inform these next phase of HELCA's development.

7. Conclusions

7.1. Introduction

The method by which a biomaterial is disposed of influences its carbon footprint, this research has shown it can make up around 10% of the overall emissions if they are sent to landfill but also that disposal options can make biomaterials effectively carbon neutral (through offsetting other consumption elsewhere).

This research has investigated the extent to which this is the case; how the industry incorporates consideration of this in their operations; and has outlined a way forward to aid decision making. It is a well-established principle, and a cause of concern, that bio-based products are not necessarily low carbon. Efforts are being made by many sectors to address this issue, though currently, despite some understanding of disposal's contribution to emissions, the majority of effort is put into measuring and reducing the impacts of producing and sourcing feedstock. It is the contention of this thesis that if future economies are to be more bio-based *and* low carbon it is imperative that the importance of disposal is better understood.

As far as was known at the time of writing, this is the first study to apply mixed methods research to biomaterial waste emissions and produce a decision support tool specifically for biomaterial waste disposal. This concluding chapter summarises the empirical findings of the study, and provides comments on theoretical and policy implications of the results, before outlining the limitations and future work that may complement that presented here.

7.2. Empirical findings and implications

The objectives outlined in the three published papers presented in this thesis build a strong case for promoting the importance of end of life scenarios to the development of truly low carbon biomaterials. The main findings and discussion points are presented below.

7.2.1. Disposal is important

The first paper presented in this thesis as chapter 3 concurs with previous research on the waste hierarchy showing that recycling, and energy recovery can reduce the GHG emissions of a product. Specifically, it compared various disposal options, initially confirming the idea that combining reuse and recycling was preferred, but on closer inspection, a more surprising outcome was revealed. Ethanol conversion, a currently underutilised mass waste treatment technology, theoretically had similar potential to more common (and controversial) energy from waste technologies such as incineration with CHP. Although there may be more practical barriers to realising the benefits of 'ethanol from waste' such as establishing a pure biomaterial waste stream and building factory infrastructure, it challenges the idea that contemporary waste solutions are ideally suited to the waste and emissions goals of more bio-based future economies. Overall, the findings imply that there are significant advantages in making use of biomaterials that can be realised in the disposal stage of their life cycle, and that the low carbon benefits of biomaterials compared to petroleum-based equivalents can only be fully realised by considered end of life options. Indeed, carbon emissions benefits from using biomaterials are at best marginal unless disposal emissions are factored in. This finding could be used as a rallying cry to affirm that biomaterials can be low carbon alternatives to petrochemicals and as a flag to attract more research and investment into disposal in general, and into up and coming biomaterial-specific waste infrastructure development.

7.2.2. **Disposal needs to be taken more seriously**

The objective of the second paper was to assess how the biomaterials industry recognises and deals with 'disposal'. Interviews showed that companies had disparate views depending on their size, the type of products they produced or sold their place in the supply chain, the needs of their customers, their ethos, and their previous experiences with legislation. This diversity of views means that attempts to define and resolve problems associated with disposal's profile via a single approach for the whole industry is likely to fail. In general it is fair to say that all the stakeholders viewed disposal's role favourably in the wider remit of sustainability. Specifically though, they identified many barriers that afford disposal only a back seat in the drive towards sustainability and a low profile in their company's psyche. Interest in disposal was often based on the direct benefits that a company could receive, and so it may be the case that only where a free resource or a green image is the reward will disposal be taken seriously in the absence of outside persuasion. Stakeholders were cautious about embracing disposal as a low carbon opportunity, partly because of inadequate awareness and understanding in the industry and amongst their customers but also, perhaps surprisingly, due to a lack of direction and advice from the authorities. Discussions with experts over appropriate interventions showed that multiple soft-touch approaches could be complementary if they stimulated demand (e.g. government procurement) and increased access to and the quality of the supply of waste biomaterials (via e.g. product purity incentives and investment in infrastructure). The size and complexity of biomaterials and the lack of existing facilities meant that more stringent legislation that has proved successful in other waste streams and which draws on targets and mandatory recovery schemes, were not deemed suitable for biomaterials. Although there was no consensus on how important disposal is or how to make real changes to the status quo, there was a palpable sense that disposal is important and currently neglected by the biomaterials industry. This could mean that without future intervention in terms of policy it may be difficult for the industry to address disposal emissions. Without this piece of the jigsaw being in place, it may not be possible to fully realise the potential carbon savings of biomaterials over petrochemicals.

7.2.3. Disposal needs to be better integrated into decision making

The final piece of research in this thesis developed a tool that could be used to help the biomaterials industry better understand how important a product's end of life is on its overall carbon footprint and help them better design products and processes with this in mind. The approach looked in detail at biomaterials deriving from one feedstock (hemp) considering both the range of co-products in a biorefinery, and how the overall GHG emissions changed when different disposal options were selected. The disposal options were ranked for each type of co-product in order to guide the industry on which end of life options are suited to which co-products. This approach and tool could allow companies to design their products to fit in with low carbon disposal options (by being easy to recycle) or advertise to customers what to do with products once they are finished with them so as to minimise emissions. In accordance with existing studies on the waste hierarchy, this research found that reuse and recycling should be favoured regardless of the product, since some additional benefit can be extracted prior to ultimate disposal thereby avoiding the use of virgin resources. Similarly, incineration with CHP, or incineration for power only, are the next best options in GHG terms ahead of AD, producing ethanol from waste and composting. When the cost effectiveness of the low carbon technologies is considered however, a different picture emerged. Unexpectedly, producing ethanol was found to be one of the most profitable ways of reducing GHG (behind reuse and recycle). This shows that conventional wisdom on how to reduce carbon emissions in waste disposal may be turned on its head when considering biomaterial-only waste streams. This research suggests that in future bio-based economies, more investment into infrastructure to support novel biomaterial-specific treatment of waste seems to be the most cost effective (and therefore most likely) route to minimising carbon emissions.

7.3. Theoretical and methodological implications

7.3.1. Hybrid LCA

The first paper presented (chapter 3) uses an integrated hybrid LCA as opposed to a process or IO LCA to compare the GHG emissions of natural fibre and foam mattresses. Hybrid LCA is an emerging tool being popularised by the ability to perform specific process style assessments with the complete IO system boundaries. In this instance, the hybrid LCA reported around a quarter more emissions than the process LCA. The implication is that in biomaterials research, as in other areas, the continued use of process LCA may be resulting in an under-reporting of GHG emissions. This is a concern where accuracy is paramount, and so there may be some advantage in hybrid LCA becoming the norm to replace process LCA.

7.3.2. **Mixed methods**

Employing mixed methods and engaging a range of stakeholders in qualitative research results in a more thorough investigation and allows areas of uncertainty to be raised. The expert focus group in the second paper dismissed the idea of ‘do nothing’ and had this been the only research method employed or group consulted, one could surmise that there was consensus on the need for intervention. Opposition to intervention was however heartily raised in the stakeholder interviews which provided an insightful addition to the research narrative. The insights gained from this work highlights the usefulness of mixed method approaches in biomaterials research. Similarly, the mixing of qualitative and quantitative research has been shown to be useful, because without the insights gained from the interviews and focus groups, the need for more transparency in assessing which disposal options were suited to which biomaterial could not have become apparent, and the focus of the final decision making tool could not have been set with as much confidence to its usefulness. This implies that when decision support tools or perhaps even LCA in general are being used, it is useful to get an understanding of the needs of the potential user community.

7.3.3. **LCA harmonisation**

There is a precedent of harmonisation in LCA to be used where industry standards and guidance are a key goal, where simplicity is required, and where it will be used on a large scale, as in the case of the EU RED legislation. The research in this thesis appears to be the first to expand the principle of harmonisation to the case of disposal to allow companies to understand how their products will be affected by particular end of life scenarios. This was carried out because there are many ways in

which the emissions from recycling or energy recovery can be allocated in LCA and because data is currently so scarce that default data is required. Standardisation has the advantage that all stakeholders are then ‘singing from the same hymn sheet’.
The wider application of harmonisation raises questions as to the direction and purpose of LCA in general. Restricting content detail and methodological freedom to expand the audience and usefulness of LCA does so at the cost of accuracy. There are costs and benefits to both harmonisation and non-harmonisation approaches. This research shows that there can be a niche for both, as long as they are properly focussed and interpreted.

7.4. Policy implications

7.4.1. Climate change

This research presented here has quantified the impact of disposal on cradle to grave carbon foot printing of biomaterials, and has demonstrated that in some instances, the disposal choice can help biomaterials to be ‘carbon neutral’, potentially reducing emissions by 100%. Since GHG emissions are such a key issue in the acceptability of bio-based products, it seems imprudent to not incorporate them into policies that aim to mitigate climate change. Comprehensive policy should widen the focus beyond the sourcing of bio-feedstock to also incorporate disposal considerations.

7.4.2. Waste

Waste legislation tends to address MSW or sub-classifications based on hazardous material or product types, for example, cars or packaging. It may not be appropriate to extend this existing type of policy to biomaterials since they are so often just one component within a larger product and because they are so diverse and numerous.

Biofuels targets may well be extended to regulate the emissions resulting from sourcing all biomaterial feedstock but this would not help resolve the issue of disposal emissions. Moreover, extending target-based biofuel policy to encompass all biomaterials may not be appropriate as it is difficult to say who is responsible for the waste disposal and would therefore be unmanageable. Specific waste policy for biomaterials may however be possible if it takes a different form. This research suggests it could take a softer approach to encourage a market solution.

7.4.3. Research

Developing a decision support tool to assess the disposal GHG emissions for biomaterials highlighted that there is currently a dearth of detailed data on which to base decisions. Presently, data on waste refers to either MSW or broad material classifications like plastics, paper or textiles. The research has hinted at the unique potential of different products to reduce GHG emissions according to their disposal so that for example, cardboard can be differentiated from paper. Refining data for waste GHG emissions may be more important for biomaterials and future bio-based economies than for existing MSW composition and it may be that in order to make competent policy decisions, more refined waste GHG data will be needed.

7.5. Future research and limitations

Opportunities for future research and the limitations of the work presented have been identified throughout this thesis. In general terms expanding this research to explore other impacts of sustainability beyond GHG emissions may broaden the understanding of how important disposal is in the biomaterials industry. As identified in the literature review a particularly pertinent issue regarding agricultural products such as biomaterials is that of LUC and iLUC since their impacts are inherently diverse and difficult to regulate. While these challenges are already being addressed by organisations like the RSB, understanding their relationship with disposal options requires further attention.

Making overarching claims about policy and technologies that may be favourable in reducing the GHG emissions of biomaterials is inherently difficult since biomaterials are so diverse. For example, the barriers and solutions for waste bio plastic may not be the same as for used timber. There may be some merit therefore in comprehensively investigating one material at a time and designing policy or recommending technology in a more refined manner. When more data are available, this may be a useful undertaking. Another barrier blocking action from the industry was the uncertainty in ownership of waste emissions. From the interviews and stakeholder group discussion presented in the thesis, there emerged a clear desire for 'something to be done' and so one practical way to approach this may be to establish where the boundaries of ownership lie, using this as a basis for future policy and investment.

There is much debate on carbon footprinting methodologies, discussions over the merits and problems of process, IO or hybrid LCA and conflicting opinion over the use of harmonisation by those undertaking assessments. It is generally accepted that the aim of the assessment directs the choice of methodology. Investigations into the perceived role of carbon footprints according to different stakeholders may therefore be a reasonable next step. Understanding what the general public, consumers, politicians or business leaders want from a carbon footprint may be a means by which to inform where to allocate resources and effort regarding different methodologies.

7.6. Conclusion

The method of disposal of biomaterials can be crucial to validating their low carbon status. Currently there is a lack of industry awareness of the importance of disposal, as well as little consensus on how to address this legislatively or logistically or how to capture this in assessments. All of these issues, in addition to the shadow cast by interest in feedstock sourcing, ultimately diminish disposal's profile in the carbon footprinting of biomaterials. The significance of this study is that it exposes the key barriers and proposes future solutions in terms of possible policy direction and carbon footprinting assessment methodology. The work also identifies that existing policy for waste (taxes and bans) and biofuels (GHG targets) may not be suitable for biomaterials and that there are gaps in existing waste infrastructure (ethanol conversion facilities) which may be revealed if confronted with biomaterial-based waste profiles in more bio-based economies. The work carried out here suggests how disposal could be captured in carbon footprint assessments using user-friendly harmonised support tools and reveals the disposal techniques that rank among the most efficient in reducing GHG emissions (reuse, recycle and incineration with CHP) as well as the most cost effective (reuse, recycle and ethanol from waste). This raises the idea that the conventional waste hierarchy may be slightly different for biomaterial and MSW waste streams. These results may inform companies wishing to design products with low carbon disposal options in mind and policy makers wanting to ensure a future bio-based economy may also be a low carbon one. Critically, ill-informed decisions regarding biomaterials can result in higher GHG emissions than petrochemical alternatives. This was shown to be the case when problems surfaced over the sourcing of first generation feedstock for biofuels, and it is imperative that the same mistakes are not made regarding the disposal of biomaterials.

Appendices

Appendix I, Glew et. al., 2012, How Do End of Life Scenarios Influence the Environmental Impact of Product Supply Chains? Comparing Biomaterial and Petrochemical Products, Journal of Cleaner Production

Appendix II, Glew et. al., 2013, Achieving sustainable biomaterials by maximising waste recovery, Journal of Waste Management

Appendix III, Concept Note for Potential Interviewees

Appendix IV, Post Analysis Summary for Interviewees

Appendix V, Concept Note for Potential Experts

Appendix VI, Interview Results Summary for Potential Experts

Appendix VII, Summary of Focus Group Findings for Experts

Appendix VIII, Biomaterials and environmental sustainability: Predicting disposal-stage GHG emissions via a harmonised life cycle assessment tool (HELCA), as submitted to Journal of Industrial Ecology

Appendix I

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How do end of life scenarios influence the environmental impact of product supply chains? comparing biomaterial and petrochemical products

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ABSTRACT

In this research natural fibre (biomaterial) pocket spring mattresses are shown to emit marginally less greenhouse gasses (GHG) than foam (petrochemical) pocket spring mattresses. However, when end of life scenarios are considered, the results suggest much larger GHG emission reductions for natural fibre than foam mattresses. Refurbishing natural fibre mattresses and reusing the springs, coupled with recycling the waste components, can reduce GHG emissions by 90% compared to sending the mattresses to landfill. Incinerating mattresses via combined heat and power plants for electricity production and converting the waste textiles to ethanol are also shown to reduce GHG emissions, though to a lesser extent than refurbishment and recycling. Mattresses are normally disposed of via landfill however designing for reuse and recycling, coupled with supportive policy and legislation, may encourage more natural fibre mattresses and recycling. Such changes could save between 210 and 2092 thousand tCO_2-eq in the European Union annually.

1. Introduction

1.1. Biomaterials

Given that petrochemicals are polluting, have potentially unstable production and are ultimately finite it is not surprising that alternatives to petrochemicals are being sought globally by industry and governments. Scarlat and Dallemand (2011) summarises initiatives that encourage 'biomaterial' alternatives to petrochemicals including legislation and certification schemes. Biomaterials broadly include plastics, fibres, oils, chemicals, and fuels; all refined from plants and animals (Cherubini and Ulgiati, 2010). Biomaterial markets are growing and currently contribute €2 trillion to the EU economy annually (Geoghegan-Quinn, 2010).

Using biomaterials has in some instances been demonstrated to cause lower greenhouse gas (GHG) emissions than using equivalent petrochemicals (González-García et al., 2010), though there are concerns regarding land use change (LUC) caused by growing biomaterial feedstock on land which has high carbon stocks or on areas that have significant social value (Searchinger et al., 2008). In addition, biomaterial feedstock may need irrigating and often requires fertilizer and pesticides in order to achieve economically viable yields. Competition with food crops is also a major concern especially in developing countries which can lack robust social and environmental safeguards (Tilman et al., 2009). Sustainability assessments can help quantify these impacts, and in doing so, identify situations in which biomaterials demonstrate benefits over petrochemicals.

1.2. Sustainability assessments

Many sustainability assessments on biomaterial feedstock have been undertaken in recent years and reviews have been performed which conclude that their sustainability impacts can range hugely; some performing much better, others several times worse than their petrochemical counterparts (Von Blottnitz and Curran, 2007). The majority of assessments address only GHG emissions, though this could be because assessments often only focus on biofuels for which energy (and hence GHG emissions) are an appropriate unit of measurement. Fahd et al. (2011) used a multi-method assessment to compare energy efficiency, acidification, economics and land use impacts, and found that producing biochemicals along with biofuel is more economically competitive and energy efficient. Despite this advantage, relatively few studies address biomaterials beyond biofuels. This research therefore occupies an important and growing niche in sustainability assessments.

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Assessing all three dimensions of sustainability (environmental, societal and economic) within one single study can be problematic especially regarding issues of subjectivity and data collection (Robert et al., 2002). This research paper therefore focuses only on quantifying GHG emissions as its unit of measurement, using life cycle assessment (LCA). LCAs record the GHG emissions, among other potential impact categories, caused by extracting raw materials as well as processing, manufacturing, transporting and disposing of a product. They are used by industry to identify hotspots in supply chains where efficiency savings can be targeted. The results of LCA are also useful for governments and consumers in the form of setting legislative targets and providing certification for the environmental impact of products. 'Attribution' and 'consequential' are types of LCA that describe a product's current impact and its potential impact under different assumptions respectively (Finnveden et al., 2009). This study uses the latter.

Despite having an international standard (ISO 14040-44), LCA give notoriously inconsistent results; Cherubini et al. (2009) show that different results can be achieved from a single LCA depending on the method and assumptions used. Excellent critiques on the uncertainties in data quality and dealing with allocation methods can be found in Chiaramonti and Recchia (2010). There are three LCA techniques. The first is the 'Process LCA' which uses LCA databases that contain data on the average emissions for commonplace raw materials. These are then added together to provide the total emissions for a product and give good indications of specific products' emissions, however, it can be difficult to find appropriate data.

The second technique is the 'Input Output' (IO) LCA. This LCA applies an emissions factor to economy-wide economic data to give average emissions per unit in monetary value spent by each sector in the economy. This ensures IO LCA includes indirect emissions that can be missed by process LCA. They are also much quicker to undertake though they can only give general indications of the emission hotspots of product types, and it is more difficult to compare similar products using this technique. The third technique is the 'Hybrid LCA' which combines both Process and IO approaches, making the assessment more holistic and yet still specific to the product (Wiedmann et al., 2011). The research presented here uses a hybrid, consequential LCA.

LCA is commonly 'cradle to gate' and therefore omit end of life stages unlike 'cradle to grave' assessments as shown in Fig. 1. However, ignoring end of life emissions makes it difficult for policy makers and industry to understand the overall GHG emissions of a product and therefore to target appropriate policy and legislation to further reduce emissions. This research is the first to investigate how end of life scenarios influence the GHG emissions of biomaterials and petrochemicals using hybrid LCA. The aim of this paper is to undertake a comparative assessment between (biomaterial) natural fibre and (petrochemical) foam pocket spring mattresses using a hybrid LCA to identify how end of life scenarios impact GHG emissions.

2. Materials and methods

2.1. LCA product: mattresses

Pocket spring mattresses have been selected in this LCA as a model product as they can be made from petrochemicals (foam) or biomaterials (natural fibre fillings). Over 35 million mattresses are sold in the EU annually, most of which are made from combinations of springs, foam and natural fibres; very few mattresses are foam free (AFNOR, 2006). The UK mattress market is worth between £388 and £776 million annually (Centre for industrial studies, 2010; GFK, 2010). The mattress industry is a useful target for reducing GHG emissions since it is a large sector with several producers that have already begun to use 'green' marketing. Pocket spring mattresses also have similarities with other furniture industries such as car seats, office chairs and domestic three piece suites for which transferable lessons may be learned.

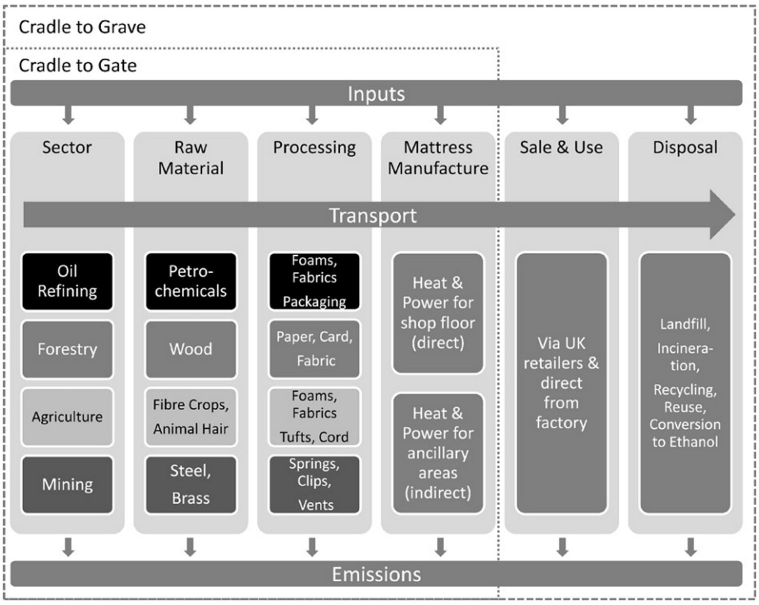

Fig. 1. System boundary for pocket spring mattress LCA.

2.2. LCA assessment

The assessment used in this paper is consistent with the four ISO standards for LCA:

i) scope and goal definition;
ii) lifecycle inventory (LCI) which quantifies the inputs;
iii) inventory analysis (LCIA) which converts the inputs to emissions; and
iv) interpretation of results.

2.2.1. Scope and goal

This LCA investigates the GHG of memory foam (petrochemical) and natural fibre (biomaterial) mattress under twelve end of life scenarios (see Section 2.3 for list of scenarios). Data is taken from a manufacturer in Yorkshire, UK, that constructs handmade beds and pocket spring mattresses using compressed air powered tools, and has machines on site to make springs and frames and for carding natural fibre fillings. Contribution reports were provided to identify the quantities of materials that make up the mattresses. Records of the heat, power and transport diesel consumed during manufacture were provided for the calendar year 2010. The steel, foam, fabrics, fillings and other mattress components are sourced from the UK and overseas, mainly China and Europe. The LCA is a cradle to grave assessment and has the system boundary shown in Fig. 1.

Functional units are used in LCA to ensure like with like comparisons and in this study the mattress functional unit considers six variables; 1) durability, 2) lifespan, 3) quality, 4) comfort, 5) costs and 6) size. All mattresses sold in the EU are made to the standard BS 1957:2000 (British Standards Institution, 2000) and are therefore deemed to have equal durability. Mattress lifespan nevertheless remains subjective. According to Deliege et al. (2010) in their European Commission (EC) report, mattress lifespan could be between 3 and 15 years, though 10 years is a more reasonable approximation. This study therefore also adopts 10 years as an appropriate lifespan. Mattresses are not standardised by quality or comfort, making like for like comparisons difficult. This is overcome to an extent by comparing the same type of mattress: pocket spring, and by selecting mattresses of roughly the same price – a high quality price bracket of £1500 per mattress. The mattresses to be considered are also the same size: 2000 mm × 1500 mm × 270 mm. In order to scale up the results to any size of mattress, the functional unit will consider 1 m^2 of mattress. The functional unit is therefore 'm^2 of pocket spring mattress over 10 years in the £1500 price bracket'.

2.2.2. Lifecycle inventory (LCI): input–output and process data

Two main types of LCI and data are used: the process LCI (primary and secondary data) and the environmental IO data. Whereas the process LCI is used to systematically compute physical inputs and outputs of the mattress supply chain within the process LCA system boundary, the environmental IO data enables the completion of the analysis by enumerating upstream indirect inputs from outside the process system boundary. The use of IO data and environmental extensions has been widely applied in EIO LCA (Joshi, 1999). The Office of National Statistics (ONS) produces tables reporting the total annual financial transactions that each economic sector spends on another. They also report the total GHG emissions for each sector, thus, the GHG caused by each sector's spending on other sectors can be calculated. Allocating the cost of a product's components to various sectors therefore provides its cumulative IO GHG emissions.

This study uses the ONS data for the calendar year 2008 which has been expanded upon by Wiedmann et al. (2010) to include multiregional IO tables split between the UK and rest of the world. This disaggregation and refinement takes into account imports and exports since different nations' economies have different embodied GHG to the UK in their energy mix. The mattress IO GHG emissions are extracted by inputting the cost of each component into the EIO spreadsheet.

The process GHG emissions of each component was derived using the LCA database ecoinvent v2.2 (www.ecoinvent.org/database/) which gives cradle to gate GHG emissions per kg of material produced; this is presented in Table 1. To check that these data were robust, additional GHG values for each component were gathered from other literature. Where data was not available from ecoinvent the mean value of the other literature searches were used. In the case of mohair and cashmere where no data could be found, wool's GHG emissions were assumed. Although rearing sheep is often more intensive and takes place in different regions to rearing goats, the agricultural inputs required for animal husbandry and the extraction of animal wool are similar. The GHG emissions of composite materials such as woven cotton and viscose fabric were calculated by summing the GHG of the source materials according to the ratio of each material in the composite.

Electricity and heating emissions from the factory for the calendar year 2010 were allocated to each mattress type based on information provided by the manufacturers. The number of shop floor employees and supporting staff in administration and advertising for example, was 134 and 61, each working an average of 8.25 and 7.5 h for 232 days per annum respectively. The total electricity and gas consumption was 1,356,505 and 119,530 KWh respectively. Given that it takes 3.64 h to make the natural fibre mattress and 3.74 h for the memory foam one, plus an additional 2.14 h of supporting indirect hours for both mattresses, the combined GHG for electricity and gas use needed to make a natural fibre and memory foam mattress were calculated to be 37.78 and 38.45 $kgCO_2$-eq respectively. This was calculated using GHG data from the Carbon Trust (www.carbontrust.co.uk) and using Eq. (1):

$$M_{en} = \left(\frac{T_{en}}{T_{hr}}\right) * (M_{hr}\mathrm{In} + M_{hr}\mathrm{Di}) \tag{1}$$

Where:

M_{en} = Electricity and gas emissions allocated to 1 mattress
T_{en} = Total electricity and gas
T_h = Total hours
M_{hr}In = Indirect hours attributed to 1 mattress (receptionists, administration, etc)
M_{hr}Di = Direct hours attributed to 1 mattress (shop floor)

Transport emissions caused by delivering mattresses to shops were attributed to the individual mattresses based on their respective weights. The total consumption of diesel was 250,824 L in 2010 which according to Carbon Trust data causes 667,193 $kgCO_2$-eq. The total number of mattresses produced that year was approximately 55,000. Given that the natural fibre and memory foam mattress weigh 61.61 kg and 67.79 kg it is assumed that they are responsible for 11.55 and 12.71 $kgCO_2$-eq respectively. This was calculated using Eq. (2):

$$M_{tr} = \left(\frac{T_{tr}}{T_m}\right) \Big/ (M_{tw} * M_w) \tag{2}$$

Where:

M_{tr} = Transport emissions allocated to 1 mattress
T_{tr} = Total transport emissions
T_m = Total number of mattresses

Table 1
Embodied GHG of pocket spring mattress components (kgCO_2-eq/kg).

Material	Ecoinvent	Other literature						Mean
PUR foam	4.32	3[a]						3.66
Polypropylene fabric	1.98	2.7[a]	2.8[b]	1.9[c]	1.7[d]			2.22
Polythene (excluding HDPE and LDPE)	2.70	1.94[a]	4.1[b]	3.1[c]	2.8[d]	4.3[e]	4.66[k]	3.37
Steel wire	0.397	2.83[a]						1.61
Steel frame (excluding recycled content)	0.37	1.77[a]						1.07
Woven cotton	27.09	6.78[a]	10.1[f]					14.66
Hemp	n/a	1.6[g]	0.35[h]	0.31[h]	0.26[h]	0.33[h]		0.57
Wool	19.84	16.69[i]	6.58[i]	15.26[i]				14.59
Cotton fibres	3.07	1.28[a]	2.2[d]	6[d]	4.7[f]	3[f]	2[b]	3.18
Viscose	4.80	3.8[b]	9[d]					5.87
Card	0.66							0.66
Paper	0.85	1.5[a]						1.17
Brass	2.45	2.42[a]						2.43
Latex	2.63	1.63[a]						2.13
Tencel	n/a	1.1[b]						1.10
Silk	n/a	5.1[j]						5.10
Mohair	n/a							n/a
Cashmere	n/a							n/a
Horse hair	n/a	0.96[a]						0.96
Nylon	9.28	5.5[a]	5.6[a]	5.6[d]	5.5[d]	5.6[e]		6.16
Poly wrapping (extrusion film*, HDPE** & LDPE***)	0.52*	1.7[c]**	1.9[c]***	1.6[a]**	1.7[a]***			1.48
Cotton cord	14.34							14.34

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M_{tw} = Combined weight of both mattresses
M_w = Weight of individual mattress

2.2.3. Lifecycle impact assessment (LCIA)

The LCIA approach adopted in this research is the integrated hybrid LCA method which combines the process LCA inventories and Environmental IO data (EIO) (Suh and Huppes, 2005; Acquaye et al., 2011). Generally, the hybrid LCA method combines process LCA based on the PAS 2050 LCA methodology (BSI Group, 2008; Crawford, 2008) and Environmental Input–Output (IO) LCA (Hendrickson et al., 1998; Acquaye and Duffy, 2010).

Using lifecycle inventories, the process LCA can be defined by Eq. (3) as:

$$\text{Process LCA} = \sum_{i=1}^{n} A_{pi} \cdot E_{pi} \tag{3}$$

Where:

A_p: Inputs (i) into a product's (in this instance, mattress) supply chain such as raw materials, energy usage, transport, etc
n: Total number of input (i) into a product's supply chain
E_p: Emissions intensity for each input (i) into a product's supply chain

Environmental IO LCA on the other hand is evaluated using national IO tables combined with direct industrial emissions intensities. National IO tables are a matrix model of the economy describing the inter-relationship of all products and service requirements by all industries in an economy (Miller and Blair, 1985). Given that (A_{io}) represents the technical coefficient IO matrix (Ten Raa, 2007) and the identity matrix (E_{io}) the direct emissions intensities for each input–output industry and (y) the final demand then the Environmental IO LCA is defined by Eq. (4) as:

$$\text{Environment Input} - \text{Output LCA} = E_{io} \cdot (I - A)^{-1} \cdot y \tag{4}$$

$E_{io}(I - A_{io})^{-1}$: represents the total (direct and indirect) emissions intensities of each industry required to produce a unit of output. The integrated form of the hybrid LCA used in this paper combines the process and environmental IO LCA by integrating them using matrix algebra (Suh and Huppes, 2005) shown below in Eq. (5):

$$\text{Hybrid LCA} = \begin{bmatrix} E_p & 0 \\ 0 & E_{io} \end{bmatrix} \begin{bmatrix} A_p & -D \\ -U & (I - A_{io}) \end{bmatrix}^{-1} \begin{bmatrix} y \\ 0 \end{bmatrix} \tag{5}$$

Many forms of hybrid LCA have been used in literature (Joshi, 1999; Crawford, 2008), but the integrated hybrid LCA was used in this study because it allows for the combination of the matrix representation of the process LCA system and the EIO LCA system in a consistent mathematical framework (Heijungs et al., 2006). This allows for both the upstream and the downstream linkages between the two LCA systems (Suh and Huppes, 2005). It also ensures that double counting of emissions in the hybrid LCA is avoided by removing inputs from IO subsectors that are already provided by the process LCA (Wiedmann et al., 2011) for example 'growing of fibre crops'.

Additional IO subsectors are also removed if they are deemed to be inappropriate for the product, for example 'weapons and ammunition' which are included in the raw IO data due to aggregation of industries within subsectors. The LCIA presented in Table 2 shows the components listed in the contribution reports combined with the LCI emissions data from Table 1 to give the specific emissions for the mattress components. This is added to the transport and energy process emissions for each mattress as well as the EIO data in the final interpretation stage of the LCA which takes place in Section 3 Results and discussion.

2.3. End of life scenarios

The emissions calculated thus far are cradle to gate. To expand the hybrid LCA to cradle to grave, these are complemented with the emissions from different end of life scenarios. The waste hierarchy prioritises end of life scenarios thus: reduce, reuse and recycle, then incineration with combined heat and power recovery (CHP) and finally landfill (Häkkinen and Vares, 2010). Emerging technologies can convert waste biomaterials into ethanol and so create an additional scenario. This study therefore assesses twelve end of life scenarios:

i) Landfill
ii) Landfill and reuse
iii) Landfill and ethanol conversion
iv) Landfill, reuse and ethanol conversion
v) Recycle
vi) Recycle and reuse
vii) Recycle and ethanol conversion
viii) Recycle, reuse and ethanol conversion
ix) Incineration CHP
x) Incineration CHP and reuse
xi) Incineration CHP and ethanol conversion
xii) Incineration CHP, reuse and ethanol conversion

2.3.1. Landfill, recycling, incineration and ethanol conversion

Data published by the European Commission (2001) on the GHGs arising from materials that are sent to landfill, recycled, or sent for mass incinerated with CHP, are presented in Table 3. Converting lignocellulosic material into ethanol for transport fuel

Table 2
LCIA of pocket spring mattress components.

Component	Raw material	Carbon emissions ($kgCO_2$-eq)	Memory foam mattress (kg)	Natural fibre mattress (kg)	Memory foam mattress ($kgCO_2$-eq)	Natural fibre mattress ($kgCO_2$-eq)
60 g Spunbond	Polypropylene	1.98	3.93	3.93	7.78	7.78
Spunbond sheet	Polypropylene	1.98	0.15	0.00	0.30	0.00
Web backing	Polypropylene	1.98	0.06	0.06	0.12	0.12
Springs	Wire drawing steel	0.40	36.15	34.03	14.36	13.51
Hog rings	Wire drawing steel	0.40	0.24	0.24	0.09	0.09
Vertex clips	Wire drawing steel	0.40	0.01	0.01	0.01	0.01
Frame	Scaleless blue oil Hardened	0.37	1.13	1.13	0.41	0.41
375 g Flexbond	Polyethylene terephthalate	2.70	0.18	0.18	0.47	0.47
Memory foam	PUR	4.32	4.83	0.00	20.86	0.00
Contura foam	PUR	4.32	7.50	0.00	32.36	0.00
Latex foam	Latex	2.63	2.10	0.00	5.52	0.00
450 g Flexbond	Polyethylene terephthalate	2.70	1.34	0.00	3.63	0.00
70 g Spunbond	Polyethylene terephthalate	1.98	0.04	0.00	0.09	0.00
Cotton	Cotton fibres	3.07	0.00	9.01	0.00	27.64
Egyptian cotton	Cotton fibres	3.07	0.00	0.36	0.00	1.10
Hemp	Hemp	0.84	0.00	1.90	0.00	1.60
Wool	Wool	19.84	0.00	1.87	0.00	37.16
Mohair	Wool	19.84	0.00	0.95	0.00	18.90
Horsehair	Animal hair	0.96	0.00	0.24	0.00	4.71
Silk	Silk	5.10	0.00	0.24	0.00	6.44
Cashmere	Wool	19.84	2.13	0.00	42.18	0.00
Natural weave	39% Woven cotton & 61% Viscose	13.50	3.25	0.00	29.03	0.00
Natural soft	39% Woven cotton & 61% Viscose	13.50	0.00	1.55	0.00	14.44
Silk thistledown	Viscose	4.80	0.00	1.14	0.00	5.46
Labels/Cards	Paper	0.85	0.04	0.09	0.04	0.08
Brass vents	Brass	2.45	0.04	0.04	0.11	0.11
Foam corner	PUR	4.32	0.07	0.07	0.30	0.30
340 g Flexbond	Polypropylene	2.70	0.08	0.08	0.21	0.21
Cord	Yarn	14.34	0.05	0.02	0.66	0.33
Tuft	60% Wool & 40% Polyethylene	12.99	1.10	0.00	14.30	0.00
Wool tuft	Wool	19.84	0.00	1.10	0.00	21.84
Tape edging	Polypropylene	1.98	0.02	0.02	0.05	0.05
Kevlar tape	Nylon 6	9.28	0.02	0.02	0.22	0.22
Poly bag	Extrusion film	0.52	1.04	1.04	0.54	0.54
Corner protector	Cardboard	0.66	1.85	1.85	1.22	1.22
Bubble wrap	Polyethylene terephthalate	2.70	0.14	0.14	0.38	0.38
V21 foam	PUR	4.32	0.30	0.30	1.29	1.29
Total component [$kgCO_2$-eq]					176.53	166.43

Table 3
Net GHG emissions arising from disposal scenarios.

	Landfill ($kgCO_2$-eq/kg)	Recycling ($kgCO_2$-eq/kg)	Incineration CHP ($kgCO_2$-eq/kg)	Conversion to ethanol ($kgCO_2$-eq/kg)
Textiles	0.015	−3.169	−0.162	−2.587
Plastics HDPE	0.008	−1.761	0.31	n/a
Plastics PET	0.008	−0.253	0.31	n/a
Paper	0.223	−0.6	−0.691	n/a
Metal	0.008	−1.487	−1.346	n/a

has been shown to avoid GHG. The savings achieved by using waste textiles from the mattresses to produce ethanol, shown in Table 3 are estimated by adapting data published by Jeihanipour et al. (2010) and Macedo et al. (2008) which suggests that 81% of the potential 0.56 g of ethanol per gram of cellulose can be extracted from textiles, and that using ethanol as a transport fuel can avoid an average of 2041 $kgCO_2$-eq per m^3 of ethanol.

Transport emissions, carbon sequestration in landfill over 100 years and EU-wide averages of methane collected from landfills and used for electricity generation are embedded in the EC's data. Data on the pollution and energy expended during the process of landfill, recycling and incineration is also included. Credit is given according to the EC for electricity production via methane incineration at landfills, from mass incineration at CHP plants, for transport fuel made through ethanol conversion and in the case of recycling the net benefit is given for avoiding the use of virgin materials minus the energetic costs of collecting and processing the recycled materials (European Commission, 2001).

2.3.2. Refurbish and reuse

Estimates from the manufacturer suggest one third of 'shop floor' and 'indirect' hours taken to build a mattress are required to strip down and refurbish a mattress, reusing the springs and frame. The additional emissions arising from this extra work were added to the process section of the LCA. The additional components required to rebuild the mattress were also added. The new lifespan of the mattress will now be 20 years, thus, to fit the LCA functional unit of 10 years, the cumulative GHG must be halved.

3. Results and discussion

3.1. End of life scenarios

Fig. 2 indicates that recycling the components is by far the most effective single end of life scenario emitting only 39 and 56 $kgCO_2$-eq per m^2 of pocket spring mattress averaged over 10 years for the natural fibre and memory foam pocket spring mattresses respectively. Despite having a build time that is only one third of that of a new mattress, reusing the mattress springs receives perhaps a surprisingly minor reduction in GHG of less than 10% in the landfill scenario, though this increases accordingly if the treatment of the non-reused waste avoids landfill. The reason for this modest reduction may be that the steel springs (which are reused) make up less than 10% by weight and less than 6% by GHG emissions of the overall mattress components, whilst the remaining components must still be sourced from virgin materials.

In addition, electricity and heat are not major contributors to the total GHG emissions of the mattress (around 5% of the total) thus the time saved by rebuilding a mattress versus building a brand new mattress has a relatively small effect on the overall emissions. Landfill is the most polluting end of life option, being

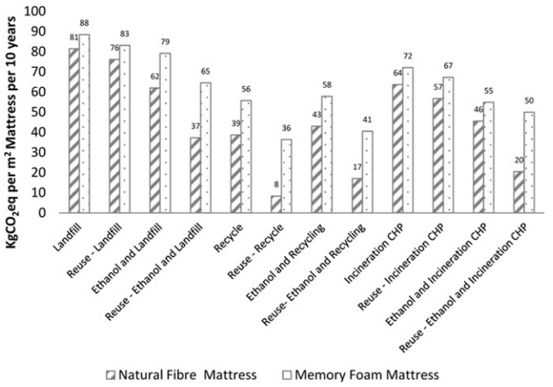

Fig. 2. End of life influence on the hybrid LCA for a natural fibre and a memory foam pocket spring mattress.

81 and 88 $kgCO_2$-eq, whereas incineration and ethanol production lie in the middle with 64 and 72 $kgCO_2$-eq, and 62 and 79 $kgCO_2$-eq for the natural fibre and memory foam mattresses respectively.

The end of life scenarios are therefore ranked in the order: 1) recycle, 2) ethanol conversion, 3) incineration, 4) reuse and 5) landfill. However, combining compatible scenarios such as ethanol conversion and incineration reduces GHGs further; the reuse scenario reduces GHGs of the mattress in every scenario by between 6% in the case of landfilling and most notably 90% when combined with recycling. Converting the textiles into ethanol as an additional end of life treatment also reduces GHG emissions by around a quarter in the landfill and incineration scenarios, however ethanol conversion is not as effective as recycling since it is shown to increases GHG emissions of the mattress by around 10% when coupled with recycling compared to the recycling only scenario.

Table 4 scales up the predicted impacts of the end of life options of the mattresses to the 35 million mattresses sold in EU each year. Industry research suggests that half of all mattresses being replaced are disposed of via landfill every year; the remainder are put to use in spare rooms or given away and sold second hand; 17.5 million mattresses are therefore assumed to be sent to landfill annually. Very few mattresses are currently made with natural fibre; around 70% are composites of springs, foam and fibre, similar to the memory foam mattress in this study. The remaining 30% are foam slab mattresses (GFK, 2010). In scaling up, we assume 95% of the mattresses are equivalent to the memory foam mattress and 5% to natural fibre. The mattresses in this study are at the high quality end of the market thus as a conservative estimate the mattresses are assumed to have only 75% of the materials and therefore emissions that are 25% lower.

When the results from the study are scaled up to 2000mm × 1500mm × 270mm mattresses, the potential estimated savings range from over 210,000 tCO_2-eq for the reuse scenario to 2,092,000 tCO_2-eq for reuse and recycling. Currently reuse schemes are very rare. The most widely achievable and therefore significant end of life option is to recycle the mattresses. Recycling alone could save over 648,000 tCO_2-eq annually which would otherwise be emitted during landfill, equivalent to 0.08% of the UK's 783 million tons CO_2-eq annual emissions as reported by Department of energy and climate change (2011). Mattress recycling schemes are growing in popularity in the UK though they are not currently incorporated in the EU waste directives. However these estimates assume mattress homogeneity and equal access to end of life options that are currently unrealistic across the EU. This means it is difficult to

Table 4
Potential annual avoided GHG emissions in the EU from different mattress end of life scenarios (tCO_2-eq/m^2).

	Avoided GHG natural fibre	Avoided GHG memory foam	Total avoided GHG
Landfill	0	0	0
Reuse – landfill	3432	66,879	70,311
Ethanol and landfill	12,756	115,825	128,581
Reuse – ethanol and landfill	28,944	298,530	327,473
Recycle	28,044	408,150	436,194
Reuse – recycle	47,927	649,734	697,661
Ethanol and recycling	25,191	382,245	407,436
Reuse – ethanol and recycling	42,221	597,923	640,144
Incineration CHP	11,636	204,370	216,006
Reuse – incineration CHP	16,204	264,191	280,395
Ethanol and incineration CHP	23,524	419,271	442,795
Reuse – ethanol and incineration CHP	39,980	480,084	520,064

have confidence in such extrapolations, though they are useful in providing context to the research.

3.2. Biomaterial versus petrochemical mattresses

It is observed from Fig. 2 that in each scenario the natural fibre mattress emits lower amounts of GHGs than the memory foam mattress, however, the difference between the two changes dramatically depending on the end of life scenario. Differences are quite minor under landfill (8%) due to the potential for GHG emissions from the degradation of natural fibres, yet are very noticeable (78%) in the reuse and recycling scenario since recycling textiles avoids more GHG than recycling foam. Converting textiles to ethanol also increases the disparity between natural fibre and memory foam mattresses since there are fewer natural fibres in the memory foam mattress. The natural fibre mattress has 60% lower GHG emissions than the memory foam mattress when combining ethanol conversion with incineration and CHP which is shown to be the second most effective combination to reduce GHG with ethanol conversion.

3.3. Input output versus process LCA

The process LCA captured 74% of the GHGs emitted during the construction of the natural fibre mattress, i.e. direct impacts that the mattress manufacturer has control of such as the heat and power they use and their choice of materials. This number was 78% for the memory foam mattress. As illustrated in Fig. 3, the natural fibre mattress causes more upstream IO GHG, though only in the Chemicals and Business Services sector. The most significant upstream emissions came from Agriculture and Business Services and then the Chemicals sectors for both the mattresses.

The major sources of GHG that contribute at least 1% to the overall emissions of each mattress are shown in Fig. 4. Natural fibres such as wool, cotton, animal hair and viscose are the biggest contributors to the LCA of the natural fibre mattress making up 51% of the total. The foam in the memory foam mattress contributes a total of 23% whereas natural fibre in the fabric of the memory foam mattress actually cause higher GHG emissions, 30% of the total. This is due to woven cotton being by far the most GHG dense material which is combined in the fabric with viscose and causes 13.5 $kgCO_2$-eq per kg. This suggests that the selection of the specific biomaterial can also greatly influence the overall GHG of the mattress. Where possible, materials with lower GHG densities such as hemp may therefore be favoured over equivalent high GHG

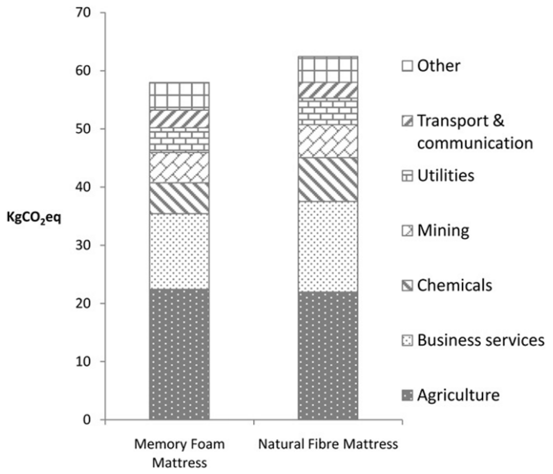

Fig. 3. Pocket spring mattress input output LCA GHG emissions by sector.

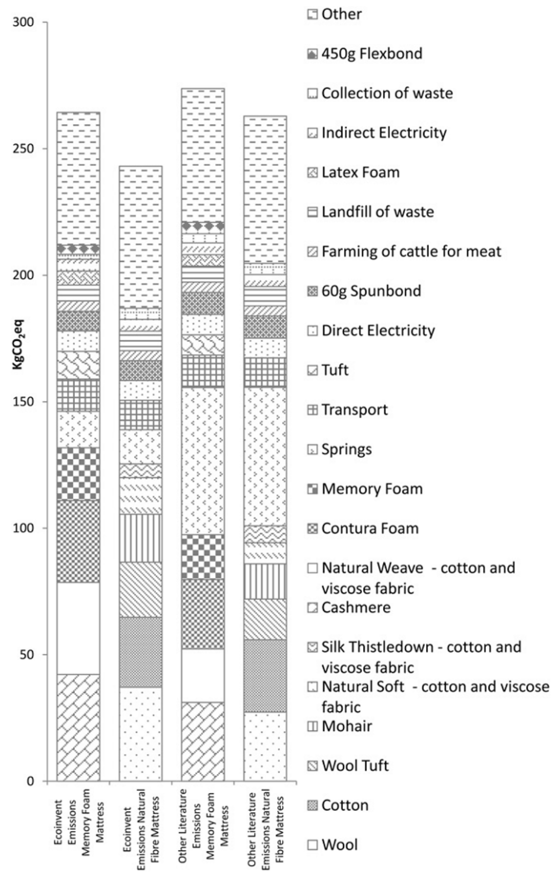

Fig. 4. Cradle to gate LCA for a natural fibre and memory foam pocket spring mattress.

density materials such as wool, provided functionality is not compromised.

The springs contribute only 6% and 5% of the GHG for the natural fibre and memory foam mattress respectively, despite being among the biggest components by weight. Electricity used in the production of the foam and natural fibre mattress was also around 5% of the total GHG, as was transport of the mattresses to the shops. It may be therefore that effort from the mattress manufacturers to reduce GHG may be most effective if focused on the emissions embedded in the supply chain rather than their own activities, though this may not necessarily be the most cost effective approach.

Upstream impacts picked up by the EIO that were surprising include farming of cattle for meat and the collecting and landfilling of waste; each contributed just over 2% to GHG. This illustrates the importance of the hybrid LCA and EIO data to pick up hotspots potentially hidden in the supply chain. The implication of this is that if economies in general reduce their consumption of meat and improve their recycling rates, the GHG density of all goods produced by the economy may potentially be reduced.

3.4. Sensitivity analysis

3.4.1. Process LCA data quality

Data quality is an often cited area of uncertainty in LCA (Chiaramonti and Recchia, 2010). Fig. 4 shows the mattress components' contributions to the cradle to gate LCA which were established using the LCA database ecoinvent compared to an equivalent LCA using data taken from the other literature searches listed in Table 1. The major changes are that steel springs take on a much more prominent role in both mattresses compared to the ecoinvent results and that the importance of the foam and natural fibres is diminished. These changes impact the total GHG emissions as shown in Fig. 5.

The difference in emissions caused by changing the data sets remains roughly between 5% and 15% of the ecoinvent results depending on the end of life scenario. Data from 'other literature' generally results in more emissions, the exception being the 'reuse' scenarios. This may be because the 'other literature' weights the impact of steel more heavily and therefore gives more credit for reusing the springs.

3.4.2. End of life data assumptions

This study assumes that the credits identified in Table 3 can be achieved in every case. It is likely however that in some instances, due to contamination and cleaning, or problems with collection and sorting, it will not be possible to recycle all the materials, to refurbish the mattress, or to produce ethanol from all the textiles. There may even be a lack of commercial biorefineries or CHP plants available to convert textiles to ethanol or incinerate waste. A sensitivity analysis was undertaken therefore assuming that it is only possible to claim 50% of the credits given to recycling, conversion to ethanol and incineration from Table 3.

As expected the reductions in GHG are much less pronounced than when assuming all the hypothetical savings could be made. Also, the scenarios became relatively less distinctive in their relative GHG emissions. GHG emissions in reuse and recycling became less remarkable, jumping from 8 to 43 $kgCO_2$-eq, whereas Incineration CHP with reuse and ethanol conversion rose only to 48 from 20 $kgCO_2$-eq. Nevertheless, the waste hierarchy has not altered, remaining as: 1) recycle, 2) ethanol conversion, 3) incineration, 4) reuse and 5) landfill. Combining compatible scenarios, especially the reuse and recycle scenario, also remains the most effective method of reducing GHG despite only potentially half of the mattress being diverted from landfill.

3.4.3. Functional unit

Inferring the results of this mattress LCA to similar furniture industries such as sofas or car seats may be difficult since our assessment is based on a per m^2 functional unit. A functional unit that considers mass may be more useful. Fig. 6 shows the results from Fig. 2 but with a functional unit that uses kg not m^2. Despite the change in functional unit, this assessment ranks the end of life options similarly to the original LCA. However unlike the initial assessment, there are certain scenarios where the memory foam mattress appears to have lower GHG emissions than the natural fibre mattress. Since the memory foam is a heavier mattress, 67.79 kg compared to 61.61 kg for the natural fibre, its cumulative emissions are higher. However the emissions per kg are lower,

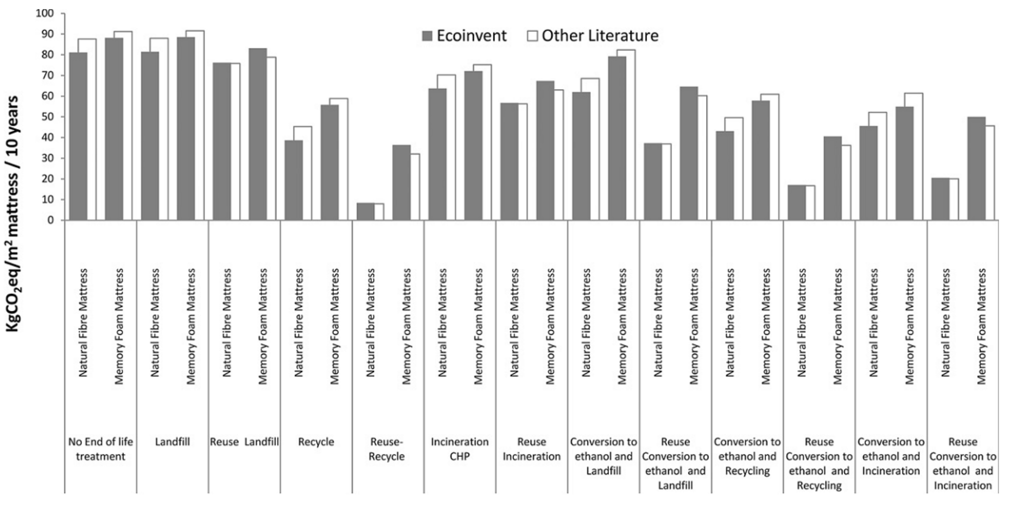

Fig. 5. End of life influence on the hybrid LCA for a natural fibre and a memory foam pocket spring mattress; comparing ecoinvent data with average literature data.

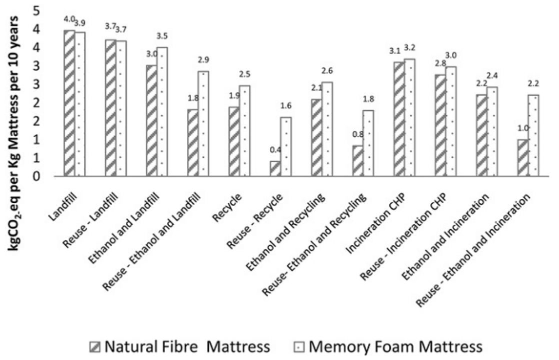

Fig. 6. End of life influence on the hybrid LCA for a natural fibre and a memory foam pocket spring mattress; using mass as a functional unit.

meaning the memory foam mattress appears less polluting as a feature of the functional unit bias.

3.5. Limitations

3.5.1. Uncertainty in process LCA data

Despite the 'other literature' sensitivity analysis, uncertainty remains in the process LCA data. Often data used was known not to be optimal, for example data could not be found for the processing stages required to make fabrics from the polypropylene and polyethylene granulate foam. Also data for wool assumes sheep production in the USA not Yorkshire. In addition, all the ecoinvent data were cradle to gate so omitted emissions caused by delivering raw materials to the mattress factory, meaning transport emissions for products from China were comparable to products from the UK. Although emissions are generally attributed to co-products in ecoinvent using economic allocation this may not always be the case or the most appropriate method.

3.5.2. Uncertainty in the EIO LCA data

Despite division into import and export tables and disaggregation into subsectors, large amounts of aggregation still exist. For example, no distinction can be made between the recycled steel versus the virgin steel industries or between a product from a modern efficient supply chain versus the same product made in inefficient Victorian factories. Also, owing to the lag between generation and publication of data, the IO tables used are 5 years old at the time of writing.

3.5.3. Subjectivity and indirect effects

The choice of including or excluding inventories from the EIO LCA to account for missing inputs and to avoid double counting of inputs are inherently subjective. Making correct decisions requires in depth knowledge of the supply chain and process LCA data. LUC and rebound effects are not considered in this assessment, nor are the longer term effects on supply chains such as economies of scale that may occur by pursuing different end of life scenarios.

3.6. Future research

There are several areas that could be explored in future research: Mattresses are composites of many different materials, therefore assessments on more mattresses including non-pocket spring mattresses, i.e. foam slabs which may have more limited end of life scenarios, would be required to ascertain if the conclusions drawn here are common to all mattress types or other furniture items or if they were statistically significant. The two mattresses assessed in this study, both occupy the luxury end of the mattress market. The sustainability of a luxury versus a standard mattress which uses fewer components and which is therefore likely to have lower GHG but arguably provide similar functions, takes on an ethical dimension that has not been considered in this paper. This research has reported exclusively on GHG since only GHG environmental extensions for IO sectors (and consequently for the hybrid LCA) have been defined. Process LCA can report on more environmental issues such as impact on human health, acidity of waterways and eutrophication risk among others. Finally it is not known if benefits identified here (for example by recycling) are economically viable; nor are the social impacts of using more land to grow natural fibres quantified. Both of these areas need assessing to be confident in claiming the sustainability of a mattress or end of life scenario.

4. Conclusions

The waste hierarchy for mattresses considered here is: recycle, ethanol conversion, incineration, reuse and landfill. Combining scenarios provides greater GHG savings: reuse and recycling can reduce emissions by 90%. Natural fibre mattresses emit marginally less GHG than foam under the *status quo* disposal to landfill and have greater potential to reduce GHG under different disposal options. Designing for reuse and recycling should be prioritised along with favouring biomaterials over petrochemicals. The case for legislative support for this is that 210,000–2,092,000 tCO_2-eq could be saved in the EU annually should this be widely adopted.

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Appendix II

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Achieving sustainable biomaterials by maximising waste recovery

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ABSTRACT

The waste hierarchy of 'reduce, reuse, recycle, recover' can be followed to improve the sustainability of a product, yet it is not applied in any meaningful way in the biomaterials industry which focuses more on sustainable sourcing of inputs. This paper presents the results of industry interviews and a focus group with experts to understand how waste recovery of biomaterials could become more widespread. Interview findings were used to develop three scenarios: (1) do nothing; (2) develop legislation; and (3) develop certification standards. These scenarios formed the basis for discussions at an expert focus group. Experts considered that action was required, rejecting the first scenario. No preference was apparent for scenarios (2) and (3). Experts agreed that there should be collaboration on collection logistics, promotion of demand through choice editing, product 'purity' could be championed though certification and there should be significant investment and research into recovery technologies. These considerations were incorporated into the development of a model for policy makers and industry to help increase biomaterial waste recovery.

1. Introduction

The biomaterial industry in its broadest sense includes all products derived from plants and animals including natural fibres, oils and waxes, bio plastics and biofuels. According to industry surveys, biomaterials will play a prominent role in future global economies (Vandermeulen et al., 2012). Based on the assumption that they have fewer negative impacts and can be replenished from a wider range of sources, they were historically hailed as ideal replacements for petrochemicals (OECD, 2001). However, questions soon surfaced regarding their sustainability, with key concerns including emissions from land use change (LUC) in shifts towards biomaterial production, as well as those linked to excessive fertilizer, pesticide and water use, and displacement of people and food (Tilman et al., 2009; Searchinger et al., 2010). These concerns are especially important because despite on-going debate surrounding its definition, 'sustainability' has momentum in industry as a business principle, a marketing tool and a legislative requirement. As such, it is imperative that biomaterials are seen to be sustainable (Boer, 2003; Golden et al., 2010).

In response to these concerns, sustainability assessments were developed including e.g. the European Union's (EU) Renewable Energy Directive (RED) and the Roundtable on Sustainable Palm Oil's (RSPO) sustainability standard which target consumable biomaterials (fuel and food) and focus on the impacts of sourcing, processing and transporting feedstock. Such schemes are nevertheless inadequate in terms of capturing a complete picture of the impacts of non-consumable biomaterials like bio-plastics and natural fibres, which also need to factor in the impacts of disposal.

The waste hierarchy sets out a pathway of options to reduce the impact of waste. This study focuses on the 'recovery' aspect of the waste hierarchy to identify how waste recovery of biomaterials could be made more widespread. The term 'biomaterials' is used in this research only to refer to plant based products such as natural fibres, paper, and bioplastics and everything in between. Fuels, food and garden waste are outside the scope of the research.

1.1. Biomaterials

Combined, the biomaterials industry is vast, contributing a turnover of 2 trillion Euros to the EU economy per annum (Lieten, 2010), so it is important to define with which part of the industry this research is concerned. Compostable bio-waste such as food and garden waste is part of the biomaterials landscape. However this has a relatively mature waste management strategy within European Union policy[1] and it is the subject of significant academic research even having academic journals devoted to it[2]. As such, compostable bio-waste poses different challenges to other less regulated biomaterials, and is therefore not discussed in this paper.

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[1] http://ec.europa.eu/environment/waste/compost/index.htm.
[2] http://www.journals.elsevier.com/international-biodeterioration-and-biodegradation/.

Table 1
Chronology of EU Waste Legislation.

Year	EU legislation	Summary
1994	Packaging and packaging waste directive	'Producer responsibility' principle founded, set out targets for reducing packaging and to recover 80% of packaging (including incineration)
1999	Landfill directive	Regulations for what can be admitted to landfill, restricting biodegradable waste but permitting all other biomaterials
2000	Waste incineration directive	Regulated the emissions caused by the incineration of waste to produce electricity including biomaterials like textiles, etc.
2003	End of life Vehicle Directive (ELV)	Fines for producers not achieving recovery targets of up to 90% prompting companies to use more easily recoverable biomaterials
2006	Waste Electrical and Electronic Equipment Directive (WEEE)	Similar to ELV resulting in incentives for design for disassembly
2008	Waste framework directive	Clarified responsibility for governments, waste producers and managers to promote prevention, preparing for re-use, recycling and other recovery (no explicit reference to biomaterials)

Despite representing a relatively small proportion of the overall market, the overwhelming majority of research into biomaterials focusses on biofuels, partly because biofuels are becoming more mainstream but also because of the RED (Gallagher, 2008). The research presented here concerns only the lesser studied non-consumable biomaterial products.

Biomaterials have not been comprehensively studied within the sustainability literature. However, predictions by the National Non-Food Crops Centre (NNFCC, 2012) suggest that the UK biomaterial market could triple over the period 2012–2015. A cavalcade of research on non-consumable biomaterials may therefore be expected, and so establishing a framework for designing interventions to promote their waste recovery, and therefore improve their sustainability, is both a timely and vital exercise.

1.2. Biomaterial waste recovery

'Recovery' is used in this paper to refer to disposal options that avoid landfill as per the waste hierarchy; reuse, recycling, incineration with energy recovery, conversion into a liquid fuel like bioethanol and composting. Research suggests that whether a biomaterial is sent to landfill or is recovered through any of these methods can influence its life cycle impact on CO_2 emissions up to the same degree as other more conventionally studied issues such as the amount of fertilizer used or LUC (Glew et al., 2012; Shen et al., 2010; Ross and Evans, 2003). Currently the UK recycles less than 32% of its textiles and plastics (including natural fibres and bioplastics) yet it manages to recycle 42%, 44% and 75% respectively of glass, paper and steel packaging (European Commission, 2009). Further recovery via incineration of municipal solid waste (including biomaterials) in the UK is only around 10% according to the Chartered Institute of Waste Management[3], virtually no biomaterials are currently converted to ethanol since the technology is still embryonic (Schmitt et al., 2012) and only food and gardening wastes are commonly composted, all of which indicates there is room for improvement in biomaterials recovery.

Recovering waste products can improve supply chain security and have cost savings (Lynes and Andrachuk, 2008; Sacramento-Rivero, 2012). The recovery of waste is therefore taken seriously, as can be seen in Table 1, which gives a summary of European Union (EU) waste legislation that has been variously enshrined into UK law. No specific legislation to tackle biomaterials has been developed as of October 2012.

2. Research design and methods

This research uses a qualitative, mixed methods approach comprising interviews with biomaterials industry representatives, and an expert focus group. Findings from interviews were used to construct three scenarios to promote the recovery of waste biomaterials, which were then evaluated during the focus group. Each of the methods used is outlined in detail below, and complied with the Economic and Social Research Council's (ESRC) *Six Key Principles*[4] for research projects, ensuring an ethical approach appropriate to the nature of the study.

2.1. Interview method

Opportunities and barriers to biomaterial recovery are difficult to explore with quantitative assessments and so qualitative, semi-structured interviews were used (Neuman, 2004), allowing questions to be asked around pre-determined themes in a conversational manner (Gillham, 2005). The biomaterial industry in the UK was chosen as the focus of data collection because this is where the researchers were located, because waste legislation and sustainability assessments are relatively common, and because the UK comprises a range of representatives of this diverse market: from small independent companies to large multi-nationals. Products made from biomaterials are as diverse as cotton T-shirts to car panels, so it was important to collect the views of a wide range of industry stakeholders to cover this spectrum. The choice of the UK industry provides a useful case study, although the different waste profiles of EU member states mean that specific results may differ from country to country.

Non-probability sampling was employed, gathering the insights of company representatives with specific insider knowledge (Flowerdew and Martin, 2005). There were no existing networks of biomaterial industry-research collaborations available, so leading companies in the industry were contacted directly and from these initial contacts snowball sampling was then used, taking recommendations to widen the sample and avoid further cold calling (Neuman, 2004). The sample size was defined when new interviews unearthed little novel information (Flowerdew and Martin, 2005).

Target industry groups were based on considerations in the WEEE and the ELV where 'producer responsibility' is assumed, manufacturers must pay for waste recovery, and retailers may facilitate take back schemes (European Commission, 2003, 2000). Therefore, manufacturers and retailers were invited to take part in the research. Engaging with employees that have strategic understandings of companies has been shown to be important, so operational or sustainability managers were approached (Pagell, 2004). Feedstock growers are inherently involved in the sustainability of biomaterials so growers were also invited to participate (Black et al., 2011; Gallagher, 2008). Attitudes of consumers are important as they play a role in product disposal. However, since this falls outside the remit of producer responsibility, collecting consumer opinions was outside the scope of this study. The sample thus constituted a wide selection of stakeholders, so conclusions with multi-stakeholder implications may be drawn. A summary of the company profiles is shown in Table 2.

[3] http://www.ciwm.co.uk/CIWM/InformationCentre/AtoZ/IPages/Incineration.aspx.

[4] http://www.esrc.ac.uk/_images/Framework_for_Research_Ethics_tcm8-4586.pdf.

Table 2
Interview sample demographic.

Company classification	Description
Growers ($n = 4$)	Small scale less than 1000 acres, both food and biomaterial feedstock
Small manufacturers ($n = 5$)	Use raw feedstock or processed biomaterials, sell to UK consumers and industry, less than 500 employees
Large manufacturers ($n = 3$)	Use raw feedstock or processed biomaterials, sell to UK and international consumers and industry, more than 500 employees, multinational supply chains
Large retailers ($n = 2$)	Sell a range of processed biomaterials and non-biomaterial products in the UK, over 1000 employees, multinational supply chains

Interviews took place in spring 2012. Preference was for face-to-face interviews or video or telephone interviews if it was not possible to meet in person. Participation was encouraged by providing a concept note via an email invitation, followed by telephone reminders. During the interviews notes were made and written up afterwards, in addition to an audio recording being taken where permission was granted, in order to enable fact checking. The interview protocol was iteratively upgraded with each interview without altering the focus or content. For example, a standard introduction to the research was given to each interviewee after the first interview revealed this would be helpful. Forty-one companies were contacted and fourteen agreed to an interview, giving a response rate of 34%. Appendix A identifies the role of each interviewee and their sector.

Literature on response rates applies mainly to probability sampling where rates range from 30% to 85% depending on the number of reminders sent, respondent age and occupation, etc. (Hocking et al., 2006; Regula-Herzog and Rodgers, 1988). Data on non-probability interview response rates similar to this research are not found, since biases resulting from low response rates are less likely to influence non-random sampling. Non-respondents were not from any one group in particular and respondents came from each of the main categories of retailers, manufacturers and growers, in addition to there being representatives from large multinational and smaller organisations. Despite this, there were a substantial number of non-respondents which could have resulted in some degree of selection bias.

Following the final interview, a post analysis summary was sent to each interviewee and they were encouraged to identify any changes needed to the record of their responses (Brenner et al., 1985). All interviewees were content with their documented answers and no changes were suggested as a result.

2.2. Focus group method

Following analysis of the interviews (described in detail in Section 2.3) three scenarios were developed which were then presented to an expert focus group. Scenario-based stakeholder engagement is a useful tool for qualitative analysis comparing preferences between groups (De Lange et al., 2012; Morgan-Davies and Waterhouse, 2010; Tompkins et al., 2008).

The focus group was held in summer 2012 and targeted UK experts with experience in the biomaterial, waste and sustainability sectors. Focus group participants were identified by conducting an online review of research and government organisations active in the field of biomaterial recovery. Following this, snowball sampling was employed to widen the pool of contacts. Experts had a strategic understanding of their organisation as characterised in Table 3.

Table 3
Focus group sample demographic.

Organisation type	Expert's role
Research facility for deriving high value biomaterials from plants and bio waste	Director
University department for sustainability research	Director
Consultant to government departments and Co-founder of a sustainability certification scheme	Consultant
Government funded waste organisation	Project manager
Consultancy advising the UK government departments specifically DEFRA on waste and textiles	Technical consultant
University environment department	Teaching fellow in environmental economics
University department for industrial uses of plants (biomaterials)	Research chair
Not for profit research institute promoting global sustainable development	Director
Not for profit research institute promoting global sustainable development	Senior research associate

The focus group experts were introduced to the research via a concept note and a two-page summary of the interview findings. In total, nine experts attended (a response rate of 26%) which is a useful size for data collection in exploratory research (Billson, 2006; Tang and Davis, 1995). The three scenarios: (1) do nothing; (2) develop legislation; and (3) develop certification, were discussed over a period of 2.5 h. Despite differences of opinion between the experts, consensus was reached on the views to be recorded. Following the focus group, a summary of the outputs from the session was sent to all attendees who were asked to provide feedback. Detailed comments were received from one expert. A further nine experts unable to attend the day but who showed an interest in the research were sent a copy of the output summary from the focus group and were asked to comment via a telephone interview or by email. Two replies were received.

2.3. Data analysis

The use of coding to categorise comments from interviews and focus groups forms the core of the analytical techniques used in this research (Neuman, 2004). Codes were chosen because they reflected the purpose of the research and were both etic and emic, meaning key words and common themes were used in categorisation (Holsti, 1969; Flowerdew and Martin, 2005). Coded comments were organised hierarchically using axial coding according to the book title, chapter and sub heading analogy proposed by Gillham (2005). Once the coding of the interview data had been done, descriptive quantifications of the number of times particular codes were raised could be undertaken. Beyond this, semiotic clustering and a semiotic square was used so that related codes could be defined into to more distinct classifications to identify mutually exclusive and duplicate codes, to align opinions with specific company traits and allow the identification of the scenarios (Flowerdew and Martin, 2005).

To analyse the focus group data, experts' discussions on the scenarios were noted and their comments were similarly grouped into codes to identify the underlying themes, the areas of consensus and the variation of opinions that existed regarding the scenarios.

3. Results and discussion

3.1. Interviews

Fig. 1 presents a summary of the interview findings according to the number of times a particular theme was mentioned. This

Fig. 1. Key themes emerging from interviews.

quantitative assessment is useful to introduce the issues that were raised and to group them under broad headings e.g. "uncertainty", "markets", "ethics" and "cost". It is important to note that the number of mentions is not an indication of ranked importance and many contradictions were apparent. For example, "government support" was mentioned frequently in some form, though those mentioning it differed in their opinion as to whether it was necessary or not.

Certain trends are apparent when attributing the frequency of mentions to respondents' stakeholder groups (Fig. 2). For example, those selling to the public had a greater preoccupation with 'greenwash' and addressing holistic sustainability; they noted the uncertainty of distinguishing 'good and bad' biomaterials; and felt their supply chains were difficult to influence compared to those who only sold to other industries. Manufacturers often mentioned costs, were most vocal on rejecting the need for government involvement and said they would only use biomaterials because they served a particular function, not because of their perceived sustainability.

There are clear differences in priorities for stakeholders and picking out the interesting trends beyond these prosaic patterns requires qualitative analysis. During the analysis of the interview data it became apparent that the interview responses could be usefully presented under the following two headings: the need for intervention and possible interventions.

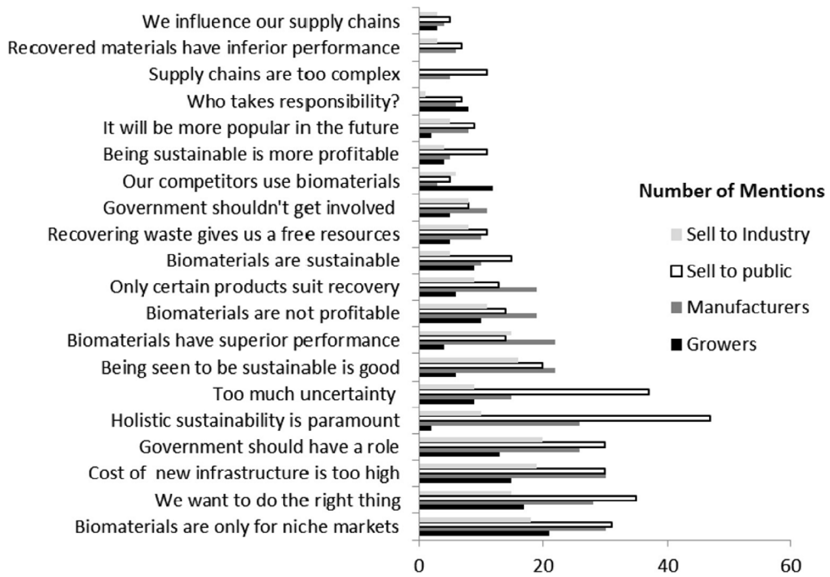

Fig. 2. Key themes in interviews according to company type.

3.1.1. The need for intervention

According to the interviews, companies' main concerns were financial sustainability, followed by issues including product quality, risks and environmental footprints. After these common priorities there was some divergence, for example, concerns over stable supply chains, social welfare, habitat destruction, climate change and depleting resources were recorded mainly by companies with international operations. Only a few large retailers and small manufacturers considered waste recovery to be important and these were companies that had an economic or marketing interest in it. A lack of priority for recovery was especially evident for companies selling products that use energy, such as cars, houses or washing powder, whose main life cycle impacts were the in-use energy consumption of their products. It was common that life cycle assessments (LCAs) on individual products had not been performed, either because it was too expensive, because companies did not see the need to, or because it was too great a task, especially for those who sold thousands of different products. Several larger firms had performed holistic LCA of their entire company operations but the majority of respondents were confident, even without having done product or company wide LCA, that disposal represented one of the smaller, if not the smallest, environmental impact of their operations. Despite this, several respondents claimed to be interested in disposal and almost all anticipated that it would become more significant to their business in the future. However, in the short-term at least, more pressing problems push waste disposal further down their agenda.

Recovering biomaterials can be profitable, for example, where it provides a free resource in the case of reconstructing natural fibre carpet tiles. Generally it was suggested that recovery is rare because of the low economic value of recycled biomaterials compared to synthetic alternatives. There were also concerns that the reprocessed biomaterials may not have sufficient quality. For example, a retailer investigating the sale of clothes made from recycled natural fibres was concerned they are not always comparatively comfortable, and was reluctant to offer a lower quality product to consumers. This finding confirms that of Nicolli et al. (2012) who also established quality was a barrier to finding markets for recycled products. Similarly, car manufacturers claimed they were restricted in using recycled products in components such as seat belts due to health and safety legislation. Interviewees felt that technological advancements may be needed to produce cheaper, higher quality recovered biomaterials before they become profitable and desirable enough to be mainstream products.

Companies with many sites, large shop footprints, car parks and who may already be providing recycling facilities for e.g. glass and plastic were particularly concerned that if biomaterial recovery was forced upon them, they would have to take the brunt of the logistical burdens for the rest of the industry. One such respondent stated "we are not a waste management company" and smaller companies even confirmed that allowing larger companies to host their take back schemes for them would be more practical than collecting material on their own smaller premises. A fear of the risks and burdens means large retailers that could arguably benefit the most from recovering large quantities of waste biomaterials to put back into their supply chains, are put off, and are least likely to actually recover any material. Growers appeared most positive about taking back waste, suggesting they drop off raw materials to factories and could simply bring back the waste biomaterial (presumably in composted form) to "put it back on the land and complete the cycle". Fairness and responsibilities are important issues and how these are shared seems a common barrier that prevents biomaterial recovery rising up the agenda.

Producer responsibility is embedded in waste legislation, yet consumers influence waste recovery too and this was reflected in interview comments ranging from "consumer education is key" through to the notion that any scheme will fail if it places additional cost on "penny pinching customers". Those accustomed to using various sustainability labels felt that having many schemes running in parallel can be confusing for consumers, and they were not keen on using more labels to promote recovery. The reluctance to place responsibility or cost on consumers seems another reason for the lack of experience and growth in the recovery of biomaterials.

In summary, there are significant barriers to generating interest in recovering biomaterials. These include competing priorities, unknown potential costs and benefits, insufficient knowledge and technical capability, a lack of proven nationwide logistics, uncertainty over responsibilities for recovery and collection, and trepidation about consumer responses. These issues are difficult to tackle with strict intervention and overall, suggested that 'do nothing' was a realistic scenario to include in the focus group discussion.

3.1.2. Possible interventions

Although "do nothing" may be a desirable scenario from the perspective of some companies it has thus far not led to high rates of biomaterial waste recovery. "Intervention" is used here to refer to any form of legislation, investment, law or certification scheme that may stimulate waste recovery. Generally there was concern about government intervention resulting in 'yet more red tape' especially from farmers and small companies who had experiences of burdensome requirements. A cautious overall agreement was nevertheless put forward from larger companies and those accustomed to regulation, suggesting that intervention may be useful. According to an interviewee from the construction industry, intervention would make it easier to "differentiate good from bad". Almost all interviewees across the different stakeholder groups agreed that before intervention on a mass scale is implemented (either from within the industry or from outside), there should be a greater understanding of the risks, logistical requirements and benefits of recovering different biomaterials in different ways.

The interviews revealed that four companies were currently involved in voluntary recovery schemes driven by the desire to "do the right thing" but also in some instances to take advantage of a "free resource". These were: (1) a refurbishment schemes for mattresses though "[they] only do the take back [scheme] on the top of the range models"; (2) leasing schemes for carpet tiles; (3) removal of large bulky items when replacements are being delivered; and finally, (4) a voucher system to encourage consumers to return their clothes to a partner charity shop. These voluntary recovery schemes are in various stages of maturity but all are relatively new, small-scale and not necessarily suitable for all biomaterials. Although the positive impact of voluntary agreements is hinted at by the respondents it has not been conclusively shown in this research. However, this suggestion does align with others studies that have suggested they are particularly critical in spurring technological advancement specifically in the automotive sector (Nicolli et al., 2012).

It was generally agreed by those not partaking in a voluntary scheme that they would require some form of support, such as subsidised costs of infrastructure for collecting, transporting and processing waste, or collective action on a nationwide collection scheme in order to benefit from economies of scale to persuade them to embark on a recovery scheme. Incentives have justification in fixing the market failure of technological externalities. For example manufacturers may not invest in making products easy to recover by other companies at another point in time despite the net benefit to society this may bring since they receive no reward for this, incentives address this (Nemoto and Goto, 2004). Yet beyond the potential benefits of recovered materials being free resources there was no mention by either industry or expert

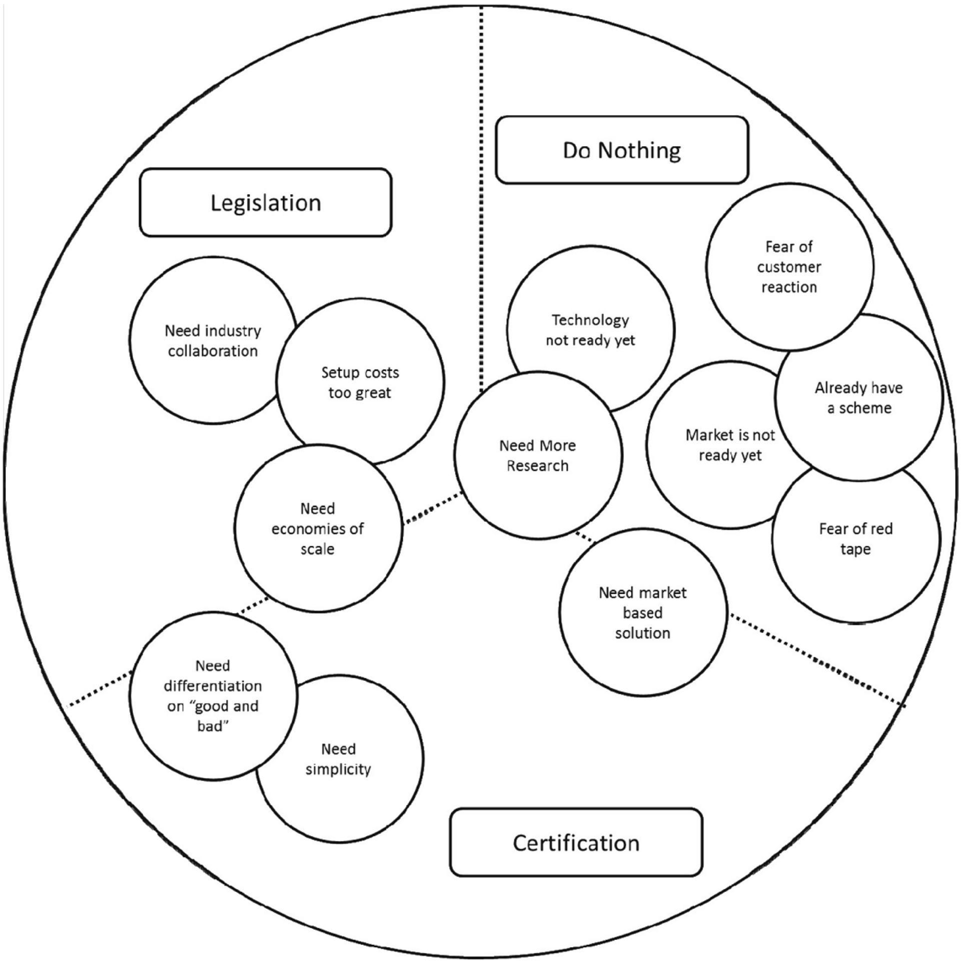

Fig. 3. Development of intervention scenarios from interview comments.

stakeholders that vertical integration of biomaterial producing and recovering companies would be beneficial, which is the case in some other markets for example the electricity industry where technological externalities have been observed.

Those companies already involved in a scheme enjoyed their uniqueness and did not crave participation by their competitors, some referring to themselves as "leaders" and enjoying competitive advantage. Thus, although incentives exist to set up recovery schemes, these are unlikely to be sufficient to stimulate recovery on a national scale. As such, "developing legislation" of some kind represents a reasonable scenario to include in the focus groups.

Fear of innovation being stifled by intervention was mentioned by several smaller companies. However, this may be a misconception of the 'hands-off' approach, since innovation seemed to be most advanced in the automotive industry where prototypes using biomaterials to increase recovery rates and reduce environmental footprints were more common. At the same time, this is a sector in which waste recovery is heavily regulated (to combat the negative external of sending used cars to landfill), though research and development budgets are generally higher in the automobile industry. The interviews seem to support the assertion that certainty of legislation can stimulate innovation (Office of Fair Trading, 2009), especially where there is momentum behind the technology (Luiten et al., 2006). In the case of the ELV directive, the metals recovered are valuable and so a profitable recycling network collects, sorts and processes end of life vehicles. Biomaterials may not have similarly high market values and individuals from the automotive industry suggested that if other biomaterial producers were forced to recover their products along the lines of the ELV directive, they may end up out of pocket. Assisting recycling companies to extend their capabilities to process all sorts of disparate biomaterial products more cheaply may be helpful yet according to those interviewed one of the benefits of recovering biomaterials is that they provide a cheap feedstock. This means that if they themselves do not directly benefit from recovering the biomaterials, they may not be incentivised to design in recoverability, preferring cheaper petrochemical alternatives.

There was concern from retailers that customers are already faced with multiple forms of labelling and that they may not be ready for additional certification schemes around biomaterial waste recovery, yet the need to segregate biomaterials from synthetics was identified to be a problem by companies from each

Table 4
Scenarios for discussion.

	Expand or develop new legislation	Voluntary certification schemes	Do nothing
For	The automotive industry is subject to waste regulations which have greatly increased its recovery of materials as a result. The certainty that legislation has brought has spurred on more innovation and could be successful in the biomaterial industry too	There is a market for sustainable biomaterials that cannot be tapped because of uncertainty. Certification could provide clarity, inform the market and promote best practice within the biomaterials industry	Change should be allowed to grow organically from within the industry without being hindered by external influences
Against	There is no ready-made recycling industry to deal with logistical problems of collecting biomaterials as there was for metal in cars	Additional certification will confuse consumers adding more labels to already crowded packaging and will not guarantee customers will actually take part in waste recovery	The costs of setting up a recovery program for mass biomaterial markets are prohibitive, collective burden sharing represents the highest possibility of success and needs some market intervention to make it happen
	Biomaterials are too diverse to have a one size fits all approach and legislation risks lumbering huge costs onto an emerging market		

stakeholder group. For example, a company selling textiles argued there was a need for products to be designed with disassembly in mind, making it easier to break down fibres to their constituent parts without contamination from synthetics before large-scale recovery programs would be worthwhile. Linked to this are the barriers of providing access to collection points and the complexity of self-sorting; challenges that were almost unanimously mentioned. Recovering materials at a large scale is therefore less likely while biomaterials are complex, heterogeneous and difficult to separate. A final scenario for the focus group discussions may therefore be "developing certification", which may incentivise the use of pure biomaterials which will simplify sorting and improve the efficiency of technology.

Fig. 3 captures some of the main threads discussed in the interviews. Overlapping circles reflect related themes which are each located in the "legislation", "certification" or "do nothing" scenarios or some combination of all three.

In summary, biomaterial industry representatives presented mixed views on the need for intervention. Currently, recovery is being held back because products are not 'pure', the technical challenges and costs of mass recovery are thought to be too great, and there is no guaranteed market for recovered biomaterials, so economies of scale are being missed. Existing schemes are irregular and small scale, though they are indicative of the potential that exists. Despite opposition from some smaller manufacturers there is agreement across the other stakeholder groups that intervention could play a useful role. The scenarios of "do nothing", "develop legislation" and "develop certification" were developed from the interviews and used in the focus group discussions.

3.2. Focus groups

The intervention scenarios taken from the interviews in Fig. 3 were presented to the focus group as a starting point for discussion as shown in Table 4.

Coding of the focus group discussions revealed several overarching principles which held consensus with all the experts. These were: (i) that increasing the recovery of biomaterial waste will increase efficiency and sustainability in the industry; (ii) that intervention was a reasonable next step to encourage more biomaterials recovery; (iii) that interventions should target biomaterials according to their product type not as an overall group (thus recovering textiles in clothes should be approached differently to recovering textiles in furniture and so forth); and (iv) that holistic sustainability (not just recommending a particular end of life option) should be promoted. There was also consensus on the general approach of tackling the 'easy wins' first, so that effort can be targeted to where it is most effective. Specific blueprints of schemes were not explicitly suggested by the experts, though the following sections discuss their comments on different intervention options.

3.2.1. Do nothing

Allowing the market to act can be an effective means of change yet the option of *do nothing* was discussed very little in the focus group, despite it being a starting scenario and a relatively well represented stance within the interviews. This may be because of a bias in the sample where only those who had an interest in intervention possibilities that encouraged more biomaterial waste recovery chose to attend the focus group. In concurrence with the majority of the interviewees, the experts generally regarded that something needed to be done to stimulate more waste recovery and that the market alone was not able to bring about the necessary shift in increasing recovery rates.

3.2.2. Legislation

There were palpable concerns for the 'perverse consequences' of legislation, where good intentions can bring about unknown damage. Detailed discussions on the various legislative options that the experts identified are summarised below.

Targets set for recycling and energy recovery have been successful in the ELV directive. However, given the differing waste collection infrastructure, and that cars represent relatively valuable products compared to biomaterials, it was thought that recovery targets and the possibility of financial penalties would be unsuitable for the biomaterial industry.

Incentives were discussed positively for their ability to reward design for disassembly and purer products, especially important when consumers self-sort the products. Specific proposals such as tax relief or direct payments for 100% natural fibre T-shirts for example were not discussed, but the principle of incentives was preferred to that of setting targets.

Bans and taxes were thought to be a hostile form of legislation, though it was mentioned that they have been implemented in some EU member states to penalise those not engaging in biomaterial waste recovery. A case study in France was noted, where textiles companies must either pay a levy on each product they make to help cover the costs of recycling infrastructure, or they must directly fund a recovery scheme with a waste management partner company. The results of this trial were not published at the time of writing.[5] A blanket ban on certain biomaterials being sent to landfill was suggested in the focus group. However, it would be very difficult to differentiate between e.g. plastic and bioplastic bags, and this may result in inequality where biomaterials are penalised more than synthetic products.

[5] http://www.ecotlc.fr/.

Government procurement was suggested as means to stimulate demand for recovered biomaterial products. For example, all carpets and uniforms made from natural fibres could be required to be 'pure', easily recoverable, or sourced from recovered textiles. This proposal was popular in that it provided a relatively unobtrusive approach to legislation, while accommodating the freedom of the market to satisfy demand. It was also seen to assist economies of scale and add a degree of certainty within the market. Having a list of approved products has the appeal of simplicity and is already used by EU governments to ensure 'green procurement' exemplified by the UK Government's Buying Standards that ensure energy efficient appliances are preferred in government departments (European Commission, 2011). It follows that given a government lead, it could be more likely that other organisations would follow suit and apply choice editing to their operations.

3.2.3. Certification

Initially, focus group discussions demonstrated limited support for certification because it was felt that each biomaterial would need its own scheme. Multiple certification schemes were thought to introduce excessive complexity for consumers. In addition bio-based certification seen in the USA[6] that ensures a minimum percentage of biomaterial content in products fails to give an indication of potential contamination or the ease recovery or even the most appropriate method of recovery. Support nevertheless grew for the idea as discussions progressed and ideas such as using existing schemes like the European Union's eco label certification scheme were discussed. This scheme was already in the consumer landscape and provides an example of a single scheme that covered multiple products. This idea also appeased the requirement to be inclusive of wider sustainability issues which consumers would instinctively expect. Certification was also seen to work well with other complementary forms of intervention, especially government procurement. The inherent complexity of sustainability was mentioned as a potential problem for certification (especially when the purpose of certification is usually to promote single issues). However, it was suggested with little opposition that experts could set the standards behind the scenes and consumers would only need to see the 'logo'. Problems nevertheless remain with this approach; problems that were not mentioned during the focus group. These include the disempowerment of consumers, who may not be aware why a product has been certified. Also, situations may arise where products designed to be recovered easily may not achieve certification if they fall foul of other sustainability obstacles, which could be a disincentive for companies to 'play along'. In addition to not being discussed in the focus group, they were not raised when the experts were asked to comment on a post analysis summary, indicating they perhaps were not important.

One problem that was discussed was that it could not be guaranteed that consumers would actually dispose of their certified biomaterials appropriately. Certification was therefore suggested to be limited to issues such as purity not compostability, which has already seen to cause significant problems for the plastic bag industry. However, it was felt that certification could be effective if targeting the percentage purity or recycled content of a product, and if it is used in conjunction with other legislation (such as government procurement) along with improving access to recovery facilities.

3.2.4. Other intervention: more research

Beyond these scenarios other interventions were proposed in the focus group which can mostly be classified as calls for more research. Whether the source of funding should be from government or industry or a combination of both was not discussed. This section describes the types of research that were suggested would be needed prior to intervention.

Logistical knowledge and infrastructure was currently thought to be inadequate to support wider recovery of biomaterials, and research to quantify the amounts of waste for different biomaterials was perceived to be important. Companies do not currently know if they would be inundated with waste if recovery schemes were employed, or if a lack of material would make investment in recovery infrastructure futile. This information could be used in conjunction with research on the relative impacts of different end of life scenarios (recycle, energy recovery, producing fuel, etc.) to compile a list of preferred disposal options for common types of biomaterials, as well as enabling cost benefit analyses. It was thought this would assist the compilation of a list of 'easy wins' which would provide simplicity and help focus effort efficiently, being especially useful for government procurement.

A lack of *technical knowledge* was cited as an important challenge, and improving recovery technologies and capacities was thought to be vital in improving the quality and quantities of recovered biomaterials. An expert from the research sector had experience in running a demonstration plant to investigate new ways of dealing with waste biomaterials with companies who often were unaware of the possibilities. This participant also explained that the research facilities in the UK were still only functioning at a demonstration scale and although demonstration plants are widely used as a means of establishing proof of principle techniques and to improve collective knowledge, commercial companies were needed to invest to take infrastructure to the next useful scale. Once greater awareness and capability is established, costs are likely to fall, increasing the profitability of recovering biomaterials and the quantities consumed. Experts in involved in existing kerb side recycling nevertheless expressed concerns that even advanced recycling facilities and technologies struggle severely with contamination issues, so they may not be able to cope with mixed biomaterials. This hints that technical solutions may not be a panacea.

Public knowledge of the potential for recovering biomaterials was perceived to be low. It was suggested that the majority of consumers would "throw their old holey socks in the bin" without thinking, instead of taking them to a collection bank for reuse or recovery. It was suggested this was down to both limited availability of facilities but also a lack of understanding of the value of waste textiles as new fuels or new fabrics. An education campaign to widen this understanding was tentatively suggested but the unpreparedness of the waste and biomaterial industry to cope with large-scale collections meant that this idea was not thought to be suitable until the industry was better prepared.

In summary, several areas of consensus were identified regarding the design of a proposed intervention: it should be simple, product specific, have few burdens and be economically profitable. Schemes that were discussed are not necessarily mutually exclusive and it may well be advantageous to employ a multi-pronged approach to achieve maximum biomaterial waste recovery. The policy scenario "do nothing" received very little consideration unlike the other two scenarios. "Developing legislation" was seen to have many problems but it found some support where approaches were less strict. The final scenario "develop certification" also received positive comments and was thought to be a useful tool. In addition to evaluating the scenarios, this section has identified useful areas for future investigation. The following section outlines the recommendations that may be drawn from this research.

4. Recommendations

Despite the array of different biomaterial products and companies, and the diversity of comments and opinions collected, this

[6] http://www.biopreferred.gov/.

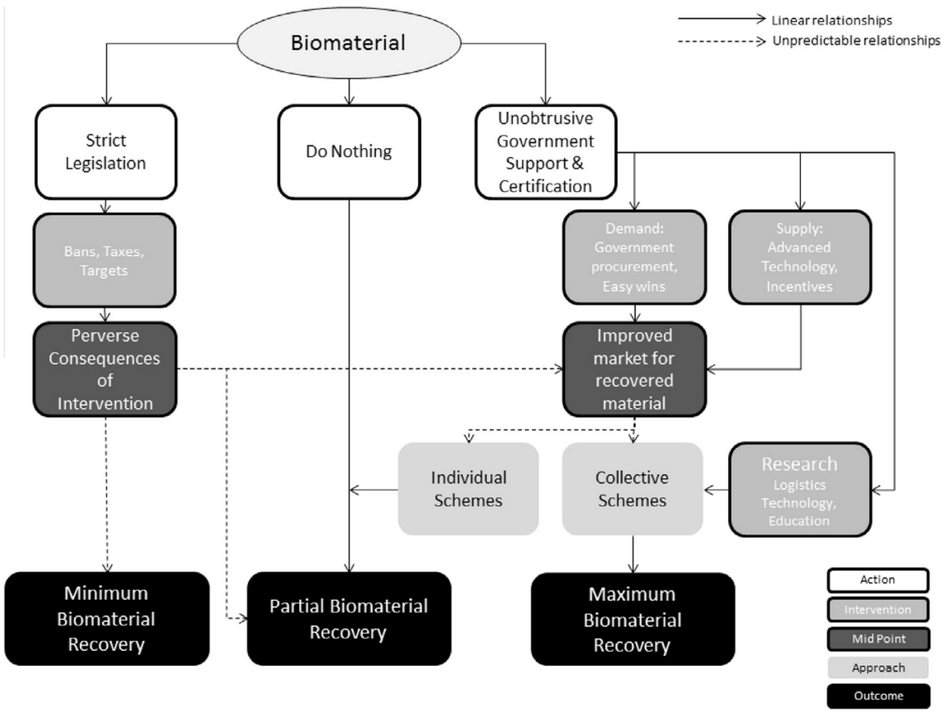

Fig. 4. Model for maximising biomaterial waste recovery.

research established a concrete foundation on which to encourage more biomaterials recovery through intervention. This is described in Fig. 4.

Fig. 4 describes the predicted outcomes; minimum, partial or maximum biomaterial recovery of the intervention scenarios, based on the focus group consensus. The "ideal" outcome of maximum recovery is shown to only be delivered by multiple interventions; promoting demand for pure biomaterials through government procurement or certification, increasing the supply of quality recycled materials by developing technology or introducing incentives and finally addressing logistical problems though industry agreements or legislation.

As can be seen, depending on the biomaterial, there may be no intervention required to achieve some amount of waste biomaterial recovery, though this is unlikely to maximise waste recovery. Fig. 4 also suggests that improving market conditions for recovered biomaterials may not in itself necessarily achieve the ideal outcome, since logistical and infrastructural issues can still be a barrier.

Strict legislation was less clear in its outcomes. There was uncertainty over the legislation trialled in France and yet it was an unpopular approach with both interview respondents and experts who predicted it should be a tool of last resort. It is likely that strict legislation may achieve some increase in recovery rates but that it is not the preferred route and so is shown to either produce minimum or partial recovery.

The model in Fig. 4 may be especially useful for companies or governments embarking on recovery schemes, as it identifies steps that could be taken (i.e. to improve supply, demand *and* logistics). It also highlights that although certain biomaterials may not require any form of intervention to promote recovery, in general, multiple unobtrusive interventions may be beneficial, and collaboration, especially regarding the logistics of a nationwide collection scheme, may underpin attempts to maximise biomaterial waste recovery in the industry as a whole.

5. Conclusions

This research has revealed that biomaterial recovery is not currently seen to be an important issue, even though biomaterial waste is highly likely to become more important in the future. Significant barriers to improving recovery rates have been identified which are not being adequately addressed by industry, indicating that some form of intervention may be required. This research has produced a model for policy and decision makers concerned with promoting biomaterial recovery. It suggests the policy scenario "do nothing" may not be appropriate for the entire industry despite its support from the minority already undertaking voluntary activities and that strong regulation such as taxation, fines and targets like those found in the WEEE and ELV directives may have limited and unpredictable success. This is due to the unknown potential market for recovered biomaterials, immaturity of technology and public attitudes, logistical difficulties in collecting biomaterial waste and contamination with synthetics. This research suggests that a lighter touch multi-pronged approach to boost supply through increasing purity of products and the capacity of recovery technology and to stimulate demand through certification or government procurement is perceived to offer an effective way to encourage more biomaterial waste recovery. In addition this study has found that simply influencing the market conditions may not be enough. It is vital in the case of biomaterials to organise and support recovery and collection infrastructure since the diversity of biomaterial products and their particular challenges make spontaneous solutions unlikely, even with a lucrative market.

Acknowledgements

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Appendix A. Interview respondent backgrounds

	Interview respondent role	Classification	Sector
1	Manager	Grower	Agriculture
2	Manager	Grower	Agriculture
3	Manager	Grower	Agriculture
4	Manager	Grower	Agriculture/ building materials
5	Consultant	Manufacturing	Carpets and textiles
6	Director	Manufacturing	Chemicals and plastics
7	Director	Manufacturing	Textiles
8	Consultant	Manufacturing	Building materials
9	Research and development	Manufacturing	Chemicals and plastics
10	CSR manager	Manufacturing	Automotive
11	Executive materials engineer	Manufacturing	Automotive
12	Senior sustainability manager	Manufacturing	Building and construction
13	Head of corporate social responsibility	Retail	Household and consumer products
14	Sustainability specialist	Retail	Household and consumer products

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Appendix III Concept Note for Potential Interviewees

Are End of Life Scenarios Necessary in Improving the Sustainability of the UK's Biomaterials Industry?

Why Biomaterials?

Biomaterials (products made from plants and animals) can be as simple as the textiles in clothes and furniture and the fibres that go into insulation and composite boards, or as complex as oils that make cosmetics and medicines or starches which can be turned into plastics. Their percieved benefits to society, beyond being from natural origins, are that they are said to be environmentally sustainable; usually meaning they have the potential to cause less damage to the environment than alternative products that are made from oil (petroleum).

What is not always clear is whether use of a biomaterial automatically makes a product more sustainable. Some biomaterials have even been shown to be more damaging to the environment than their oil equivelents. Considerable research is taking place to develop the framework and criteria to define what constitutes a sustainable biomaterial, which is where this project contributes.

Why End of Life Impacts?

This research project provides valuable data to the study of sustainable biomaterials in an area that is yet to receive much attention: end of life impacts. These refer to how the product is disposed of, which in other industries (electrical equipment and cars) has mature legislation devoted to it. Research has been already undertaken in the initial stages of this PhD to show that without considering end of life scenarios like recycling and converting organic wastes into vehicle fuels, the relative sustainability of a biomaterial compared to its oil equivalent may be put in doubt.

Identifying the attitudes and experience of major players in the UK's biomaterials industry is the next stage of this research and is vital if the industry is to develop and disseminate a holistic value of the sustainability of the biomaterials it produces.

Project aim

To identify the challenges and opportunities associated with incorporating end of life impacts into the UK biomaterials industry's practices.

Example of Biomaterial; Corn based plastic bottle (www.primabottle.com)

Example of Biomaterial: Wool insulation (www.thermafleece.com)

This project is part of a PhD funded through the White Rose University Consortium of Yorkshire

The White Rose Consortium's role is to ensure effective collaboration between the universities of Leeds, Sheffield and York.

Are End of Life Scenarios a Necessary Part of the Sustainability of the UK's Biomaterials Industry?

Research Process

1. Interviews

The research will involve interviews with growers, manufacturers and retailers in the biomaterials industry. Anonymity will be maintained if requested and there is no requirement to answer all or any of the questions. Interviews will last for less than 30 to 60 minutes and can be conducted at the farm, factory, office or store, or by telephone. All the interviews will take place during April–June 2012.

The interviewer will be the PhD student, David Glew. Participating interviewees should be a person within the organisation that has an understanding of the oragnisation's overall actvities. Ideally they should be involved with assessing the impact of the organisation's activities or reponsible for achieving compliance with environmental standards. This may include, for example, Farm, Sustainability or Compliance Managers.

2. Feedback Session

Following the interviews, all those who took part will be invited to attend a feedback meeting where there will be a presentation of the findings and chance to contribute any further thoughts. Being able to attending the feedback meeting is not a prerequisite for involvement in the initial interviews and written feedback will be provided to all interviewees after the feedback meeting whether they attended or not.

The meeting will be held in York in July 2012 over a half-day. Lunch will be provided.

Outputs

In addition to the feedback session, a paper will be produced that will be submitted for publication in an academic journal will be which will provide integrity and open access to the research.

Analysis of the data collected will provide information to policy makers on a national level and sustainability managers on an organisational level, identifying barriers to and opportunities to incorporating end of life impacts into the biomaterial industry's processes, as well as the risks of not doing so.

Example of Biomaterial: Film packaging
(www.organics-recycling.org.uk)

Academic Supervisors:

Dr Lindsay Stringer School of Earth and Environment, University of Leeds, UK

Professor Simon McQueen-Mason, Centre for Novel Agricultural Products, University of York, UK

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Appendix IV Post Analysis Summary for Interviewees

Industry views on how to encourage more waste recovery in the UK's biomaterial industry

1. What was the Research?

Biomaterials may prove to be useful alternatives to petrochemical alternatives, but it is less clear whether or not they are more sustainable and disposal options are highly influential in this regard. In this research a range of biomaterial feedstock growers, manufacturers and retailers were interviewed on their sustainability and waste recovery strategy and ethos. The results from this are presented here.

2. What did we find?

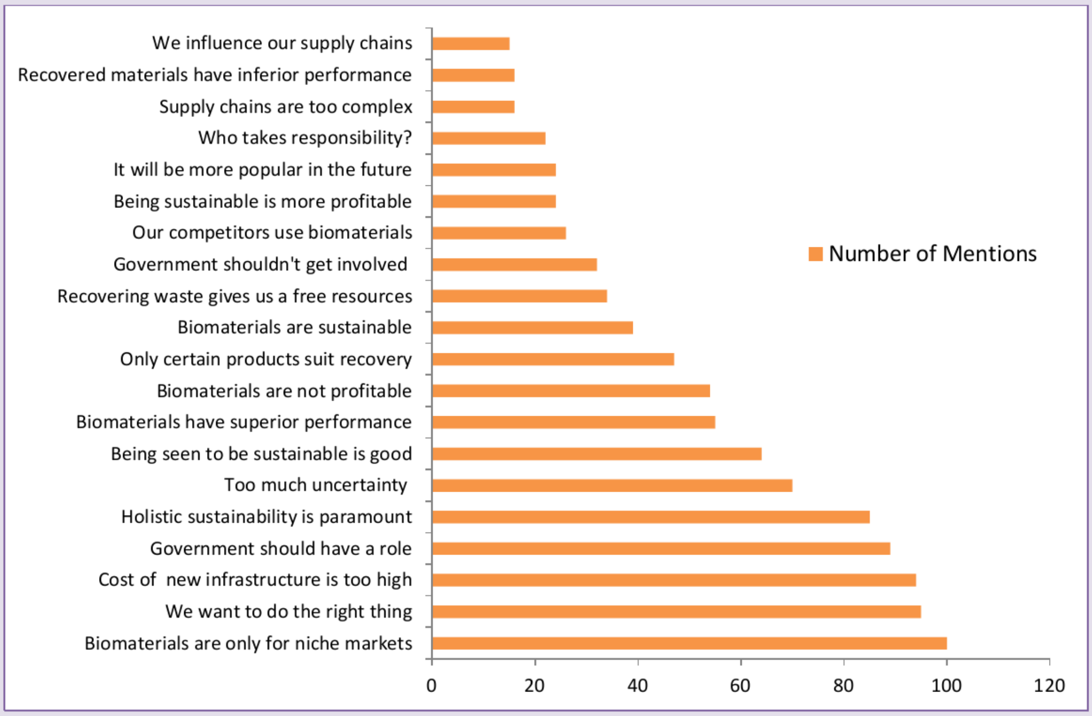
This research is qualitative and so all comments were grouped into umbrella topics with the aim of understanding what trends were emerging and why. Fig 1 and 2 are graphics used to illustrate the main topics that were mentioned in the interviews however they only refer to the number of times a topic was mentioned, not their rank order of importance.

3. Discussion

Several issues yielded consensus:

- Biomaterials are mainly for niche products that serve particular functions.
- Selling the story of sustainability is desirable though being local or recycled are more palatable for consumers than the idea of biomaterials.
- Biomaterials have great potential but the market is not yet ready.
- Few companies were currently making a profit out of recovering biomaterials.

Fig 1. Number of Mentions for Interview Topics



**This project is part of a PhD funded through the White Rose University Consortium of Yorkshire
The White Rose Consortium's role is to ensure effective collaboration
between the universities of Leeds, Sheffield and York.**

Industry views on how to encourage more waste recovery in the UK's biomaterial industry

Fig 2. Number of Mentions for Interview Topics per Company Type

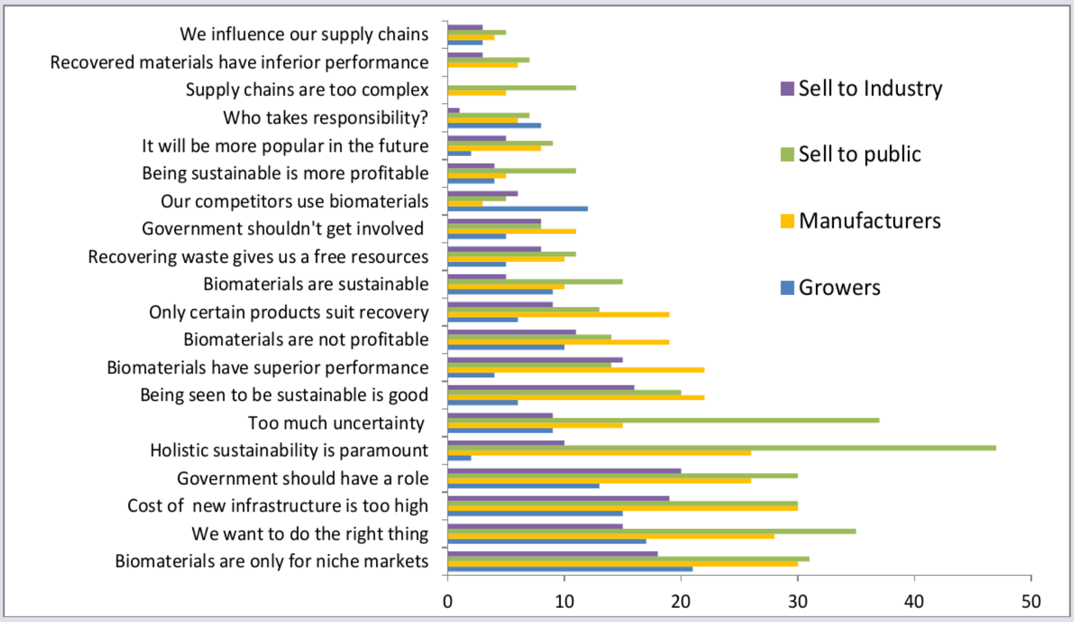


Figure 2 illustrates where there were some differences of opinions, below discusses several more:

- Recovering biomaterials has huge challenges and uncertainty, there were various levels of concern on issues like costs, potential volumes of waste materials, logistics of collection, profitability of recovered materials, degrees proven technology needed and who should take responsibility for collecting the waste.
- Specialist companies (making only a few different products) were not as cautious as multinationals in making claims about the sustainability of their products.
- Companies that had established some form of recovery scheme already were less inclined to want any regulatory pressure from government, whereas companies that were accustomed to operating in regulated markets were more positive about the potential of intervention.
- Companies were more confident about their ability to influence supply chains if it was small or relatively simple.
- Companies generally ranked waste recovery below other issues in the life cycle of their products. The certainty in this view was greatest for products that consume energy when used such as cars, houses and washing powders and which therefore have much large overall life cycle impacts than other products

4. Conclusion

In the interviews there were many areas of consensus especially that biomaterials will be very important in the future and waste recovery will become second-nature. How to get to this future point was less obvious. There was support and derision for ideas to use legislation, certification or rely on the market to recover more biomaterial waste with almost equal candour. This indicates that several different approaches may be best suited to several different biomaterial products and no overarching scheme is preferred.

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Appendix V Concept Note for Potential Experts

Encouraging more waste recovery in the UK's biomaterials industry

1. What is the Research?

It is clear that biomaterials may be a useful alternative resource to oil, but it is also known that they are not necessarily always more sustainable. Certification validating the chain of custody of timber and regulations on the GHG emissions of biofuels in the EU are just two examples of society's concerted efforts to steer towards a sustainable bio-based economy.

These schemes may be considered successful by various individuals and groups. However, there are inevitably still question marks over the sustainability of the majority of the many other biomaterials which fall outside of the remit of existing schemes. These include textiles in clothes and furniture, the fibres that go into insulation and composite boards, the complex oils that make cosmetics and medicines and the starches which can be turned into plastics.

This project funded by the White Rose University Consortium and based at the Universities of York and Leeds is undertaking research into more generic approaches to sustainability in the wider biomaterials industry. A range of biomaterial industry representatives have been interviewed ranging from the growers of feedstock and manufacturers of niche products in the UK to multinational corporations selling and sourcing biomaterials around the world.

The research focuses generally on the sustainability of biomaterials and specifically on a missing piece in the current landscape of sustainability and that which ties all these many disparate biomaterials together: their end of life impacts. The views of biomaterial industry representatives were recorded on these issues in a series of interviews by researchers at the university, especially on what they perceived the challenges and opportunities of sustainability to be.

2. Project aim

To identify the challenges and opportunities associated with voluntary schemes and policy that could be designed to support sustainable biomaterials.

3. Policy and Research Experts

Industry representatives were questioned on their understanding of the importance of recovering used biomaterial products on their company's overall sustainability. They also described their experiences with waste and environmental policy and with voluntary schemes that affect their businesses.

Policy makers and research institutions have a wealth of experience and it is therefore important to the research to arrange a focus group with policy and research institution representatives. Their critical appraisal of the interview findings and the industry generally can guide the research on appropriate ways to promote waste recovery in the biomaterials industry.

Participants at the focus group will be asked to critically appraise three possible pathways 1; creating new or adapting existing policy, 2; using certification schemes, 3: do nothing. These are explained further in the accompanying information sheet. The outcome of this focus group will ultimately be fed back to the interviewees at a later date

Example of Biomaterial; Corn based plastic bottle (www.primabottle.com)

This project is part of a PhD funded through the White Rose University Consortium of Yorkshire

The White Rose Consortium's role is to ensure effective collaboration between the universities of Leeds, Sheffield and York.

Encouraging more waste recovery in the UK's biomaterials industry

4. Research Process

You are invited to take part in this research which involves attending a focus group with other members from the policy and research communities concerned with biomaterials.

The meeting will be held in York in August 2012 over a half-day. Lunch will be provided. In addition to the meeting there will be the opportunity to have a guided tour of the new Biorenewables Development Centre (BDC) at the University of York (see attached leaflet for more information). The itinerary will be as follows:

11:00 Welcome, tea and coffee
11.30 Presentation of interview findings
12.00 Lunch
12.30 Attendees discussions
13.00 Presentation by attendees of discussions
13.30 Summary
14.00 End / Optional tour of BDC

The event will be hosted at the Biorenewables Development Centre (BDC) at the University of York www.biorenewables.org

In recognition of the cost of attending the day up to £100 towards travel costs is available to all attendees and may be claimed back by completing a claim form on the day.

Academic supervisors on the project:
Professor Simon McQueen-Mason, Centre for Novel Agricultural Products, University of York
Dr Lindsay Stringer, School of Earth and Environment, University of Leeds

Researcher: David Glew, University of York

For further information please contact David Glew:
email; dwg501@york.ac.uk
telephone; 01904 328787

5. Outputs

All comments made on the day will be kept anonymous and a journal paper will be produced as a result of the research that will be submitted for publication in an academic journal which will provide integrity and open access to the research. The work will also contribute towards a PhD Thesis.

The findings will provide information for policy makers on a national level and sustainability managers on an organisational level on the barriers and opportunities to incorporating sustainability (and specifically end of life impacts) in the biomaterial industry's operations, as well as outlining the risks of not doing so.

Example of Biomaterial: Film packaging (www.organics-recycling.org.uk)

Example of Biomaterial: Wool insulation (www.thermafleece.com)

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Appendix VI Interview Results Summary for Potential Experts

Introductory Information for the Policy and Research Focus Group on Waste Recovery in the Biomaterial Industry

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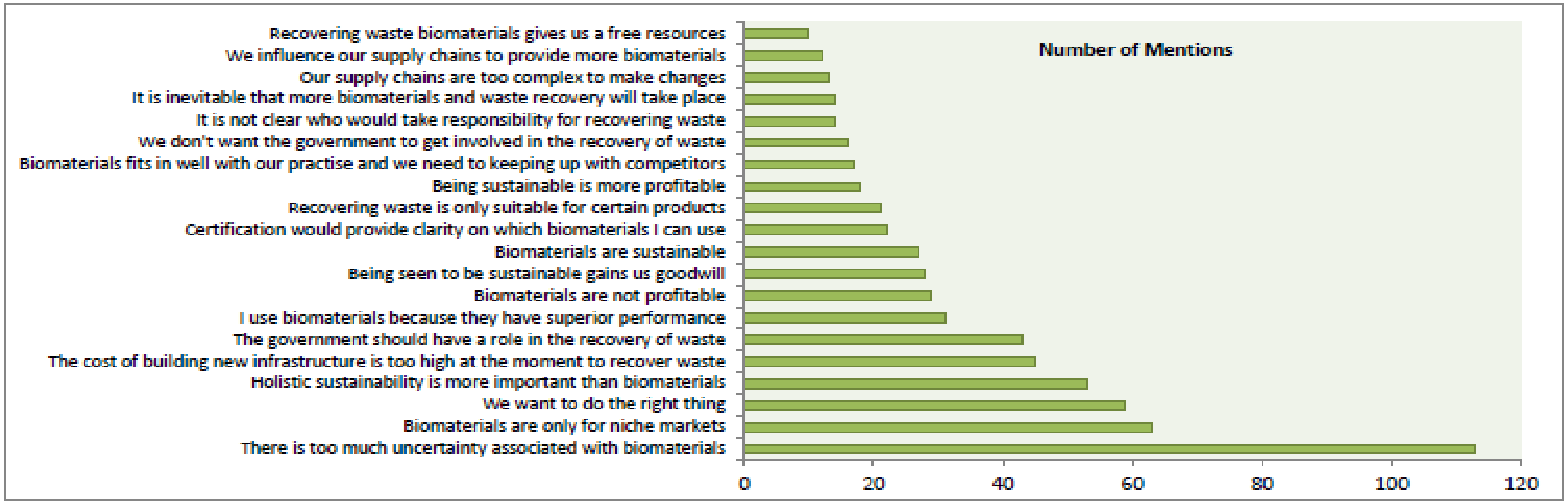

Table 1, Summary findings from biomaterial industry interviews; Key Themes Mentioned

	1. Expand or Develop New Legislation	2. Voluntary Certification Schemes	3. Do Nothing
For	The automotive industry is subject to waste regulations and has greatly increased its recovery of materials as a result. The certainty that legislation has brought has spurred on more innovation and could be successful in the biomaterial industry too.	There is a market for sustainable biomaterials that cannot be tapped because of uncertainty. Certification could provide clarity, inform the market and promote best practice within the biomaterials industry.	Change should be allowed to grow organically from within the industry without being hindered by external influences.
Against	There is no ready-made recycling industry to deal with logistical problems of collecting biomaterials as there was for metal in cars. Biomaterials are too diverse to have a one size fits all approach and legislation risks lumbering huge costs onto an emerging market.	Additional certification will confuse consumers adding more labels to already crowded packaging and will not guarantee customers will actually take part in waste recovery.	The costs of setting up a recovery program for mass biomaterial markets are prohibitive, collective burden sharing represents the highest possibility of success and needs some market intervention to make it happen.

Table 2, Indicative Arguments For and Against Proposed Schemes for Discussion at Focus Group

Appendix VII Summary of Focus Group Findings for Experts

Expert views on how to encourage more waste recovery in the UK's biomaterial industry

1. What was the Research?

Biomaterials may prove to be useful alternatives to petrochemical alternatives, but it is less clear whether or not they are more sustainable and disposal options are highly influential in this regard. In this research a range of biomaterial feedstock growers, manufacturers and retailers were interviewed on their sustainability and waste recovery strategy and ethos. The results from this were been presented via a focus group to experts on sustainability, waste, biomaterials, policy and consumer behaviour to identify the challenges and opportunities of intervention designed to promote more biomaterial waste recovery.

2. What did we find?

This research is qualitative and so all comments were grouped into umbrella topics with the aim of understanding what trends were emerging and why. Fig 1 and 2 are graphics used to illustrate the main topics that were mentioned in the interviews and focus group respectively, however they only refer to the number of times a topic was mentioned, not their rank order of importance.

3. Discussion

Several issues yielded consensus from the focus group experts. For example, interventions should target biomaterials according to their product type not as an overall group, thus, textiles should be treated differently to bio-plastics and so forth. Having product specific intervention rules out overarching biomaterial legislation but may help complement another commonly proposed theme; tackle the 'easy wins' first so that effort can be targeted to where it is most effective.

There may be some conflicts with these proposals however, and participants felt simplicity is key to the success of any intervention. Multiple schemes could introduce excessive complexity especially since many consumer labels already exist. Industry-led certification schemes were cited to already be in use in France for example, and well-designed legislation need not be complex.

Fig 1. Number of Mentions for Interview Topics

This project is part of a PhD funded through the White Rose University Consortium of Yorkshire
The White Rose Consortium's role is to ensure effective collaboration between the universities of Leeds, Sheffield and York.

Expert views on how to encourage more waste recovery in the UK's biomaterial industry

Fig 2. Word Cloud of Focus Group Topics Mentioned more than Once

There was no single favoured type of intervention though innovation around designing purer products that reduce contamination in disassembly stages was propounded by experts involved in the current waste management industry and engaged in wider sustainability research.

There were palpable concerns for the 'perverse consequences' of intervention of any kind and the availability of a profitable market for recycled biomaterials was deemed an essential starting point from which companies could compete and innovate. The angle of inquiry shifted in this regard to discuss how government procurement may be used to both stimulate demand for recycled biomaterials, as well as ensure a relatively unobtrusive approach to intervention.

One key issue identified in the interviews was 'insufficient knowledge' about how much material there was, how much it was worth or how it could be collected and used.

Experts similarly talked of more research being needed though mainly to prioritise and clarify preferred disposal options for different products and provide better technology to promote the quality and quantities of recycled materials. Experts were less confident in determining who should lead interventions or how.

Large retailers were recognised as having a major role, while a ban on sending biomaterials to landfill and a tax on products with biomaterial content that was reimbursed on its appropriate disposal were all briefly proposed, as were payments to incentivise companies to use recycled materials. Moreover specific blueprints of schemes were not explicitly suggested though the need to promote holistic sustainability (not just recommending a particular end of life option) was seen as an essential criterion to any scheme, especially one designed for biomaterials.

The final issue that raised notable collective concern was that of consumer behaviour and education. This was especially important to experts working with existing waste legislation. The potential to expand existing schemes like the European Union's Eco Label that are part of the consumers' existing sustainability landscape were thought to be good vehicles to achieve change (in conjunction with other complementary 'behind the scenes' schemes) and since experts would deal with the complexities of waste recovery and sustainability, consumer understanding may not necessarily be a barrier, though such advancements could take some time yet.

4. Conclusion

There were several areas of consensus over the design of a proposed intervention; it should be simple, product specific, and profitable, yet there is much wiggle room. Schemes are not necessarily mutually exclusive and it may well be advantageous to employ a multi-pronged approach to achieve maximum bio waste recovery.

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Appendix VII, As submitted to Journal of Industrial Ecology

Biomaterials and environmental sustainability: Predicting disposal-stage GHG emissions via a harmonised life cycle assessment tool (HELCA)

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Summary

The carbon footprint of biofuels and biomaterials is one of the major barriers to their acceptance, yet the greenhouse gas (GHG) emissions associated with disposing of biomaterials are frequently omitted from analyses. This paper develops a harmonised life cycle assessment (LCA) tool to identify the importance of end of life scenarios. There is a precedent for harmonised LCA industry tools in European biofuels legislation and it is used here to quantify the GHG emissions of two hemp biorefineries and rank disposal options according to their GHG emissions and cost effectiveness. In general the greatest GHG emission benefits are obtained through reuse and recycling is preferred in combination with incineration and combined heat and power (CHP) generation. Using energy onsite also reduces GHG emissions significantly and anaerobic digestion with CHP, ethanol conversion and anaerobic digestion with electricity generation have the next largest GHG reductions above composting. Despite its emissions savings, incineration is among the most costly ways of reducing GHG emissions, along with composting. The net costs of anaerobic digestion are negligible and onsite energy production and ethanol conversion may provide net revenue. Incorporating disposal emissions and ranking between scenarios in the tool presented here gives credibility to complete GHG assessments. It provides benefits for companies who want to promote disposal options to customers and it may direct them towards more easy to deconstruct designs. Our tool could also allow policy makers to understand the consequences of different waste disposal options and thereby help ensure waste policy is GHG and cost efficient. These themes will become increasingly important in a future bio-based economy.

Keywords

Waste, Life Cycle Assessment (LCA), Greenhouse Gas (GHG), Hemp, Biorefineries, Disposal

1 Introduction

Predictions show that nations are generating more greenhouse gas (GHG) emissions, in the case of the UK consumption based emissions could increase by around 30% (Barrett and Scott, 2012). Companies are held responsible for their environmental and social performance (Seuring et al., 2008). In some instances GHG emission reductions are required for legislative targets, certification, or simply for a company to be seen to be responsible (Golden et al., 2010, Lynes and Andrachuk, 2008, Nawrocka et al., 2009, de Boer, 2003, Pawelzik et al., 2013). This is especially true of biomaterial products like natural fibres, bioplastics and biofuels (Ragauskas et al., 2006) which have bad press regarding their GHG emissions (Searchinger, 2010, Gallagher, 2008). The biomaterials market in the UK alone may triple over the period 2012-2015 (NNFCC, 2012) exemplifying European growth (Carus, 2012) so the need to ensure biomaterial products are 'low carbon' is pressing.

Traditionally biomaterial carbon footprints focus on those emissions associated with the production of feedstock (Black et al., 2011, Cherubini and Jungmeier, 2010, Seguin et al., 2007, Falloon and Betts, 2010) for which life cycle assessment (LCA) is the preferred tool (Fahd et al., 2011, Cherubini and Ulgiati, 2010, Harding et al., 2007), providing holistic supply chains evaluations (Roy et al., 2009, Acquaye et al., 2012b, Guinee et al., 2001). Few studies consider emissions from biomaterial disposal though GHG emissions can be reduced via the waste hierarchy through reusing, recycling and energy recovery (UNEP, 2010, Wang et al., 2012, Stichnothe and Azapagic, 2009, Shen et al., 2010, Glew et al., 2012, Ross and Evans, 2003).

The disposal end of a product's life is less-well studied because: 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 if a company cannot claim benefits (of making GHG reductions), they have little motive to design for deconstruction (Nemoto and Goto, 2004); 2) LCA can be a costly undertaking and carbon footprinting is not always well received (Golden et al., 2010); 3) there is no approved methodology for measuring disposal emissions (Ekvall and Tillman, 1997) leaving a company vulnerable to criticisms or competitors commissioning alternative, more favourable assessments; 4) LCA are usually product-specific and not used to compare emissions across a range of products; and finally 5) consumers may not chose to dispose of products in preferred, low-carbon ways. Despite these barriers research suggests there may be demand for industry guidelines on low carbon disposal for biomaterials (Glew et al., 2013).

This paper presents a consequential harmonised LCA tool taking its lead from the EU Renewable Energy Directive's (RED) GHG calculations for biofuels. RED deliberately simplifies and standardises the LCA procedure, adopting a range of standardised assumptions on system boundaries and allocation procedures and using default values for controversial or difficult to calculate inputs such as land use change (LUC) factors and impacts from improved agricultural practices. There are two important benefits to harmonisation: 1) to include small producers who cannot fund a full LCA in the regulated EU biofuels market and 2) to set GHG targets so allowing comparisons between companies and feedstock performance. This solves some barriers identified above however, as shown in Figure 1, it does so at loss to exactitude.

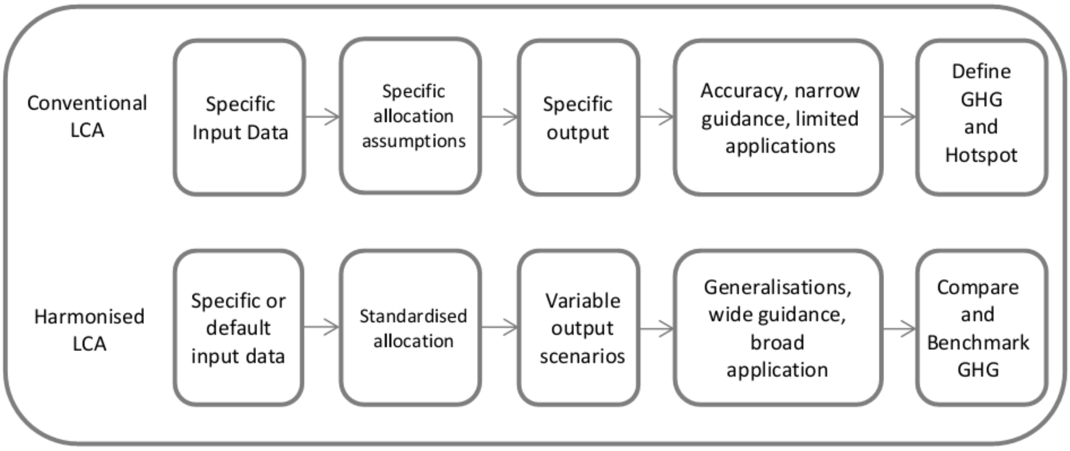

Figure 1 Comparison of Conventional and Harmonised approaches to LCA

Choosing system boundaries, allocation methods and data sources in conventional LCA benefits accuracy and enables a range of specific problems to be solved such as locating energy hot spots (Koh et al., 2012, Acquaye et al., 2012a). Since variations in the assumptions of an LCA can have profound impacts on its results, being able to manipulate inputs in is a double-edged-sword, allowing greater refinement but reducing pertinence (Whittaker et al., 2011, Cherubini, 2010b, Wiedmann, 2009). Thus, harmonisation was ultimately preferred in RED because it allowed benchmarking and multilateral targets to be implemented.

The principle of harmonisation is generally accepted as useful and some would like to see it extended (Schlegel and Kaphengst, 2007). The philosophy is adopted in this research; we have developed a LCA tool that companies could use to understand the best disposal options for their products without commissioning a full LCA and so results may be compared to others with confidence that the same allocation methods, data sources and system boundaries are used. Our tool takes a 'Harmonised' approach to consider 'End of life' impacts using 'LCA' and is henceforth referred to as 'HELCA'.

Decision support tools are interactive systems that produce data and information to promote understanding and problem solving (Georgilakis, 2006). HELCA is intended to be used to provide evidence by gathering, processing, analysing and presenting data on product end of life scenarios. In addition data on costs and revenues of end of life scenarios collected from the literature are included to indicate the cost-effectiveness of reducing GHG emissions by different disposal options. This may aid sound decision making by prompting companies to change their design processes to align with particular disposal options and inform customer guidance regarding low carbon disposal of their products. It may also allow policy makers to predict GHG emissions reductions of disposal stages of biomaterials and advise where to direct funding into disposal technologies to reduce GHG emissions.

The term 'biomaterials' captures a diverse range of products such as natural fibres, oils and bio plastics which are touted as sustainable alternatives to petrochemicals (Carus, 2012, Vandermeulen et al., 2012). This paper investigates a European hemp (*Cannabis sativa*) biorefinery as an archetypal up-and-coming biomaterial producing a range of products from paper and plastics to textiles, fuel and food (van der Werf and Turunen, 2008) and provides a good context for demonstrating the use of HELCA.

2 Method, LCA Goal and Scope

LCA frameworks (ISO, 2006) first set the study's objective, system boundary and functional unit (1: Goal and Scope) then quantifies supply chain inputs (2: Life Cycle Inventory) and applies environmental burdens to these (3: Life Cycle Impact Analysis) before interpreting the results (4: Interpretation). There is no requirement to consider disposal although it is recommended (Ekvall and Tillman, 1997). Conversely in HELCA it is only the inputs and outputs of disposal that are tested. GHG emissions are used in HELCA since they are commonly cited environmental impacts and data are readily available for emissions caused by disposal (UNEP, 2010, European Commission, 2001).

The goal of an LCA defines its detail and application (ISO, 2006). HELCA's goal is to predict emissions resulting from different end of life scenarios for biorefinery feedstock. HELCA does not investigate GHG embedded in sourcing feedstock or the nuances of farming practices, this is examined extensively elsewhere in the literature (Hoogwijk et al., 2003, Khoo et al., 2010). In addition inclusion of such considerations would not allow a true comparison between disposal options' GHG emissions of two different biorefinery systems since only one variable (i.e. disposal) should be altered at a time in a fair test. . Having said this, it may be useful for a company to add cradle to gate GHG emissions to HELCA to contextualise disposal GHG savings, although this is not the purpose of this research.

2.1.1 Functional Unit and Allocation

Mass is a useful unit to measure GHG emissions against and is suitable for most biomaterial co-products since it gives an indication of their relative importance. One exception is high-value, low-quantity chemicals and oils, where an economic unit may be more appropriate, though this can lead to a phenomenon where different co-products appear more or less polluting if their price changes. Mass is unchanging and if desired can be converted to cost or energy (if fuels are a co-product) hence, the functional unit in HELCA is "$gCO_2eq\ per\ kg\ of\ feedstock$".

HELCA uses kg of feedstock not kg of co-product because it considers holistic biorefinery emissions and does not treat any co-product as a 'lead' since modern biorefineries produce a spectrum of marketable products all providing some revenue (Cherubini, 2010a). Lead products complicate allocation and weighting amongst co-products which is much debated in LCA (Finnveden et al., 2009), standardising this in HELCA removes this problem.

2.1.2 HELCA Outputs and Calculations

Decision support tools are valued for their ability to compare scenarios (Hubacek et al., 2009), in HELCA it is possible to change the amount and type of co-products assumed in the biorefinery as well as altering the disposal scenarios to assess how GHG emissions change. Default emissions savings data applied to each disposal scenario adopts the 'system expansion' approach, providing data on additional outputs of disposal and crediting these as GHG savings, for example by avoided emissions from energy-from-waste or recycling (Ekvall and Tillman, 1997). In addition HELCA uses default costs and revenues data to calculate the cost effectiveness of each disposal option at reducing GHG emissions. Figure 2 presents the main stages that make up HELCA.

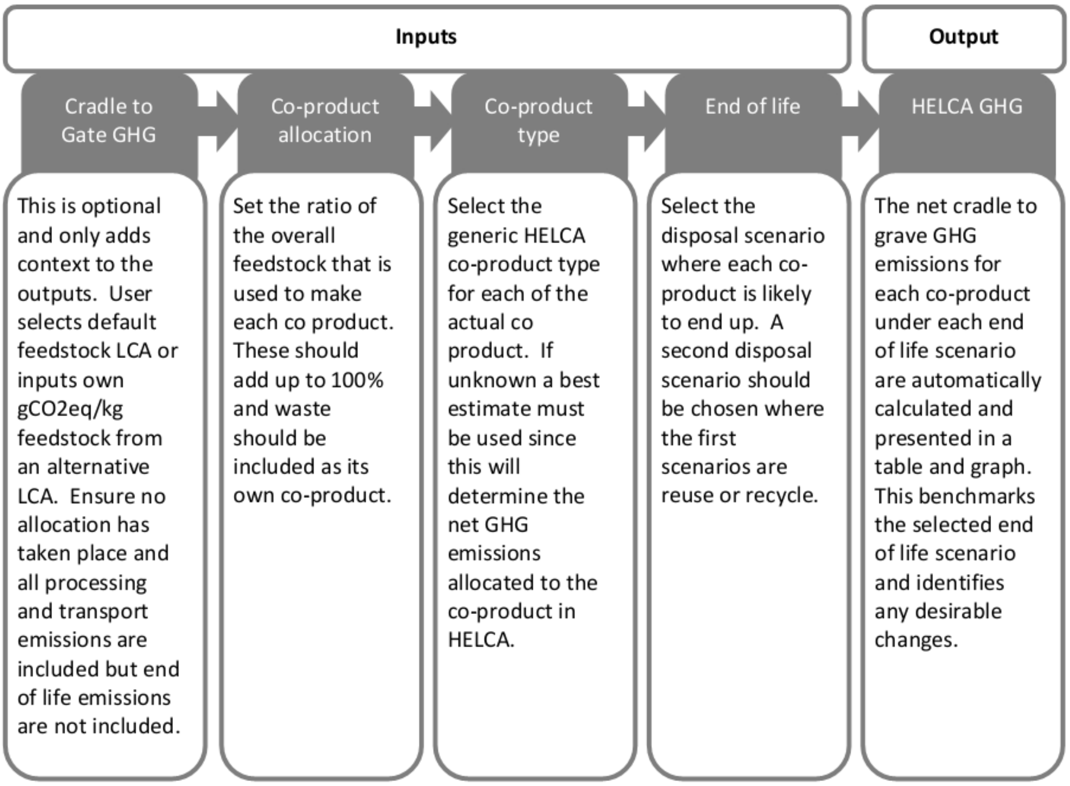

Figure 2 Flow chart of HELCA data inputs and outputs

HELCA is based on the following equation:

$$HELCA = \{(a_1 \times b_1) + (a_2 \times b_2) + (a_3 \times b_3) + (a_n \times b_n) \ldots\}$$

Where: a = Co-product$_{1, 2, 3, n\ldots}$ % of total feedstock yield (allocated by mass)

b = End of life net GHG emissions (gCO_2eq/kg co-product$_{1, 2, 3, n\ldots}$)

The optimum disposal scenario in terms of GHG emissions and cost effectiveness is automatically identified and ranked comparing GHG emissions if all the products were sent to landfill or energy from waste plants for example. This inherent ranking gives a complete picture of disposal's influence so users can quickly see which disposal options are most sustainable in terms of carbon emissions. The HELCA tool can be viewed in the supporting information for more details.

2.2 Life Cycle Inventory: Data Collection

2.2.1 Cradle to Gate LCA

HELCA only calculates GHG emission resulting from disposal. However, a feedstock cradle to gate LCA may be added to contextualise the findings. In this paper emissions for growing hemp in Europe assume 1kg of hemp fibre results in 1,600 gCO_2eq (González-García et al., 2010), weaving 1kg of fibre results in 406.7 gCO_2eq[1] and assuming that the fibres are only 33% of the hemp feedstock (EIHA, 2012) a total of 662.2 gCO_2eq per kg of hemp may be emitted in the cradle to gate stage. Ecoinvent[1] was selected to provide GHG values for weaving of bast fibres as it is a widely used LCA database though does not include data on the cultivation of hemp so an alternative value was used (Frischknecht and Rebitzer, 2005). Multiple data sources raise the possibility of different system boundaries and assumptions, double counting or omitted information adding uncertainty however this value is only used to contextualise HELCA's findings on disposal emissions.

A limitation with these data is that they assume identical processing inputs regardless of the outputs from the biorefinery. In reality, it is likely that when more ethanol is made instead of more paper for example, different amounts of energy will be consumed and therefore different amounts of GHGs will be emitted by the biorefinery. However, HELCA is concerned only with the end of life emissions so this is not deemed a significant limitation and biorefinery operators may use their own input data to overcome this.

2.2.2 Biorefinery Output Scenarios

The choice of co-products made in a biorefinery is often dictated by profit and shifts in market prices stimulate different outputs (Dornburg et al., 2006). HELCA allows users to alter co-product ratios to investigate resulting changes in GHG emissions that accompany such shifts in co-product ratios. This study tests the influence of co-product choice via two hypothetical biorefineries A[2] and B[3], shown in Table 1.

[1] http://www.ecoinvent.org/database/
[2] www.cetiom.fr
[3] www.eiha.org

Table 1 Co-Products of Hemp Biorefineries

Hemp Biorefinery A	HELCA	Mass	Hemp Biorefinery B	HELCA	Mass
Bird Feed	Food /	10%	Flower Essential Oils	Food /	8%
Dust	Lignocellulosic	9%	Seed Feed & Food	Food /	7%
Fish Feed	Food /	1%	Dust	Lignocellulosic	2%
Plastics	Bio Plastic	1%	Shivs for Animal Bedding	Textiles /	29%
Paper	Paper	19%	Shivs in Construction	Textiles /	7%
Insulation	Textiles /	7%	Fibres in Bio Composites	Bio Plastic	4%
Animal Bedding	Textiles /	34%	Fibres in Insulation	Textiles /	7%
Composite Boards	Textiles /	15%	Fibres in Mulch	Textiles /	1%
Waste	Putrescible /	4%	Fibres in Pulp and Paper	Paper	15%
	Total	100%	Waste	Putrescible /	21%
				Total	100%

Data on GHG emissions caused by disposal of biomaterials is not comprehensive, so categories in Table 1 have been developed to aggregate different biomaterials under umbrella categories for which GHG emissions data is available. Deciding in which category a product should be placed is highly influential to the ultimate GHG emissions that HELCA will allocate to it and. No systematic way to make these distinctions exists so the user must make a competent selection and this represents an area of uncertainty. The difference between GHG emissions of each category is presented in Table 2

Table 2 GHG Emissions from End of Life Scenarios (gCO2eq/kg Co-product)

HELCA Category	Textiles / Fibres	Putrescible / Biowaste	Paper	Bio Plastic	Food / Consumable	Lignocellulosic Waste
Landfill	46	762	255	187		349
Landfill with gas recovery for electricity	30	730	223	171		327
Recycling	10		10	10		
Reuse	34		34	34		
Incineration for Electricity	-303	-66	-235	205		-10
Incineration for CHP	-880	-224	-691	-426		-348
Ethanol Production	-47	-81	-72			-47
Anaerobic Digestion for Electricity		-104	-104			
Anaerobic Digestion for CHP		-185	-184			
Compost	-37	-37	-37	-29		-37
Onsite Heat / Briquettes		-690				-690
Onsite Electricity		-1010				-1010

2.2.3 End of Life GHG Scenarios

Ten disposal scenarios are considered by HELCA chosen to align with the waste hierarchy and incorporate innovative up-and-coming as well as mainstream disposal technology. The disposal GHG values for different co-products are also shown in Table 2. These are sourced predominantly from the European Commission's (EC) report on waste and climate change (European Commission, 2001) unless otherwise stated. The EC report includes Europe-wide average emissions for transport and processing and credits energy recovered and recycled products, both of which avoid consumption of virgin resources elsewhere (system expansion). The EC's data were used since they represent a transparent reference point from a trusted source and are relevant to the geographical location of this study.

Bioplastics data were not given in the EC's report, instead, data are provided by Madival et al. (2009) although no data for bioplastic conversion into ethanol could be found. All ethanol conversion data are based on Schmitt et al. (2012) where 'Lignocellulosic Waste' and 'Textiles / Fibres' are considered to be similar to woody organic 'Yard Waste' and 'Putrescible / Biowaste' is assumed to equate to 'Municipal Solid Waste'. It is worth noting that 'Textiles / Fibres', 'Lignocellulosic Waste' and 'Bioplastics' are not suitable for compost or anaerobic digestion (AD) as stated in UK guidelines (Environment Agency, 2008a, Environment Agency, 2008b) and neither 'Waste' categories are suitable for recycling or reuse. Using different data sources is a decision taken out of necessity due to lack of a comprehensive data set and so must be accepted as an area of uncertainty. This paper thus highlights the need for the development of such a dataset should disposal emissions be given a significant policy platform. These values underpin HELCA's output, so conclusions should be interpreted with reference to these.

Emissions associated with product reuse are calculated using the EC's reported default transport and mobilisation emissions (10gCO_2eq/kg) to account for collecting the products and distributing them to new users. Emissions associated with recycling also include this in addition to emissions from processing energy (24gCO_2eq/kg) to convert the used items into new products. Emissions saved are equivalent to the cradle to gate GHG emissions used in HELCA to make the virgin product. If it was decided not to include a cradle to gate LCA then EC default data for emissions savings achieved by reuse and recycle may be used. HELCA assumes that recycling neither downgrades the product, nor offsets the production of more polluting products.

Although it is technically feasible to burn all the co-products to produce energy, data for onsite energy production are only allowed in HELCA for 'Lignocellulosic Waste' and 'Putrescible / Biowaste'. If co-products were all burned the economic incentive for the biorefinery would be lost. Onsite energy emissions are calculated using the lower heating value of hemp, 15.9MJ/kg (Prade, 2011). A conversion efficiency of 85% is set for heat production[4] (European Commission, 2010) and a power to heat ratio of 1:3 is used[5] to calculate the productivity of onsite electricity production (EPA, 2002) to offset 0.2407 kgCO_2eq per kWh for gas and 0.5246 kgCO_2eq per kWh for electricity (Carbon Trust, 2011).

The data presented in Table 2 are averages and default values to achieve harmonisation. Using these as exact values may therefore be misleading. For example, contamination in compost will affect the savings stated here, similarly, differences in disposal infrastructure efficiencies across the EU will affect actual savings achieved. Another limitation are the broad products classifications, for example there is no distinction between 'cardboard' and types of 'paper' and there are no data for 'Bio-composite', 'Insulation' or some other hemp co-products. This is due to lack of data to allow further refinement of categories, thus, these are useful to guide decision making where the exact values are not the main aim, though the defaults can be updated as more accurate data are published

Table 3 shows typical UK disposal fates for each co-product (DEFRA, 2011). It suggests the majority of UK landfill sites operate without gas recovery[6] and the most common waste sent to incineration plants (usually electricity generation only) is municipal solid waste (MSW) including 'Other / Waste' and 'Bio-plastics'. 'Paper' is shown to be recycled more often than any other product so is the only material with this end of life scenario followed by incineration. All non-MSW, like building materials are sent to landfill and all agricultural and horticultural products are assumed to be composted.

[4] Middle of range estimate for district heating conversion efficiency
[5] Small steam CHP assumed
[6] http://www.environment-agency.gov.uk/cy/ymchwil/llyfrgell/data/34423.aspx

Table 3 Typical UK End of Life Scenarios for Hemp Products

Hemp Biorefinery A	Selected End of life Treatment	Hemp Biorefinery B	Selected End of life Treatment
Bird Feed	n/a	Flower Essential Oils	n/a
Dust	Briquettes for heat	Seed Feed	n/a
Fish Feed	n/a	Dust	Briquettes for heat
Bio-Plastics	Incineration with Electricity Recovery	Shivs for Animal Bedding	Composted
Fibres for Paper	Recycled followed by Incineration with Electricity Recovery	Shivs in Construction	Landfill
Fibres in Insulation	Landfill	Fibres in Bio composites	Landfill
Shivs for Animal Bedding	Composted	Fibres in Insulation	Landfill
Fibres in Bio composites	Landfill	Fibres for Mulch	Composted
Other / Waste	Incineration with Electricity Recovery	Fibres for Pulp and Paper	Recycled followed by Incineration with Electricity Recovery
		Other / Waste	Incineration with Electricity Recovery

2.2.4 Cost Effectiveness

Data on costs of waste treatment are derived from the UK Waste Resource Action Plan's (WRAP) 2012 report on costs of alternative waste treatment (WRAP, 2012a). Where WRAP does not publish data, other sources are referenced in Table 4. It is worth noting 'reuse' and 'onsite energy production' currently assume no costs are incurred, since no data could be found.

Table 4 Costs and Revenues of Treating 1kg of Feedstock via Different Disposal Scenarios

	Worst Case (£)			Best Case (£)		
	Cost	Revenue	Balance	Cost	Revenue	Balance
Landfill	0.1270	0.00	-0.1270	0.0720	0.00	-0.0720
Landfill with Energy Recovery	0.1270	0.0009	-0.1261	0.0720	0.0009	-0.0711
Reuse	0.0000	0.0400	0.0400	0.000	0.5200	0.5200
Recycling	0.1450[7]	0.0400	-0.1022	0.0550[7]	0.5200	0.4650
Incineration Electricity	0.1310	0.0288	-0.1022	0.0560	0.0288	-0.0272
Incineration CHP	0.1310	0.0480	-0.0830	0.0560	0.0480	-0.0080
Ethanol	0.0906[8]	0.0950	-0.0045	0.0350[8]	0.1163	0.0801
Anaerobic Digestion Electricity	0.0600	0.0130	-0.0470	0.0350	0.0130	-0.0220
Anaerobic Digestion CHP	0.0600	0.0245	-0.0355	0.0350	0.0245	-0.0105
Composting	0.0600	0.0000	-0.0600	0.0150	0.0100	-0.0050
Onsite Heat	0.0000	0.0717	0.0717	0.0000	0.0717	0.0717
Onsite CHP	0.0000	0.0986	0.0986	0.0000	0.0986	0.0986

The EC publishes regular quarterly data on energy prices used to calculate revenue data in HELCA as in Table 4. It assumes 1 kWh of electricity and natural gas generates £0.0405[9] and £0.0162[10] respectively based on a Sterling to Euro exchange rate of 0.81[11] and that Renewable Obligation Certificates (ROC) were available for onsite electrical production at £0.027 per kWh[12]. Prices for ethanol are closely related to corn and wheat prices, which fluctuate greatly. A study undertaken by the financial markets information providers Kingman suggests 650 Euro per cubic meter (£0.40 per litre) is reasonable[13]. Embedded in the EC data are energy yield values of 427, 1185, 120, 286 kWh per tonne of waste for Incineration Electric, Incineration Heat, AD Electric and AD Heat production respectively (European Commission, 2001). Best and worst case ethanol yields listed in HELCA of 237 and 249 litres per tonne are taken from work by Littlewood (2013). The revenue data for recycling and composting shown in Table 4 are taken from a report commissioned for Friends of the Earth, UK Waste and Waste Watch and it is assumed that reuse yields the same revenue as recycling (ECOTEC, 2000).

[7] WRAP 2012b. Materials Pricing Report: full listings August 2012 – week 2. *In:* PLAN, W. R. A. (ed.). Waste Resource Action Plan.
[8] Littlewood et. al (2013)
[9] ec.europa.eu/energy/observatory/electricity/doc/qreem_2012_quarter1.pdf
[10] ec.europa.eu/energy/observatory/gas/doc/qregam_2011_quarter4_2012_quarter1.pdf
[11] Based on market rates on 09/01/2013
[12] REA 2008. Energy from Waste – A Guide for Decision Makers. *In:* ASSOCIATION, R. E. (ed.). Renewable Energy Association.
[13] http://www.kingsman.com/ last accessed January 10th 2013

The HELCA tool with all its default data and assumptions and a guidance booklet can be accessed in the supporting information.

3 Results and Discussion: Life Cycle Impact Analysis and Interpretation

Table 5 presents the GHG emissions of two hemp biorefineries assuming UK typical disposal scenarios in addition to the complete range of disposal possible scenarios in turn. Highlighted numbers represent the lowest GHG disposal option for each co-product. In general it is clear to see that larger co-products by mass such as 'Paper' and 'Animal Bedding' have the biggest impact on emissions. The choice of co-products alters the GHG emissions of the biorefinery because it opens up possibilities for different end of life scenarios to be exploited. The main points of interest in this research surround the change in GHG caused by disposal. For example it is shown in Table 5 that if onsite energy production was maximised, both biorefineries may be able to operate 'carbon neutrally' though this will depend on the contextualising cradle to gate LCA (if one has been selected).

Table 5 End of Life GHG Emissions for Hemp Biorefinery A and B (gCO2eq/kg)

	Co-Product	Cradle to Gate	Typical UK Case	Landfill	Landfill with Energy Recovery	Reuse	Recycling	Incineration Electricity	Incineration CHP	Ethanol	Anaerobic Digestion Electricity	Anaerobic Digestion CHP	Composting	Briquettes / onsite heat	Onsite CHP
A	Bird Feed	66	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dust	60	-62	31	29	0	0	-1	-31	-4	0	0	-3	-62	-91
	Fish Feed	7	0	0	0	0	0	0	0	0	0	0	0	0	0
	Bio Plastics	7	2	2	2	0	0	2	-4	0	0	0	0	0	0
	Fibres in Pulp and Paper	126	-62	48	42	-22	-17	-45	-131	-14	-20	-35	-7	0	0
	Fibres in Insulation	46	3	3	2	-3	-1	-21	-62	-3	0	0	-3	0	0
	Shivs for Animal Bedding	225	-13	16	10	-73	-65	-103	-299	-16	0	0	-13	0	0
	Fibres in Bio Composites	99	7	7	5	-13	-10	-45	-132	-7	0	0	-6	0	0
	Waste	26	-3	30	29	0	0	-3	-9	-3	-4	-7	-1	-28	-40
	Gross Additional GHG	0	-127	138	120	-111	-93	-216	-669	-47	-24	-42	-33	-90	-131
	Net GHG emissions	662	535	800	781	551	569	446	-7	615	638	620	629	572	531
B	Flower Essential Oils	53	0	0	0	0	0	0	0	0	0	0	0	0	0
	Seed Feed & Food	44	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dust	13	-14	7	7	0	0	0	-7	-1	0	0	-1	-14	-20
	Shivs for Animal Bedding	192	-11	13	9	-53	-46	-88	-255	-14	0	0	-11	0	0
	Shivs in Construction	46	3	3	2	-3	-1	-21	-62	-3	0	0	-3	0	0
	Fibres in Bio Composites	26	2	2	1	-1	0	-12	-35	-2	0	0	-1	0	0
	Fibres in Insulation	46	3	3	2	-3	-1	-21	-62	-3	0	0	-3	0	0
	Fibres in Mulch	7	0	0	0	0	0	-3	-9	0	0	0	0	0	0
	Fibres in Pulp and Paper	97	-44	37	33	-13	-9	-34	-101	-10	-15	-27	-5	0	0
	Waste	137	-14	158	151	0	0	-14	-46	-17	-22	-38	-8	-143	-209
	Gross Additional GHG	0	-74	224	205	-71	-56	-194	-577	-51	-37	-65	-32	-157	-229
	Net GHG emissions	662	588	886	867	591	606	469	85	612	626	597	631	506	433

3.1.1 Impact of Disposal on GHG Emissions

'Feed' and 'Food' are consumable so have no end of life emissions and the two landfill scenarios have positive emissions whereas the other disposal options show negative emissions representing a reduction in life cycle emissions. The 'Typical UK' case in both biorefineries shows reduction of around 10% to 20% (depending on the co-products selected) of the cradle to gate GHG emissions, and reductions of up to a third compared to landfill emissions. This indicates that it is disingenuous and deleterious to the biomaterials cause to undertake carbon footprinting without reference to disposal emissions. The 'Typical UK' case is well short of the respective 'Best Case' shown to reduce emissions by 90 % to 100% compared to landfill.

Recycling and reuse by themselves each achieve roughly 30% emissions reductions compared to landfill and although these are not the largest reductions they are still the preferred end of life scenarios because they are the only options that can be paired with an alternative disposal (once the recycled product is used) and so provides a cumulative benefit. Incineration with CHP is consistently the single most effective means of reducing GHG emissions due to the offset grid electricity and heat consumption, saving 90 % to 100% compared to landfill. Generating both heat and power on-site is shown to be even more preferable than connecting to the national grid, though of course is only possible for the left over waste. Despite it being the chief means to reduce GHG emissions there is no large scale CHP infrastructure in the UK, partly because of the large distances heat must be transported to reach homes from power stations. CHP is better suited to nations with a history of district heating systems. Incineration for electricity is in most instances the next best alternative scenario saving almost 50% emissions compared to landfill and is also more suited to the UK energy-from-waste plants. This research confirms that proposed energy-from-waste would be more effective at reducing GHG emissions if they were required to be located where they can incorporate heat utilisation, for example near industrial clusters.

In the UK, existing disposal facilities may not be compatible with energy-from-waste and it is not always a popular local planning issue so decision makers may look towards next best alternatives. One of the most common waste treatments for biomaterials is composting, which is shown to achieve a reduction in GHG emissions of around a quarter compared to landfill. AD with electricity-only is as effective as composting though in general AD is unlikely to be possible on such a large scale as current composting rates. Identifying locations for AD with CHP would achieve slightly higher GHG reductions of around 25 to 30% compared to landfill. Scaling up demonstration-scale ethanol-from-waste plants may be more practical than AD or incineration CHP and would still achieve between 25 to 30% reduction in GHG compared to landfill, although this is an embryonic technology. If reducing GHG emissions is the priority, this research shows that making any use of the technologies mentioned could achieve a significant saving, especially if paired with reuse or recycling and where CHP can be exploited.

3.1.2 Cost effectiveness of Disposal Options in Reducing GHG Emissions

These generalisations may already be widely suspected, if not quantified, for biomaterial specific waste. To aid decision makers further, the cost of predicted GHG savings is presented by HELCA and shown in Figure 3. These data are net costs to society over the whole disposal process and so because there are many stakeholders including waste management companies, consumers, manufacturers and local councils, it is not easy to determine which group may benefit or lose out. Given that there is heterogeneous technology and a lack of cost data published by companies due to confidentiality, 'best' and 'worst' cost case estimates are made using a range of data sources; it is noteworthy how much estimates vary.

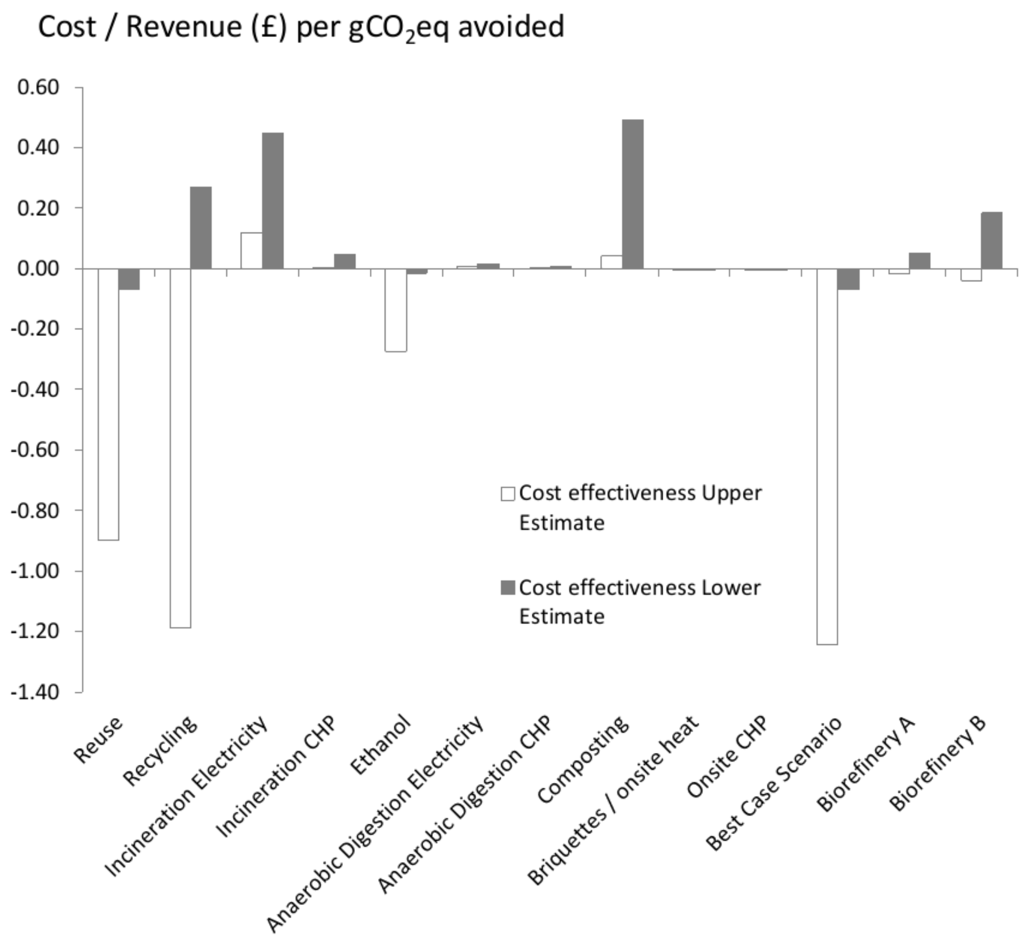

Figure 3 Cost Effectiveness of Reducing GHG emissions through Disposal options

When considering UK typical disposal scenarios we see cost effective options are not being pursued. ‘Incineration with electricity’ was among the most effective ways to reduce GHG emissions and was part of the typical UK disposal mix, yet it is potentially one of the most costly ways of reducing emissions. This could dramatically alter the way it is ranked by decision makers. These unfavourable economics may be one reason for ROCs however it may also be seen as a justification to not direct investment into electricity only energy-from-waste plants. The price achieved for the electricity produced will dramatically affect this evaluation and the abundance of energy-from-waste plants in Europe suggests these may be conservative estimates. Similarly the worst case scenario for recycling shows that there are still products for which recycling is still not yet cost effective, though given the size of the recycling industry, recycling in the main is profitable.

Clearly 'Reuse' and 'Recycling' are shown to make economic sense since there is no or a low cost of processing. More surprisingly 'Ethanol' is shown as the next most profitable means of reducing GHG emissions. Conventionally first generation biofuels are more profitable than second generation biofuels made from lignocellulosic material like hemp. This is the main reason why investment to date has mainly focussed on first generation crops. However, in a biorefinery the question perhaps should not be whether second generation biofuel made from waste is more profitable than first generation biofuels, but whether or not it is more profitable than other forms of waste management. The high price of ethanol is one reason why this evaluation is favourable, though this technology is in an embryonic stage so there is no established infrastructure for waste to ethanol. In reality, this means upfront investment would be required to build factories prior to achieving these savings. This uncertainty is further mirrored by the high variability between ethanol's best and worst case cost estimates. In reality, success is likely to vary based on the quality of the feedstock, yet the result here could be used to encourage more investment in this field highlighting its potential as a cost-effective low carbon disposal option for biomaterial waste. Companies may even be encouraged to design the biomaterials so that they are more suitable for ethanol conversion, for example by encouraging purity like removing synthetic thread in natural fibre clothes.

'AD' and 'Incineration with CHP' have some net costs so may be less preferred despite reducing GHG emissions. This may because the 'AD' often only occurs on a small scale and the inputs into the process itself are expensive. Incineration with the energy yield quoted is for an average EU facility in 2001 and it is likely that plants coming online now will have much greater efficiencies. When reviews of these up-to-date technologies are published HELCA may be updated and CHP may appear more cost effective. Generally, profit will depend on the price achieved for the heat and electricity, and this can vary over time and space. Thus, if the same assessments were redone with different unit energy prices their cost effectiveness may be very different.

Finally 'Composting' is shown to be a relatively expensive means of reducing GHG emissions perhaps because compost as a product has a very low market value, often zero. Composting still remains a common function performed by local governments and private companies. It is appealing since it allows large volumes of vegetative waste from landscape maintenance and organic waste collection to be dealt with without going to landfill and attracting large taxes. This research shows that technologies like ethanol conversion can greatly reduce GHG emissions and be more profitable than composting. This finding suggests that perhaps decision makers should have more confidence to make the upfront investment in waste-to-ethanol plants rather than expanding composting facilities.

3.1.3 Implications

Our research indicates that the current typical UK waste disposal of biomaterials may be neither 'low carbon' nor cost efficient. Simply identifying GHG emissions as a barometer of success may not be practical; a wider focus on cost in addition to GHG can expand the range of technologies deemed useful specifically in highlighting the potential of waste to ethanol. Changing course in policy terms may require the use of broad harmonised tools like HELCA to stimulate a rethink on which disposal infrastructure to support, for example, rethinking the use of ROCs for waste incineration but not ethanol production.

The use of specific LCA by individual companies to identify the influence of disposal emissions presents barriers of cost, comparability and the fear of acting alone contributing to disposal's low profile. The biomaterial industry may benefit from harmonised tools like HELCA to identify which 'low carbon' disposal fates should be kept in mind when designing products or advising customers since it is cheaper and provides more credibility than a company's own study. There may be some value in linking manufacturers with waste managers so that they may have some of the benefits from improving product purity to improve ethanol yields or in designing for deconstruction to facilitate reuse and recycling though this research does not seek to resolve the many questions about who has the responsibility or right to claim benefits of low carbon disposal.

Harmonised tools like HELCA can be updated when more comprehensive data are available thus becoming more accurate and they provide a fair playing field which may encourage stakeholder engagement. HELCA has the added benefit of ranking disposal options for each product by their ability to reduce GHG emissions and by their cost effectiveness at reducing GHG emissions. It can even be used to predict the impact of future technological advancements, for example, if ethanol conversion efficiencies improve as the industry matures. This can help to direct potential investment.

3.1.4 Limitations

This research only investigates disposal emissions. Any attempt to incorporate differences in the inputs of two biorefineries earlier in their life cycle (LUC, fertilizer use etc) would mean differences in emissions could not be said to be caused wholly by their different disposal fates. Thus, they are not interrogated in this research and are only used to give context to the scale of emissions reductions achieved by disposal.

More economic and GHG appraisals of the technology mentioned here would be useful, and the data that are used are the most up to date that could be found. A further refinement in these data, for example to allow the distinction between cardboard and paper, would be useful. Ensuring the unit price for electricity and heat is accurate is a difficult task given business confidentiality. Specific technology efficiency can severely influence the results given by tools using defaults like HELCA. This research focusses only on GHG emissions omitting entirely other environmental impacts of waste management which can affect ecosystems and local communities.

4 Conclusions

As environmental impacts of products become important the need arises for tools to support decision making, operational practices and policies. For biomaterials, such tools must address the whole lifecycle impacts including end of life disposal scenarios. HELCA presents a tool based on LCA to help the understanding of end of life GHG emissions for the biomaterial industry and its policy landscape. HELCA not only supports the generic claims of the waste hierarchy that reuse and recycling should be prioritised but also shows that excluding the impact of end of life scenarios is especially bad practice in sustainability assessments of biomaterials. ‘Low carbon’ disposal can even help biomaterials to be ‘carbon neutral’, for example if biorefineries use waste to provide onsite energy and if products are disposed of via energy-from-waste CHP plants. Conversely, HELCA’s appraisal of cost effectiveness shows that ethanol conversion may be a preferred waste treatment to CHP. AD and composting are lower carbon alternatives to landfill but are not particularly cost effective.

Policy makers can use decision support tools like HELCA to decide on their levels of support for particular technologies and whether to prioritise cost effectiveness or emissions reductions. Using this tool would allow them to rank options to fit in with their local circumstances (for example, there may not be any incineration CHP plants in a particular region). Use of HELCA may also prompt companies to investigate ways of making their products easier to fit in with a particularly promising or profitable end of life scenario, especially if they have a share of any future benefit. HELCA provides a quick at-a-glance guide which may even prompt companies to make further investigations and undertake a detailed LCA to answer specific questions. Harmonised decision support tools give companies credibility in promoting particular disposal options, which individual specific LCA may not necessarily achieve.

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Glossary

CO_2	Carbon Dioxide
CBA	Cost benefit analysis
CHP	Combined Heat and Power
CH_4	Methane
DDOC	Dissimilable biodegradable carbon
DEFRA	Department of Environment Food and Rural Affairs
EC	European Commission
EIO	Environmental Input Output Life Cycle Assessment
EU	European Union
GHG	Greenhouse Gas
Ha	Hectare
HELCA	Harmonised, End of life, Life Cycle Assessment
IO	Input Output Life Cycle Assessment
IPCC	Intergovernmental Panel on Climate Change
iLUC	Indirect Land Use Change
Kg	Kilograms
Km	Kilometres
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Analysis
LUC	Land Use Change

MSW	Municipal Solid Waste
MBOE	Million Barrels of Oil Equivalent
N_2O	Nitrous Oxide
ONS	Office of National Statistics
PUR	Polyurethane
RED	European Union Directive on Renewable Energy
UK	United Kingdom

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