Extending the attributional-consequential distinction to provide a categorical framework for greenhouse gas accounting methods

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Declaration

The candidate confirms that:

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b. the work submitted is his/her own, except where work which has formed part of jointly-authored publications has been included. The contribution of the candidate and the other authors to this work has been explicitly indicated below;
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Signed:

Matthew Brander

Paper 1 within this thesis (*The attributional-consequential distinction and its applicability to corporate carbon accounting*) was co-authored by the candidate and Dr Francisco Ascui. The work directly attributable Dr Francisco Ascui is primarily the content relating to the concept of framing, and the use of attributional corporate inventories for understanding regulatory risk. The remainder of Paper 1 is directly attributable to the candidate.

Signed:

Dr Francisco Ascui

Abstract

As part of the response to the threat of dangerous climate change a variety of methods have emerged for measuring greenhouse gas emissions to the atmosphere, assigning responsibility for those emissions, and informing decisions on mitigation actions. Many of these greenhouse gas accounting methods have developed in semi-isolated fields of practice, and this raises questions about how these different methods relate to each other, and whether they form 'families' of conceptually similar approaches.

A useful distinction has developed within the field of life cycle assessment (LCA) between attributional and consequential methods, and this thesis explores the possibility of extending that distinction to categorise other forms of greenhouse gas accounting. Broadly, attributional methods are inventories of emissions/removals for a defined inventory boundary, while consequential methods aim to estimate system-wide changes in emissions that result from a decision or action.

This thesis suggests that national greenhouse gas inventories, city inventories, corporate inventories, and attributional LCA are all attributional in nature, while project-level assessments, policy-level assessments, and consequential LCA are all consequential in nature. The potential benefits from creating this categorical framework include ensuring that individual methods are conceptually coherent, transposing lessons between methods of the same categorical type, and ensuring that the correct type of method is used for a given purpose.

These various benefits are explored conceptually through the analysis of existing greenhouse gas accounting standards, and also empirically with the use of a bioenergy case study. The findings suggest that the attributional-consequential distinction is highly useful for conceptualising and developing greenhouse gas

accounting methods, which is important, ultimately, for addressing dangerous climate change.

Lay Summary

Climate change, which is caused by greenhouse gas emissions, is generally considered to be one of the greatest threats facing humanity. In order to address this threat a large number of different methods have been developed for measuring greenhouse gas emissions to the atmosphere, assigning responsibility for those emissions, and informing decisions on how to reduce emissions. Many of these greenhouse gas accounting methods have been developed by different groups of users, and this creates an opportunity for sharing lessons and ideas between those different groups. One idea that has developed within the product 'life cycle assessment' community is the distinction between what are called 'attributional' and 'consequential' methods. Broadly, 'attributional' methods are inventories of greenhouse gas emissions, while 'consequential' methods aim to estimate the total change in emissions that result from a decision or action.

The research in this thesis aims to develop the attributional-consequential distinction as a way of categorising all other forms of greenhouse gas accounting. One possible benefit from doing this is to make sure that incompatible methods are not mixed together. Another benefit is the possibility of sharing lessons between methods of the same type (e.g. between different attributional methods, or between different consequential methods). A further possible benefit is to ensure that the correct type of method is used for a given purpose, e.g. if the purpose is to inform a decision, then the best type of method is one that gives information on all the consequences or impacts from the decision.

As a first step the research sets out the most important characteristics of attributional and consequential methods, i.e. the research develops definitions for attributional and consequential methods. The next step builds on this foundation to

categorise all existing greenhouse gas accounting methods as either attributional or consequential in nature.

The main part of the research then provides a real-world example of the different results that different methods provide, using the example of a bioheat plant (i.e. a boiler that burns wood). The results show that attributional methods do not necessarily show the full impacts of the bioheat plant, and so are not useful for deciding whether the bioheat plant is a good or bad idea. The results also show that consequential methods are better at estimating the total impacts of the bioheat plant, and are better at informing decisions. The bioheat example also shows that it is important to look at the timing of emissions and forest growth (which absorbs greenhouse gases from the atmosphere), as burning trees can cause an initial increase in emissions which takes a long time to be compensated by the regrowth of forests. The results also show that in some cases the overall effect of the bioheat plant is to actually increase emissions rather than reduce them.

The final part of the research provides a further illustration of how lessons can be shared between methods of the same type. Overall the research shows that using the attributional-consequential distinction is useful for understanding the strengths and weaknesses of different forms of greenhouse gas accounting, and for identifying ways of improving the methods that currently exist.

Acknowledgements

First and foremost I would like to thank my supervisors, Francisco Ascui and Craig MacKenzie, who have provided a superb combination of interest, encouragement, critique, attention to detail, and intellectual challenge throughout the development of this thesis. It is often useful to have to a particular person or audience in one's mind's eye while writing, and Francisco and Craig have been figuratively sitting on my shoulders for much of the last three years (though my neck issues are more likely related to bad posture and staring at spreadsheets). Thank you also to Francisco for the excruciating games of squash, which helped to put all other physical and mental struggles into perspective.

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Last, but by no means least, thank you to my wonderful wife Alice and children Sam, Emma, and Lucy, for their inexplicable indifference to the attributional-consequential distinction, and the essential balance they provide.

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Introduction

1. Initial Overview, Motivations, and Research Questions

Climate change is recognised as one of the greatest threats facing humanity, and the biosphere more generally (Stern 2006; IPCC 2014). Largely in response to this threat, a growing number of methods and practices have been developed to inform, understand, interpret, and assign responsibility for human-induced greenhouse gas emissions to (and removals from) the atmosphere (Ascui & Lovell 2011). Examples of these methods and practices include national greenhouse gas inventories (IPCC 2006; United Nations 1992), community or city-level inventories (British Standards Institute 2013; GHG Protocol 2014), corporate-level inventories (WBCSD/WRI 2004; ISO 2006c), product-level life cycle assessment (British Standards Institute 2008; ISO 2013c), project-level assessments (WBCSD/WRI 2005; ISO 2006d), and policy-level assessments (WRI 2014c). This list is in no way exhaustive, and there is an ever-growing number of sector-specific, national, regional, and other forms of guidance for measuring and reporting greenhouse gas related information.

This proliferation of different greenhouse gas accounting methods is interesting from a research perspective for a number of reasons. Firstly, given the threat of dangerous climate change, important questions arise about whether these methods and practices are sufficient for managing greenhouse gas emissions. Secondly, many of these methods have developed in semi-isolated fields of practice, and this raises questions about their similarities and differences, and whether they have shared conceptual foundations. This situation may also create opportunities for sharing lessons and innovations that have developed in one area, but have not yet appeared elsewhere.

One apparent instance of such an innovation, which has developed within the field of life cycle assessment (LCA), is the distinction between what are termed attributional and consequential approaches (Curran et al. 2005). Broadly, attributional methods provide an inventory of emissions/removals for a defined inventory boundary, while consequential methods aim to estimate the total system-wide change in emissions/removals caused by a given action or intervention. The fact that attributional methods do not capture changes in emissions outside the inventory boundary means that their use for decision-making can result in decisions which increase rather than decrease emissions (R. J. Plevin et al. 2014b). A particularly striking example of this is provided by Searchinger et al.'s (2008) critique of the use of attributional LCA to support the US Government's promotion of corn-based biofuel. This study shows that an attributional LCA does not include the market-mediated effects of the biofuel policy on world commodity prices, which may increase rates of deforestation. Once the greenhouse gas emissions from deforestation are attributed to the corn biofuel, as they would be using a consequential LCA, the biofuel policy is shown to substantially increase rather than decrease global emissions. Clearly, selecting the appropriate method is highly important for achieving effective climate change mitigation.

Given the usefulness of the attributional-consequential distinction for conceptualising and understanding the distinct forms of life cycle assessment, this thesis explores the possibility of extending the distinction to categorise other forms of greenhouse gas accounting. Such a categorical framework may have a number of benefits. Firstly, the categorisation helps to identify the appropriate use of different methods, e.g. if a method is attributional in nature it can be inferred that it is not sufficient for decision-making. Secondly, the categorical framework can be used to ensure that individual methods are methodologically coherent, and do not mix incompatible elements within a single approach. Thirdly, identifying 'families' of methods, with shared purposes or conceptual underpinnings, allows methodological lessons to be transposed between methods of the same type. The practical motivation for the present research is therefore ultimately to improve existing greenhouse gas accounting practice, and to contribute to the effective mitigation of dangerous climate change.

Aligned with the distinct fields of greenhouse gas accounting practice, much of the academic literature aimed at developing these accounting methods also tends to focus on specific areas, such as product life cycle assessment or national inventory accounting, with very little dialogue between different fields. The academic motivation for the present thesis is to address this gap in the literature, and one of the novel contributions is to offer a more holistic perspective across the diverse range of different greenhouse gas accounting methods, and to explore the relationships between them. To this purpose, the attributional-consequential distinction is developed as a generic categorical framework for classifying and conceptualising the different existing accounting methods, and the development of this framework forms the primary theoretical contribution of the present research.

These initial remarks are intended to provide a brief overview of the themes explored in this thesis, with further discussion on the background literature and relationship to theory given later in this chapter. The overall format for the thesis is a portfolio of four papers, accompanied by introductory and concluding chapters, which set-out and develop the overarching narrative of the thesis. The papers in this portfolio are listed in Table 1, below.

Table 1. Papers in portfolio, authors, publication, and publication status

Paper Number	Title	Authors	Publication	Publication Status
Paper 1	The attributional-consequential distinction and its applicability to corporate carbon accounting	Brander and Ascui (forthcoming)	Book chapter in Corporate Carbon and Climate Accounting	Accepted, forthcoming
Paper 2	Transposing lessons between different forms of consequential greenhouse gas accounting: lessons for consequential life cycle assessment, project-level accounting, and policy-level accounting	Brander (2015b)	Journal of Cleaner Production	Published
Paper 3	Comparative analysis of attributional corporate greenhouse gas accounting, consequential life cycle assessment, and project/policy level accounting: a bioenergy case study	Brander (under review)	Journal of Cleaner Production	Under review
Paper 4	Response to "Attributional life cycle assessment: is a land-use baseline necessary?" – Appreciation, renouncement, and further discussion	Brander (Brander 2015a)	The International Journal of Life Cycle Assessment	Published

The research contained in Papers 1, 2, and 3 was planned with the intention that it should form a coherent whole, with each paper providing outputs that would then be utilised or illustrated by the subsequent paper(s). Paper 4 is distinct, in that it arose in response to a recent journal paper (Soimakallio et al. (2015)), but the issues it explores are highly relevant the present thesis, and further develop the main themes of the research. The research questions addressed by Papers 1, 2 and 3, which underpin the logical flow of the thesis, can be stated as follows:

1. What is the attributional-consequential distinction and what is its applicability to other forms of GHG accounting?

This is intended to identify the defining characteristics of attributional and consequential methods, and to develop the distinction as a classificatory scheme for categorizing other forms of physical greenhouse gas accounting.

2. What are the different forms of *consequential* greenhouse gas accounting method, and what methodological lessons might be shared between them?

This question is necessary for identifying the different consequential methods that are available, which are then subsequently used in addressing Question 3.

3. Do attributional inventories and the different consequential methods provide different results, and what are the implications for decision-making?

The 'decision-making' in Question 3 is that specifically related to actions aimed at mitigating climate change, and answering this question is the core empirical component of the research. The approach adopted is to apply the various consequential methods that are available (identified in answering Question 2), and to compare the results to those from a conventional attributional method. Papers 1, 2 and 3 broadly correspond, though not completely, to the three research questions above, and the papers and their interconnectedness are outlined in more detail in Section 2 below.

The papers presented in this portfolio are the versions published or submitted for publication, with the exception of minor changes, such as the renumbering of figures and tables to ensure a sequential order in this thesis. The main reason for maintaining the published versions is that later papers in the sequence, particularly Paper 4, refer to mistakes or shortcomings in the earlier papers, and revising the earlier papers would then obscure or invalidate such points. The fact that the papers were written at different points in time, and represent an evolving understanding of the attributional-consequential distinction, is illuminating in itself, and this is reflected on further in the Conclusions chapter.

It is worth noting that each of the papers include their own introductory sections, details of the relevant literature, descriptions of their methodologies etc., and the intention is not to replicate that content within this introductory chapter, unless doing so is particularly useful for setting out the overarching narrative of the thesis. The remainder of this introductory chapter has the following structure: Section 2 provides, for orientation purposes, a more detailed overview of each of the papers, and how they link together; Section 3 sets out the role of theory in the thesis; Section 4 discusses a number of methodological issues that deserve some additional explanation; and Section 5 provides an overview of the literature on greenhouse gas accounting, and the position of the present research within it.

Before proceeding, it is worth briefly addressing a terminological issue. The term 'greenhouse gas accounting' is used throughout the Introduction and Conclusion chapters of this thesis to refer collectively to the practices and methods concerned with quantifying greenhouse gas emissions and removals. A synonymous term is 'carbon accounting', however, greenhouse gas accounting is adopted for two reasons. Firstly, it is the term generally used in, or aligned with, the relevant international standards and guidance documents (e.g. ISO (2006c; 2006d; 2006e), and the GHG Protocol standards (2004; 2005; 2011a; 2014c)). Secondly, a number of the greenhouse gases normally included within accounting and reporting requirements (e.g. for the Kyoto Protocol, ISO, and GHG Protocol) do not actually contain the atomic element *carbon* (e.g. nitrous oxide, sulphur hexafluoride, and nitrogen trifluoride), and therefore 'carbon' accounting may be something of a misnomer. The term 'carbon accounting' can serve as a useful shorthand, and appears to be the preferred term within the social and environmental accounting literature (Ascui 2014), and within the field of forest and soil carbon modelling, and it also happens to be used throughout the Paper 1 within this portfolio.

A final point to note is that the present research focuses on what can be termed *physical* greenhouse gas accounting, i.e. where the unit of measurement is a mass unit of greenhouse gas. This is distinct from *financial* accounting for carbon-based assets and liabilities, such as tradable pollution permits in an emissions trading scheme, where the unit of measurement is in monetary terms. The reason for this focus is that the attributional-consequential distinction has evolved within the field of life cycle assessment, which is a form of physical greenhouse gas accounting, and therefore a natural first step is to extend the distinction to other forms of physical greenhouse gas accounting. A potentially interesting subsequent step would be to explore extending the distinction further still, to other forms of environmental accounting, financial greenhouse gas accounting, or financial accounting more generally. These possibilities are taken up again in the Conclusions to this thesis.

2. Summary and Relationships between the Papers

To provide a sense of the overarching themes and interconnectedness of the portfolio as a whole, this section provides a more detailed summary of the papers, and the relationships between them. Figure 1, which follows the narrative account below, provides a graphical representation of the relationship between the papers.

2.1. Paper 1.

"The attributional-consequential distinction and its applicability to corporate carbon accounting." Accepted for publication as a chapter in S. Schaltegger, D. Zvezdov, I. Alvarez, M. Csutora, & E. Günther (Eds.), *Corporate Carbon and Climate Accounting*. Dordrecht: Springer.

Paper 1 starts from the premise that the attributional-consequential distinction has proved useful within the field of life cycle assessment, and may prove similarly useful for understanding and conceptualising other forms of greenhouse gas accounting. The LCA literature suggests that attributional inventories do not reflect the system-wide impacts of decisions and can lead to unintended/undesired consequences, and that consequential methods are therefore necessary for decision-making. If corporate/organisational-level greenhouse gas inventories can be characterised as being *attributional* in nature, thereby extending the attributional-consequential distinction beyond the field of life cycle assessment, then it can be inferred that the use of corporate inventories for decision-making may result in similar unintended consequences, and are not sufficient as a decision-making tool.

The paper describes the evolution of the attributional-consequential distinction; the key defining features of attributional and consequential approaches; examples of the results obtained from each method (from the life cycle assessment literature); and an overview of the critical discussion concerning the distinction within the life cycle assessment community. The paper then goes on to consider the applicability and implications of the distinction for corporate greenhouse gas accounting, and the potential usefulness of the distinction for academic research on social and environmental accounting. Although the motivation for focusing on corporate greenhouse gas accounting was partly based on considerations of expediency, as the editorial remit for the book to which the chapter was submitted was corporate climate accounting, this focus is also justified by the widespread use of corporate-level accounting (CDP 2015; Scottish Government 2015), and the novelty of applying the attributional-consequential distinction in this area. Notwithstanding the focus on corporate inventories, much of the discussion is applicable to any form of attributional inventory, such as national or community inventories. The key output from Paper 1, i.e. the identification of the defining characteristics of attributional and consequential methods to enable the further development of the categorical scheme in Paper 2, is independent of the specific focus on corporate-level accounting.

A further point to note is that there is a separate explanatory thread within Paper 1, which is returned to in several places in the paper. This is the use of the concept of ‘framing’ to explain why the attributional-consequential distinction appears to have evolved in one field of practice (i.e. life cycle assessment) but not in another (i.e. corporate-level accounting). Framing refers to ‘the processes by which people construct interpretations’ (Rein & Schon 1993, p.147), and is also used within Paper 1 to explain the ‘observed pattern of resistance and recognition in the development of the distinction in the field of LCA’ (Brander & Ascui forthcoming). This is the element of the paper provided by the co-author, Francisco Ascui, and the distinction between the technical/methodological elements of Paper 1 and the social/explanatory elements is explored further in Section 3 (theoretical framework) and Section 5 (relationship to the literature) in this Introduction chapter, and in the Conclusions chapter.

2.2. Paper 2

“Transposing lessons between different forms of consequential greenhouse gas accounting: lessons for consequential life cycle assessment, project-level accounting, and policy-level accounting”. Published in the *Journal of Cleaner Production* (2015).

While Paper 1 focuses on the lessons that can be shared between two forms of attributional greenhouse gas accounting method, i.e. attributional LCA and corporate-level greenhouse gas accounting, Paper 2 focuses on the lessons that can be shared between the different forms of consequential method. Paper 2 starts by using the defining characteristics of the attributional and consequential distinction identified in Paper 1 as a categorical scheme for classifying other identified forms of physical greenhouse gas accounting as either attributional or consequential. National inventories, community/city inventories, corporate inventories, and attributional product life cycle assessment are categorised as attributional, while consequential product life cycle assessment, project-level, and policy-level accounting are categorised as consequential.

The guidance documents and standards for the three identified consequential methods were then analysed to determine the key elements and structure of each method, and to identify lessons that could be transposed between one method and another. The findings suggest that the project and policy-level methods share very similar structures, and are essentially the same method. In addition, consequential life cycle assessment could be enhanced by adopting a number of elements used by the project-policy method, i.e. a time-series of impacts, aggregate level analysis, and a transparent baseline and decision scenario structure.

2.3. Paper 3

"Comparative analysis of attributional corporate greenhouse gas accounting, consequential life cycle assessment, and project/policy level accounting: a bioenergy case study". Submitted to the *Journal of Cleaner Production* (in review).

Papers 1 and 2 provide a largely conceptual discussion on the characteristics, structure, and expected limitations associated with corporate-level greenhouse gas inventories (Paper 1), and the three identified consequential methods of consequential LCA, project-level, and policy-level accounting (Paper 2). A possible critical response to this analysis might be that although there are conceptual or methodological differences between the approaches, the difference in the results they provide may be immaterial, i.e. they all support the same decision-making outcomes. Paper 3 therefore builds on the conceptual analysis in Paper 1 and Paper 2 by providing an empirical case study to illustrate the magnitude of difference in the results and information provided by an attributional corporate inventory, a consequential LCA, and the project/policy level method. The paper applies each of these accounting methods to the same bioenergy decision-scenario, and provides a three-way comparative analysis of the results.

The findings demonstrate that attributional corporate greenhouse gas inventories will typically not capture the full impacts of decisions/actions, and that using such inventories to inform decision-making can lead to unintended consequences (i.e. illustrating the conceptual discussion in Paper 1). The results of the case study also demonstrate that, although consequential LCA and the project/policy method both aim to capture the total consequences of decisions, there are important methodological advantages to the project/policy approach (i.e. illustrating the conceptual discussion in Paper 2). The lessons that can be transposed include the provision of a transparent baseline-decision scenario structure, and the distribution of impacts over time. This latter point is particularly relevant to bioenergy mitigation actions, as the potentially long regrowth periods for harvested forests means that bioenergy may cause net increases in emissions during the timeframe for most reduction targets (e.g. 2050), and could contribute to a near-term climate tipping point (Lenton et al. 2008).

A further contribution from Paper 3 is the use of normative decision theory to interpret the uncertainties associated with the case study decision. A large number of existing studies already illustrate the range of possible impacts from bioenergy policy (Stephenson & MacKay 2014; Adams et al. 2013; Cherubini et al. 2009; Jonker et al. 2014; Lippke et al. 2011; Repo et al. 2014; Zanchi et al. 2012; Chum et al. 2011; Matthews et al. 2014; Marland & Schlamadinger 1997; Agostini et al. 2013), however, they tend not to interpret the implications of this uncertainty for decision-making. Using a number of the concepts and principles present in normative decision theory, Paper 3 suggests that the emissions outcomes from bioenergy interventions are characterised by *Knightian* uncertainty, i.e. the probabilities of the different plausible outcomes are not known. Furthermore, based on the principle that decisions should be justified based on their expected outcomes, and the finding that we do not know what the probable outcomes from a bioenergy intervention will be, we cannot justify the implementation of bioenergy interventions. In this way the *uncertainty* of the outcomes from bioenergy interventions should be treated as the *finding,* and should be recognised as highly decision-relevant information.

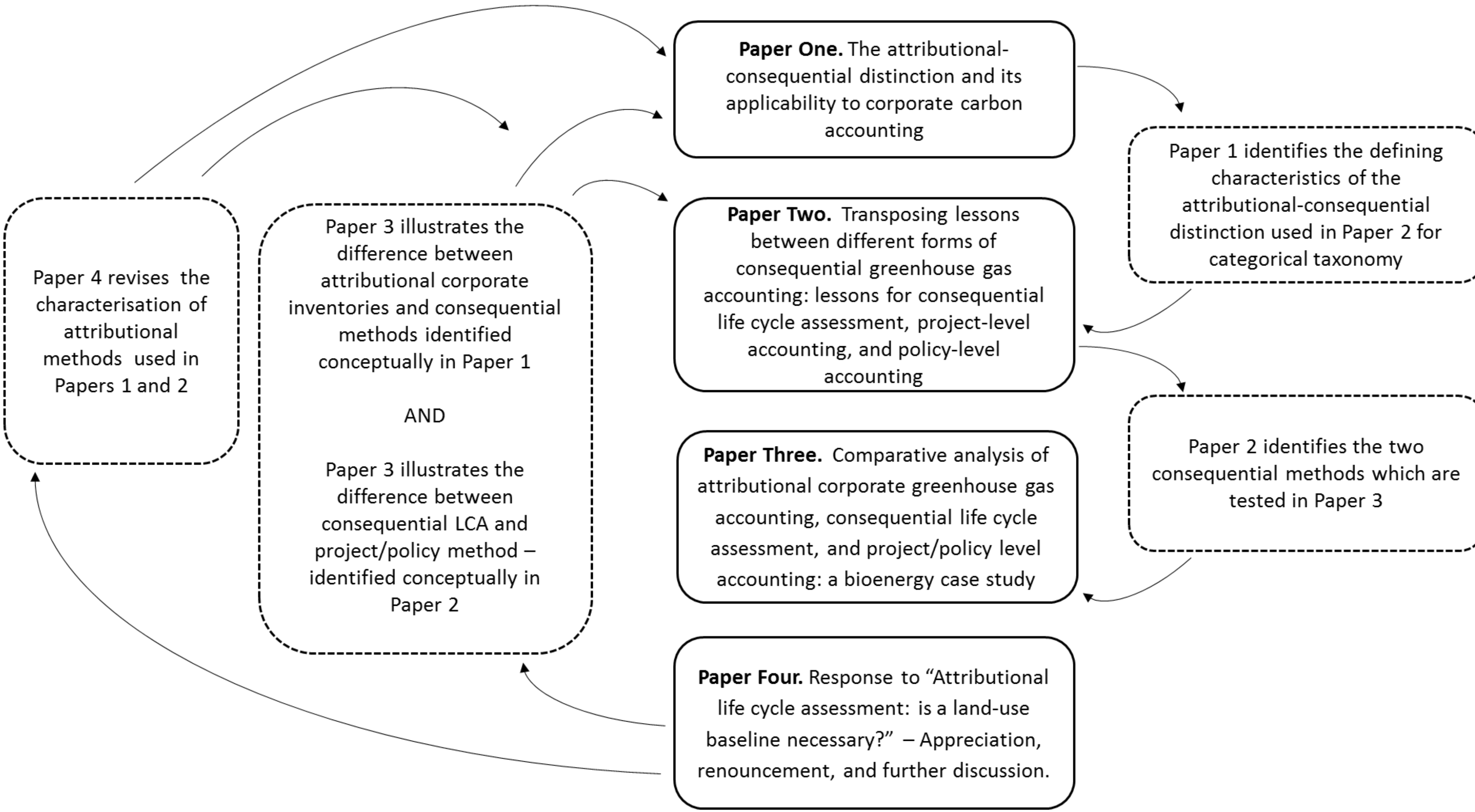
2.4. Paper 4

'Response to "Attributional life cycle assessment: is a land-use baseline necessary?" – appreciation, renouncement, and further discussion'. Published in *The International Journal of Life Cycle Assessment* (2015).

As mentioned above, Paper 4 provides a response to a journal paper by Soimakallio et al. (2015), which argues that, contrary to a number of previous papers (including Brander et al. (2009), and Brander and Wylie (2012)), attributional life cycle assessments are *not* inventories of absolute environmental flows. The relevance of this to the present portfolio is that Papers 1 and 2 both describe attributional methods in this way, i.e. as inventories of absolute emissions/removals. Paper 4 acknowledges the correctness of Soimakallio et al.'s assertion and renounces the previous characterisation of attributional inventories, and therefore represents an evolving position within this portfolio of research. However, Paper 4 also discusses the implications of the re-characterisation, and argues that the taxonomy of greenhouse gas accounting methods presented in Paper 2 is unchanged.

Paper 4 also provides a further exploration and discussion of some of the key themes within this portfolio, i.e. the applicability of the attributional-consequential distinction to other forms of greenhouse gas accounting (i.e. particularly national greenhouse gas inventories), and the potential for sharing lessons across different fields of practice (i.e. between attributional life cycle assessment and national inventories). Paper 4 also identifies a number of possible misconceptions of the nature and use of attributional inventories within Soimakallio et al. (2015), which serves to refine the conceptualisation of the attributional-consequential distinction further still.

Figure 1. Overview of relationship between the papers



3. Theoretical Framework

This section discusses a number of different issues related to theory and its relationship to the present thesis. Readers from academic disciplines outside the social sciences or humanities may not expect such a discussion within a doctoral thesis, and the presence of the discussion here deserves some explanation, with that potential audience in mind.

This portfolio of research has been undertaken within the context of a business school, where the majority of research can be characterised as some form of social science, and within this field of discourse an apparent norm or expectation is that research should be 'theorised' or make a 'contribution to theory'. This explicit emphasis on 'theory' appears to be less prevalent in other academic contexts, such as geosciences, engineering, or the environmental sciences. As an illustration (which may or may not be representative), an environmental science thesis on the greenhouse gas impacts of biochar contains the word 'theory' once (Hammond 2009); an engineering thesis on the carbon payback period of variable renewable generation contains the word 'theory' three times; while a social science thesis on greenhouse gas policies and social network influences on acceptability contains the word 'theory' 306 times (Holland 2013). One of the main challenges encountered whilst undertaking the current research has been to understand the role of theory in research, and to understand why some specific forms of theory are not relevant to the kinds of questions that are addressed in this thesis.

Given that this portfolio of research was produced within the social context of a business school, with its institutionalised expectation for theoretical discussion, a number of theory-related issues are explored below: the manifold meanings of 'theory' and the role of theory in the present research; the ontological and epistemological presuppositions of the present research; and the ‘tyranny’ of theory within social science research.

3.1. The meaning of ‘theory’ and its role in the research

The term ‘theory’ appears to denote a wide variety of different concepts, and has numerous shades of meaning. ‘Theory’ may variously mean: an explanation of causes and effects (Malmi & Granlund 2009); a categorical system for classifying phenomena (Denzin 1970); a set of normative or prescriptive principles (Blaikie 2000); the identification of regularities that can be extended from one context to another (Malmi & Granlund 2009); an overarching perspective on the world (Menzies 1982); some form of conceptual intervention aimed at creating enlightenment (Dimaggio 1995); a narrative which accounts for observations (Dimaggio 1995); a set of philosophical assumptions regarding the nature of existence and our knowledge of the world (Hopper & Powell 1985); and possibly many others besides. Many of these interpretations overlap with one another, and ‘theory’ could be characterised as a ‘family resemblance’ concept (Wittgenstein 1997, para.67), in that its meaning has various different characteristics or elements, not all of which are present in any individual context of use. This accords with Malmi and Granlund’s observation that ‘accounting academics seem to have very different perceptions of what is to be regarded as theory’ (2009, p.599); a statement which may be equally true without the qualification ‘accounting’.

Given this spectrum of meanings it is unsurprising that theory can also play numerous different *roles* within research, including: providing the context in which research questions are framed and approached; the set of statements from which hypotheses are deduced for testing (i.e. a deductive relationship) (Popper 1968); the end point of generalisations from observed data (i.e. an inductive relationship); the basis for explaining observed phenomena, which in turn corroborates the theory in question (i.e. an abductive relationship); underpinning the meaning of observational terms; and the basis for generating or interpreting data from instrumentation (Feyerabend 1975).

Given this plurality of meanings and roles, it is possible to identify a number of different ways in which theory informs or is relevant to the present research:

1. *Literature as a proxy for theory*. The background literature can be viewed as a proxy for theory in the sense that it establishes the focus and context for the research (Bryman & Bell 2007; Malmi & Granlund 2009). The present research is largely motivated and contextualised by the existing academic literature, qua theory, from the field of life cycle assessment. The majority of Paper 1 is concerned with exploring and analysing this background literature, and the context it provides is carried over to the rest of this thesis.

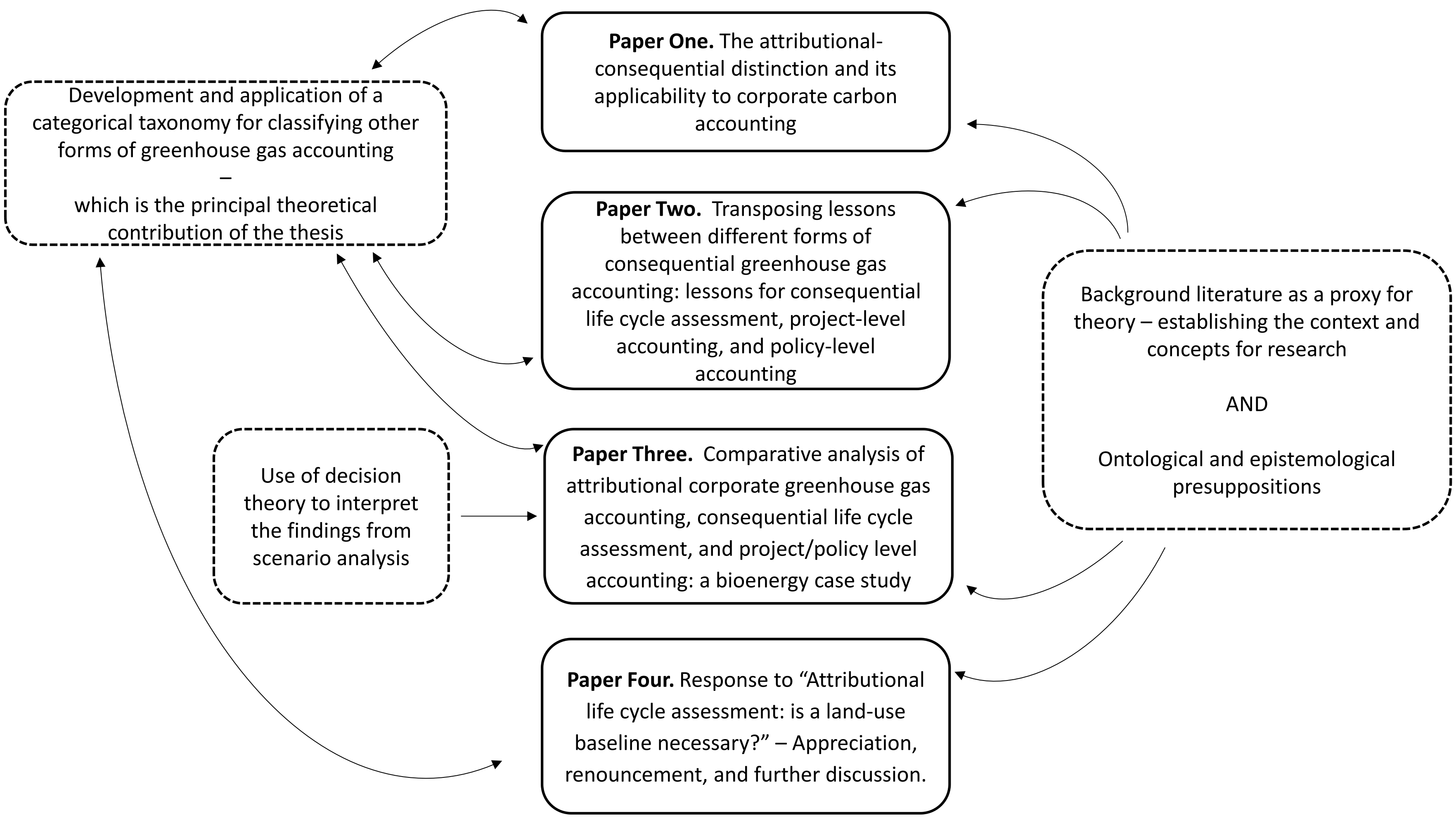
2. *Categorical system*. The use of the attributional-consequential distinction to categorise all forms of physical greenhouse gas accounting tallies with Denzin's identification of 'categorical taxonomy' as a form of theory (Denzin 1970). One of the principal theoretical contributions of the present research is therefore the development of the attributional-consequential distinction as a framework for 'categorical taxonomy'. An important benefit of categorical taxonomy is that it facilitates inferences or connections between methods which are identified as being of the same or different categorical types. For instance, if discrete greenhouse gas accounting methods are categorised as the same type, e.g. attributional, then the uses and limitations associated with attributional-type methods can be inferred to apply to each discrete method within that category (which is essentially the argument presented for corporate inventories in Paper 1). A further benefit is that it creates opportunities to transpose techniques and innovations between methods of the same type (which is the argument presented in Paper 2).

3. *Normative principles*. Normative decision theory is used within Paper 3 to develop and justify the interpretation of the findings from the bioheat case study. One key normative principle adopted from decision theory is that decisions *should* be based on consideration of the consequences of that decision. The concept of Knightian uncertainty, which is employed within decision theory, is also used to understand the nature of the information produced from the scenario analysis presented in Paper 3 (i.e. that the probability of each scenario occurring is unknown). The use of these principles and concepts does not contribute to the development of decision theory itself, however, it does constitute a potentially important contribution to the existing literature on bioenergy, which has generally not identified the uncertainty of outcomes as a decision-relevant *finding*, in its own right.

4. *Ontology and epistemology.* All research involves a theoretical framework, in the sense of the underlying conceptual scheme or presuppositions of the researcher (Feyeraband 1975). One form of presupposition relates to the nature of reality, or what can be termed 'ontological' presuppositions. They are the researcher's implicit or explicit assumptions about the kinds of entities that exist, and whether their existence and nature are dependent or independent of human cognition. Another form of presupposition relates to the nature and justifications for claims to knowledge, or what are termed 'epistemological' presuppositions. The ontological and epistemological presuppositions that would be otherwise implicit in the current research are explored in greater detail in the next section.

Figure 2, below, presents an overview of the role of theory in this thesis.

Figure 2. Overview of the role of theory in this thesis.



A further possible form of theory relevant to this thesis is the development of normative or prescriptive principles (Blaikie 2000) for greenhouse gas accounting, such as ‘attributional inventories are not sufficient on their own for mitigation decision-making’. The development of such general principles could conceivably be described as a contribution to greenhouse gas accounting theory. However, to date such methodological principles have not been labelled as ‘theory’ in the academic literature, and it may be overly presumptuous to coin and claim a contribution to ‘greenhouse gas accounting theory’ here. However, this thesis does seek to develop methodological principles for greenhouse gas accounting, but the label ‘normative method development’ is used to describe this undertaking.

3.2. Ontology and epistemology

Beginning with ontology, there is a spectrum of possible positions between the two opposing poles of realism and anti-realism. Broadly, realism holds that reality is made up of objects that exist independently of human cognition, while anti-realism holds that objects and their properties are essentially ‘constructed’ through human cognition, and do not exist independently. The philosophical debate between these broad positions is extensive, covering at least 2,400 years of the western philosophical tradition, and cannot be resolved or done full justice in this thesis. Instead, the approach taken here is to identify the ontological presuppositions that are evident in the present research, to make these transparent, and to offer some justifications for the position adopted.

Firstly, it appears that some form of realism is generally present in the practice of greenhouse gas accounting, i.e. the implicit ontological ‘model’ is that the atmosphere exists whether we perceive it or not. In addition, there is an ontological presupposition that human activities, such as the combustion of fossil fuels, create certain quantities of greenhouse gas emissions which our methods of measurement may (or may not) accurately represent. For example, if a litre of diesel is combusted we presume that a certain quantity of greenhouse gas is released at the point-of-combustion. We may seek to measure or represent that quantity of greenhouse gas, but the actual amount released is independent of our representations (i.e. it is real). Such an ontological model is clearly present in Swart et al.'s (2007) discussion on the scientific validity of national greenhouse gas inventories, which contrasts 'real world emissions' with the 'emission inventory'. A realist ontology also appears to be implicit in the accounting principle of 'accuracy' which is stipulated in a number of greenhouse gas accounting standards (e.g. WBCSD/WRI (2004) and ISO (2006c)). Accuracy is the degree to which a measured or reported value corresponds to the actual value, i.e. the 'actual' value is conceived of as something distinct and independent from the measured or perceived value.

The same realist ontology is presupposed in much of the research presented in this portfolio. In Paper 3 a number of greenhouse gas accounting methods are applied to a case study in order to undertake a comparative analysis of the results, and for this exercise a realist ontology is implicitly adopted, following the ontology implicit within the greenhouse gas accountings practices that are used. It is presupposed that the case study activity (the decision to build a bioheat plant) will cause a certain change in greenhouse gas emissions, and that the different quantification methods tested may be more or less accurate in terms of estimating what that change in emissions is. It may be that the actual change that occurs is inherently unknowable, as the change will always be measured relative to a hypothetical or counter-factual baseline, but nevertheless, from a realist perspective, there will be an actual change in emissions that exists independently of our ability to know what that change is.

Similarly, normative decision theory, which is used to interpret the findings from the case study in Paper 3, also appears to involve a realist ontology. It is presumed that there is an independently existing effect from the implementation of the bioheat plant, but we do not know what that effect is likely to be. In contrast, a *thorough-going* anti-realist approach may assert that the change in greenhouse gas emissions is 'constructed' by the processes and methods used to account for the change, and the change does not exist as an independent fact. Many anti-realist positions are often of a weaker variety, and merely emphasize the social construction and contingency of *social* reality, e.g. the contingency of our modes of representation. Such forms of weak anti-realism appear to be largely compatible with the realist model of a physical reality, which exists independently of our representations.

One justification for adhering to a realist ontology in the present research is that a strong anti-realist perspective is radically different from that implicit within greenhouse gas accounting practice itself, and the outputs from the anti-realist perspective may not be immediately accessible or comprehensible to the greenhouse gas accounting community. Given this point, and that one of the primary motivations for the present research is to inform and develop current greenhouse gas accounting practice, it is important to maintain the discussion within the conceptual scheme of the greenhouse gas accounting community in order for the research to be meaningful, and consequentially impactful. A further justification for adopting a realist ontology is that it appears to underpin our concern with, and response to, climate change as a threat to human wellbeing, and the biosphere more generally. We rely on climate and ecosystem science, and their implicit realist ontology, to provide information on the expected impacts of climate change, and we view climate change itself as something real, which is not contingent upon our belief (or disbelief) in its existence.

Moving on to epistemology, a similar reflective exercise can be undertaken for the approaches used to generate knowledge in the present research. A common meta-theoretical distinction for describing the epistemological approaches used in social research is that between positivism and interpretivism. Positivism advocates for the methods of the natural sciences, e.g. hypothesis testing, identifying regularities, and prediction, whilst interpretivism covers a range of approaches often characterised by qualitative data and the interpretation of subjective meaning (Chua 1986). However, this distinction does not appear to be a helpful one for characterising the epistemological approach in the present research for a number of reasons. Firstly, the present research combines elements of both natural science (e.g. forest carbon modelling) and the modelling of social behaviour (e.g. market responses to biomass demand), and so does not fit neatly within the bracket of social research, to which the positivist/interpretivist distinction is generally applied. Secondly, although it could be argued that the 'positivist' label can be applied to the aspects of the research which are concerned with natural science, the label 'positivism' appears to be something of a caricature of knowledge generation within the natural sciences, which may not have a unified methodological approach in any case (Feyeraband 1975). A further reason for rejecting the positivist/interpretivist distinction for social research more generally is that it creates a false opposition between two approaches that answer different kinds of question, create different kinds of information (e.g. prediction or understanding), and which may be complementary in terms of providing a plurality of views on the same subject.

An alternative meta-perspective for reflecting on the epistemological presuppositions of the present research, which recognises the diversity of knowledge-generating practices in a way that the positivist-interpretivist dichotomy does not, is a descriptive or Wittgensteinian approach. This approach suggests that the ascription of 'knowledge' should be considered within the context or 'language game' in which the ascription takes place (Wittgenstein 1975). Following this descriptive or Wittgensteinian meta-perspective, at least three different contexts for knowledge generation can be identified in the present research.

Firstly, conceptual analysis is used in Paper 1 to answer the questions 'What is the attributional-consequential distinction?' and 'What is its applicability to corporate GHG accounting?', and in Paper 2 to answer 'What are the different forms of consequential method?'. A literature review and document analysis were undertaken in order to address these questions, and the ascription of 'knowledge' in such cases is based on adherence to practices such as the provision of supporting statements with referenced documentation, demonstrating the breadth of the supporting literature, and logical argument. The presence or absence of such practices are the conditions by which 'knowledge' generation is ascribed.

Secondly, in Paper 3, a number of different greenhouse gas accounting methods are used to generate results for answering the question 'Do attributional inventories and the different consequential methods provide different results, and what are the implications for decision-making?', alongside a comparative analysis, and interpretation of the results using normative decision theory. The generation of 'knowledge' may be ascribed when the requirements of recognised greenhouse gas accounting standards or methods are followed correctly; high quality data are sourced; the comparative analysis is based on the same research object or case study to ensure comparability; and the application of theoretical or normative principles is supported by rational argument.

Thirdly, the 'knowledge' generated in Paper 4 is of a largely conceptual nature, and uses a comparison between attributional life cycle assessment and national greenhouse gas inventories, and also an illustrative example, to discuss the appropriateness of natural regeneration baselines for attributional inventories. The generation of 'knowledge' may be ascribed when the comparison is between methods of the same categorical type, and the illustrative case is recognised as a counter-example, warranting the rejection of the proposed baseline approach in question. That is, these are some of the 'criteria' by which 'knowledge' may be ascribed or justified, within the context in which the word 'knowledge' may be used. A further important criterion, common across the four papers, is the peer-review process for publications, which often plays a central role in the language game through which academic knowledge is ascribed.

The above provides a brief overview of the methodological activities or procedures used to generate ‘knowledge’ in order to illustrate the meta-theoretical point that the epistemologies, qua justifications of knowledge, used in the present research are simply those recognised within the context of the procedures and methods employed. The above overview also indicates the plurality of different practices and procedures associated with ‘knowledge’ generation, and indicates the inadequacy or over-simplification of knowledge-generation descriptors such as ‘positivism’ or ‘interpretivism’, which do not appear to map onto the plurality of practices actually used. For example, the conceptual analysis of existing consequential methods (Paper 2) generates new knowledge about the similarities and differences between those methods, but it is not clear that either ‘positivism’ or ‘interpretivism’ would be helpful in describing the conditions under which this ‘knowledge’ is ascribed. The descriptive approach for articulating the actual procedures through which ‘knowledge’ is ascribed allows for, or captures, the multiple ways in which knowledge is generated.

One potential pitfall that arises when taking a descriptive approach to epistemology is that it is often used to support some form of epistemological relativism, i.e. all knowledge is contingent on the thought-style/paradigm/frame adopted (Kuhn 1962; Bird 2000; Fleck 1979). An issue that then arises is how to reconcile any conflict between the relativism of the second-order or meta-perspective (i.e. the descriptive account of epistemology) and any implicit absolutism presupposed by the first-order perspective or practices that are being described. If we seek to revise or reformulate the first-order perspective (i.e. the perspective being described) then we are no longer simply describing it, and in any case, from a relativist perspective it is not clear on what basis the meta-perspective has a greater claim to being ‘true’ (if all truth is relative to a perspective).

One argument for not going beyond describing first-order epistemological practices is that, as with ontology, this maintains the description within the terms and concepts that are meaningful to the first-order perspective. As soon as the description is presented using a second-order framing an abstraction has occurred, and any debate on epistemological justification is no longer recognisable to the field of practice in question. In the case of greenhouse accounting for instance, if practitioners are told that their methods create knowledge through social agreement (rather than because the methods accurately reflect reality), this might not be viewed as an intelligible or helpful justification *within* that field of practice. Parallel concerns are expressed by Hilary Putnam in his critique of naturalised epistemology, which draws on psychology to provide the description of the conditions under which knowledge is ascribed (Quine 2008). Putnam argues that once we have abstracted ourselves from our normative practices then there is no 'justification, rational acceptability [or] warranted assertibility' (Putnam 1982, p.20). For this reason, the epistemological and ontological commitments presupposed in the present research are described, as above, without going on to adopt alternative meta-theoretical positions on epistemology or ontology.

3.3. The tyranny of theory

As noted above, the role of theory appears to have a particularly prominent position within the social sciences, with a strong presumption that research should be theoretically informed, in the sense that explicit use of operational theories should be present in the research. This presumption can manifest itself in a number of ways, through formal channels such as doctoral research review boards/examinations and the editorial policies of academic journals, or informally through discussions with colleagues/peers and pejorative labels such as 'naïve empiricism' (Bryman & Bell 2007, p.10) or 'consultancy' research. One possible implication of this predilection for theory is that it tends drive research either towards some form of conceptual interventionism (Dimaggio 1995), or towards explanatory ‘Why?’ questions which require theoretically informed explanations (Blaikie 2000), to the exclusion of exploratory or descriptive research questions (e.g. ‘What?’ questions), and other types of research such as normative method development. It is important to note that these other non-explicitly theorised forms of research still involve ‘theory’ in some form, e.g. in the sense of an underlying conceptual scheme, or for justifying normative principles, but the ‘theory’ tends to play a background or framing role.

Chua (1986) suggests that accounting research, at the time, was dominated by a single paradigm or world-view which ‘restricted the range of problems studied’. A similar argument can be applied to the current situation, where the obligation to ask questions which require explicitly theorised or explanatory answers also restricts the range of problems studied. Tuttle and Dillard’s (2007) account of the isomorphic processes (mimetic, coercive, and normative) which restrict the diversity of academic accounting research also seems particularly pertinent.

The predilection for theory can be viewed as a norm within the social institution or ‘paradigm’ of academic social research, the contingency of which may be seen by considering practice in other academic disciplines, such as environmental science or geosciences, where explicit focus on theory is less evident. In the case of greenhouse gas accounting, the preference for theorised social research may explain the absence of normative method development within the social and environmental accounting literature (Thomson 2007). This omission can be contrasted with the presence of method development within the environmental sciences or life cycle assessment literature, where theory is allowed to play a background role, and explanatory ‘Why?’ questions are less dominant. A distinction which may help in conceptualising these different forms of research is that between *studying* greenhouse gas accounting, qua social phenomenon, and *doing* greenhouse gas accounting, in terms of implementing, conceptualising, and developing different accounting methods. This distinction is taken up again in

Section 5, below, when discussing the position of the present research within the greenhouse gas accounting literature.

It is worth emphasizing that the above discussion in no way suggests that highly theorised research is unimportant, but rather that the preference for theory-focused research may unintentionally result in the exclusion of other forms of valuable research. With some irony, theorised social research often emphasizes the plurality of possible perspectives on the world, whilst at the same time it excludes other forms of research which do not adopt the same theorised approach. Again with some irony, an interesting area for research may be the social norms and routines that reinforce the preference for theoretically informed research, i.e. social theory can be used to explain the dominance or hegemony of social theory research itself.

This issue is discussed here as the present research is largely focused on 'What?' questions, and normative method development, and does not therefore have the same explicit emphasis on theory as that present in explanatory social research. As noted above, this is not to suggest that explanatory social theory is unimportant, but is simply relevant to different types of question, some of which are discussed as ideas for further research in the Conclusions chapter. Also noted earlier, the present research is related to different forms of theory in a number of different ways, with the principal theoretical contribution being the development of the attributional-consequential distinction as categorical framework.

4. Methodology

Although each paper within this portfolio provides information on the methods used, there are some details which are not discussed fully in the papers, particularly in Paper 3, and this Methodology section offers an opportunity to provide some further detail. More specifically, it may be helpful to provide some further discussion on the reasons for choosing a case study as the methodological approach for Paper 3. It is also worth discussing the reasons for undertaking a single case study for this thesis, rather than multiple case studies, and the choice of the bioheat case study in particular.

Before proceeding, the sense in which the term 'case study' is used here should be briefly clarified, as it can be used in a wide variety of different ways (Tight 2010), and can also carry the connotation of favouring a qualitative form of research (Bryman & Bell 2007). For the present research the term is used to indicate an in-depth study of a single situation, i.e. a single decision scenario, to which various different greenhouse gas accounting methods are applied. The analysis is predominantly *quantitative* rather than qualitative, as the greenhouse gas accounting methods are quantitative, and the comparative analysis in Paper 3 focuses on the numerical/quantitative results from those methods.

The purpose of Paper 3 is to explore the difference in the results and information provided by an attributional method (corporate-level greenhouse gas accounting) and the two consequential methods available (consequential LCA and the project/policy method). A case study approach was selected as it allows each of these greenhouse gas accounting methods to be applied to the same decision scenario. This controls for differences in results due to differences in the scenario investigated, and therefore any differences in results will be due to the accounting methods employed. This approach approximates to an experimental method, as other variables are controlled for or held constant, and only the variable investigated (i.e. the accounting method used) is varied. There are precedents for using this approach, particularly in the field of life cycle assessment where case studies have been used to investigate the difference between attributional and consequential life cycle assessments (Ekvall & Andræ 2006; Thomassen et al. 2008; Dalgaard et al. 2008; Searchinger et al. 2008). The present research takes a similar approach, but, in contrast to existing studies, it appears to be the first to compare attributional *corporate* greenhouse gas accounting and different consequential methods.

One commonly cited weakness with the use of case studies is the limited evidence provided for making generalised statements or inferences (Gerring 2004; Lijphart 2011), i.e. it is not possible to infer from a sample of one to the whole population. However, this limitation is not relevant when the purpose of the case study is to disconfirm a general statement or hypothesis (Gerring 2004), i.e. it is deductively sufficient to show that attributional accounting is not always reliable for decision-making if there is one instance where this is the case. In addition, it is only possible to know whether a particular attributional account is reliable by checking it with a consequential method, in which case the initial attributional account would be redundant. A single case study will not be sufficient for estimating the *probability* that attributional accounts provide incomplete information (Lijphart 2011), but will be sufficient for disconfirming the general reliability of such accounts. This use of case studies is variously described as disproving 'invariate relationships' (Gerring 2004), 'nomothetic' case studies (Bryman & Bell 2007), or 'critical' case studies (Yin 2003).

Given the above, it was decided that the use of a single case study would be sufficient for the purposes of the research project. In addition, once the data collection and modelling for the bioheat case study was underway it became apparent that the depth of detail and number of possible modelling scenarios would be considerable, and that a single thorough case study would be more insightful than a larger number of less intensive studies. A further reason for focusing on a single case study is that the novelty of a first case study would lend itself to publication in an academic journal, but that subsequent case studies showing the same generalised conclusion would not be novel, and would be of less interest to potential reviewers or editors.

In terms of the choice of a bioheat plant for the case study, a number of reasons were taken into consideration. Firstly, given the deductive structure of the research, the first selection criterion was that the case study should be likely to provide a ‘crucial’ case, rather than necessarily a representative one (Gerring 2004), i.e. a case that would provide a counter-example to the general sufficiency of an attributional corporate inventory for decision-making. Bioheat appeared likely to provide such a case, based on previous studies comparing attributional and consequential LCAs for biofuels (e.g. Searchinger (2008)). The long time-horizons for the emissions and removals associated with forest growth also appeared likely to illustrate the difference between consequential LCA and the project/policy approach, in terms of their ability to model the temporal distribution of impacts.

A second selection criterion was the availability of data, as implementing the different greenhouse gas accounting methods would be difficult without data with which to populate the methods. The present research owes a large debt of gratitude to the participation of the developers of a 6 MW bioheat plant, and their provision of data, documents, and access to key personnel. The relationship with the organisation commissioning the bioheat plant was extremely timely and serendipitous, and although alternative approaches such as the use of secondary or proxy data to construct ‘shadow’ or ‘silent’ accounts (Dey 2007) may have been possible, the access to primary data greatly facilitated the research.

A third selection criterion was topicality, i.e. the case study should be relevant to current policy or decision-making, in order to ensure that the research has relevance to current practice. Again, the bioheat case study fulfils this criterion, given the high levels of policy support for bioenergy, and corporate investment in this area (e.g. Diageo (2015); European Parliament and Council of the European Union (2009); UK Government (2012); US Department of Energy (2015)). As a result of such support bioenergy is expected to increase from its current global energy output of ~10 EJ to ~56 EJ by 2050 (derived from IEA (2015)), an increase of over 450%. As a further indication of the relevance and topicality of the bioheat case study, the European Commission is currently undertaking a study into the environmental implications of the increased reliance on biomass imported from North America (Kittler et al. 2015), to which a draft version of Paper 3 has been submitted. There is also a live and on-going debate in the academic literature on the impacts of bioenergy (Bernier & Paré 2013; Bright et al. 2012; Schulze et al. 2012; Edrisi & Abhilash 2015; Searchinger 2012; Haberl et al. 2012; Upham & Smith 2014; Cherubini et al. 2009; Favero & Mendelsohn 2013; Haberl et al. 2013), to which the bioheat case study provides a further contribution.

Table 2 provides an overview of the reasons for selecting the bioheat case study, and the details of two further options which were considered, and which could be developed in future research.

Table 2. Possible case studies and selection criteria

Possible case studies	Likelihood of providing a 'crucial' case	Data availability	Topicality	Variation in case studies
Bioheat plant.	There are likely to be market-mediated or displacement effects from sourcing biomass for the heat plant, and these effects will not be reflected in an attributional account.	The bioheat plant developer is willing to participate in the case study.	Biomass policy is highly topical at present, particularly with the Drax conversion to biomass. In addition, there is an on-going debate on the impact of UK and EU biomass policy on forests in the US (Schlesinger 2014).	Bioenergy topic.
Increased use of aluminium in vehicles.	One of the other major uses of aluminium is in food packaging, where there are GHG emission benefits from enhanced preservation of food/reduced food wastage. If the packaging market is sensitive to price then increased use of aluminium in vehicles may decrease the use in packaging, and increase emissions from food wastage.	There is a large amount of data available on production volumes and capacity. Further scoping is needed to check the availability of data on the cross-price elasticity of demand for aluminium packaging.	The use of aluminium in vehicles is topical as it is one of the main options for meeting regulatory fuel efficiency targets (European Commission 2009).	Material use topic.
Purchase of renewable energy certificates.	The purchase of renewable energy certificates shows as a reduction in emissions in attributional corporate greenhouse accounts, whereas a consequential assessment would show whether actual emissions have reduced or not.	This case study would require data on the cost structure of renewable energy projects, which is available from previous studies on support mechanisms for renewable generation. The US energy company Bloom Energy may also be interested in participating in this case study.	The Greenhouse Gas Protocol published new guidance on scope 2 (electricity) reporting in 2015 (WRI 2015), however the guidance may not ensure accurate or relevant GHG accounting, and should be subject to detailed scrutiny and critique.	Renewable energy topic.

The background context and precise details of the bioheat plant are not given in Paper 3, largely in order to maintain the anonymity of the organisation commissioning the bioheat plant, and because the contextual details are not considered immediately relevant to the broader methodological conclusions drawn from the case study. However, for further context, it is worth noting that the bioheat plant was at an advanced stage of planning, with finance in place, but not yet implemented at the time the quantitative analysis in Paper 3 was undertaken. The analysis in Paper 3 was not intended to inform the decision on whether to implement the bioheat plant in the case study, but rather to draw broader conclusions about the most appropriate methods for informing mitigation decisions, and about the range of possible outcomes from bioenergy, more generally. The organisation commissioning the bioheat plant was keen to use the development as a learning opportunity, and although the outcomes from the study were not intended to inform the implementation of the plant, the results could potentially guide the operational management and sourcing of the biomass used.

As noted earlier, Paper 3 seeks to appraise a number of different greenhouse gas accounting methods based on a comparative analysis of the information they provide. Although discussed within Paper 3, the principles or criteria by which this appraisal is undertaken are left largely implicit within the paper, and it is therefore worth articulating them in more detail here. One of the underpinning principles used for this appraisal is that decisions (e.g. decisions aimed at mitigating climate change) should be based on the consequences of the decision in question, which is a key tenet of normative decision theory (Hansson 2005), and also has its theoretical roots within ethical consequentialism (Shafer-Landau 2013). Although it is largely to be expected that the consequential methods tested will better reflect the consequences of the case study decision, the intention in Paper 3 is to understand the potential magnitude of difference between the methods, and whether the attributional comparator method provides a reasonable proxy for the consequences of the decision. A further underlying principle used in the appraisal of the methods is that of *relevance* to decision-making, which is an established principle within many greenhouse gas accounting standards (e.g. the GHG Protocol *Corporate Accounting and Reporting Standard* (WBCSD/WRI 2004) or ISO 14064-1 (ISO 2006c). The principle of relevance holds that greenhouse gas information should serve the 'decision-making needs of the user' (WBCSD/WRI 2004, p.7). This principle is used within Paper 3 to appraise the absence of information on the temporal distribution of emissions in consequential LCA, i.e. consequential LCA does not provide this information, despite its relevance to decision-making, and this is taken as a justification for preferring methods that do provide such information.

A final point to make within this section, for the purposes of clarity, is that the present research treats the various *methods* for greenhouse gas accounting as the *objects* of the research, and investigates them using research methods such as conceptual analysis, literature review, document analysis, and the empirical case study. This is worth highlighting to avoid any potential confusion from the dual use of the term 'methods' within this thesis, i.e. the research objects to which the methodology is applied are themselves methods.

5. Relationship to the Literature

Each of the papers in this portfolio provides its own overview and discussion of the relevant background literature, with Paper 1 in particular providing a detailed review of the literature on the attributional-consequential distinction. This section is not intended to replicate those discussions, but instead aims to provide a broader overview of the types of existing research on greenhouse gas accounting, and where the present research fits within that literature. This section also provides an opportunity to revisit and extend parts of the literature review presented in Paper 1, and to offer a number of other remarks on literature-related issues.

One initial distinction that can be made within the literature on greenhouse gas accounting is between what might be described as 'technical' research or normative method development on one side, and research focused on the *social* practices and *social* implications of greenhouse gas accounting on the other. This distinction largely parallels the one made earlier between research for *doing* greenhouse gas accounting, and research on greenhouse gas accounting practices, qua social phenomena. On the technical side, this research often involves the straightforward implementation of accounting methods, such as life cycle assessment, and the reporting of the empirical findings. For example, Maxineasa et al. (2015) provide a life cycle assessment for carbon fibre-reinforced polymer flexural strengthening solutions for reinforced concrete beams, and they simply report their findings from implementing the method (though, as an aside, they fail to clarify whether it is an attributional or consequential LCA, which indicates the still evolving recognition of the distinction within the LCA community). In addition to such straightforward empirical exercises, the technical academic literature on greenhouse gas accounting is also characterised by numerous studies which propose, critique, and discuss methodological issues, with the aim of improving or developing existing methods, standards and practice, i.e. they offer prescriptions for how greenhouse gas accounting *should* be done. There are occasionally papers that are wholly devoted to methodological techniques and procedures, such as Ekvall and Weidema (2004), but more often there is a tendency to advance a methodological proposal with an empirical example or application (e.g. Weidema et al. (1999), Ekvall and Andræ (2006), Schmidt (Schmidt 2008), Brander and Wylie (2012), or Chalmers et al. (2015)). It is worth noting this is essentially the approach used in Paper 3, i.e. a number of methodological proposals are illustrated with the use of an empirical case study.

A further strand within the 'technical' literature is the occasional instance of an almost purely conceptual discussion, such as Ekvall et al. (2005), which explores the parallels between method choice, i.e. between either attributional or consequential

LCA, and theories of normative moral philosophy, i.e. utilitarian or deontological ethics. However, even in this highly conceptual discussion the expectation of some empirical content is evident, and the prosaic example of a conference centre is included (Ekvall et al. 2005, p.1229)).

A final observation on the 'technical' side is that the academic literature appears to be dominated to a large extent by life cycle assessment, which has a dedicated journal (i.e. *The International Journal of Life Cycle Assessment*) and international conferences (e.g. the Society of Environmental Toxicology and Chemistry conferences). It is interesting to reflect that other forms of greenhouse gas accounting, such as corporate-level greenhouse gas accounting or project-level assessment, which are both very widely implemented in practice, do not have a correspondingly large academic community which debates and advances methodological issues, in the way that LCA academics do for LCA practice. There are of course instances of technical academic literature for corporate greenhouse gas accounting (e.g. Huang et al. (2009), Huang et al. (2009), Trexler and Schendler (2015)), community-level accounting (e.g. Erickson & Lazarus (Erickson & Lazarus 2012), Brander et al. (2014)), project-level accounting (e.g. Gustavsson et al. (2000), Vöhringer et al. (2006), Trexler et al. (2006), Kartha et al. (2004)) and policy-level accounting (e.g. Okubo et al. (2011)), and national inventories (e.g. Peters and Hertwich (2008), Swart et al. (2007), Davis & Caldeira (2010)), but not with the same level of community identity or cohesion as that evident for life cycle assessment. This observation on the fragmented social groups engaged in 'techncial' greenhouse gas accounting research aligns with the idea that many of these methods and practices have developed in semi-isolation of one another, giving rise to the present opportunity for sharing lessons between them, which, as mentioned above, constitutes one of the underlying premises for this thesis.

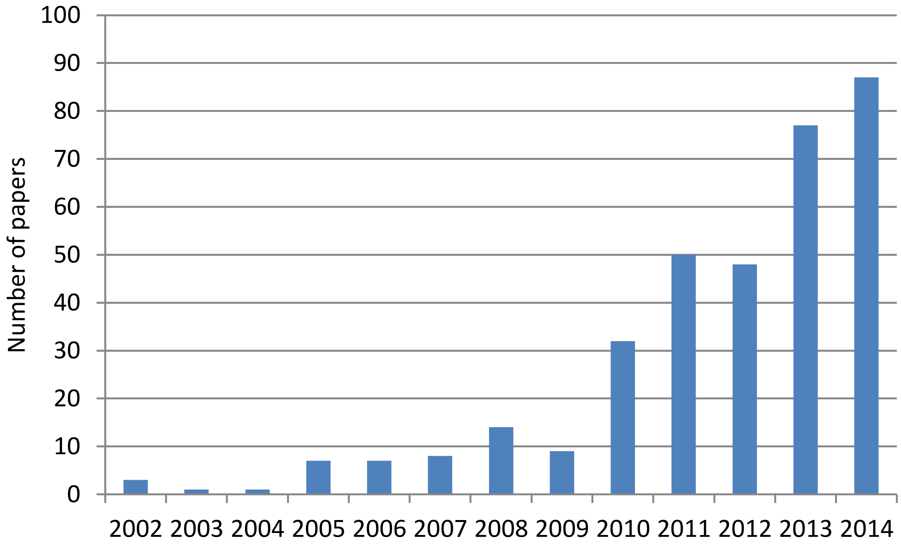
The above observation on social context provides a useful segue for turning to the other side of the academic literature on greenhouse gas accounting, which is concerned with the social practices that constitute greenhouse gas accounting. Ascui (Ascui 2014) provides a comprehensive literature review for this form of *carbon* accounting research, which includes research on both the financial and physical forms of carbon accounting. The review suggests a further useful distinction specifically within this type of academic research, that between 'critical, philosophical, and normative discussions *about* carbon accounting' and 'empirical studies *of* carbon accounting' (Ascui 2014). An example of the former is Lohmann (2009), which emphasizes the social construction of carbon accounting practices, and provides a critical perspective on the effects of those practices, including the way in which they privilege certain groups and embed existing power structures. An example of the latter is Gallego-Álvarez et al. (2011), which provides a quantitative empirical study of the drivers for corporate disclosure on climate change opportunities, using an econometric model to identify the explanatory variables for disclosure. It is worth noting that both these examples of social research explicitly use some form of social theory as an explanatory device, in Lohmann (2009) it is the concept of 'framing' (which is also present in Paper 1), and in Gallego-Álvarez et al. (2011) it is legitimacy theory, which serves to further illustrate the earlier point that the explicit use of theory is an expected characteristic within the field of social research.

The main purpose for providing this brief overview of the academic literature is to orientate the present research within it. The present research is primarily concerned with the conceptualisation and development of methods for greenhouse gas accounting, and not with the social context or drivers for those methods, and therefore sits within the technical academic literature (with the brief exception of the use of 'framing' within Paper 1). It is interesting to note that much of the social research, in contrast with the technical research, does take a holistic perspective across different forms of greenhouse gas accounting. For instance, Ascui and Lovell (2011) show the immense range of different activities that fall under the label 'carbon' accounting. However, the social research agenda has not been concerned with normative method development *for* greenhouse gas accounting, as evidenced by Thomson's (2007) review of the social research into sustainability accounting more generally, and this means that the opportunity for sharing methodological lessons across fields of practice, which motivates the present research, has remained largely untapped.

All this is not to say that the present research is not of potential relevance to the themes and issues explored by the critical or empirical social research on greenhouse gas accounting. As will be suggested in Paper 1, the attributional-consequential distinction may be relevant for critiquing existing practice, e.g. by exploring the way in which attributional methods limit the visibility of, and responsibility for, the impacts of corporate activities. Similarly, the attributional-consequential distinction may also be useful to empirical studies of greenhouse gas accounting, e.g. for characterising forms of accounting and investigating, empirically, the presence or absence of those forms of practice. The attributional-consequential distinction could also be useful to other undertakings within the social research literature, such as the attempt to define 'carbon' accounting. Stechemesser and Guenther (2012, p.35) suggest that 'carbon accounting' comprises 'the recognition, the non-monetary and monetary evaluation and the monitoring of greenhouse gas emissions on all levels of the value chain and the recognition, evaluation and monitoring of the effects of these emissions on the carbon cycle of ecosystems'. Interestingly, this definition does not distinguish between inventories of emissions (attributional methods) and assessments of *changes* in emissions (consequential methods), and does not appear to include the possibility of accounting for emissions beyond the value chain (i.e. life cycle) of the entity in question. The definition of 'carbon' accounting provided in Ascui and Lovell (2011) is considerably more inclusive, and does recognise the both inventories and assessments of change, but does not categorise the broad array of practices identified according to this distinction. Awareness of the attributional-consequential distinction could therefore help to broaden the definition given, in the case of Stechemesser and Guenther (2012), and to group conceptually related forms of practice into distinct categories, in the case of Ascui and Lovell (2011). The present research also raises further research questions, such as *why* seemingly inappropriate accounting methods are used to inform decision-making, as appears to be the case in the bioheat case study, with such questions lending themselves to some form of socially theorised explanation. The topic of possible further research questions generated by this thesis is discussed in more detail in the Conclusions chapter.

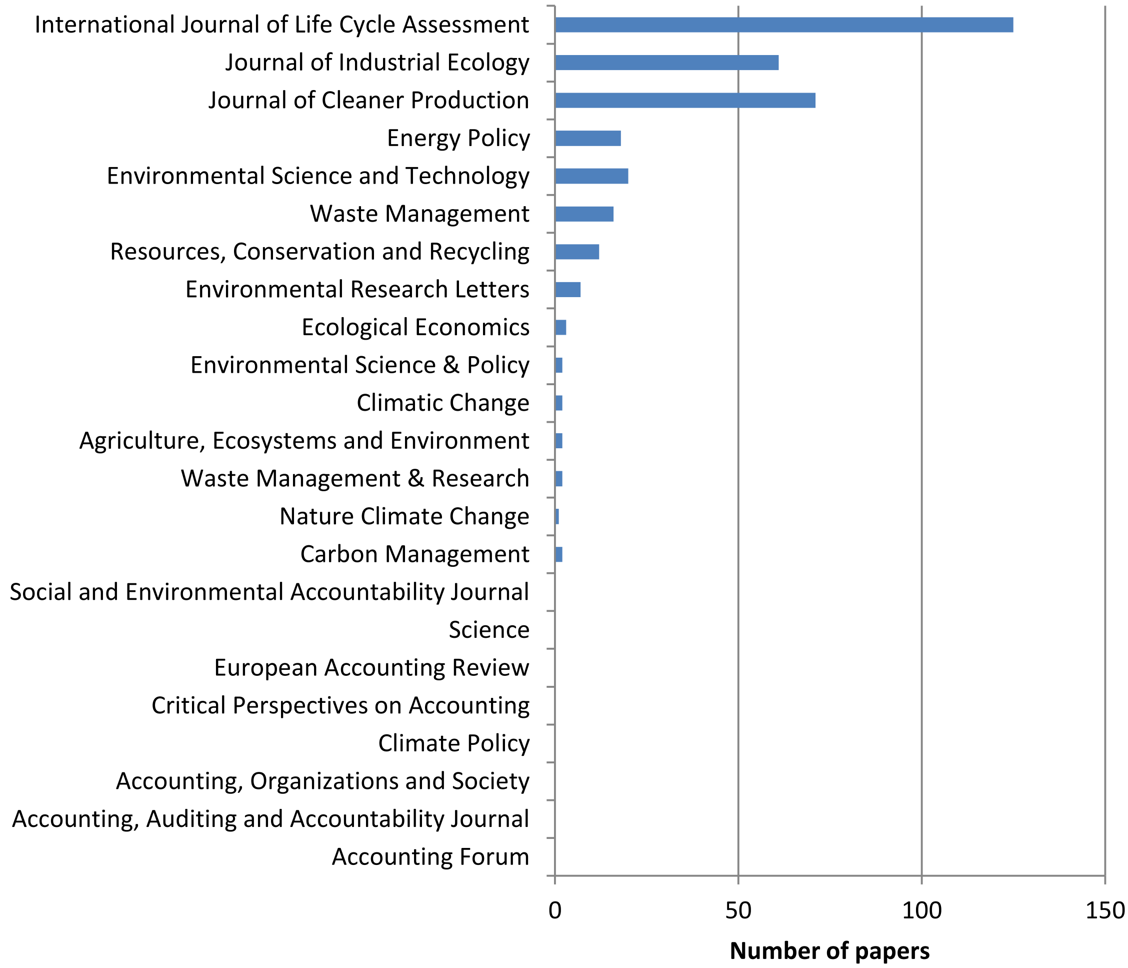
In addition to situating the present research within the broader greenhouse gas accounting literature, this section also provides an opportunity to update parts of the literature review presented in Paper 1, which was undertaken in 2013 and early 2014. Figure 3 below provides an updated version of Figure 5 in Paper 1, to include data for 2014. It is also useful to show this within the context of this Introduction, as the growth in the use of the distinction indicates its utility as a conceptual innovation. It is also useful to highlight that the publications using the distinction, shown in Figure 3 below, are almost entirely from the life cycle assessment literature, which again emphasizes the considerable opportunity for extending the distinction to other fields of practice.

Figure 3. Number of papers referring to the attributional-consequential distinction by year of publication.



An additional chart, which was not included in Paper 1 due to space limitations, shows the number of papers using the attributional-consequential distinction by journal. Figure 4 below is an updated version of this omitted chart, which indicates that, as mentioned above, the attributional-consequential distinction has not yet been used within the social research literature on greenhouse gas accounting (the proxy for which are the journals included in Ascui's (Ascui 2014) review of the social and environmental accounting literature, i.e. *European Accounting Review*, *Critical Perspectives on Accounting*, *Accounting, Organizations and Society*, *Accounting Auditing and Accountability Journal*, and *Accounting Forum,* but excluding the *Journal for Cleaner Production* which publishes predominantly technical research, but also some social research). This reinforces the point that in addition to extending the attributional-consequential distinction beyond the field of life cycle assessment to other parts of the technical academic literature on greenhouse gas accounting, there also appears to be considerable opportunity to extend it to the social research literature as well.

Figure 4. Number of papers using the attributional-consequential distinction by journal



A final introductory point to highlight with regard to the literature is that the debate on the nature of attributional and consequential life cycle assessment is still a highly active one, evidenced by the recent and lively exchange of views over whether the use of attributional LCA misleads policy-makers (R. J. Plevin et al. 2014b; Brandão et al. 2014; Hertwich 2014; Suh & Yang 2014; Anex & Lifset 2014; Dale & Kim 2014; R. J. Plevin et al. 2014a; R. Plevin et al. 2014). Similarly, there is an on-going debate on the nature of attributional LCA, and whether baselines (i.e. counterfactual scenarios) are conceptually or methodologically appropriate (Soimakallio et al. 2015), to which Paper 4 makes a direct contribution. This indicates that the conceptualisation of the attributional-consequential distinction is not a wholly settled matter, even within the field of life cycle assessment itself. The present research exercise of extending the attributional-consequential distinction appears likely to yield opportunities for feeding-back lessons from other forms of greenhouse gas accounting, and for contributing to the evolving conceptualisation of the distinction.

Paper 1 - The attributional-consequential distinction and its applicability to corporate carbon accounting

Abstract

Methods of carbon accounting have developed in a number of semi-isolated fields of practice, such as national inventory accounting, corporate carbon accounting, project level accounting, and product life cycle assessment, and there appears to be considerable potential for learning across these different fields. One methodological distinction that has emerged within the field of life cycle assessment (LCA), and which has been highly useful there, is that between attributional and consequential methods. However, this distinction has not been fully developed or explored within the field of corporate carbon accounting. Attributional methods provide static inventories of emissions allocated or attributed to a defined scope of responsibility, while consequential methods attempt to measure the total system-wide change in emissions that occurs as the result of a decision or action, such as the decision to produce one extra unit of a given product. Numerous LCA studies show that attributional inventories can ignore important indirect or market-mediated effects that occur outside the scope of the analysis, and thus decisions based on attributional information can result in unintended consequences. Given that the most widely recognised form of corporate carbon accounting (the organisation-level greenhouse gas inventory) is attributional in nature, it is probable that decisions based on such inventories may also result in unintended consequences. This paper explores the nature of the attributional-consequential distinction and its applicability to corporate carbon accounting. In addition, the concept of framing is used to help explain how the distinction developed within the field of LCA, and to highlight the conceptual work required to achieve a degree of consensus around the distinction within that community, which in turn may be helpful when considering its applicability beyond the field of life cycle assessment.

1. Introduction

Methods of carbon accounting (used here as shorthand for all forms of greenhouse gas related accounting) have developed in a number of semi-isolated fields of practice, such as national inventory accounting, corporate carbon accounting, project level accounting, and product life cycle assessment, and there appears to be considerable potential for learning across these different fields (Ascui & Lovell 2011). One methodological distinction that has emerged within the field of life cycle assessment (LCA), and which has been highly useful there, is the distinction between attributional and consequential methods (Finnveden et al. 2009). However, this distinction has not yet been widely appreciated or explored within the field of corporate carbon accounting. Attributional methods provide static inventories of emissions allocated or attributed to a defined scope of responsibility, while consequential methods attempt to measure the total system-wide change in emissions that occurs as the result of a decision or action, such as the decision to produce one extra unit of a given product (Ekvall & Weidema 2004; Curran et al. 2005). Numerous LCA studies show that attributional inventories can ignore important indirect or market-mediated effects that occur outside the scope of the analysis, and thus decisions based on attributional information can result in unintended consequences (Searchinger et al. 2008; Hertel et al. 2010). While organisations collect many different types of both monetary and physical carbon-related information (Burritt et al. 2011), the most widely recognised form of corporate carbon account is the organisation-level inventory of physical greenhouse gas emissions, typically produced for the purposes of voluntary carbon disclosure (but which may also be produced for mandatory reporting, participation in emissions trading schemes or internal management purposes), following standards such as the GHG Protocol (WBCSD/WRI 2004), Defra reporting guidance (Defra 2009; Defra 2013) or ISO14064-1 (ISO 2006c). These standards guide the production of corporate carbon accounts that are attributional in nature (CDP 2013; Brander & Wylie 2012) and thus it is probable that decisions based on such inventories may, like attributional LCAs, result in unintended consequences. Applying the attributional-consequential distinction to corporate carbon accounting may therefore be useful for choosing appropriate methods to inform decision-making, and for understanding the nature and limitations of mainstream (attributional) corporate carbon accounting more generally.

This chapter is structured in two parts. The first part provides an introduction to the attributional-consequential distinction, including a chronology of the development of the distinction, an analysis of the core features of attributional and consequential approaches, examples of the results obtained from each method, and an overview of the critical discussion in the literature concerning the distinction. The second part of the chapter then considers the applicability of the distinction to corporate level accounting, and discusses the utility of the distinction for designing coherent corporate carbon accounting methods, the implications for corporate-level accounting, and the potential usefulness of the distinction for academic research on social and environmental accounting.

The existence of the attributional-consequential distinction in one field (LCA) and its absence in a cognate field (corporate carbon accounting) begs the broader question of *why* this should be the case. While we cannot offer a definitive answer to this question, we believe that the history of the emergence of the distinction in LCA demonstrates that thinking in terms of the systemic consequences of a decision or action, rather than in terms of attributing responsibility for a given situation, involves a conceptual shift – a subtle change of emphasis with far-reaching implications – that is challenging and difficult to introduce when the dominant thinking is attributional. The change of emphasis has different disciplinary roots and is clearly self-evident in one tradition and not in another. This suggests that attributional and consequential methods are not equally available methodological alternatives, but rather that they are bound up with a broader set of preconceptions about how the world works, what matters and how we should respond to a given situation, or what scholars across a range of disciplines would call ‘framing’. Framing refers to ‘the processes by which people construct interpretations of problematic situations, making them coherent from various perspectives and providing users with evaluative frameworks within which to judge how to act. …Framing is problematic because it leads to different views of the world and creates multiple social realities.’ (Rein & Schon 1993, p.147). This does not mean that differences in framing, such as the difference between attributional and consequential accounting methods, are irreconcilable; rather, recognition of frames facilitates the more effective use of different approaches in their appropriate contexts. The concept of framing is an additional explanatory thread that we return to in a number of places in the chapter. In particular, we believe this level of analysis helps to explain the observed pattern of resistance and recognition in the development of the distinction in the field of LCA, which in turn suggests that recognising its implications for corporate-level carbon accounting may be similarly challenging, yet ultimately highly beneficial for both academic research and practice in this area.

2. The Attributional-Consequential Distinction

2.1. Chronology of the distinction

In order to understand the development of the attributional-consequential distinction it is useful to first look briefly at the development of LCA more generally, as it is in LCA that the distinction first developed and is still primarily employed.

LCA can be defined as the ‘compilation and evaluation of the inputs and outputs and the potential environmental impacts of a product system throughout its life cycle’ (ISO 2006b). One of the motivations for the development of LCA was the recognition that for a complete account of a product’s environmental impact it is necessary to look at all its life cycle stages (i.e. material extraction, manufacturing, transportation, use phase, and end-of-life disposal), rather than only individual stages, such as the use phase (Guinee et al. 2011). LCA typically includes a number of environmental impact categories, such as human toxicity, resource deletion, eutrophication, greenhouse gas emissions etc., and therefore has a broader scope than carbon accounting. However, the development of the attributional-consequential distinction relates equally to the carbon impact category, as to any other impact category, and the multi-impact nature of LCA does not seem to pose any fundamental limitation on the lessons that can be transposed from this field of practice to 'pure' carbon accounting.

LCA emerged in the 1960s and 1970s, and initially focused on resource use, energy, and waste (Guinee et al. 2011). Following a number of initial studies, which were primarily undertaken by companies (Hunt & Franklin 1996; Jensen et al. 1997), the practice of LCA was formalised in a number of guidance documents, for example, the Hand-book of Industrial Energy Analysis (Boustead & Hancock 1979); and later, the Manual for the Environmental Life Cycle Analysis of Products (Guinee et al. 1991); the Environmental Life Cycle Assessment of Products: Guide and Backgrounds (Heijungs et al. 1992), and Life Cycle Assessment: Inventory Guidelines and Principles (Vigon et al. 1993).

In 1993 Weidema noted that none of the recently published guidance or manuals 'adequately reflects the importance that market aspects and the economic disciplines may have in life cycle inventory methodology' (Weidema 1993, p.161). Weidema suggested that 'the use of environmental data on the marginal production reflects most correctly the actual environmental impact' (Weidema 1993, p.163), and that inventories should reflect 'to the largest extent possible, the actual consequences of implementing the results of the investigation' (Weidema 1993, p.166). This emphasis on quantifying the consequences of a decision or action, as distinct from quantifying the total environmental burdens associated with the processes directly used by or connected with the entity studied, is the essence of the 'consequential' approach.

The fact that the attributional-consequential distinction did not appear until some 30 years into the development of LCA demonstrates that it was not initially self-evident to those involved. A full examination of the communities involved in LCA is beyond the scope of this chapter, but Weidema's mention of 'market aspects and the economic disciplines' strongly suggests a new recognition of a different framing of the world (by economists) within a field previously dominated by engineers and natural scientists: as Earles and Halog (2011, p.445) put it: 'CLCA [consequential LCA] represents the convergence of LCA and economic modelling methods'. In fact, Weidema (2003, p.166) explicitly calls for 'an interdisciplinary approach… where technical experts, market experts and economists join forces'.

After this call to action, it still took time for this elucidation of an alternative framing to be adopted by other members of the LCA community. During the 1990s a small number of studies began to identify and model the processes that change as a result of a decision (the so called 'marginal processes' – terminology explicitly borrowed from economics). For example, Ekvall et al. (1998) used marginal data to study the environmental impact of different forms of packaging for beer and soft drinks, and Frischknecht states that to 'reflect the consequences of decisions, models capable of representing changes within the economic system shall consist of processes represented by marginal technologies, the technologies put in or out of operation next' (Frischknecht 1998, p.67).

The formalisation of both the consequential and attributional methods has developed since the late 1990s through the publication of further standards, guidance, and a number of key journal articles on methodological issues. On the consequential side, the key publications include: Weidema (1999) which proposes a 5-step process for identifying the marginal technologies that change as a result of a decision; Weidema (2003) which is a detailed report for the Danish Environmental Protection Agency on the use of market information in life cycle assessment; Ekvall & Weidema (2004) which brings together guidance from various sources on how to determine which technological processes to study and how to identify marginal data; and Schmidt (2008) on system delimitation for agricultural products.

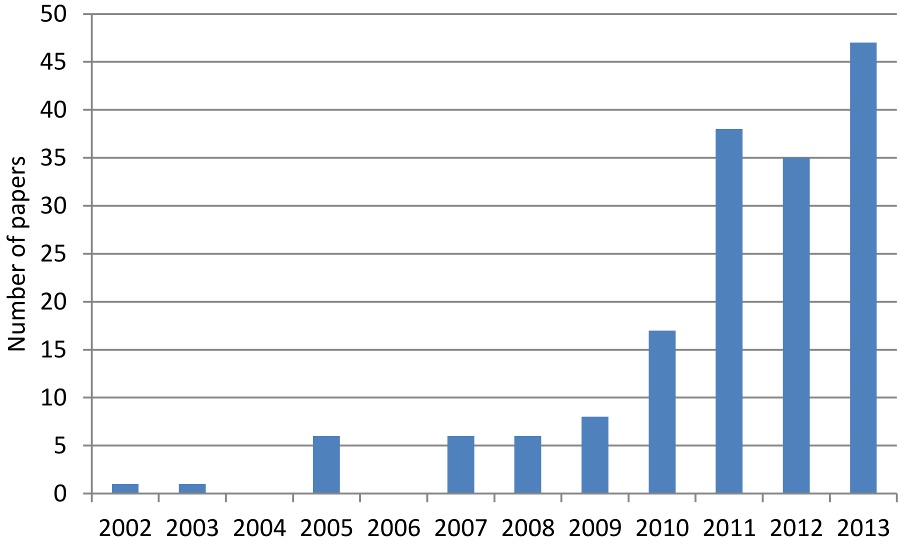
Nevertheless, despite the work represented by this proliferation of guidance, the distinction has not always been clear to different members of the LCA community. Even at the level of basic terminology, convergence on the terms 'attributional' and 'consequential' required a process of deliberate consensus-building. These terms were adopted in 2001 at an international multi-stakeholder workshop on electricity data (Ekvall & Weidema 2004; Curran et al. 2005), but prior to that, and in fact until as recently as 2009 (see Nielsen and Høier (2009)), authors had used a variety of terms to refer to essentially the same distinction. Attributional methods have been variously denoted by the terms 'retrospective', 'accounting', 'descriptive', 'book-keeping' and 'traditional', while consequential methods have been denoted by the terms 'prospective', 'market-based', 'decision-based', 'change-oriented' or 'marginal' (European Commission et al. 2010).

In addition to confusion created by the use of different terms, awareness of the attributional-consequential distinction has also not been helped by the existence of influential standards which do not mention the distinction (ISO 2006a; ISO 2006b), fail to clarify whether the standards are intended to cover attributional or consequential approaches, or both (Geyer 2008; Ekvall & Finnveden 2001; Ekvall 1999; Tillman 2000; Brander & Wylie 2012). The International Reference Life Cycle Database System (ILCD) handbook (European Commission et al. 2010) is another internationally recognised source of guidance on LCA, which has contributed to the confusion in a slightly different way. The handbook clearly acknowledges and discusses the attributional-consequential distinction, but it also identifies four distinct application contexts, and structures the guidance on methods accordingly. Unfortunately, it is not always transparent which of the distinct application contexts correspond to attributional or consequential methods, or whether a mixture of methods is at play in the different application contexts.

At this point in time in the LCA community there appears to be growing consensus; Finnveden et al. (2008, p.365) state that there 'is today a general agreement within the life cycle assessment (LCA) community that there are two types of LCA… These are often called attributional and consequential LCA'. However, even with this 'general agreement' and the increasing usage of the distinction in the academic literature (as shown in Figure 5 below), there is still a continuing and lively debate on a number of issues, such as the correct purpose of each method, and the relative advantages of each (as will be discussed later in the chapter).

In summary, this brief history of the development of the attributional-consequential distinction within the field of LCA shows that the distinction was not initially self-evident to those involved (only emerging some 30 years into the development of the field) and that acceptance of the distinction since the early 1990s has been gradual, uneven and contested, all the way from the level of basic terminology up to fully developed international standards and guidance manuals. The alternatives have different disciplinary origins, with the consequential approach bringing concepts borrowed from economics to a field previously dominated by engineers and natural scientists. Deliberate consensus-building efforts have led to convergence on the terms 'attributional' and 'consequential', but there is still a lack of coherence on the distinction in various international standards and guidance manuals. These features all suggest that attributional and consequential methods are not simply two equally available methodological alternatives, but rather two different ways of framing an accounting problem. Nevertheless, a process of reframing, that is still underway, has led to wider appreciation of the distinction and hence to a better understanding of the applicability of each of the alternatives in different contexts.

Figure 5. Number of papers referring to the attributional-consequential distinction by year of publication.[1]



2.2. Key characteristics of attributional and consequential approaches

Many authors (Ekvall 2002; Ekvall & Weidema 2004; Curran et al. 2005; Ekvall et al. 2005; Schmidt 2008; Earles & Halog 2011) have proposed definitions or descriptions of attributional and consequential forms of life cycle assessment.

Two key characteristics can be drawn from these various descriptions:

[1] Word searches for the terms "attributional" and "consequential" were conducted on the web sites for the journals listed below. Articles that used the terms in ways other than to refer to the attributional-consequential distinction were excluded. The journals included in the search were: International Journal of Life Cycle Assessment; Journal of Industrial Ecology; Journal of Cleaner Production; Energy Policy; Environmental Science and Technology; Waste Management Resources, Conservation and Recycling; Environmental Research Letters; Ecological Economics; Environmental Science & Policy; Climatic Change; Agriculture, Ecosystems and Environment; Waste Management Research; Nature Climate Change; Greenhouse Gas Measurement and Management; Social and Environmental Accountability Journal; Science; European Accounting Review; Critical Perspectives on Accounting; Climate Policy; Accounting, Organisations and Society; Accounting, Auditing and Accountability; Accounting Forum.

1. Firstly, consequential assessments are concerned with describing change, whereas attributional assessments are a description of a static state. The results from a consequential assessment represent the amount by which emissions change between one state or scenario and another, while the results from an attributional assessment are for absolute quantities of environmental impacts, for a single given state or scenario (Ekvall 2002; Curran et al. 2005; Ekvall et al. 2005).

2. Secondly, consequential assessments are concerned with total changes wherever they occur, whereas attributional assessments are only concerned with the environmental impacts physically used by or produced by the life cycle under analysis (Ekvall & Weidema 2004; Earles & Halog 2011).

A number of other subsidiary features that distinguish consequential from attributional methods can also be identified in the definitions in the literature, but these can be understood as methodological techniques for fulfilling the two key characteristics identified, rather than being essential characteristics in their own right. These subsidiary features include:

1. *Use of economic modelling*. Economic modelling methods are often a characteristic of consequential LCA and are used to identify the processes affected by changes in demand and supply. For example, in Ekvall & Andræ (2006), reduced demand for lead due to the promotion of lead-free solder reduces the price of lead and increases its use elsewhere – and this additional usage is then included in the analysis. Most consequential assessments model market-mediated effects, but with varying degrees of sophistication, ranging from simple identification of market trends and the most/least competitive technologies (e.g. Weidema et al. (1999)) to the use of sophisticated computable general equilibrium models (Searchinger et al. 2008; Hertel et al. 2010).

Attributional LCA, in contrast to consequential LCA, only considers the physical flow of resources to, and impacts from, the physical processes used during the life cycle of the product, and does not model market-mediated effects.

2. *System expansion*. System expansion (or 'substitution') is a method for dealing with co-products (or other forms of multi-functionality) whereby credit is given to the product studied for the environmental impacts that are avoided due to its co-products replacing alternative forms of production (Heijungs & Guinée 2007). For example, beef co-products from the dairy industry replace dedicated beef and pork production, and the avoided impacts due to the avoided production are credited to the production of milk (Thomassen et al. 2008).

   In contrast, attributional LCA tends to allocate emissions between co-products, on the basis of physical characteristics such as mass or energy content, or alternatively, economic value (Thomassen et al. 2008; Schmidt 2008). There is continuing debate within the LCA community as to whether attributional LCA can also use system expansion, with some standards allowing its use (WBCSD/WRI 2011c; European Commission et al. 2010). However, Brander & Wylie (2011) suggest that doing so introduces values for avoided emissions into what should only be an inventory of actual physical emissions or removals.

3. *Source of data*. Attributional LCA uses either data for the specific physical processes used in the life cycle of the product, or average data, such as average emissions from grid electricity (Curran et al. 2005). Consequential LCA only considers marginal data, which provide information on the processes that change, rather than the processes that are physically used in the life cycle of the product studied (Schmidt 2008). For example, if there is an additional unit of demand for electricity then the generation technology that is deployed to meet that demand is the marginal process. Similarly, if there is one less unit of

demand then the generation technology that is reduced is the marginal process (Curran et al. 2005).

A summary of the key and subsidiary characteristics of the attributional-consequential distinction is presented in Table 3 (modified from a similar table in Thomassen et al. (2008)).

Table 3. Key characteristics of the attributional-consequential distinction

		Attributional	Consequential
Key characteristics	What is described or modelled?	Static inventory of absolute emissions and removals	Change in emissions or removals caused by a specific decision/action
	System boundary	Physical processes used in life cycle under analysis	Any process that changes as a result of the decision studied
Subsidiary characteristics	Economic modelling	Not used	Often used
	Treatment of co-products/multi-functionality	Disagreement on whether system expansion (substitution) should be used	System expansion (substitution) used
	Data	Process-specific or average	Marginal

2.3. Significance of the difference between the methods

An important question is whether using an attributional method rather than a consequential method produces materially different results, to the extent that different decisions would be taken had the alternative method been used. If the methods tend to produce similar results then there is little practical significance to the attributional-consequential distinction. However, if the methods produce very different results, then there is at least a possibility that different decisions would be made if information from the other method was available.

A number of papers have applied both attributional and consequential LCA methods to the same product in order to understand the difference in results generated. Ekvall & Andræ (2006) found very little difference between the attributional and consequential results for lead-free solder, and the assessment by Dalgaard et al.

(2008) of soybean meal found that the results from using the consequential method ($721gCO_2e$/kg of soy meal) were only trivially different to the attributional results ($726gCO_2e$/kg of soy meal). However a number of key impacts were not included in the latter study, such as the avoided emissions from deforestation due to the soy oil co-product replacing palm oil, which could have lowered the consequential results considerably (see Schmidt (2010) for the significance of land use change on the emissions from palm oil).

Thomassen et al. (2008) found that the consequential results for milk production ($901gCO_2e$/kg of milk) were significantly lower than the attributional results ($1{,}560gCO_2e$/kg of milk) due to beef co-products from the dairy industry replacing dedicated beef and pork production, and thereby avoiding large quantities of emissions. Similarly, Viera and Horvath (2008) found that the attributional results for concrete were higher than the consequential results, but that the decision supported by the information would be the same, i.e. recycling concrete is shown to reduce emissions in both cases.

However, there are cases where the decision supported by an attributional account is markedly different from that supported by a consequential assessment. A seminal paper for consequential LCA is a study by Searchinger et al. (2008) on indirect land use change caused by increased demand for biofuel crops. US government biofuel policy was predicated on the fact that attributional LCAs show corn ethanol to have lower emissions than conventional gasoline ($74gCO_2e$/MJ of corn ethanol compared to $92gCO_2e$/MJ of gasoline). However, using cropland for biofuels displaces food production elsewhere in the world, and some of the new cropland is likely to be converted from ecosystems such as forests or grasslands, resulting in high losses of stored carbon. Searchinger et al. showed that if the emissions from indirect land use change are taken into account the emissions for corn ethanol are in the region of $177gCO_2e$/MJ of fuel, or 93% higher than gasoline. Other studies, such as Hertel et al. (2010) have since replicated this work and produced lower emission estimates, but still found that US biofuel policy is likely to increase greenhouse gas emissions.

The magnitude of difference between attributional and consequential LCA results clearly depends on the specific product that is studied. However, it is also clear that in some cases the difference can be very large, and using a single method for a given purpose (such as using attributional methods to inform policy-making) can result in unintended or negative outcomes, as with US biofuel policy. The appropriate uses for each method is one of the main contentions in the literature, and this is discussed next.

2.4. Contentions in the literature

A long-running debate in the LCA literature is over whether there is any purpose for which an attributional approach is more appropriate than a consequential one (and if the distinction can be transposed to corporate-level accounting, the question is whether there is any purpose for mainstream attributional corporate carbon accounting that could not be better served by an alternative consequential method). Wenzel (1998) suggests that the only purpose of an LCA is to inform decision-making, which implies that the only appropriate method is a consequential approach, as it is this approach that explicitly aims to quantify the total consequences of decisions. There does appear to be a strong case for favouring consequential over attributional LCAs in most application contexts, such as those listed in ISO 14044 (ISO 2006b):

a. identifying opportunities to improve environmental performance;
b. informing decision-makers for priority setting and process design;
c. selecting indicators of environmental performance; and
d. marketing

Each of these application contexts either implicitly or explicitly involves decision-making, and therefore warrants a consequential approach. For example 'marketing' suggests that consumers may use the information to make decisions about which product to buy, and if consumers want to choose a product that causes the lowest environmental impact, then they will need a consequential assessment. Similar considerations apply to the other application contexts as well.

Tillman (2000) agrees that all LCA is either directly or indirectly concerned with change, but argues that there is still a role for attributional accounts, for instance, in identifying emissions ‘hot spots’ that can be targeted with abatement actions. However, this appears to beg the question, ‘How do we know that our actions to manage hot spots don’t have unintended consequences?’ which suggests that a consequential assessment is still needed to operationalize the attributional information.

An alternative attempt to carve out a role for attributional LCA is made by Ekvall et al. (2005). They give the example of a Swedish energy user that could be incentivised to isolate its hydropower plant from the electricity grid in order to avoid having to account for electricity consumption using the emission factor for the marginal technology in the Nordic electricity grid, which is coal. Doing so would increase emissions at the system level, as excess hydropower from the plant would no longer be supplied to the grid. Attributional accounting would not incentivise this behaviour, as the energy user could report the low emissions from its hydropower. However, it is not clear that consequential accounting would truly incentivise an increase in emissions. If a consequential approach were applied to the question, ‘What will happen if the energy user isolates its hydropower from the grid?’ the answer would be ‘It will increase emissions’, and decision-making based on minimising total system emissions would lead the energy user to maintain the connection of its hydropower plant to the grid.

Arguments about ease of application are often intermingled with arguments about the purpose of attributional and consequential approaches. For instance, many of the arguments in favour of attributional accounting in Tillman (2000) centre on the difficulties in identifying marginal processes or in undertaking system expansion (rather than arguing that attributional methods are conceptually more appropriate). Advocates of consequential LCA, particularly Weidema (2003), have argued that the consequential approach is actually simpler, as only those processes that change need to be modelled, and once the marginal product for a sector has been identified, this can be used for all other assessments that involve that sector. For instance, all consequential assessments involving changes in demand for vegetable oil only need to consider palm oil, as this is the marginal form of vegetable oil (Schmidt & Weidema 2008).

However, the claim that consequential accounting is simpler is not borne out by the take-up of the approach. Despite the apparent superiority of a consequential approach in most application contexts, attributional LCA continues to have greater levels of usage. For instance, the Carbon Trust's Carbon Footprint Label (Carbon Trust 2013) is based on the PAS 2050 (British Standards Institute 2011) and GHG Protocol (WBCSD/WRI 2011c) methodologies, both of which are attributional in nature. The main reason for the continued preference for attributional accounting appears to be the complexity of consequential modelling, and the difficulty in sourcing marginal data (which was the experience reported by Ekvall & Andræ (2006) in their case study for lead-free solder). Another suggested reason is the greater comprehensibility of attributional results, as users may struggle with conceptualising market-mediated or system wide impacts (Thomassen et al. 2008).

There is one application context for LCA where an attributional approach could genuinely be more appropriate (and not just easier to apply), and that is in measuring absolute environmental burdens, such as the total emissions associated with consumption (Zamagni et al. 2012). The results from attributional LCAs are additive and do not double-count the impacts included in other product life cycles, thus the sum of attributional results should approximate total actual impacts

(Tillman 2000). In contrast, the results from consequential assessments are non-additive, and reflect changes in emissions rather than absolute emissions. Attributional accounts could therefore be used for setting consumption ‘budgets’ in order to meet normative targets for absolute total emissions (such as 450ppm atmospheric CO_2 concentrations), whereas consequential accounts cannot be used in this way. However, as with the ‘hot spot’ application context discussed above, consequential methods would still be needed to inform decisions on changes in consumption practices, product design etc., to avoid unintended consequences outside of individual attributional budgets.

These ongoing debates illustrate that even after a process of what Rein & Schon (1993, p.159) call ‘frame-reflective discourse’ leading to a degree of shared understanding within a given community, disagreements can remain, either between different sub-groups within the community, or between the community and others, as illustrated by the problem of making consequential results as easily comprehensible as attributional results, to users of this information. Again, this does not imply irreducible conflict, but rather highlights the fact that frame-reflective discourse needs to be iterative and responsive to changing situations. After all, eternal consensus may be just as undesirable as eternal conflict or misunderstanding.

3. Application of the Attributional-Consequential Distinction to Corporate Carbon Accounting

We now turn to the question of whether the attributional-consequential distinction, which has developed within the LCA literature, may also be useful within the field of corporate-level carbon accounting. A first point to make is that the characteristics of attributional accounting identified in Table 3 (i.e. providing an inventory of actual emissions and removals; and the inventory boundary defined in terms of the processes physically or directly connected with the reporting entity) match the characteristics of corporate level carbon accounting, as prescribed by accounting standards such as the GHG Protocol *Corporate Standard* (WBCSD/WRI 2004) or *ISO 14064-1* (ISO 2006c). In other words, these standards provide guidance for the production of accounts which can be described as being attributional in nature (Brander & Wylie 2012; CDP 2013).

To the best of our knowledge, no direct equivalent to consequential (versus attributional) LCA exists as a methodology or standard for corporate-level carbon accounting, in the sense of guiding the production of a consequential version of the typical organisational greenhouse gas inventory. Indeed, it may be impossible to hope to capture all of the possible consequences of a company's actions or to define baselines against which change can be measured in a meaningful way, particularly for companies operating in competitive markets, within cap-and-trade schemes, or dealing in relatively uniform commodities. Decisions to create, re-design or cease manufacturing a single product are routinely made and offer relatively clearly defined alternatives for comparison, whereas change at a corporate level is rarely so simple. Nevertheless, clearly companies do also routinely make decisions or choices between different alternatives – at a range of levels from strategic to tactical and operational – which may have different greenhouse gas implications, even if these alternatives may be more complex and difficult to define than alternatives at a product level. Therefore, in principle, there is no reason why a consequential assessment could not be undertaken to evaluate the systemic consequences of any particular action or choice made by an organisation, rather than relying solely on attributional information to make the same evaluation. Differences in the unit of analysis might help to explain the earlier and wider acceptance of the distinction in LCA, but this does not seem sufficient to explain its near total absence in corporate carbon accounting.

This section identifies a number of areas where greater awareness of the distinction may be beneficial to corporate level accounting: promoting coherence in corporate carbon accounting standards; clarifying the most appropriate choice of accounting method to answer specific types of question; and informing carbon accounting research more generally.

3.1. Issues with coherence in corporate carbon accounting standards

Although the literature on the attributional-consequential distinction focuses almost exclusively on LCA, there are some instances in standards or guidance documents where the distinction is recognised in relation to corporate carbon accounting. One example is in the CDP's guidance note on corporate reporting of emissions from electricity consumption:

> The attributional approach is the approach adopted by the GHG Protocol *Corporate Standard* for corporate inventories. A consequential approach, on the other hand, tries to answer the question 'What are the systemic consequences (changes) in total (system) emissions from given policy decisions at product/entity level?' (CDP 2013, pp.12–13).

Here the distinction is applied to corporate carbon accounting rather than product LCA, and the context of its use is to ensure that consequential methods are not confused with, or introduced into, attributional accounts. A similar provision is made in the GHG Protocol *Corporate Standard* itself, though without explicit reference to the attributional-consequential distinction:

> These reductions [i.e. reductions in emission sources not included in the inventory boundary] may be separately quantified, for example using the GHG Protocol *Project Quantification Standard*, and reported in a company's public GHG report under optional information… (WBCSD/WRI 2004).

Despite these instances where the attributional-consequential distinction has been used to ensure the methodological coherence of greenhouse gas accounting practice, there are also cases where greater awareness of the distinction would have been useful. Although the European Commission's Organisation Environmental Footprint method (European Commission 2013) is a multi-impact method rather than being solely focused on carbon accounting, it nevertheless provides an example of a standard that mixes attributional and consequential elements as it includes credits for avoided emissions (or other environmental burdens) within what would otherwise be an attributional inventory (see Pelletier et al. (2013)). Organisational inventories based on this method will be neither an account of absolute emissions and removals (or other environmental burdens), nor an account of the total consequences from the reporting company's activities. A more thorough understanding of the attributional-consequential distinction could help to avoid such methodological mix-ups. While perhaps the European Commission's method represents an attempt to merge or reconcile the attributional and consequential approaches, we suggest a conceptually more coherent approach would explicitly recognise, rather than try to remove, their differences.

A further example of confusion is provided by the GHG Protocol's recently proposed guidance on reporting emissions associated with electricity generation, known as 'scope 2' emissions (WRI 2014a). As shown in Table 3 above, the processes included in attributional accounts are based on a physical relationship with the reporting entity in question. However, the GHG Protocol guidance allows the use of contractual emission factors that do not reflect any physical relationship between the reporting company and the contracted emissions rate. In addition, the suggested justification for using contractual emission factors for scope 2 reporting is to promote a *change* in the total amount of renewable generation (although the guidance also allows the use of contractual emission factors even if there is no evidence of change in the amount of renewable electricity generated). If *change* in renewable generation is the desired outcome then this could be better supported and accounted for separately using a *change*-oriented method (i.e. a consequential method), such as project level accounting, rather than mixing this into what would otherwise be purely attributional accounts.

3.2. Clarifying the most appropriate choice of accounting method

In addition to promoting conceptually coherent carbon accounting standards, the attributional-consequential distinction could be useful in choosing the appropriate method for a given application. As was shown in the field of life cycle assessment, consequential methods appear to be the most appropriate for decision-making contexts (such as comparing two alternatives with respect to a desired outcome) as they explicitly aim to quantify the total consequences of decisions. In the Searchinger et al. (2008) example, decisions based on attributional methods can result in system-level outcomes that are the exact opposite to those intended. Given that corporate level carbon accounting is attributional in nature (CDP 2013; Brander & Wylie 2012), it is probable that such accounts will be similarly unreliable for good decision making. This fundamental shortcoming, which is due to the fact that attributional accounts do not capture the full system-level impacts of a given alternative, should be understood as distinct from other limitations on comparability which have to do with a lack of consistency in accounting and reporting, as observed by many authors (Kolk et al. `2008; Solomon et al. 2011; Andrew & Cortese 2011; Dragomir 2012; Sullivan & Gouldson 2012). A greater appreciation of the attributional-consequential distinction could encourage the use of consequential methods, such as project level accounting or consequential LCA, to inform or appraise corporate decision making.

One possible application context for attributional corporate level accounting is to provide information on exposure to regulatory risk. Given that many regulatory measures such as carbon taxes and emissions trading schemes impose responsibilities on emitters based on attributional accounting methods, one reason for companies reporting such information might be to indicate the risk of such liabilities in future. Attributional accounts will then be decision-useful for investors interested in the financial impacts of such impositions on the valuation of corporate assets (Hassel et al. 2005; Kolk et al. 2008). However, questions would remain about the efficacy of regulation based on attributional methods, precisely for the reason that they do not show the total consequences of a corporation's activities, nor the impact of regulating those activities. Out-sourcing of emissions-intensive activity to a country not covered by such regulation might be an example of a perverse outcome that would only be recognised with a consequential assessment. Companies and investors relying on attributional accounting should therefore also consider the risk that policy-makers could change the emphasis of regulation to capture more non-attributional impacts in future, if these consequences are material at a systemic level.

A common explanation for the different application contexts for attributional and consequential methods is the scope of the decision under analysis. It may be assumed that consequential methods are only necessary if whole markets or industries are affected by the decision in question, and where this is not the case then attributional methods are sufficient. However, it is possible to conceive of micro-level decision scenarios that do not affect whole markets, but nevertheless require a consequential approach to capture systemic impacts. For example, if a farmer purchases straw from a neighbour to use as a fuel, the neighbour may have to use more fertiliser as they are no longer ploughing the straw back into the soil. A conventional corporate level inventory for the farmer would not capture the indirect effect of the neighbour's increased fertiliser use, and, moreover, it would only be through undertaking a consequential assessment and comparing it with an attributional assessment that the systemic adequacy of using an attributional approach could be known.

It could also be suggested that expanding the scope of attributional corporate inventories will help to capture more of the consequences of corporate activities, thereby mitigating the problem of missing system-level impacts. Indeed, this appears to be part of the rationale for the provision of guidance on including all scope 3 sources (i.e. sources of emissions not controlled by the reporting company, but occurring either upstream or downstream in their value chain) in corporate accounts (WBCSD/WRI 2011b). However, emission consequences, especially where they are mediated by markets, can occur well beyond the value-chain of the reporting entity in question, and so will not necessarily be captured even by whole value-chain inventories. For example, consuming an additional unit of a product in one country may affect production of the marginal unit in another country; value-chain analysis will only capture the upstream and downstream impacts from the former unit and not the latter. In contrast, a consequential approach will attempt to provide 'complete' information, as it specifically aims to identify all the emission sources that are affected by a decision or action.

In some sectors, there is already some awareness of this limitation to attributional corporate accounting. For instance, the telecommunications industry makes the case that although its own value chain emissions may be increasing, the industry's services reduce emissions in other sectors, such as transportation (e.g. by video conferencing replacing business travel). A recent report commissioned by the telecommunications industry calculates the abatement potential from telecommunications to be approximately seven times larger than emissions from the sector (Global e-Sustainability Initiative 2012). Similar consequential impacts, which would not be captured in a conventional attributional inventory, are reported by BASF who claim a 246 million tonne reduction in CO_2e emissions due to their sustainable building products (BASF 2014). Awareness of the limitations with value-chain attributional accounting is also evidenced by the GHG Protocol's proposal to develop a standard specifically focused on product-enabled reductions (WBCSD/WRI 2014b). It is interesting (but not surprising, given the self-regulatory nature of most such initiatives in corporate carbon accounting) to note that the current focus of these initiatives is on the beneficial *reductions* caused by company activities, with little interest yet shown in understanding the possible *increases* in emissions that may occur outside conventional attributional inventory boundaries.

It is possible that companies may be using consequential methods to support *internal* decision-making, but not public reporting, and therefore their use is not evident. However, a more likely possibility is that companies are using attributional methods to inform their internal decision-making, as it is on the basis of these accounts that companies will be judged by their external stakeholders, because the attributional approach currently dominates public reporting. In addition, attributional corporate accounting standards, such as the widely used GHG Protocol *Corporate Standard*, state that the information provided by such inventories should be relevant to *decision making* (WBCSD/WRI 2004, pp.7 – 8), which clearly suggests that attributional accounts will be used in this way. The question of whether companies are using attributional or consequential methods to support their internal decision-making is a subject for further empirical research, but it is worth noting that such research is only likely to take place if the attributional-consequential distinction becomes more widely recognised within the carbon accounting research community, which in turn requires the sort of frame-reflective dialogue and consensus-building that previously occurred within the LCA community.

Despite the limitations with attributional corporate accounting, one feature of attributional accounting which appears to be lacking with consequential methods is the sense of 'ownership' conferred on emissions within a company's attributional inventory. Conventional attributional accounts provide a starting point for companies to recognise a set of emissions as 'theirs', which they can then seek to manage over time. In contrast, with consequential accounting, it is more difficult to identify which emission sources the company 'owns'. What may be needed is a combination of both approaches, with attributional accounts used to establish a set of emission sources to be managed, and consequential assessments used to inform decisions on how to reduce those emissions without causing unintended consequences elsewhere in the system.

A next step for companies interested in understanding and managing the total system-level greenhouse gas impacts of their decisions would be to utilise or develop consequential methods that capture both reductions and increases in emissions resulting from specific actions or choices. A number of methods already exist that can be used for this purpose. Consequential LCA can be used where the decision concerns the production or design of a specific product. Although there are no published standards dedicated to consequential LCA, helpful guidance is available in Ekvall & Weidema (2004) and in Weidema (2003). Project level accounting can be used where the decision concerns a discrete activity, e.g. the development of on-site renewables (see (WBCSD/WRI 2005) or ISO 14064-2 (ISO 2006d) for guidance). Finally, policy-level accounting can be used for assessing company policies, e.g. economy-class only business travel, or making payments for employee-owned car mileage (see the GHG Protocol's *Policy and Action Standard* (WRI 2014c) for guidance).

3.3. Utility of the distinction to academic understanding of carbon accounting

A final area where a greater awareness of the attributional-consequential distinction may be fruitful is to academic research on corporate carbon accounting. As mentioned earlier, despite its widespread use within the field of life cycle assessment, there appears to be very limited use or awareness of the attributional-consequential distinction in the academic literature for other areas of carbon accounting. The only journal article identified (based on the literature review illustrated in Figure 5) that uses the attributional-consequential distinction to categorise different fields of greenhouse gas accounting practice is Brander & Wylie (2011), which suggests that national inventories and corporate greenhouse gas accounting are attributional in nature, and that project and policy-level accounting are consequential. The lack of other literature suggests that utilising the distinction to understand the different forms of greenhouse gas accounting is relatively underdeveloped at present. Likewise, the attributional-consequential distinction only appears in one of the eight journals covered by Ascui's (2014) review of the 'social and environmental accounting' (SEA) literature on carbon accounting (the exception being the *Journal of Cleaner Production*, which has a strong focus on LCA). Ascui (2014) distinguishes between critical/normative discussions *about* carbon accounting, and empirical studies *of* carbon accounting. It appears likely that the attributional-consequential distinction could be pertinent to both these areas of research.

It is worth noting that the attributional-consequential distinction is not equivalent to the distinction between financial accounting (external reporting) and management accounting (internal decision-making) (Ratnatunga 2008; Burritt et al. 2011; Stechemesser & Guenther 2012). Internal decision-making may or may not be based on attributional accounts, and external reporting may provide either attributional or consequential information. As previously observed, an important area for further research is the extent to which current corporate level decision-making is based on attributional information, and whether this leads to sub-optimal outcomes at the system level.

4. Conclusions

There appears to be considerable potential for wider learning from the conceptual and methodological development of the attributional-consequential distinction in the LCA literature. With respect to corporate carbon accounting, the potential benefits include the development of more coherent carbon accounting standards, and a better understanding of the appropriateness of relying on attributional versus consequential accounts to answer different kinds of query. In short, attributional accounts provide a snapshot of a particular scope of assumed responsibility, which may be relevant to corporations concerned only with regulatory liabilities based on attributional accounting. However, given the global, systemic nature of climate change as a problem, consequential accounts are appropriate for informing decision-making where the objective is genuine mitigation of the problem (i.e. based on a much wider sense of responsibility). Attributional accounts may also be useful for managing absolute carbon budgets, and for creating a sense of ‘ownership’ for a specific set of emissions. However, if actions aimed at reducing emissions within an attributional budget or inventory are not informed by a consequential assessment, it will be impossible to know whether the actions also cause unintended consequences elsewhere in the system. Further research is required to develop heuristics or simplified methods to understand when such consequences may be material or not. The use of consequential assessment for corporate level decision-making could also be greatly facilitated by a standardised methodology, potentially bringing together aspects of consequential LCA, project-level accounting, and policy-level accounting. The challenges of developing such a methodology, however, should not be underestimated.

Despite superficial similarities (e.g. presenting results in the same metric, such as carbon dioxide equivalents), attributional and consequential accounts are not alternative methods for answering the same question, but rather, methods suitable for answering fundamentally different questions, informed by different disciplinary perspectives and conceptual frames. ‘[H]ow you account for CO_2 emissions and the answer you get depend on the questions you ask, the framework of the query.’ (Marland et al. 2013). Problems arise, however, when this distinction is not appreciated. It is hoped that this chapter provides an initial contribution to further frame-reflective debate on the nature and utility of the attributional-consequential distinction for corporate carbon accounting, which may facilitate more rapid adoption of common terminology, standards and associated conceptual understanding than was the case with its earlier emergence in the field of LCA.

Finally, this chapter has focussed on the potential application of the distinction to corporate carbon accounting, but there is considerable scope for further research to explore the application of the concept and methods to other forms of carbon accounting. It may be helpful, for example, to understand that national inventories are also generally attributional in nature, while project and policy-level carbon accounting are consequential (Brander & Wylie 2012). Policies aimed at managing national emissions may well create effects that are not captured in attributional national inventories, and alternatives such as consumption-based accounting (Barrett et al. 2013), while including more consequences, will not necessarily be sufficient to fully capture system-wide marginal impacts. Beyond this, the distinction may prove fruitful to other forms of social and environmental accounting: we suspect that similar issues would be raised in accounting for water, biodiversity, health or employment impacts, for example. In each case, appreciating the distinction may support a better understanding of possible alternatives and the appropriateness of using different alternatives to answer different questions.

Paper 2 - Transposing lessons between different forms of consequential greenhouse gas accounting: lessons for consequential life cycle assessment, project-level accounting, and policy-level accounting

Abstract

Greenhouse gas accounting has developed in a number of semi-isolated fields of practice and there appears to be considerable opportunity for transposing methodological innovations and lessons between these different fields. This research paper identifies three consequential forms of greenhouse gas accounting: consequential life cycle assessment; project-level accounting; and policy-level accounting. These methods are described in detail and then compared in order to identify the key methodological differences and the potential lessons that can be transposed between them. Analysis of the substantive methodological differences suggests that consequential life cycle assessment could be enhanced by adopting the same structure used in project and policy-level accounting, which provides a time-series of impacts, aggregate level analysis, and a transparent specification of the baseline and decision scenarios. There is a case for conceptualising a unified form of consequential time-series assessment, of which project, policy and product assessments would be sub-types.

1. Introduction

Greenhouse gas accounting has developed in a number of distinct fields of practice (Ascui & Lovell 2011; Marland et al. 2013), and as a result there appears to be considerable potential for transposing conceptual or methodological innovations from one field of practice to others. Greenhouse gas accounting methods have developed at the national level (IPCC 2006), the organisational level (WBCSD/WRI 2004), the product level (British Standards Institute 2011; WBCSD/WRI 2011c), the project level (ISO 2006d; WBCSD/WRI 2005), in addition to others. It may be assumed that when such methods have similar purposes but employ different methodological approaches, there is an opportunity for comparing those approaches and generating lessons for potential methodological development.

One grouping of methods, which forms the focus of this paper, is the set of greenhouse gas accounting methods that can be described as 'consequential' in nature. The term 'consequential' originates within the field of life cycle assessment (LCA) (Curran et al. 2005; Russell et al. 2005), but the concept can be used more broadly to denote any form of assessment which aims to quantify the total *change* in impacts that results from a given decision or intervention (Brander & Wylie 2012). Consequential methods are often contrasted with 'attributional' methods (Reinhard & Zah 2009; Tufvesson et al. 2013; Finnveden et al. 2009), which can be defined in a broad sense to denote any inventory of *absolute impacts* attributed to a given entity, such as a country, organisation, or product (Brander & Wylie 2012; CDP 2013), with attribution normally based on some form of physical connectedness. The focus of this paper is on the lessons that can be shared between different *consequential* methods, though some discussion of attributional methods will also be provided where this helps to explain certain features of the consequential approaches in question.

The novel contribution of this paper is the identification of *methodological* lessons that can be shared across different fields of greenhouse gas accounting practice. The academic literature on greenhouse gas accounting *methods* tends to exist within narrow communities of practice, such as the life cycle assessment community or the project accounting community, and there appears to be a significant lack of *methodological* dialogue between such fields. For example, the recent development of dynamic life cycle assessment (Beloin-Saint-Pierre et al. 2014; Collet et al. 2013) can be viewed as a reinvention of time-series assessment but without reference to, and some years after, project-level accounting. Greater awareness of the methodological innovations within other areas of practice may be fruitful in guiding and facilitating similar methodological developments. The existing literature that does take a more holistic view across different fields of greenhouse gas accounting practice has tended to take a social theory perspective, and considers issues such as the distinct social purposes of greenhouse gas accounting (Ascui & Lovell 2011; Schaltegger & Csutora 2012), or how accounting practices and competence are socially constructed (MacKenzie 2009; Ascui & Lovell 2012; Burritt & Tingey-Holyoak 2012). However, as yet there is very little research on transposing methodological lessons, notwithstanding the *prima facie* likelihood that there is much to be learned.

The primary contribution of this paper is the identification of methodological lessons that can be transposed between different forms of consequential greenhouse gas accounting, however, in pursuing this end the paper also provides some supplementary outputs: a classification of current greenhouse gas accounting methods according to whether they are consequential or attributional in nature; and a detailed discussion on the core and superficial methodological characteristics of the identified consequential methods. Although this paper is primarily focused on greenhouse gas accounting, the findings are relevant to consequential methods that consider other impact categories as well.

2. Methodology

This paper proceeds by identifying the existing forms of greenhouse gas accounting through a review of the current accounting standards and guidance, and classifies these methods as being either consequential or attributional in nature.

A list of published standards and guidance for physical greenhouse gas accounting was compiled based on existing knowledge of the main organisations publishing such guidance, such as the International Organization for Standardization, the Greenhouse Gas Protocol, and the Intergovernmental Panel on Climate Change, and also an internet search for ‘greenhouse gas guidance’, ‘carbon guidance’, ‘GHG guidance’ and ‘LCA guidance’. An initial list of standards was compiled in early 2014, and was updated in early 2015 to achieve a more complete list at the time of publication. The list of standards collected is not intended to be exhaustive, and given the proliferation of standards and sector-specific guidance any list would become incomplete rapidly. However, the list of collected documents is sufficient for the present purpose of identifying the main consequential forms of greenhouse gas accounting and their methodological features.

Only standards and guidance for physical greenhouse gas accounting, as distinct from financial greenhouse gas accounting, were included as the purpose of financial accounting was considered sufficiently different that the transposition of methodological lessons would be unlikely. Physical greenhouse gas accounting is concerned with flows or changes in greenhouse gases in mass units, such as tonnes of CO_2e, while in contrast financial greenhouse gas accounting is concerned with the financial value of carbon-based assets and liabilities, such as tradable emission permits or reduction credits, measured in monetary units.

The collected standards were then classified as being either consequential or attributional in nature. The defining characteristics of consequential greenhouse gas accounting methods are taken to be: 1. the method aims to quantify *change* in emissions/removals, resulting from a decision or action; 2. the method aims to quantify *system-wide* change (i.e. not only change within a limited boundary). The criterion used to identify attributional methods is: the method aims to quantify and allocate absolute emissions/removals to a given entity or item. These defining characteristics are those identified in Brander and Ascui (2015), which collates a number of definitions for the 'consequential' and 'attributional' approaches in the LCA literature, and provides an analysis of the essential and supplementary features of the two types of approach.

As with many conceptual distinctions, there is ongoing debate as to its precise nature and implications (Suh & Yang 2014; R. J. Plevin et al. 2014a; Brander & Ascui 2015). Nevertheless, the nuances of that debate are sufficiently fine-grained that any alternative interpretations are highly unlikely to yield alternative classifications of the published greenhouse gas accounting standards. In the instances where classification did prove difficult, this tended to arise because the standard in question mixes both consequential and attributional elements, rather than because the classification criteria are unclear. It is worth noting that this situation can be distinguished from cases where the standard in question clearly intends to address both methods separately, within a single document (e.g. the ILCD handbook (European Commission et al. 2010)). The instances where classification was uncertain are discussed further in Section 3.1.

Some of the standards and guidance documents identified cover a wider range of impact categories than just greenhouse gas emissions, but were nevertheless included in the analysis if they covered greenhouse gas emissions as an impact category. The standards and guidance documents were then grouped by the type of entity or action they primarily relate to, e.g. national level, community level, product level etc. Table 4 in Section 3.1 presents the guidance and standards reviewed, and their categorisation by type.

The identified consequential methods are then described in detail, setting out the key steps and structure of each method. This information is then used to analyse any substantive differences between the methods and to identify the potential lessons for methodological development.

3. Results and Discussion

This section presents the findings from the review and classification of existing greenhouse gas accounting methods, a detailed description of each of the consequential methods identified, and an analysis of the main methodological differences and potential lessons for the development of the methods.

3.1. Review and classification of existing greenhouse gas accounting methods

Table 4. Categorisation of greenhouse gas accounting methods

Consequential Methods			Attributional Methods	
Entity/Action	Guidance/Standard		Entity/Action	Guidance/Standard
Product (consequential LCA)	1. International Reference Life Cycle Data System Handbook (European Commission et al. 2010) 2. Market information in life cycle assessment (Weidema 2003) 3. Guidelines for application of deepened and broadened LCA (Weidema et al. 2009)		Product (attributional LCA)	1. International Reference Life Cycle Data System Handbook (European Commission et al. 2010) 2. PAS 2050:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services (British Standards Institute 2011) 3. Greenhouse Gas Protocol: Product Life Cycle Accounting and Reporting Standard (WBCSD/WRI 2011c) 4. ISO 14040:2006 (ISO 2006a) 5. ISO14044:2006 (ISO 2006b) 6. ISO/TS 14067:2013 (ISO 2013b) 7. Product Environmental Footprint (European Commission 2013)

Consequential Methods			Attributional Methods	
Entity/Action	Guidance/Standard		Entity/Action	Guidance/Standard
Project	1. GHG Protocol for Project Accounting (WBCSD/WRI 2005) 2. ISO14064-2:2006 (ISO 2006d) 3. Clean Development Mechanism methodologies (UNFCCC 2014) 4. Verified Carbon Standard methodologies (Verified Carbon Standard 2014)		Organisational	1. Greenhouse Gas Protocol: A Corporate Accounting and Reporting Standard (WBCSD/WRI 2004) 2. Greenhouse Gas Protocol: Corporate Value Chain (Scope 3) Accounting and Reporting Standard (WBCSD/WRI 2011b) 3. ISO14064-1:2006 (ISO 2006c) 4. ISO 14069:2013 (ISO 2013a) 5. Organisation Environmental Footprint (European Commission 2013)
Policy	1. Greenhouse Gas Protocol: Policy and Action Standard – Final Draft (WBCSD/WRI 2014a)		Community	1. Global Protocol for Community-Scale Greenhouse Gas Emissions (GPC) (Schultz et al. 2014) 2. PAS 2070: 2013 Specification for the assessment of greenhouse gas emissions of a city (British Standards Institute 2013) 3. U.S. Community Protocol for Accounting and Reporting of Greenhouse Gas Emissions - version 1.1 (ICLEI 2013)
			National	1. IPCC Guidelines for National Greenhouse Gas Inventories 2006 (IPCC 2006)

As noted above, there were a number of instances where it was more difficult to categorise a standard/guidance document as being either consequential or attributional, largely because the standard/guidance in question is ambiguous or mixes elements of both approaches in a single methodology. This is the case with the Greenhouse Gas Protocol's Product Life Cycle Accounting and Reporting Standard (WBCSD/WRI 2011c), which explicitly states that it is intended as an attributional method but allows the use of substitution when dealing with multi-functionality, though substitution is generally regarded as a consequential modelling technique (Brander & Wylie 2012). A similar issue arises with ISO 14040:2006 (ISO 2006a) and ISO 14044:2006 (ISO 2006b), though in these cases neither standard states whether it is intended to represent a consequential or attributional method, or both simultaneously. ISO 14040 uses the term 'allocation procedures' which suggests an attributional method, though ISO 14044 allows both substitution and allocation. The failure of these standards to actually standardise practice is well noted by Weidema (Weidema 2014), however, for the purposes of the current analysis these ISO standards have been classified as attributional as they contain no specifically consequential modelling requirements, other than the inclusion of substitution in ISO 14044, which appears to be an aberration similar to that in the GHG Protocol standard.

The same aberration is present, but seemingly deliberately so, in both the Organisation Environmental Footprint and the Product Environmental Footprint methods (European Commission 2013). As a general principle, attributional methods should only include values for absolute emissions and absolute removals, and should not include values for avoided emissions, which is the implication of using substitution (Brander & Wylie 2012). Combining both attributional and consequential elements in a single analysis means that the results are neither an inventory of absolute emissions/removals, nor a complete assessment of change, and are effectively incoherent (R. Plevin et al. 2014; Brander & Wylie 2012).

The review and classification exercise identified three main forms of consequential assessment: consequential life cycle assessment; project greenhouse gas accounting; and policy greenhouse gas accounting. These methods share the general 'consequential' characteristics of aiming to quantify change in emissions/removals resulting from a decision or intervention, and quantifying that change wherever it occurs (i.e. not only within a limited boundary). In the case of consequential life cycle assessment (consequential LCA) the intervention in question relates to the production or consumption of a product, or changes in the configuration of the life cycle of a product (Weidema 2003). For project accounting the intervention is the implementation of a project, which can be defined as a set of activities intended to cause a change in greenhouse gas emissions (ISO 2006d; WBCSD/WRI 2005). Lastly, in the case of policy accounting the intervention is any policy, such as a tax, payment incentive, market mechanism etc. (WBCSD/WRI 2013). Although policies are normally implemented by governments or public agencies, the method can equally be applied to policies implemented by corporations.

The grouping of the identified consequential standards as being 'product', 'project' or 'policy' methods is based on the stated level of action each standard aims to address, e.g. ISO 14064-2 (ISO 2006d) states that it is for the quantification of emissions and removals at the *project* level, and has therefore been grouped with other project level methodologies. However, it is important to note that the groupings chosen do not entail mutual exclusivity, e.g. consequential LCA is described as a *product* level method although it can be, and often is, used for policy analysis (R. J. Plevin et al. 2014b; Searchinger et al. 2008), albeit only where the policy relates to changes in the supply or configuration of products.

The following three sections provide detailed outlines of each of these methods in order to facilitate the subsequent analysis of their key differences and similarities, and the opportunities for sharing lessons between them.

3.2. Consequential life cycle assessment

Consequential LCA aims to quantify the changes in impacts that result from a change in the level of production of a product, or changes in the configuration of the life cycle of a product (Weidema 2003). It is worth noting that this method developed out of conventional attributional LCA in the 1990s (Weidema 1993; Zamagni et al. 2012), and as a result consequential LCA still contains much of the methodological structure and conceptual apparatus of its attributional forebear, which is of relevance to the later discussion in Section 4. The 'life cycle' of a product can be defined as 'the consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal' (ISO 2006b, p.2). Table 5 provides a summary of the generic steps used in implementing an LCA (largely adapted from the *International Reference Life Cycle Data System Handbook* (European Commission et al. 2010)).

Table 5. Key steps in product LCA

Step 1 – goal definition	Define the intended application of the study (e.g. to inform decision-making, make marketing claims etc.), and the intended audience for the results.
Step 2 - scope definition	Define a number of features of the study, including the product that is studied, and the 'functional unit'. The 'functional unit' is the 'quantified performance of a product system for use as a reference unit' (ISO 2006b, p.4), and is used to ensure that products are compared on a like-for-like basis (Weidema 2003).
Step 3 - inventory analysis	Identify the processes to include within the assessment, and collect data on the material and energy flows associated with those processes. For a consequential LCA the processes that are inventoried are the 'marginal' processes, i.e. the processes that change as a result of the decision in question (Schmidt & Weidema 2008). These processes are often, but not always, different to those used directly in the life cycle of the product physically produced/consumed (i.e. those that would be included in the inventory for an attributional assessment).
Step 4 - impact assessment	Convert the information on material and energy flows into impacts. In the case of greenhouse gas emissions this generally involves the use of emission factors and the conversion of non-CO_2 greenhouse gases into CO_2e by the use of global warming potentials.
Step 5 – interpretation	Identify the significant findings of the assessment and relate these to the goal of the study. This is often considered an iterative or concurrent process that feeds back into the other stages of the method as the assessment progresses (European Commission et al.

	2010).

The main difference between attributional and consequential LCA relates to the processes that are included in the inventory stage (step 3) of the assessment (Zamagni et al. 2012). Consequential life cycle inventory includes all and only the processes that *change*, wherever they occur in the system, while in contrast an attributional life cycle inventory includes the processes used directly in the life cycle stages of the product physically produced/consumed. This difference is illustrated by the use of the technique of substitution in consequential LCA, which is used to deal with co-products, or other forms of multi-functionality. Substitution involves identifying the product systems that are displaced (i.e. *changed*) by the production of co-products, and crediting the displacement of those product systems to the decision studied, as the avoidance of those systems and their associated impacts are a consequence of the decision (Ekvall & Weidema 2004; Brander & Wylie 2012).

An early formulation of a general procedure for identifying the processes that change, i.e. the 'marginal' processes, is provided by Weidema et al. (1999), in which considerations such as the time horizon of the study are taken into account:

> One should distinguish between short-term, when studying changes which take place with the existing production capacity and which are not expected to affect capital investment (installation of new machinery or phasing out of old machinery), and long-term, when studying changes that are expected to affect capital equipment. (Weidema et al. 1999, p.49)

From such initially straightforward procedures there has been a continual development of methods for identifying the processes that change, such as the inclusion of positive feed-back loops through economies of scale and learning (Sandén & Karlström 2007); procedures for determining the proportion of increased agricultural output from yield increases, land use change, or reduced consumption elsewhere (Schmidt 2008); and the use of general equilibrium modelling to predict the world regions that will respond to changes in commodity prices (Searchinger et al. 2008; Hertel et al. 2010).

A further important feature of both consequential and attributional LCA is the use of *amortisation*, which is required when there are large non-linear emission events over time, such as land-use change or the production of capital equipment (Sjödin & Grönkvist 2004; Hsu et al. 2010; Searchinger et al. 2008; Hertel et al. 2010). The need for amortisation is ultimately driven by the fact that LCA calculates impacts on a normalised per product (or per functional unit) basis, for example, Searchinger et al. (2008) present their findings in gCO_2e/MJ of fuel. Amortisation allows temporally distributed non-linear impacts, such as land use change, to be averaged by the amount of production during a specified period of time. This feature of consequential LCA will be contrasted with the approach taken by the other consequential methods in Section 4.

3.3. Project accounting

Project accounting aims to quantify the changes in emissions or removals that occur as a result of a project (WBCSD/WRI 2005), with 'project' broadly understood as a 'planned set of activities within a specific geographical location' (Watson et al. 2000, sec.5.1.2)[2]. In general, the background context for the development of project accounting is markedly different from consequential LCA, as its focus has been primarily on crediting greenhouse gas emission offsets (Gustavsson et al. 2000), rather than informing decision-making with respect to a range of possible options. One exception to this is ISO 14064-2, which was intended to also accommodate the quantification of internal abatement actions and technology choices.

[2] A more precise name for the practices commonly denoted by the label 'project accounting' would be 'consequential project accounting' as in theory attributional accounting could also be applied at a project level. However, 'project accounting' is used in this paper as it is the label commonly used to denote consequential project accounting, and to-date the need to distinguish between consequential and attributional methods at the project level has not arisen.

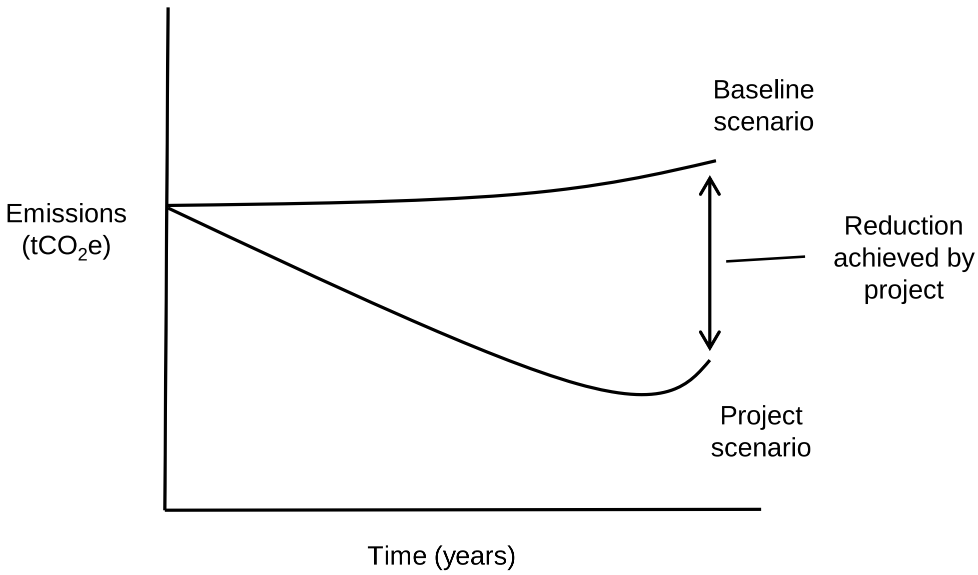
As shown in Table 4, project accounting has been formalised in a number of standards and guidance documents, including the *GHG Protocol for Project Accounting* (WBCSD/WRI 2005), and *ISO 14064-2 Specification with guidance at the project level for quantification, monitoring and reporting of greenhouse gas emission reductions or removal enhancements* (ISO 2006d). There are also numerous methodologies for specific project-types, such as those under the Clean Development Mechanism (UNFCCC 2014), or the Verified Carbon Standard (Verified Carbon Standard 2014). Table 6 presents the key steps in project accounting, based on *ISO 14064-2*, with some details also taken from the *GHG Protocol for Project Accounting*.

Table 6. Key steps in project accounting

Step 1 – describe the project	Describe the physical location of the project and the activities that will be undertaken to reduce greenhouse gas emissions (ISO 2006d, sec.5.2).
Step 2 - identify the greenhouse gas sources and sinks relevant to the project	All sources and sinks that are controlled by, related to, or affected by the project should be included in the assessment (ISO 2006d, sec.5.3). This step appears to parallel the life cycle inventory stage for consequential LCA (step 3), i.e. identifying all emission sources that change as a result of the intervention in question.
Step 3 - determine the baseline scenario	The baseline scenario can be defined as a 'hypothetical reference case that best represents the conditions most likely to occur in the absence of a proposed greenhouse gas project' (ISO 2006d, p.3). ISO 14064-2 requires that the baseline scenario be equivalent to the 'with project' scenario in terms of the supply of products and services (ISO 2006d, sec.5.4), which parallels the requirement for equivalent 'functional units' when comparing product systems in consequential LCA.
Step 4 - identify greenhouse gas sources and sinks for the baseline scenario	This is a parallel process to Step 2, but for the baseline rather than project scenario (ISO 2006d, sec.5.5)
Step 5 - quantify greenhouse gas emissions or removals	For each source or sink identified in the 'with project' and 'baseline' scenario, the level of emissions/removals should be calculated, e.g. by applying emission factors to activity data for each source and sink (ISO 2006d, sec.5.7). This is similar to the 'life cycle impact assessment' stage in consequential LCA (step 4).
Step 6 - quantify emission reductions and/or removal enhancements	This is done by subtracting the 'with project' emissions/removals from the 'baseline' emissions/removals, for each year that the project is in operation (ISO 2006d, sec.5.8; WBCSD/WRI 2005, p.77)

The schematic diagram in Figure 6 below provides a graphical illustration of the main components and overall structure of the project accounting method.

Figure 6. Illustration of the key components of the project accounting method



There are a number of key features illustrated by Figure 6 that are worth highlighting as they are particularly relevant to categorising the main differences between the various consequential methods in Section 4. Firstly, Figure 6 illustrates the way the change caused by a project is calculated as the difference between the baseline and project scenario, with the scenario in which the project is absent (the baseline) being transparently and explicitly modelled. This basic structure to the method can expressed as follows in Equation 1.

Equation 1.

$$Change\ in\ emissions = Net\ baseline\ emissions - Net\ project\ scenario\ emissions$$

A second key feature of the method is that emissions are presented as a time series. This means that the method can capture the 'shape' of the change in emissions as it occurs over time (i.e. the variation in difference between the two scenarios over time), including possible non-linear trends in baseline or project scenario emissions, which are illustrated by the curvature of the emission trajectories.

A third feature illustrated in Figure 6 is that the calculation of change is done at an aggregate level, i.e. for the project as a whole, rather than at a normalised level per unit of activity or functional unit.

There are a number of other distinctive methodological features/concepts commonly associated with project accounting, which tend not to arise with the other consequential methods. These include *crediting baselines*, *conservativeness*, *emphasis on reductions*, *leakage*, and *additionality*. These are important to discuss in order to inform the analysis in Section 4 on whether these features constitute significant methodological differences or not.

Crediting baselines and conservativeness

Crediting baselines are intended to lie between the main estimated baseline and the project scenario, and are used to ensure that the number of credits issued by a project are not over-estimated, and hence are *conservative* (Trexler et al. 2006; Gustavsson et al. 2000). The prominence of these features in project accounting, but not in other consequential methods, can be viewed as a legacy of project accounting's development for crediting carbon offset projects. The credibility of carbon offsets may be undermined by over-crediting, and therefore many programmes build-in conservativeness to avoid this possibility.

Focus on reductions

Another legacy of carbon offsetting is the emphasis in project accounting standards on quantifying emission *reductions*, rather than *changes* in emissions more generally, i.e. increases *or* reductions. For instance, the *GHG Protocol for Project Accounting* states that the standard is intended for 'quantifying and reporting GHG reductions' (WBCSD/WRI 2005, p.5), and does not acknowledge the possibility of quantifying *increases* in emissions. However, the structure of the method is such that either reductions or increases in emissions could be measured; in the case of increases in emissions, the project scenario emissions line shown in Figure 6 would be above, rather than below, the baseline.

Leakage

Leakage refers to emissions or removals caused by the project that occur outside the 'project boundary' (Vöhringer et al. 2006). However, the concept appears to be an artifice of identifying a 'project boundary'. If the aim of an assessment is to quantify the *total* change in emissions, i.e. both inside and outside the project boundary, then this is identical to simply quantifying *all* changes in emissions and dispensing with the idea of a 'project boundary' and 'leakage' altogether. Dispensing with the concepts of 'project boundaries' and 'leakage' is the approach taken by ISO 14064-2:

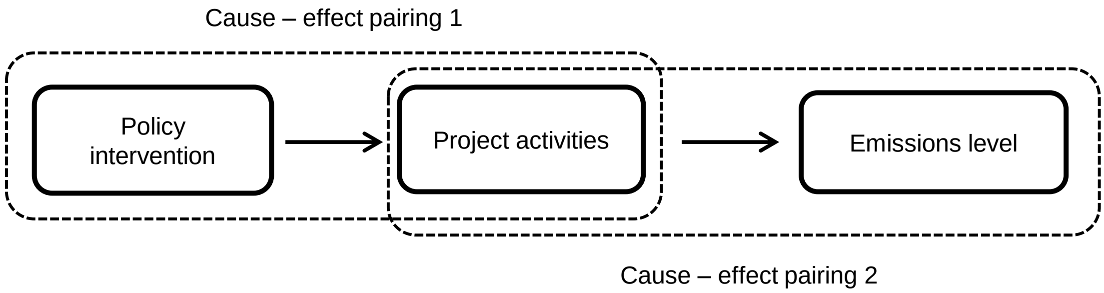
> Unlike the Kyoto mechanisms and other programmes, this part of ISO 14064 does not use the terms 'project boundary' or 'leakage'. Instead, it refers to sources, sinks and reservoirs that are 'relevant' to the project. (ISO 2006d, p.23)

Interestingly, there are some parallels between the concepts of 'project boundary' and 'leakage' on the one hand and the distinctions between 'foreground' and 'background' processes, and 'direct' and 'indirect' effects found in consequential LCA (Raadal et al. 2012; Searchinger et al. 2008). 'Foreground' processes are those that can be directly controlled by the entity undertaking the LCA, while 'background' processes are the other processes in the product's life cycle (Gaudreault et al. 2010). 'Direct' effects are those that occur in 'foreground' processes, and 'indirect' effects are those in 'background' processes (Raadal et al. 2012). However, as with project accounting, these distinctions in consequential LCA appear to be largely arbitrary, as all processes that change as a result of an entity's decision-making are subject to some degree of control (i.e. by definition they *change* as a result of the entity's decision-making). The distinctions also appear to be non-essential to the task of quantifying total change in emissions, as the same results would be achieved without using these distinctions. This is evidenced by the absence of these concepts in a number of consequential LCA studies, for example, Dalgaard et al. (2008); Schmidt (2008); and Schmidt and Weidema (2008). At most, the concepts of 'project boundary', 'leakage', 'foreground', 'background', 'direct effects', and 'indirect effects' can be viewed as presentational categories for grouping results into manageable components, but they do not appear to play a meaningful role in the actual quantification of change.

Additionality

A final concept that has been highly prominent in project accounting is that of *additionality* (Ascui 2014; Stechemesser & Guenther 2012), and it therefore also deserves some discussion. Despite the concept's prominence, it also remains an often poorly defined and misunderstood term. Gillenwater (2012) suggests that most of the definitions provided in the main project accounting standards, such the *GHG Protocol for Project Accounting*, *ISO 14062-2*, and *CDM*, are circular, i.e. projects are said to be 'additional' if they would not have occurred in a baseline scenario, and the 'baseline scenario' is in turn defined as being the absence of the project (Gillenwater 2012). This is equivalent to saying the project would not have happened (i.e. would be additional) in the absence of the project. The root of the confusion is the conflation of two distinct pairings of cause-effect relationship: firstly, that between the policy intervention, e.g. the creation of a market for offset credits, and offset projects; and secondly, that between offset projects and the level of emissions they achieve. Figure 7 is adapted from Gillenwater (2012) and illustrates this double-pairing of cause-effect relationships.

Figure 7. Double-pairing of cause-effect relationships



Gillenwater (2012) suggests that the question of additionality only arises for the first cause-effect pairing, i.e. projects are additional if they would not have occurred in the baseline scenario (with the baseline scenario defined as the absence of the policy intervention, and *not* in terms of the absence of the project). The circularity that Gillenwater identifies in many existing accounts of additionality arises because the project is treated as both the cause of the effect, and the effect that is being assessed for additionality.

Although Gillenwater's analysis does effectively diagnose and resolve the circularity evident in many proposed definitions of additionality it appears to be overly restrictive to proscribe that the concept can only be applied to the first cause-effect pairing (between a policy intervention and projects). It seems eminently possible to apply the concept of additionality to the second cause-effect pairing, and to ask whether the *level of emissions* achieved by a project is 'additional', i.e. would have been the same in the absence of the project. Furthermore, the concept of additionality appears to be a general one that can be applied to *any* cause-effect pairing, and is not restricted to the field of greenhouse gas accounting (e.g. 'Would my children have tidied their room (*the effect*) in the absence of me shouting at them (*the cause*)? Is their tidying additional to what would have happened in the absence of my action?'). The important point to take from Gillenwater's analysis is that the cause and effect in question cannot be the same thing, but the stipulation that additionality only applies to the relationship between offset markets and projects appears to be overly restrictive.

It is important to address the concept of additionality in the present discussion due to its prominence in the project accounting literature, however, the issue can also be viewed as something of a distraction from the core structure of the project accounting method. The core structure involves the quantification of baseline and project scenario emissions, and calculating the difference between the two (as shown in Figure 6 and Equation 1). The notion of additionality is already captured within that structure in the sense that if there is no difference between the two scenarios (i.e. if the two scenario lines in Figure 6 are identical), there will be no 'additional' effect. Indeed, standards such as the *GHG Protocol for Project Accounting* largely side-step the issue of additionality in exactly this way, by treating it as implicit within the method: 'Additionality is incorporated as an implicit part of the procedures used to estimate baseline emissions' (WBCSD/WRI 2005, p.8). As with the concepts of *crediting baselines*, *conservativeness*, and *emphasis on reductions*, additionality can be seen as a legacy of project accounting's development within the practice of carbon offsetting, in which concerns about non-additionality arise.

3.4. Policy-level accounting

Policy-level greenhouse gas accounting aims to quantify the total changes in emissions and removals caused by policies, such as laws, regulations, taxation, incentive schemes, investment, information instruments etc. (WBCSD/WRI 2013). Although there are many instances of policy greenhouse gas assessments (for example: Defra (2011); and US EPA (2013)), this field of practice has only recently undertaken the process of international standardisation, with the publication of the GHG Protocol's *Policy and Action Standard* in 2014 (WRI 2014c).

The structure adopted for policy accounting is essentially the same as that for project accounting, with the key steps involving the quantification of baseline and policy scenario emissions, and then calculating the difference between the two

(WRI 2014c, p.9). The lack of a clear methodological demarcation between project and policy accounting may not be wholly unexpected given the lack of a clear demarcation between what counts as a 'project' and what counts as a 'policy'. Typically a project is characterised by physical activities in a specific geographic location (Watson et al. 2000, sec.5.1.2), while policies may involve less physical interventions such as regulations, taxes, and other market-based instruments. However, these distinguishing characteristics are not always present: projects can involve less physical interventions such as information campaigns, and policies can involve location-specific physical interventions, such as transport infrastructure. The lack of a clear demarcation means that there are likely to be interventions that could be assessed using either method, and it also suggests that these two forms of accounting could potentially be merged, or treated as sub-categories within an overarching generic framework.

Although the overall structure of project and policy accounting methods is essentially the same there are some areas of detail or emphasis that differ. One area that appears to receive greater attention in policy accounting is the possibility of interactions with other policies and actions. The GHG Protocol *Policy and Action Standard* identifies three types of relationship between policies (WRI 2014c, p.41): an independent relationship whereby policies do not affect each other; a reinforcing relationship whereby policies interact and increase their overall effectiveness (e.g. an awareness campaign and a subsidy may achieve greater change when implemented jointly than if they are implemented separately); and an overlapping or counteracting relationship, whereby policies interact and achieve less change than would be expected by summing what they would achieve individually. The main issue with reinforcing or overlapping relationships is that the results from individual policy assessments cannot be summed to estimate the total effect of implementing multiple policies, and if the *total* impact of interacting policies is of interest, then the interacting policies should be assessed together as a bundle (WRI 2014c, p.44). *Project* accounting is also able to accommodate interaction effects, by including interacting projects in the baseline or also assessing projects as a bundle, but the issue of interactions between interventions is not prominent in the project accounting literature, and is not explicitly addressed in *ISO 14064-2* or the *GHG Protocol for Project Accounting*.

Another possible area of slight divergence between project and policy accounting is in the use of 'direct' *change* calculations in policy accounting. So called 'direct' change calculations bypass the quantification of baseline and policy scenario emissions, and seemingly calculate the change in emissions directly (WRI 2013). This method may be appropriate when total baseline and policy scenario emissions are unknown, but the change in activity, and therefore change in emissions, is known. For example, a transport policy may reduce vehicle mileage by two million miles per year, and in such a case it is not necessary to know what the baseline mileage is, i.e. whether it is 10 million or 20 million miles etc. However, the departure from the *baseline emissions – policy scenario emissions = change in emissions* structure (shown in Figure 6 and Equation 1) may be viewed as a largely superficial difference, as there is still an implicit baseline of '2 million miles more than the policy scenario'. In addition, the notion of '2 million fewer miles' appears to require the qualification of 'compared to the baseline', otherwise it is not known what the reduction relates to, i.e. it could be relative to 'last year', or 'country X' etc.. An interesting issue is whether consequential LCA also involves implicit baselines, and this is explored further in Section 4.

4. Discussion

The previous sections outlined the main methodological features of each of the identified consequential greenhouse gas accounting methods, with some observations on the similarities and differences between them. The following discussion now provides a more in-depth analysis of the similarities and differences, which in turn provides a basis for critiquing the different methods and identifying opportunities for transposing lessons. The analysis also seeks to distinguish between features that represent fundamental divergences, and those that are largely superficial or non-essential.

One important difference between the methods is that project and policy accounting provide a time-series of impacts, i.e. impacts can be provided year-by-year or by any other unit of time, whereas consequential LCA only provides a single normalised impact figure which is intended to be valid for a broad period of time, typically the 'long run' (Weidema 2003; Weidema et al. 2009). This treatment of time has a number of implications for the level of information provided by consequential LCA. Firstly, consequential LCA tends not to show short run impacts, and nor does it show the transition from short run impacts to long run impacts. This means that significantly different impacts from the short run product system may be overlooked if only a long run figure is provided.

Secondly, even when quantifying the impacts associated with the long run product system the temporal distribution of impacts *within* the product's life cycle tends not to be modelled in consequential LCA. However, the temporal distribution of emissions can be particularly important for understanding climate change impacts, as up-front emissions may not be equivalent to the compensatory avoidance of emissions later in the life cycle (O'Hare et al. 2009). This shortcoming is already largely recognised within the LCA community (ISO 2006b; ISO 2006a) and has given rise to the development of 'dynamic' LCA, which aims to include information on the temporal distribution of material and elementary flows (Levasseur et al. 2010; Collinge et al. 2012; Beloin-Saint-Pierre et al. 2014), which would align LCA with the time-series structure of project/policy accounting. Interestingly, although LCA appears to be playing catch-up in terms of developing a time-series structure, it may be ahead in developing temporally-explicit impact factors, such as temporally-adjusted global warming potentials (Levasseur et al. 2010), which do not appear to have been widely used in project or policy accounting. It is likely that the exchange of methodological lessons can work in both directions.

A further time-related issue for LCA, which is avoided by project/policy accounting, is the problem of how to deal with large non-linear emission events, such as land-use change or the production of capital equipment. As discussed earlier, LCA presents normalised results per functional unit, and it is therefore necessary to amortise or average large non-linear impacts over a period of time (Sjödin & Grönkvist 2004; Hsu et al. 2010; Searchinger et al. 2008; Hertel et al. 2010). One major shortcoming with this approach is that the choice of amortisation period is largely arbitrary, for example, Searchinger et al. (2008) use an amortisation period of 30 years for the emissions from land conversion caused by biofuels, while the accounting rules for the EU Renewable Energy Directive use 20 years (European Parliament and Council of the European Union 2009). In contrast, project/policy level accounting provides an assessment of change at the *aggregate* rather than normalised level, and therefore arbitrary amortisation periods are avoided.

Analysing change at the unit level can largely be seen as a legacy of attributional LCA, and there are a number of other problems that appear to come with it. Firstly, it is not always transparent what the aggregate scale of change is when the analysis is presented at the unit level. The existing guidance for consequential LCA does state that the scale of the change should be identified (Weidema et al. 2009), however consequential LCA studies rarely provide a transparent statement of what the aggregate scale is (see for example Thomassen et al. (2008), or Dalgaard et al. (2008)). Relatedly, analysis at the unit level may have a greater likelihood of missing nonlinearities of scale or cumulative impacts (Hauschild 2005), precisely because the analysis is not undertaken at the aggregate-level. Again, the existing guidance for consequential LCA does state that the scale of change should be accounted for, but it is nevertheless more likely that scale-effects will be missed due to the unit level of analysis. In addition, the question can be raised as to whether decision-makers will be better informed by understanding the aggregate impact that their individual decision is contributing to, particularly given that sustainability is a system-level property rather than a characteristic of individual practices (Gray 2010). Project/policy level methods arguably achieve greater transparency, better inclusion of scale-effects, and also greater decision-usefulness by undertaking analysis at the aggregate level. Furthermore, unit level results can always be provided when it is useful to do so by dividing aggregate level results by the number of products produced, or by any other meaningful denominator (though the provisos above regarding nonlinearities of scale should always be kept in mind).

A different but equally important area of divergence between project/policy accounting and consequential LCA is that the former explicitly model baselines, whereas the latter does so only partially or implicitly. Any assessment of change always requires a baseline from which change is measured, and this is evident in some features of consequential LCA such as the procedure of substitution which involves identifying the product system that is displaced (i.e. the baseline) by the supply of co-products. The baseline and intervention scenario structure illustrated in Figure 6 appears to have advantages in terms of increased transparency, but also has highly important benefits in terms of its conceptual robustness and the range of consequences it can accommodate. For instance, substitution is accommodated in LCA software and databases by treating the avoided product system as a negative input to the system studied (Weidema et al. 2009). However, this is conceptually awkward, in the sense that it is difficult to conceive of what a 'negative input' is. In contrast, with the baseline and intervention scenario structure the avoided product system can be accommodated in a conceptually straightforward way by including it in the baseline.

The baseline and intervention scenario structure is also able to accommodate consequences such as rebound effects and complementary products, which require ad hoc or additional procedures within consequential LCA (Weidema et al. 2009). The scenario structure is also able to model situations characterised by imperfect elasticity of supply in the long run, which is assumed not to occur in consequential

LCA (Weidema et al. 2009). In terms of transposing lessons to consequential LCA, consideration should be given to adopting a baseline and intervention scenario structure.

The remaining features that tend to characterise one or other of the consequential methods represent largely superficial or non-essential differences. Project accounting is often characterised by the notions of *crediting baselines*, *conservativeness*, and an emphasis on *reductions*, *leakage* and *additionality*, but these are non-essential features that exist due to project accounting's development within the practice of carbon offsetting. *Additionality* in particular, where it relates to the cause-effect pairing of projects and emission levels, can be subsumed into the structure of the method itself and does not constitute a separate methodological step. Similarly, the concept of *leakage* is a largely superficial or presentational one which creates an arbitrary delineation between some impacts and others (as do the notions of foreground and background processes, and direct and indirect effects, in consequential LCA).

Some of the existing guidance for both consequential LCA (European Commission et al. 2010) and project accounting (ISO 2006d) suggest that equal levels of functional output should be ensured when comparing scenarios. However, this also appears to be a non-essential requirement of either method as one of the consequences of a given decision could be an increase or decrease in the total functional output provided, i.e. our decisions do make us functionally better or worse off, and the total level of productivity is not always exogenous to the decision studied. A final superficial difference between the methods is that consequential LCA tends to be used for *ex ante* assessments, as its primary purpose is to inform decision making. However, as with project and policy accounting, it can be used for both *ex ante* appraisal or *ex post* evaluations of the change caused by past actions (Weidema 2003).

A summary table of the main methodological features of each of the consequential methods, and the similarities and differences between them, is provided in the Appendix.

5. Conclusions

A first point to make is that the *three* identified consequential methods can be re-categorised as *two* different methods, as project and policy accounting are effectively the same approach. Recognition of this fact may have implications for how these two methods should be developed going forward, and raises the question of whether there should be a unifying process in which the two are integrated in future accounting standards. It is possible to envisage a single generic form of 'consequential time-series assessment', of which project and policy accounting are two possible two sub-types.

Building on the idea of a unified 'consequential time-series' method, a case can be made for developing consequential LCA as a further sub-type. The lessons that could be transposed from project/policy accounting to consequential LCA are the adoption of a time-series approach (as suggested by dynamic LCA), quantifying impacts at an aggregate level (with normalisation as a subsequent presentational option), and the adoption of a transparent baseline and decision scenario structure. The changes typically studied by consequential LCA, i.e. changes in product demand or configuration, could be straightforwardly characterised as the 'decision' or 'intervention' modelled by a unified 'consequential time-series' method, with the absence of the decision constituting the baseline. Interestingly, Plevin et al.'s (2014b) characterisation of consequential LCA includes a description of intervention and baseline scenarios (and the subtraction of one from the other to calculate change in emissions), even though this structure is only set out in the project/policy accounting literature, and is not present in the consequential LCA standards/guidance. It appears that some reconceptualization or re-imagining of consequential LCA is already underway.

A further argument in favour of developing a unified 'consequential time-series' method is that there are likely to be cases that can be addressed using the time-series method, but which cannot be handled by consequential LCA, i.e. decisions/interventions that do not straightforwardly relate to products. For example, changes to income tax policy may have a general effect on economic activity and therefore impact on greenhouse gas emissions, but if it is not possible to identify specific product systems that are affected then the use of consequential LCA will not be appropriate. However, the reverse situation does not appear to arise, i.e. the flexibility of the time-series method means it can be applied to any product-related decision or intervention.

As an aside, it is interesting to note that the identified methodological shortcomings with consequential LCA can largely be seen as a legacy of its evolution from traditional attributional LCA, the structure of which does not include a time-series, aggregate-level quantification, or baselines. One is reminded of the traveller who asks for directions and is told 'I wouldn't start from here'. The same might be said of consequential LCA, given its attributional beginnings.

Nevertheless, the additional information and transparency from restructuring consequential LCA may come at the expense of ease of implementation or comprehensibility. This is something that could be explored in future research by applying the different consequential methods to the same case study scenarios in order to compare both the results *and* the practicality of implementation. Another consideration is whether additional information and transparency is actually required for fulfilling the goals and needs of the intended audience, i.e. there may be instances where a simplified approach is sufficient for the decision at hand. A final observation is that the process of comparing different greenhouse gas accounting methods appears to be a useful one for better understanding the nature of each method, and for identifying new avenues for methodological development.

Similar benefits may also accrue from undertaking the same exercise with attributional greenhouse gas accounting methods.

6. Appendix

Table 7 Summary of the main methodological similarities and differences between consequential methods

Characteristics of Method	Consequential LCA	Project Accounting	Policy Accounting
Key features			
1. Time-series of impacts	Consequential LCA generally does not show the distribution of impacts over time.	Project accounting generally shows the distribution of impacts over time.	Policy accounting generally shows the distribution of impacts over time.
2. Amortisation periods	Consequential LCA provides results per functional unit, and therefore large one-off emissions are amortised over a number of years. One disadvantage of this approach is that the amortisation period is largely arbitrary.	Project accounting does not have to use amortisation periods as total aggregate change is quantified.	Policy accounting does not have to use amortisation periods as total aggregate change is quantified.
3. Use of baselines	Consequential LCA does not explicitly use the concept of a baseline, though the concept is implicit in the measurement of change.	Project accounting is explicit in specifying a baseline.	Policy accounting is explicit in specifying a baseline.
Superficial features			
4. Crediting baselines, conservativeness, and emphasis on reductions	These features tend not to appear in consequential LCA.	These features often appear in project accounting and are a legacy of the method's development for carbon offsetting.	These features tend not to appear in policy accounting.
5. Additionality	Additionality is not referred to in consequential LCA, but it is implicit in the approach of only considering processes that change (i.e. processes that are additional).	Additionality is often referred to in project accounting, but can be subsumed or left implicit within the structure of calculating the difference between baseline and project scenario emissions.	Additionality is largely subsumed or left implicit within the structure of calculating the difference between baseline and project scenario emissions.

Characteristics of Method	Consequential LCA	Project Accounting	Policy Accounting
6. Leakage	The concept of leakage is not used in life cycle assessment, however, the distinctions between ‘foreground’ and ‘background’ processes, and ‘direct’ and ‘indirect’ effects are similar to that between ‘in boundary’ and ‘out of boundary’ effects (and therefore leakage).	The concept of leakage is used but is not essential to project accounting as it can be treated as an artifice of defining a project boundary. If all significant effects are quantified then the notions of a project boundary and leakage are not necessary.	The concept of leakage is generally not used in policy accounting.
7. Requirement for equal functional outputs in comparator scenarios	The comparison of different product systems generally requires the use of equivalent functional units. However, this is does not appear to be an essential requirement as one of the outcomes of a given decision may be an increase or decrease in functional output.	ISO 14064-2 requires that the project scenario should have the same level of product and service provision as the baseline scenario. However, changes in the level of product or service provision may be one of the consequences of implementing a project, and this requirement appears to be unnecessarily restrictive.	Policy accounting does not require the baseline and policy scenario to have equivalent levels of product or service provision, as one of the outcomes of a policy may be changes in product or service provision.
8. Ex post or ex ante assessment	Consequential LCA is generally undertaken as an ex ante assessment, as it is primarily used to inform decision-making. However, the method can be used ex post to estimate the change caused by past decisions.	Project accounting can be used for either ex ante estimates of expected effects or ex post estimates for implemented projects.	Policy accounting can be used for either ex ante appraisal of proposed policies, or ex post evaluation of implemented policies.

Paper 3 - Comparative Analysis of Attributional Corporate Greenhouse Gas Accounting, Consequential Life Cycle Assessment, and Project/Policy Level Accounting: a Bioenergy Case Study

Abstract

In order to avoid dangerous climate change, greenhouse gas accounting methods are needed to inform decisions on mitigation action. This paper explores the differences between 'attributional' and 'consequential' greenhouse gas accounting methods, focusing on attributional corporate greenhouse gas inventories, consequential life cycle assessment, and project/policy greenhouse gas accounting. The case study of a 6 megawatt bioheat plant is used to explore the different results and information these methods provide. The findings show that attributional corporate inventories may not capture the full consequences of the decision in question, even with full scope 3 reporting – and are therefore not sufficient for mitigation planning. Although consequential life cycle assessment and the project/policy level method both aim to show the full consequences of the decision, the project/policy level method has a number of advantages, including the provision of a transparent baseline scenario and the distribution of emissions/removals over time. The temporal distribution of emissions/removals is important as the carbon payback period for the bioheat plant can exceed 100 years, making the intervention incompatible with 2050 reduction targets. An additional novel contribution from the study is the use of normative decision theory to suggest that the uncertainty associated with bioenergy outcomes is itself a highly decision-relevant finding.

Keywords: consequential LCA, attributional LCA, corporate GHG inventory, bioenergy, project-level GHG accounting, policy-level GHG accounting

1. Introduction

Climate change poses serious global risks (Stern 2006), and greenhouse gas accounting methods are needed to understand the scale of emissions associated with different activities, and to assess the effectiveness of climate change mitigation options. A large number of different greenhouse gas accounting methods have been developed, including national inventories (IPCC 2006), community/city inventories (Schultz et al. 2014; British Standards Institute 2013), policy assessments (WRI 2014c), corporate/organisational inventories (WBCSD/WRI 2004; WBCSD/WRI 2011b; ISO 2006c; Pelletier et al. 2013), project-level methods (WBCSD/WRI 2005; ISO 2006d), and product-level life cycle assessment (British Standards Institute 2012; ISO 2013b; ISO 2006b; WBCSD/WRI 2011c; European Commission 2013), among others. Given this array of different methods it is not always clear which method(s) are the most appropriate for a given purpose.

A helpful distinction between types of method, which has developed specifically within the field of life cycle assessment (LCA), is that between what are called 'attributional' and 'consequential' approaches (Finnveden et al. 2009; Weidema 2003; Ekvall & Weidema 2004; R. J. Plevin et al. 2014b). Attributional methods can be broadly defined as inventories of anthropogenic emissions and removals for a given inventory boundary, while consequential methods aim to quantify the total change in emissions that occur as a result of a given decision or action (Brander & Ascui forthcoming; Brander 2015). The LCA literature suggests that consequential methods are more appropriate for decision-making as they capture the total consequences of the decision at hand (Weidema 1993; R. J. Plevin et al. 2014a; R. J. Plevin et al. 2014b), and empirical studies show that basing decisions on attributional LCA can result in mitigation actions which unintentionally increase rather than decrease emissions (Searchinger et al. 2008; Hertel et al. 2010).

Previous research has suggested that the attributional-consequential distinction can be extended beyond the field of life cycle assessment to create a generic categorical scheme for classifying all forms of physical greenhouse gas accounting (Brander & Wylie 2012; Brander 2015b). Brander (2015b) suggests that corporate/organisational inventories (henceforth, referred to as corporate inventories), national inventories, and community inventories, can be categorised as *attributional* in nature, while project-level and policy-level methods are *consequential* in nature. One benefit from developing this categorical scheme is to allow inferences about the appropriate use of methods of a certain categorical type. A further benefit may be the exchange of methodological lessons between approaches of the same type (Brander 2015b).

The present study illustrates the importance of selecting the correct method for decision-making by showing the magnitude of difference in the results from an attributional corporate inventory and the different consequential methods that are available (consequential LCA, and the project/policy approach). Although the study uses a corporate inventory as the attributional comparator, conclusions can also be drawn for attributional methods more generally, including the ongoing debate between attributional and consequential life cycle assessment (R. J. Plevin et al. 2014b; R. J. Plevin et al. 2014a; R. Plevin et al. 2014; Hertwich 2014; Brandão et al. 2014; Dale & Kim 2014; Suh & Yang 2014). A further contribution from the study is a comparison between the results from the two different consequential methods (consequential LCA and project/policy assessment), which has not previously been undertaken.

In addition to contributing to the conceptualisation and development of different greenhouse gas accounting methods, the study also directly contributes to the extensive debate on the greenhouse gas impacts of bioenergy (Bernier & Paré 2013; Bright et al. 2012; Schulze et al. 2012; Edrisi & Abhilash 2015; Searchinger 2012; Haberl et al. 2012; Upham & Smith 2014; Cherubini et al. 2009; Favero & Mendelsohn 2013; Haberl et al. 2013). This debate is a highly topical one, given the considerable corporate and governmental support for bioenergy as a climate change mitigation option (e.g. Diageo (2015); European Parliament and Council of the European Union (2009); UK Government (2012); US Department of Energy (2015)). The existing literature on bioenergy shows a wide range of possible outcomes (i.e. ranging between large reductions to large increases in net emissions), and an additional contribution from the present study is the application of normative decision theory to interpret such uncertainty as a highly decision-relevant finding in its own right.

2. Method

The overall approach used in this study is to apply a corporate inventory method, a consequential LCA, and a project/policy-level assessment to the same case study decision scenario, and then to undertake a comparative analysis of the results from each method. The development of a bioheat plant was selected for the case study decision scenario as data were available for a proposed 6 MW bioheat plant in the east of Scotland, and bioheat was considered likely to provide a 'crucial' case (Gerring 2004), i.e. one which illustrates the differences between the methods. A single case study will not allow the estimation of the probability that attributional methods omit important consequences, but it is sufficient for inferring that for any given decision scenario it is uncertain whether using an attributional method is sufficient. The use of a single case study in this way is variously described as 'nomothetic' (Bryman & Bell 2007) or 'critical' (Yin 2003), where it negates a generalised premise (i.e. that attributional methods are *sufficient* for managing greenhouse gas emissions). A further reason for selecting a bioenergy case study is that bioenergy is a highly topical issue, given the high level of policy support, noted above.

Corporate greenhouse gas accounting was selected as the attributional method for comparison with the consequential methods for a number of reasons. This comparison does not appear to have been undertaken before in the existing literature, whereas there are already a number of studies comparing attributional and consequential LCA (e.g. Ekvall & Andræ (2006); Thomassen et al. (2008); Dalgaard et al. (2008); and Searchinger (2008)). A second reason is that the use of corporate inventories for managing greenhouse gas impacts is widespread (CDP 2015), and is currently supported by government policy (Defra 2013; UK Government 2013; Scottish Government 2009; European Commission 2013), and it is therefore important to explore whether the use of such inventories can lead to sub-optimal decision-making. Each of the greenhouse gas accounting methods applied to the case study scenario are now described in turn.

The GHG Protocol's *Corporate Accounting and Reporting Standard* (WBCSD/WRI 2004) was used for undertaking the corporate inventory, as this is considered the most widely used standard for such inventories. ISO 14064-1:2006 *Specification with guidance at the organization level for quantification and reporting of greenhouse gas emissions and remova*ls would have yielded very similar results, although the GHG Protocol requires the quantification and reporting of CO_2 emissions from biomass, whereas for the ISO standard this is recommended but optional. The organisational boundary for the inventory is the organisation commissioning the bioheat plant, and the operational boundary is all emissions from energy use at facilities owned/operated by the organisation (termed 'scope 1' emissions); all emissions from purchased electricity, heating or cooling ('scope 2' emissions); and all other value-chain sources for which data were available ('scope 3' emissions); and emissions from the combustion of biomass and biofuels (reported separately from the scopes). The operational boundary is shown in detail in Table 8.

Table 8. Operational boundary

Scope	**Emission source**
Scope 1	Natural gas
	Diesel
	Biodiesel
	Petrol
Scope 2	UK grid electricity
Scope 3	Purchased goods and services
	Capital goods
	Fuel and energy related activities
	Waste generated in operations

	Business travel
Biogenic emissions	Biofuel component of biodiesel
	Woody biomass

Activity data were collected from the energy officer at the organisation commissioning the bioheat plant for the period August 2012 to July 2013. Emission factors were sourced from the Department for the Environment, Food and Rural Affairs and the Department for Energy and Climate Change (Defra/DECC 2015). However, the Defra/DECC emission factors are provided in units of CO_2e using global warming potentials (GWPs) from the Second Assessment Report (SAR) (IPCC 1996), whereas the GHG Protocol *Corporate Standard* requires reporting in tonnes of each greenhouse gas, and CO_2e should be calculated using the latest available 100 year global warming potentials. The published factors for CH_4 and N_2O emissions were therefore divided by the SAR GWPs to allow reporting in tonnes of CH_4 and tonnes of N_2O, and these figures were then multiplied by the Fifth Assessment Report GWPs (IPCC 2013a).

The corporate inventory was then used to assess the benefits of developing the 6 MW bioheat plant by modelling the inventory with and without the plant. It is important to note that the GHG Protocol *Corporate Standard* and ISO 14064-1 do not provide guidance on how to use greenhouse gas inventories to select mitigation actions, however, they do suggest that such inventories can be used to manage emissions. In addition, the organisation commissioning the bioheat plant used its own corporate inventory data to support its decision (i.e. corporate inventories are used in this way in practice). The level of guidance provided on the use of attributional inventories to inform decision-making is discussed further in the Discussion section (4.1).

The upstream or embodied emissions associated with the bioheat plant (the boiler, pipes, and installation activities etc.) were estimated using projected capital expenditure figures from the design team and the input-output supply chain emission factor for construction from Defra/DECC (2012). The resulting emissions estimate should be viewed as indicative only, as the factor is based on average emissions across the construction sector in the UK. The upstream emissions from the cultivation and processing of woody biomass were estimated using figures for the expected energy input to the bioheat plant and Defra/DECC's (2015) emission factor for upstream emissions from wood chips (0.01662 $kgCO_2e$/kWh of woodchips). These emissions were included as part of the scope 3 'fuel and energy-related activities (not included in scope 1 or 2)' category, while the CO_2 emissions from the combustion of the biomass itself are reported separately from scopes 1, 2, and 3, as per the requirements of the *Corporate Standard* (WBCSD/WRI 2004, p.63).

Turning to the comparator consequential methods, a consequential LCA and a combined project/policy-level assessment were undertaken (the project and policy-level methods were combined as previous research suggests that these methods have essentially the same structure and approach (Brander 2015b)).

Taking the consequential LCA first, the guidance used for implementing this method was predominantly that provided in Ekvall and Weidema (2004), and Weidema et al. (2009), with the general structure for the consequential LCA taken from the International Reference Life Cycle Data System Handbook (European Commission et al. 2010). Following this guidance, the goal and scope of the study is to estimate the change in greenhouse gas emissions/removals caused by the decision to implement a 6 MW bioheat plant in the east of Scotland, with a 200 year assessment period. The functional unit is 1 kWh of delivered heat.

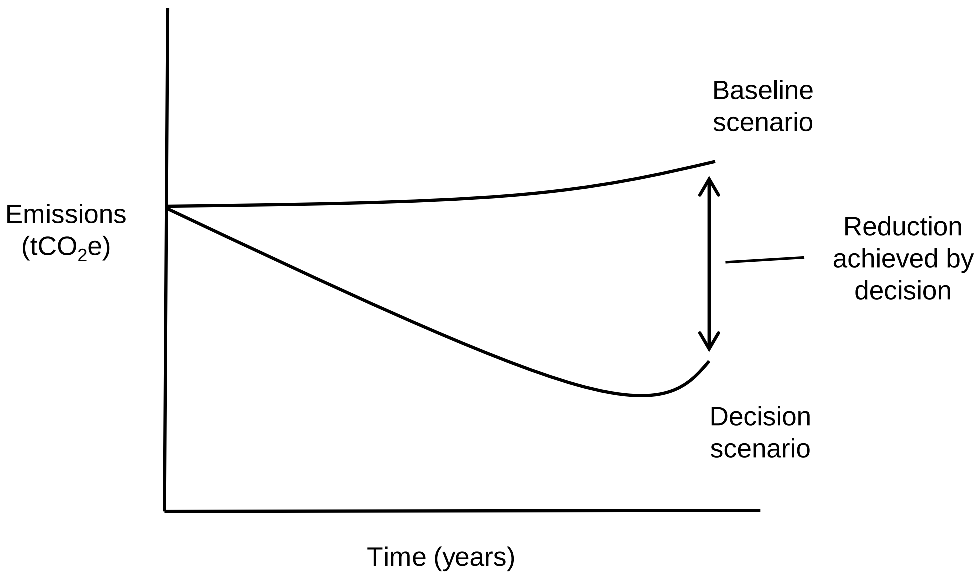
For the life cycle inventory stage, the processes included are those that change as a result of the decision, i.e. the marginal processes (Schmidt & Weidema 2008). It is worth noting that the requirement to identify all the processes that change is the same in the project/policy method, though there are differences in the structure of the methods which are discussed later. In consequential LCA, changes caused by the supply of co-products, or other instances of multi-functionality, are addressed through the technique 'substitution' (also sometimes referred to as 'system expansion'). Substitution involves identifying the product systems that are displaced (i.e. *changed*) by the production of co-products, and crediting the displacement of those product systems to the decision studied, as the avoidance of those systems and their associated impacts are a consequence of the decision (Weidema et al. 2009).

Similarly, if the decision in question causes the use of a constrained resource that would otherwise be used for an alternative purpose, then the substitute processes used to fulfil that purpose are included in the life cycle inventory, as they are affected by the decision in question (Ekvall & Weidema 2004, p.167). In the case of the present study, an example of a constrained resource is saw mill residues, the use of which for bioenergy entails that fewer residues are available for the production of medium density fibre (MDF) board, and the reduced production of MDF may be replaced by plasterboard, as a substitute. The production and other life cycle stages of *plasterboard* are therefore included in the inventory as they *change* as a result of the decision studied.

Finally, one-off emissions, such as those from the construction of the bioheat plant, were amortised over the 25 year lifetime of the plant (and the need for an amortisation period in consequential LCA is explored in the Discussion (4.2), as amortisation is absent in the project/policy method). The remaining methodological details of the consequential LCA, e.g. scenario modelling, data, emission factors etc., are shared with the project/policy method and are therefore described in conjunction, following a brief overview of the features unique to the project/policy approach.

The guidance and standards used for implementing the project/policy method are *ISO 14064-2* (ISO 2006d), the *GHG Protocol for Project Accounting* (WBCSD/WRI 2005), and the GHG Protocol's *Policy and Action Standard* (WRI 2014c). The fundamental structure of this approach is to create a time-series of emissions/removals for a baseline scenario, i.e. the scenario in which the decision has not been taken, and for a ‘with decision’ scenario. As with consequential LCA, the intention is to include all the emission source/sinks that *change*. Subtracting the baseline emissions/removals from the decision scenario emissions/removals provides the change in emissions/removals caused by the decision. This methodological structure is illustrated in Figure 8 below.

Figure 8. Illustration of the key components of the project/policy accounting method



Other than these structural differences the methodological details of the two consequential methods are largely the same. The same data, scenarios, assumptions, and emission factors were used for both methods (with the exception of the emission factors for transportation and UK grid electricity, which are expected to reduce over time, and this dynamic element is accommodated in the project/policy method’s time-series structure, but not included in the consequential LCA). Details of the input data, assumptions, forest carbon model, and emission factors are provided in the online supporting material [provided in Appendix A - Supporting Material for Paper 3 to this thesis].

A highly important and shared feature of the consequential methods is the use of scenarios for modelling the different possible marginal systems affected by the decision in question (Weidema et al. 2009; WRI 2014c). Seven scenarios, and thirteen sub-scenarios were modelled, and are summarised in Table 9 below.

Table 9. Details of scenarios for the marginal systems affected by the decision (used in the consequential modelling)

Name of scenario	Description	Name of sub-scenario	Description
1. Overseas production	Increase in demand for wood chips increases the production at the world marginal supplier of biomass. Supply in the UK is constrained and so the marginal supply is overseas production.	1.1. Sustainable forest management	The harvested forest is replanted.
		1.2. Unsustainable forest management	The harvested forest is not replanted.
2. Local production	Increase in demand for wood chips is met from local wood resources that would otherwise not be harvested/utilised, e.g. harvesting of shelter belts, small farm woodlands, wooded steep-sided gullies.	2.1. Local production without co-products	Whole trees are harvested and used for wood chips.
		2.2. Local production with co-products	Part of the tree is used for wood chips and the remainder is used for pallets and construction. In order to make the transportation of the co-products to the saw mill economically viable the trucks backhaul biomass to the bioheat plant.
3. Thinnings	Increase in demand for wood chips makes increased thinning of existing productive forestry economically viable.	3.1. Without co-products	There is no change to the proportion of harvested stem wood that can be used for pallets and saw logs.
		3.2. With co-products (marginal saw log displacement)	Thinning changes the proportion of harvested stem wood that can be used for pallets and saw logs. Reduction in plastic pallet production and marginal saw log production.

		3.3. With co-products (cement render displacement)	Thinning changes the proportion of harvested stem wood that can be used for pallets and saw logs. Reduction in plastic pallet production and use of cement render.
4. Fencing	Increase in demand for wood chips displaces the use of wood for fence posts and increases the production of concrete posts.	4.1. End of life combustion	The wooden posts would have been combusted for energy at their end of life.
		4.2. End of life decay	The wooden posts would have decayed aerobically at their end of life.
5. Pallets	Increase in demand for wood chips displaces the use of wood for pallets and increases the production of plastic pallets.		The reduced demand for wooden pallets due to the longer lifetime of plastic plastics increases biomass availability and displaces natural gas combustion.
6. MDF	Increase in demand for wood chips increases biomass market demand for wood fibre and reduces production of medium density fibreboard (MDF), and increases the production of plasterboard.		
7. Particle board	Increase in demand for wood chips increases biomass market demand for wood fibre and reduces the production of particleboard, and increases the	7.1. Breeze block lower estimate	A lower emission factor for breeze blocks is used (Hammond & Jones 2008).
		7.2. Breeze block upper	A higher emission factor for breeze blocks is

	production of breeze blocks.	estimate	used (DECC 2014).

The selection of scenarios was informed by a number of principles and heuristics from the consequential LCA guidance, e.g. the marginal processes must be unconstrained; are likely to be the least-cost form of production in an growing market; and markets are assumed to be linked unless there is evidence to the contrary (Ekvall & Weidema 2004; Weidema et al. 2009). The selection of scenarios was also based on a range of information: published studies (e.g. Lamers et al. (2015) and Lauri et al (2014)) indicate that the marginal supply will come from increased overseas production); interviews (e.g. information from the commissioning organisation and local forest managers suggested increased local production as a possible marginal system); industry reports (e.g. the Wood Panel Industries Federation (2010) suggests the marginal effect will be material displacement and substitution); and government greenhouse gas accounting tools (e.g. DECC's *Biomass Emissions and Counterfactual Model* (2014) includes both overseas production and material substitution effects).

An assessment of the probability of each of the scenarios has not been undertaken in the present study, though all of the scenarios modelled are considered to be plausible. It should be noted that the actual change caused by the decision may involve combinations of these scenarios/marginal systems, and therefore the presentation of individual scenarios is a simplification of a more complex reality. Furthermore, the scenarios modelled are not exhaustive, and alternative scenarios are also possible. The scenarios are best viewed as 'selective illustrative examples', following the approach in Zanchi et al. (2012).

An attempt was made to include all significant emission sources/sinks affected in each scenario, e.g. above ground biomass, soil carbon, whole-of-life emissions for all energy and material inputs etc. Causal chain maps were produced to provide an overview of the marginal processes and emission sources/sinks included in each scenario, and are presented in the online supporting material. The conjunction of the causal chain maps and the list of data, assumptions, forest growth model, and emission factors provides information for replicating the findings. However, it is worth providing a brief explanation of two of the more complicated scenarios: increased overseas production (scenario 1); and increased local production (scenario 2).

Increased overseas production (scenario 1) does not necessarily entail that the biomass combusted at the 6 MW bioheat plant is from overseas, but rather that this is the marginal effect of an increase in demand for woody biomass. It is assumed that the consumers/producers who would have otherwise used the biomass combusted in the bioheat plant will seek an alternative source of biomass, creating a causal chain which ultimately causes an increase in production overseas. There is considerable evidence to suggest that this is a likely scenario: UK demand for biomass is expected to exceed domestic supply (John Clegg Consulting Ltd 2006); UK forest production is expected to decline from 2030 onwards (Forestry Commission 2014); biomass is already an internationally traded commodity (FAO 2009; Lamers et al. 2015; Buongiorno et al. 2010), suggesting there is no market delimitation due to trade or geographical barriers (Weidema et al. 2009); and the international marginal supply of biomass is projected to come from the US, South America, Africa, and Asia, with only limited additional supply within Europe (Lauri et al. 2014).

An alternative possible scenario is that the increase in demand for biomass brings otherwise unmanaged local woodland, such as shelter belts and wooded gullies, into production (scenario 2). Sub-scenario 2.2 models the possibility that a proportion of the additional harvested stem wood is transported to saw mills to produce timber for construction and wooden pallets, thereby displacing marginal saw log production and plastic pallets, respectively. The cost of transportation imposes a constraint on this scenario, as in order to avoid an empty inward journey to the east of Scotland the haulage trucks are assumed to carry biomass to the bioheat plant, in proportion to the quantity of higher quality stem wood transported out. The marginal impact of the demand for inward-hauled biomass is assumed to be increased production overseas, as in scenario 1.1. The alternative local production sub-scenario (2.1) assumes that whole trees are chipped and combusted, and therefore all of the marginal supply may come from increased local production. However, the plausibility of this scenario may be questioned given that other bioenergy plants are expected to put pressure on existing local woody biomass supply (Fife Council 2013), and the costs of harvesting small and steep-sloped woodlands may restrict the viability of sourcing biomass that would not otherwise be utilised (Fife Council 2013; Walker 2009).

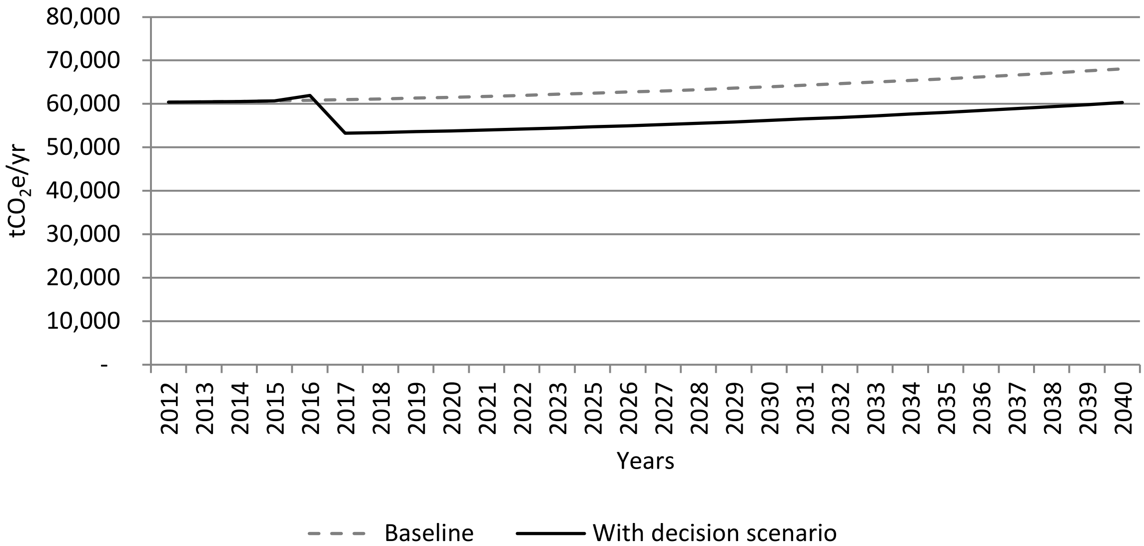
3. Findings

This section presents, in turn, the results from the attributional corporate inventory method; the consequential LCA; the project/policy method; and a comparison of the results from the different methods.

3.1. Corporate greenhouse gas inventory

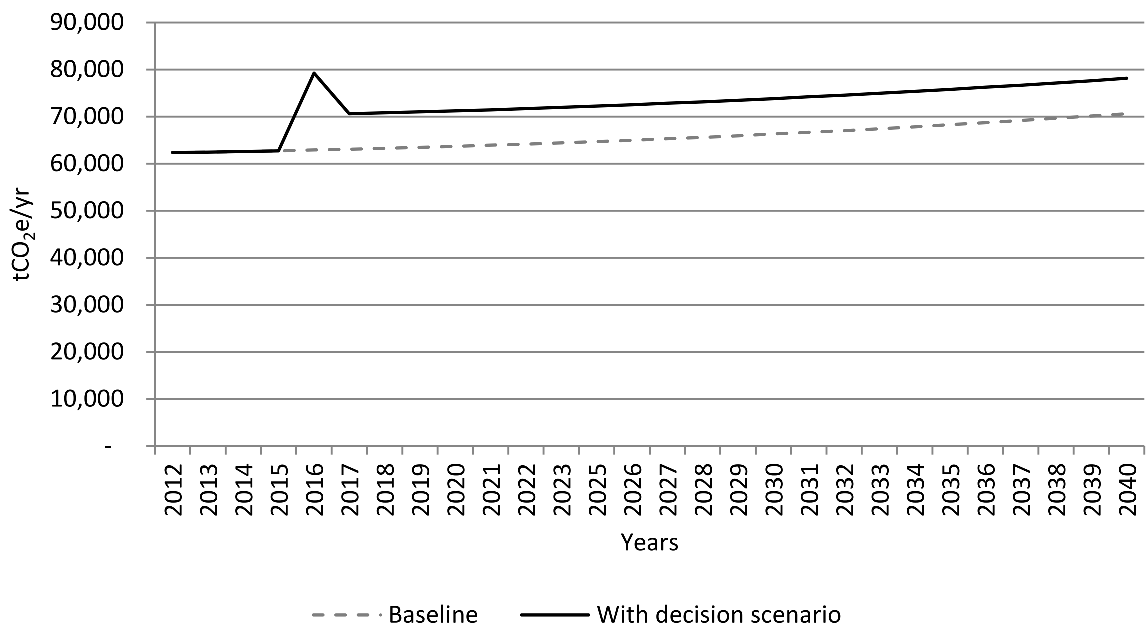
Figure 9 presents the results for scopes 1, 2, and 3 of the corporate inventory. There is a very small initial increase in emissions due to the embodied emissions and construction of the bioheat plant (reported under ‘capital goods’ in scope 3 (WBCSD/WRI 2011b)), before there is a reduction in emissions due to reduced natural gas combustion.

Figure 9. Corporate GHG inventory – scopes 1, 2 and 3



The accounting rules for corporate inventories state that biogenic CO_2 emissions (i.e. CO_2 emissions from the combustion of biomass) should not be reported within scopes 1, 2, and 3, but should be reported separately. Figure 10 presents the results for scopes 1, 2, 3, and biogenic emissions. This version of the inventory shows the same initial increase in emissions, but also an underlying increase in total greenhouse gas emissions as the release of biogenic CO_2 is greater than the baseline release of fossil CO_2 from natural gas combustion. This is because natural gas has lower point-of-combustion CO_2 emissions per unit of energy, and the overall efficiency of natural gas boilers tends to be higher than biomass boilers. However, the results in Figure 10 should be interpreted with caution as although the upstream emissions from the production of the woody biomass are included in the inventory (reported under 'fuel and energy related activities' in scope 3 (WBCSD/WRI 2011b)), the sequestration of CO_2 that occurs during the growth of the biomass is generally not included in the emission factors used for corporate greenhouse gas accounting (for example, see Defra/DECC (2015)). If this sequestration were included then the results would be identical to those in Figure 9. The overall finding is that the use of an attributional corporate inventory *would* support the decision to implement the bioheat plant, with an average reduction in emissions of 7,083 tCO_2e/yr (assuming the otherwise continued use of natural gas).

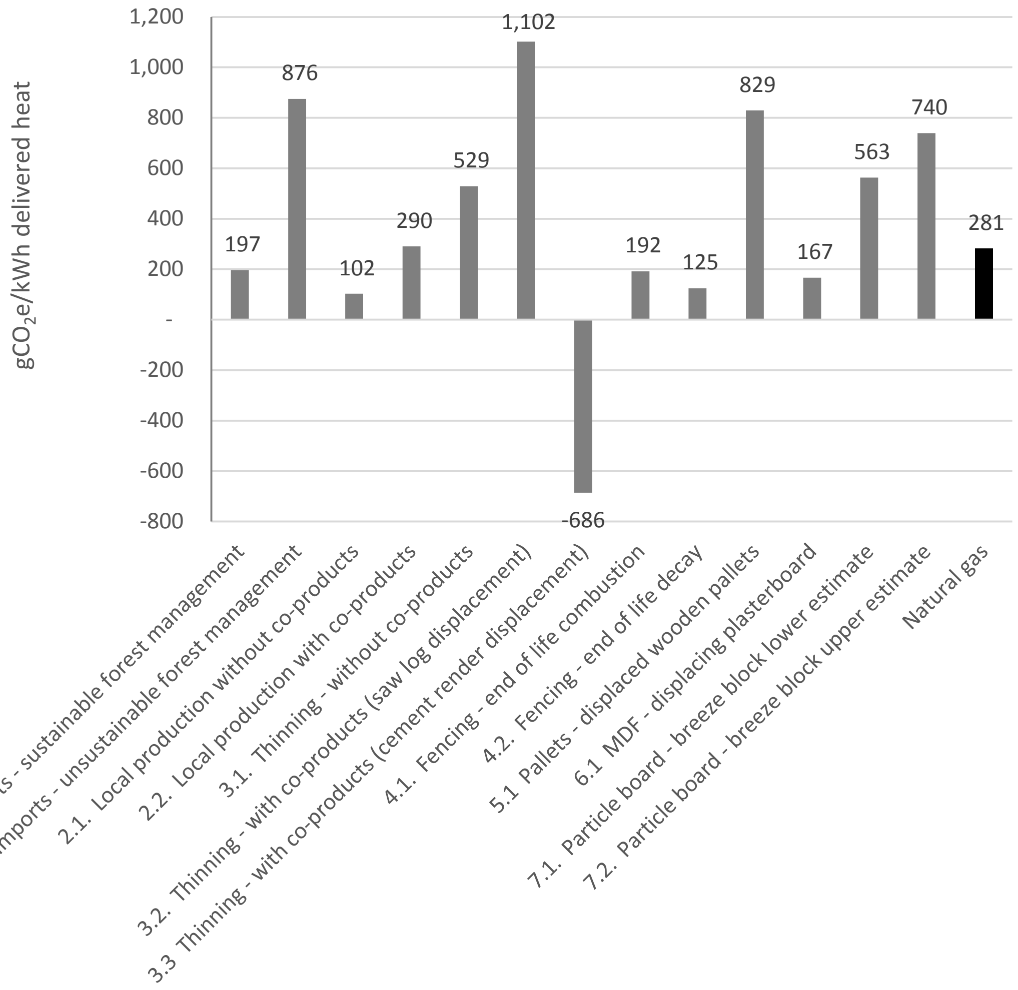
Figure 10. Corporate GHG inventory – scopes 1, 2, and 3 + biogenic CO_2



3.2. Consequential life cycle assessment

Figure 11 presents the results from the consequential LCA in gCO_2e/kWh of delivered heat (i.e. per functional unit). There is a very wide variation in the results, depending on the scenario modelled. All the scenarios with emissions lower than 281 gCO_2e/kWh (the natural gas reference case) entail that the bioheat plant will reduce emissions, and all the scenarios with emissions higher than the reference case indicate the bioheat plant will increase emissions.

Figure 11. Results from the consequential LCA

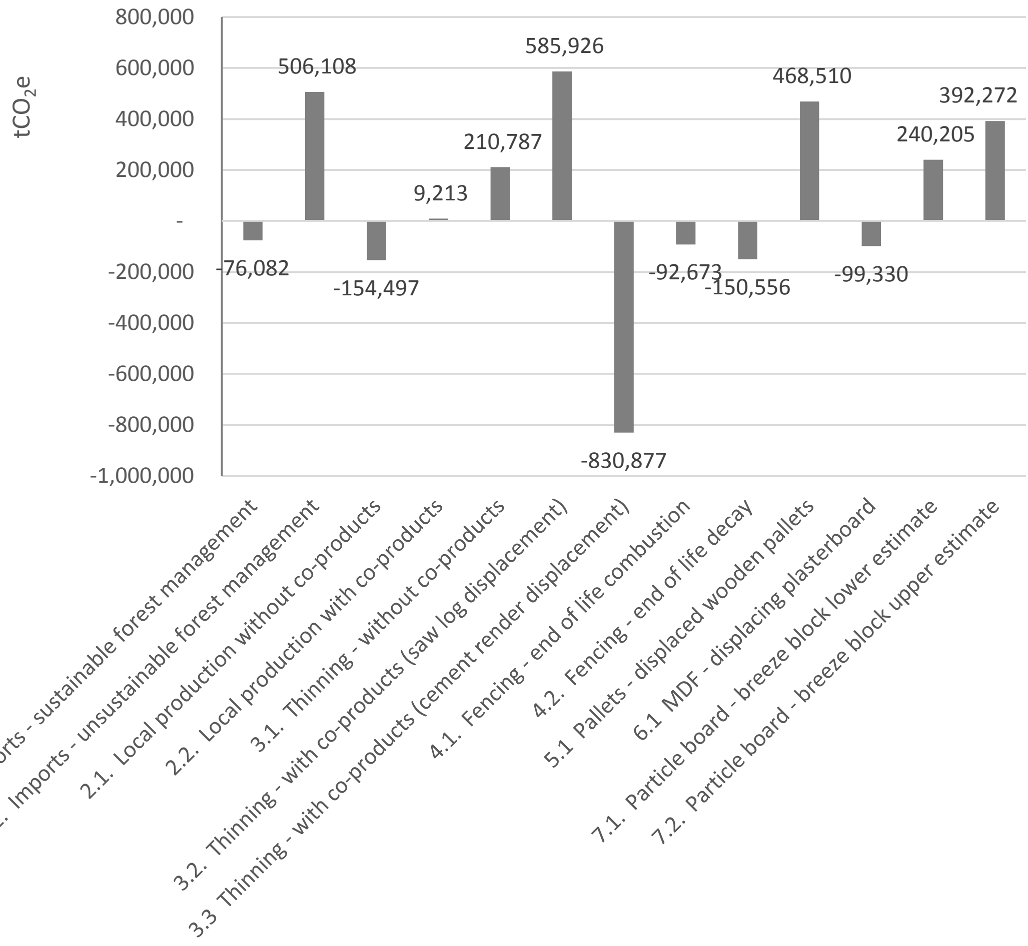


The results for scenario 3.3 (increased thinning with the additional availability of sawlogs replacing cement render) show net negative emissions as the emissions avoided by the substitution of cement render are greater than the emissions from the rest of the life cycle.

3.3. Project/policy-level accounting

Figure 12 presents the results from the project/policy-level method. The results are for the total net change in emissions/removals caused by the decision to implement the bioheat plant. Negative results (below the horizontal axis) indicate that the decision creates a net reduction in emissions, and positive results (above the horizontal axis) indicate that the decision creates a net increase in emissions. The scenarios which create increases or reductions in emissions are the same as those from the consequential LCA, though it is important to note that the presentation of the results is slightly different. The outputs from the project/policy method already show the total change in emissions caused by the decision (baseline emissions/removals minus decision scenario emissions/removals), and no further subtraction of a comparator product's emissions are required.

Figure 12. Results from the project/policy level method



In addition to the total net change in emissions/removals, the project/policy level method also provides information on the distribution of emissions and removals over time, as both baseline and decision-scenario emissions/removals are calculated as a time-series. Consideration of temporal information is proposed in dynamic LCA (Levasseur et al. 2010; Collinge et al. 2012; Collet et al. 2013; Helin et al. 2013), however conventional (i.e. static) consequential LCA is used in the present study as this is the approach set out in the existing guidance literature (Weidema et al. 2009), and the comparison of the time-series (project/policy method) and non-time-series (standard consequential LCA) approaches also serves to illustrate the importance of further developing and mainstreaming dynamic LCA.

The time-series output from the project/policy method is illustrated in Figure 13, using the example of scenario 1.1 (the time-series outputs for the other scenarios are provided in the online supporting material). There is an initial increase in emissions due to the embodied emissions of the bioheat plant, followed by a period of high emissions due to the higher point-of-combustion emissions from biomass compared to natural gas. After the assumed 25 year life-time of the bioheat plant the underlying trend in forest regrowth becomes apparent, and the level of sequestration in the decision scenario is greater than in the baseline. The emissions breakeven point (i.e. the point at which the cumulative decision scenario emissions/removals equal the cumulative level of emissions/removals in the baseline) is reached in year 75.

Figure 13. Project/policy method times-series results for scenario 1.1 (overseas production with sustainable forest management).

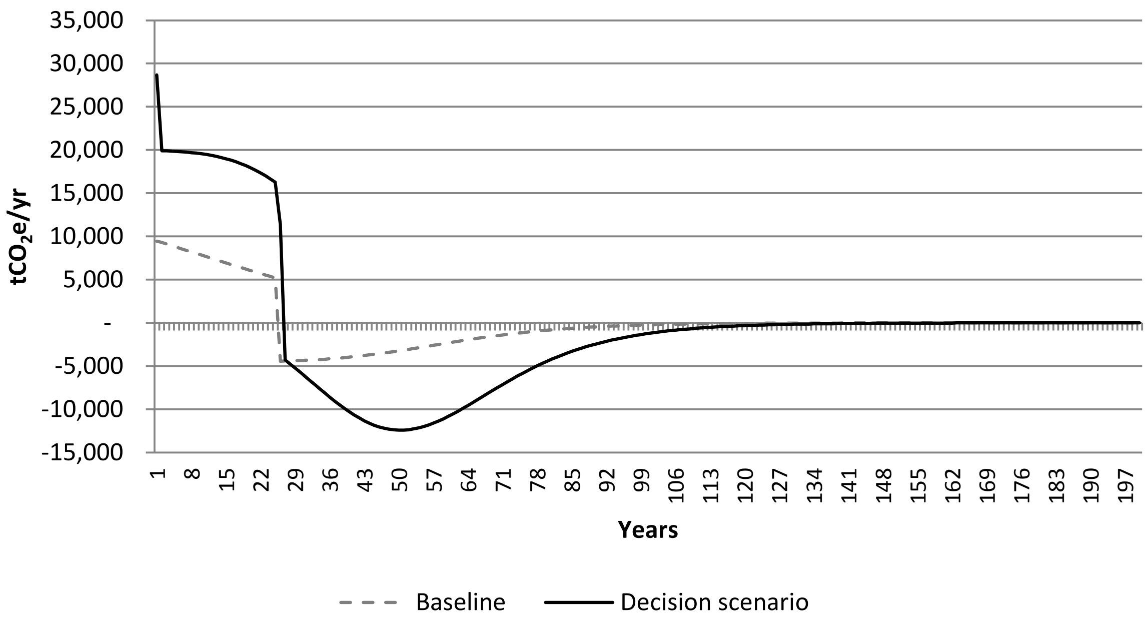


Table 10 below shows the results from the project/policy level method, including the emissions payback period for the scenarios that incur an initial carbon debt which is compensated for by subsequent reductions in emissions/enhancements in removals. The payback periods range between 1 and 103 years, and are determined by a number of factors such as the regrowth rate of the forest and the embodied emissions of the products displaced by the production of forestry co-products in the decision scenario (which is the reason for the outlier payback period of 1 year for cement render displacement in scenario 3.3).

Table 10. Net emissions and carbon payback periods from project/policy level method

Scenario	Sub-scenario	Net emissions from intervention (tCO_2e)	Emissions breakeven point (years)
1. Imports	1.1. Imports - sustainable forest management	- 76,082	75
	1.2. Imports - unsustainable forest management	506,108	NA
2. Local production	2.1. Local production without co-products	- 154,497	93
	2.2. Local production with co-products	9,213	NA
3. Thinnings	3.1. Thinning - without co-products	210,787	NA
	3.2. Thinning - with co-products (saw log displacement)	585,926	NA
	3.3 Thinning - with co-products (cement render displacement)	- 830,877	1
4. Fencing	4.1. Fencing - end of life combustion	- 92,673	56
	4.2. Fencing - end of life decay	- 150,556	58
5. Pallets	5.1 Pallets - displaced wooden pallets	468,510	30
6. MDF	6.1 MDF - displacing plasterboard	- 99,330	103
7. Particle board	7.1. Particle board - breeze block lower estimate	240,205	NA
	7.2. Particle board - breeze block upper estimate	392,272	NA

3.4. Comparison of the results from the different methods

Although the methods used tend to present their results using different metrics, Table 11 presents the results from the different methods using the common metric of total lifetime change in emissions in order to allow a direct comparison. The corporate inventory provides a single result as this method accounts for the emissions (including supply chain emissions) associated with the direct physical biomass combusted, and therefore does not model alternative scenarios for the marginal systems effected by the increased demand for biomass. It is also worth noting, as above, that the results for the consequential LCA and the project/policy method are largely the same, with small differences due to the use of temporally dynamic emission factors for the project/policy method. The corporate inventory indicates that the bioheat plant will reduce emissions, whereas the consequential methods show a range of possible outcomes, including possible increases in net emissions (the interpretation of which is explored in the Discussion (4.3)).

Table 11. Comparison of lifetime change results from the different methods

	Total change in emissions/removals (tCO_2e)		
Scenario	Corporate inventory	Consequential LCA	Project/policy method
1.1. Imports - sustainable forest management	-177,070	-72,538	-76,082
1.2. Imports - unsustainable forest management		509,653	506,108
2.1. Local production without co-products		-153,407	-154,497
2.2. Local production with co-products		7,745	9,213
3.1. Thinning - without co-products		212,158	210,787
3.2. Thinning - with co-products (saw log displacement)		704,276	585,926
3.3 Thinning - with co-products (cement render displacement)		-829,416	-830,877

4.1. Fencing - end of life combustion		-76,414	-92,673
4.2. Fencing - end of life decay		-134,298	-150,556
5.1 Pallets - displaced wooden pallets		469,691	468,510
6.1 MDF - displacing plasterboard		-98,149	-99,330
7.1. Particle board - breeze block lower estimate		241,386	240,205
7.2. Particle board - breeze block upper estimate		393,453	392,272

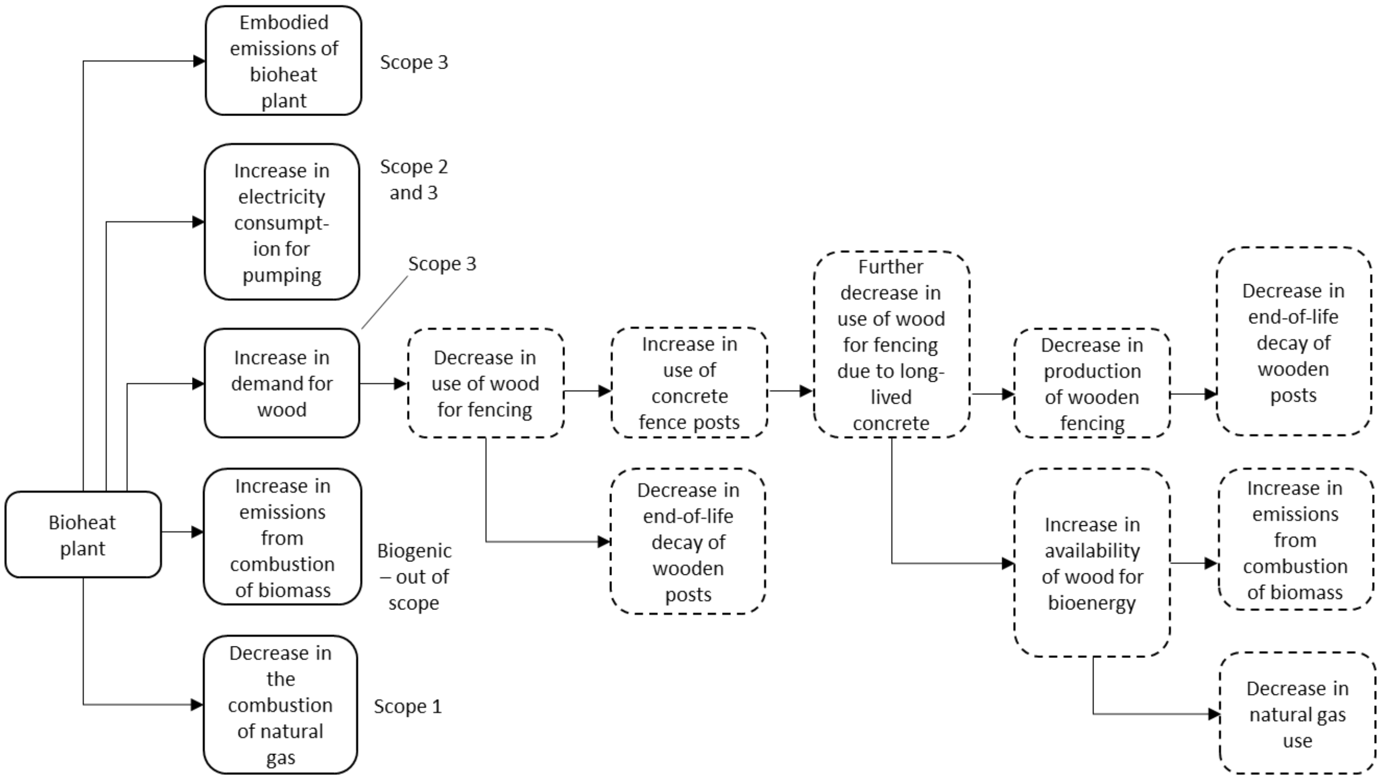
4. Discussion

The discussion in this section is structured around the following topics: the implications of the findings for attributional corporate greenhouse gas inventories (and attributional methods more generally); the relative merits of project/policy level assessment compared to consequential LCA; and the implications of the findings for the use of bioenergy as a climate change mitigation option.

4.1. Implications for corporate greenhouse gas inventories

A first point to note is that the corporate inventory method does not appear to be sufficient for informing decisions on climate change mitigation. By comparison with the consequential methods it is clear that the emission sources/sinks included in the corporate inventory do not reflect all the sources/sinks affected by the decision at hand. Figure 14 below presents the causal-chain map for scenario 4.2 (substitution of wooden fencing with concrete fencing, and assuming wooden posts would be combusted at the end-of-life) in order to illustrate the limited scope of the corporate inventory method. The emission sources/sinks indicated with the solid border are those included within the operational boundary of the corporate inventory (including all relevant scope 3 emission sources), and therefore the changes caused in the remaining sources/sinks in Figure 14 are not accounted for using the corporate inventory method. One exception to this situation is scenario 2.1. (local production with whole tree combustion), in which the sources/sinks included in the corporate inventory coincide with those identified by the consequential methods.

Figure 14. Causal-chain map for scenario 4.2



This limitation with corporate greenhouse gas inventories is recognised to some extent in the GHG Protocol *Corporate Standard* , which states that 'some companies may be able to make changes to their own operations that result in GHG emissions changes at sources not included in their own inventory boundary' (WBCSD/WRI 2004, p.61). However, the *Corporate Standard* also states that corporate GHG inventories 'provide business with information that can be used to build an effective strategy to manage and reduce GHG emissions' (WBCSD/WRI 2004, p.3) and that accounting 'for emissions can help identify the most effective reduction opportunities.' (WBCSD/WRI 2004, p.11), without the accompanying caveat that corporate inventories are not sufficient for capturing the total consequences of the reduction options under consideration.

The GHG Protocol *Corporate Value Chain (Scope 3) Standard* offers some additional clarification by stating that 'in some cases, GHG reduction opportunities lie beyond a company's scope 1, scope 2, and scope 3 inventories' and that accounting 'for avoided emissions that occur outside of a company's scope 1, scope 2, and scope 3 inventories requires a project accounting methodology' (WBCSD/WRI 2011b, p.107). In addition to the omission of biogenic emissions as part of the corporate inventory boundary (alongside scopes 1, 2 and 3), there is also no explicit recognition that company actions may also cause *increases* in emissions, as well as reductions, outside the corporate inventory, and there are many instances in the standard which imply that a scope 1, 2, and 3 inventory provides complete information for managing GHG emissions, e.g. 'increasingly companies understand the need to also account for GHG emissions along their value chains and product portfolios to comprehensively manage GHG-related risks and opportunities' (WBCSD/WRI 2011b, p.3), and a 'complete GHG inventory therefore includes scope 1, scope 2, and scope 3' (WBCSD/WRI 2011b, p.27).

The same presumption that a scope 1, 2, and 3 inventory provides complete information for decision-making is present in much of the academic literature on scope 3 (Minx et al. 2009; Downie & Stubbs 2013; Y. A. Huang et al. 2009). For instance Downie and Stubbs suggest, in their discussion of scope 3 emissions, that the application 'of HLCA [Hybrid life cycle assessment] methods has the potential to improve the validity of the respondents' GHGE [greenhouse gas emissions] assessments by ensuring they are comprehensive in capturing all relevant and material sources of emissions to the organization and removing the current subjectivity in emission source selection' (Downie & Stubbs 2013, p.162). However, the findings from the present study clearly demonstrate that even complete scope 1, 2, 3 (plus biogenic emissions) inventories do not capture all 'relevant and material sources of emissions to the organization'.

Broadening the implications of the findings, there is a strong parallel between the limitations with attributional corporate inventories and the limitations with attributional LCA. The emission sources/sinks included by both methods are based on normative boundary-setting rules, typically focused on direct physical connected-ness with the product life cycle or corporate value chain in question. This inventory boundary-setting approach means that not all of the emission sources/sinks that *change* as a result of a given decision are necessarily included within the inventory, and in such cases the inventory will not provide complete information on the consequences of the decision at hand. There is growing recognition within the life cycle assessment community of the limitations of attributional LCA for decision-making (Weidema 2003; R. J. Plevin et al. 2014b; R. Plevin et al. 2014), though attributional LCA is still highly embedded in existing practice and published standards (British Standards Institute 2011; European Commission et al. 2010; European Commission 2013). The findings from the present study show that the same limitations apply equally to corporate greenhouse gas inventories as to attributional LCA, and it appears likely that these limitations will also apply to *all* forms of attributional accounting, including national inventories and community inventories, and that the use of attributional inventories are not sufficient (on their own) to inform decisions on mitigation action.

4.2. Difference between consequential LCA and project/policy accounting

In contrast to the corporate inventory method, both consequential LCA and project/policy level assessment aim to quantify the total system-wide change in emissions caused by the decision at hand. Although both approaches reach broadly the same results (in terms of the magnitude of increase or decrease in emissions/removals for each scenario), they derive and present the results in different ways, and provide different amounts of information on the temporal distribution of emissions/removals.

One initially superficial difference, but which may obscure more significant issues, is the presentation of the results at either the unit or aggregate level, i.e. the consequential LCA results are in gCO_2e/kWh while the project/policy method shows total aggregate change in tCO_2e. This can be viewed as a superficial difference as either metric can be converted into the other, e.g. by subtracting the natural gas comparator figure from the unit level consequential LCA result and multiplying by the total delivered heat output of the plant (or the reverse for converting from the aggregate figure to the unit level). However, one potential shortcoming with focusing the analysis at the unit level is that non-linearities of scale are more likely to be missed, and despite the guidance to the contrary (Weidema et al. 2009), many consequential LCA studies do not state what the aggregate-level decision is assumed to be (Brander 2015b). Furthermore, presenting the results at the unit level may also create the misleading impression that the decision itself can be disaggregated, whereas, in the case of the bioheat plant, the decision only relates to the plant as a whole, and not to individual units of heat consumption.

Losing sight of the aggregate-level decision can also lead to the use of arbitrary amortisation periods, and therefore arbitrary aggregate output levels, for calculating unit-level results. If the unit-level results are to represent the change in emissions *caused by the decision* per unit of output *caused by the decision*, then the denominator must be based on the specific decision at hand, and not the amount of production occurring during an arbitrary or conventional amortisation period. In the case of the bioheat plant, the total expected output during a 25-year period is used, as this is the expected lifetime of the plant in question, and the production of heat during this period is amount of output caused by the decision.

Another seemingly superficial difference between the methods, but one which may also have more significant implications, is the differing structures in terms of baseline emissions/removals and decision-scenario emissions/removals. Consequential LCA results represent a combination of both decision scenario emissions/removals and credits for the avoidance of some baseline scenario emissions/removals. For example, the result of 125 gCO_2e/kWh for scenario 4.4 (displacement of wooden fencing) includes a credit for the displaced emissions from the end-of-life decay of the wooden fencing. Such results can then be compared to the consequential LCA results for other products, or if the product studied is replacing an alternative, then the total change in emissions is estimated by subtracting the results for the reference case from the results for the proposed substitute product.

This comparison of consequential LCA results for a reference case and a substitute product system is not straightforwardly equivalent to the comparison between baseline and decision scenario emissions/removals in the project/policy method, as discrete consequential LCA results represent a *mixture* of baseline and decision scenario emissions/removals, as noted above. One possible benefit of the project/policy method is that it is conceptually easier to understand. For example, the displacement of the end-of-life emissions from the wooden fencing is treated as a negative input to the product-system studied in consequential LCA (Weidema et al. 2009), but it is difficult to conceive of what a negative input is (Brander 2015b). In contrast, for the project/policy method, the displaced end-of-life emissions are simply included in the baseline, but do not occur in the decision scenario. Similarly, other effects that are awkward to accommodate in consequential LCA, such as foregone sequestration, rebound effects, and non 1:1 substitution ratios, can be straightforwardly modelled as differences between the baseline and decision scenario.

Turning to the issue of the distribution of emissions over time, conventional consequential LCA does not provide information on the temporal distribution of impacts, and moreover, is generally only concerned with quantifying normalised emissions for the long-run marginal system, based on the assumption that the long-run system will dominate the overall change caused by the decision in question (Weidema et al. 2009; Schmidt et al. 2015). In distinct contrast, the project/policy method provides a time-series of emissions/removals (illustrated in Figure 13), and this appears to constitute a major advantage over conventional static consequential LCA.

Firstly, information on the temporal distribution of emissions allows the calculation of the carbon payback period (for those scenarios that do eventually payback), which is highly decision-relevant given concerns about climate tipping points (Lenton et al. 2008) and the near and medium–term nature of most reduction targets (e.g. UK Government (2008), and European Commission (2015)). Secondly, the time-series approach allows temporally-specific emission factors to be applied to activity data. For example, in the present study the emission factors for road, rail and sea freight used in the project/policy method decline over time to reflect the expected increase in transportation fuel efficiency (and although this only makes a slight difference in the overall results, for other studies the difference could be considerable). Thirdly, the time-series approach allows the transition between different marginal systems to be modelled, e.g. the short, medium and long-term systems. For example, it is possible that the marginal system in the short-run will be increased production overseas (scenario 1) before transitioning to increased local production (scenario 2) as local capacity develops (Alexander et al. 2013). Although this transition modelling is not undertaken in the present study, the structure of the project/policy method has the inherent flexibility to allow such modelling, whereas consequential LCA does not. There is growing recognition within the LCA community for the need to include a temporal dimension to the method (Levasseur et al. 2010; Brandão et al. 2012; Collet et al. 2013; Schmidt et al. 2015), and a possible fast-track to achieving this would be to adopt the time-series structure from the project/policy approach.

Reverting briefly to the corporate inventory method, it is interesting to note that despite its other shortcomings this method does provide a partial time-series of emissions. However, corporate inventories tend to track the activities that occur in the inventory year, rather than the emissions/removals that occur in that time-period (WBCSD/WRI 2011b, p.32). For example, the total life-time emissions from landfilled waste are generally reported in the year that the waste is produced, rather than showing the distribution of emissions from the waste at the time that the emissions occur. Similarly, for some scope 3 emission sources, such as 'purchased goods and services' and 'fuel and energy related activities', attributional LCA emission factors are used to calculate emissions, and the non-temporally-explicit nature of attributional LCA is therefore imported into the corporate inventory. In the case of the 'fuel and energy related activities' for woody biomass, the attributional LCA emission factors published for corporate reporting (e.g. Defra/DECC (2015)) do not show the potentially long regrowth/sequestration period following the harvesting of the biomass.

4.3. Implications for bioenergy policy

The results from the consequential methods suggest that the case for the bioheat plant is not clear, and it is highly plausible that the decision to implement the plant will increase global CO_2e emissions rather than reduce them. Although considerable care is required in interpreting the results it is still possible to derive decision-relevant conclusions about the case for bioenergy.

However, before discussing the implications of the results, the following important caveats should be noted. Firstly, a large number of assumptions and modelling choices were made when implementing the consequential methods, and the selection of alternative parameter values will alter the results. Nevertheless, the findings from the sensitivity analysis (provided in the online supporting material) indicate that although the results for individual scenarios vary with alternative parameter values, the overall finding of large differences in the possible outcomes from the bioheat plant remains. Secondly, the range of scenarios tested is not exhaustive, and there are many other plausible scenarios that could be modelled (e.g. a scenario in which wind-blown trees are utilised, or in which increased demand for biomass increases tree planting (as suggested by Daigneault et al. (2012), Favero & Mendelsohn (2013), and Latta et al. (2013)). Thirdly, the results are presented for each individual scenario, whereas in reality there is likely to be a mix of marginal systems affected by the decision (Ekvall & Andræ 2006; Schmidt 2008; Mathiesen et al. 2009), and also a transition between combinations of scenarios over time. Fourthly, the relative probability of each scenario is not quantified, and it is not possible to infer that one scenario or outcome is more likely that another (although an initial review of the evidence suggests a strong case for increased overseas production). The development of further scenarios, and the estimation of probability should be the subject of further research.

Notwithstanding the numerous caveats with the consequential results it is still possible to draw substantive conclusions from the findings, especially when the range of possible outcomes is *itself* recognised as a key finding (Borjesson & Gustavsson 2000). Normative decision theory suggests that decision-making should be based on an understanding of the consequences of the decision in question (Lasswell & Kaplan 1950), while the results of this study, and numerous others, suggest that the emissions impact of bioenergy is unknown (Stephenson & MacKay 2014; Adams et al. 2013; Cherubini et al. 2009; Jonker et al. 2014; Lippke et al. 2011; Repo et al. 2014; Zanchi et al. 2012; Chum et al. 2011; Matthews et al. 2014; Marland & Schlamadinger 1997; Agostini et al. 2013). The situation can be characterised as one of Knightian uncertainty (Knight 1933), as the probability of the different possible outcomes are also unknown. Given the normative principle above, and the range of possible outcomes (with unknown probability of occurrence), it follows that it is not possible to justify the implementation of the bioheat plant (i.e. it is simply not known whether doing so will fulfil the aims of the decision-maker or not). To address this situation, one recommendation for future research is further exploration of the probabilities of the possible emissions outcomes (as suggested by Plevin et al. (2015)).

One decision-making strategy for dealing with situations of Knightian uncertainty is to adopt a 'maxi-min' strategy (Rapoport 1989), whereby the maximum possible loss from the decision is minimised. Given the possibility that the bioheat plant will cause large increases in emissions, alternative mitigation options that do not involve this possibility would be preferable. With this in mind, it would be useful to undertake similar consequential studies for alternative mitigation technologies, such as wind energy or ground-source heat, and to identify whether there are plausible scenarios in which these options increase emissions. If there are not, this would justify prioritising those options over bioenergy.

In addition to the above, the potentially long emission payback periods for the bioheat plant tallies with the findings of numerous other studies (Walker et al. 2010; Mitchell et al. 2012; Bernier & Paré 2013; Holtsmark 2012; Holtsmark 2013; Jonker et al. 2014; McKechnie et al. 2011; Pingoud et al. 2012; Schulze et al. 2012; Zanchi et al. 2012), and is highly relevant information to the decision at hand. The long emission payback periods entail that the bioheat plant may cause emissions to increase up to and beyond 2050, which is commonly used as the target year for reduction commitments (UK Government 2008; UK Government 2012; European Commission 2015), and may contribute to a climate tipping point before net emissions are reduced (Lenton et al. 2008).

A number of studies suggest that bioenergy does not create a carbon debt if a 'landscape' level of analysis is used, as the carbon stock of the whole forest estate will be relatively constant over time if it is sustainably managed, although the carbon stock of individual stands will change during the growth and harvesting cycle (Mitchell et al. 2012; Zanchi et al. 2012; Adams et al. 2013; Smith & Bustamante 2014). However, constant landscape-level carbon stocks are misleading as the relevant issue is whether those carbon stocks would have been higher (or lower) in the absence of the decision in question. Studies which take a properly consequential landscape-level approach still find a large carbon debt (e.g. Haberl et al. (2013)), which in some scenarios is never paid back (Hudiburg et al. 2011; Holtsmark 2013).

Although the present study focuses on the change in emissions/removals caused by the implementation of an individual bioheat plant, the key finding on the range of possible outcomes is expected to apply to any bioenergy installation using woody biomass, given the interconnected and global nature of the market for wood. One implication of this is that additional consequential assessments are not necessarily needed for each bioenergy installation within the market, as the marginal impact (or range of possible impacts) will be largely the same. This partly addresses the criticism that consequential analyses are too costly to implement (Rajagopal & Zilberman 2013)), as a single assessment may be broadly applicable to all decisions impacting the same market (Weidema 2003).

Similar findings to those from the present study are expected to apply at the level of government policy for bioenergy, where the system-wide impacts from bioenergy policies are also likely to be highly uncertain, and with long payback periods. As a further point, policy measures involving attributional supply chain reporting, such as that under the Renewable Energy Directive (European Parliament and Council of the European Union 2009)), are likely to be irrelevant for ensuring that bioenergy policies do not increase emissions, given that attributional methods do not capture the total system-wide impacts of the intervention studied.

5. Conclusions

Two main conclusions can be draw regarding greenhouse gas accounting methods. Firstly, conventional attributional corporate inventories, even with full scope 3 reporting, are not sufficient for supporting decision-making as they do not necessarily reflect the consequences of the decision in question. It is therefore recommended that existing greenhouse gas accounting standards and guidance clarify that corporate inventories should only be used for purposes such as emission reduction target setting, and that consequential methods must be used to assess possible mitigation options. The same limitations, and therefore the same recommendation, applies to the use of all forms of attributional accounting, including national greenhouse gas inventories, community-level inventories, and attributional product LCA.

Secondly, of the consequential methods studied, the project/policy method appears to have a number of advantages over consequential LCA, namely the transparent and conceptually simpler baseline and decision scenario structure, and the ability to show the distribution of impacts over time. There is already recognition within the LCA community of the need for dynamic modelling, and one option is to adopt the structure used in the project/policy approach. However, it is also worth noting that the consequential LCA literature includes numerous heuristics and techniques for identifying marginal systems, and the sharing of methodological lessons should very much be a two-way process.

A final conclusion concerns the justification for implementing bioenergy as a climate change mitigation option. The uncertainty of the emissions outcomes should *itself* be viewed as a decision-relevant finding from the present study, and further research should investigate the range of possible outcomes from alternative mitigation options, with preference then given to those without the potential for large undesirable outcomes. Furthermore, even in the scenarios where the bioheat plant achieves a net reduction in emissions, the payback period may extend beyond 100 years, thereby contributing to nearer-term cumulative emissions and a possible climate tipping-point, as well as making the intervention irrelevant to near and medium-term reduction targets.

Paper 4 - Response to “Attributional life cycle assessment: is a land-use baseline necessary?” – Appreciation, renouncement, and further discussion.

Abstract

Purpose

Soimakallio et al. (2015) establish the need for baselines in attributional life cycle assessment (LCA), and thereby provide an important milestone in the evolving conceptualisation of both attributional and consequential LCA. The purpose of this commentary is to: acknowledge Soimakallio et al.’s contribution; identify its implications for a number previously published papers; critique the use of natural regeneration baselines; and offer some further thoughts on the conceptual nature of attributional and consequential approaches.

Methods

Comparative analysis with other forms of attributional inventory, and an illustrative example of alternative ‘natural’ baselines for carbon sequestration.

Results and Discussion

The commentary concurs that attributional LCA requires baselines, and that attributional studies are not inventories of absolute emissions and removals, contrary to previous statements by the present author. Nevertheless, a number of previous statements on attributional and consequential methods remain largely unchanged: attributional studies can be aggregated to approximate total (anthropogenic) environmental impacts; substitution is conceptually inappropriate for attributional LCA; and the attributional-consequential distinction can be applied to other forms of environmental assessment such as national, corporate, and community greenhouse gas inventories (attributional), and project and policy-level greenhouse gas assessments (consequential). A further finding is that natural

regeneration baselines may not be appropriate for attributional studies, and that some arguments in their favour may be symptomatic of a misconception of attributional LCA.

Conclusions

Soimakallio et al. (2015) make an extremely useful contribution to the evolving conceptualisation of attributional and consequential approaches, which is highly important for methodological development and choosing the appropriate method for a given purpose.

Key Words

Attributional life cycle assessment; consequential life cycle assessment; baselines; substitution; national greenhouse gas inventories; sequestration; decision-making

1. Introduction

There is a lively and on-going debate within the life cycle assessment (LCA) community on the conceptual nature and relative merits of attributional and consequential LCA (R. J. Plevin et al. 2014b; Brandão et al. 2014; Hertwich 2014; Suh & Yang 2014; Anex & Lifset 2014; Dale & Kim 2014; R. J. Plevin et al. 2014a; R. Plevin et al. 2014). Clarifying the conceptual nature of these methods is highly important as it directly affects methodological issues, e.g. whether substitution should be used in attributional LCA, and the appropriate use of each approach, e.g. whether attributional LCA is sufficient to support decision-making.

The LCA community has a strong track record in conceptual debate and innovation, exemplified, not least, by the development of the attributional-consequential distinction itself (Curran et al. 2005). It is also worth noting that such innovations can be highly relevant and useful to other fields of environmental accounting beyond LCA (Brander 2015b), and this enhances the significance of the debate further. Soimakallio et al. (2015) is an important milestone in the evolving conceptualisation of attributional and consequential methods, and although the analysis presented below finds both agreement and disagreement with different aspects of the paper, all aspects help to bring further clarity to the conceptual nature of the attributional-consequential distinction.

The focus of the present paper is on greenhouse gas emissions, but the discussion applies equally to any other environmental flow or impact category.

2. Agreement on Attributional Baselines

Soimakallio et al.'s assertion that a baseline is needed in attributional LCA in order to separate out anthropogenic activities (the technosphere) from natural or non-anthropogenic processes (the ecosphere) appears to be fundamentally correct, i.e. it is not possible to achieve this separation without the use of a baseline. Additional support for this assertion can be provided through a comparison with national greenhouse gas inventories under the UNFCCC (United Nations 1992), which may be characterised as another form of attributional account (Brander 2015b). National greenhouse gas inventories are inventories of anthropogenic emissions and removals, and baselines are used, albeit implicitly, to separate out anthropogenic from non-anthropogenic emissions/removals. For example, the 'managed land proxy' assumes that all emissions/removals on unmanaged land would occur anyway in nature, i.e. they are part of the non-anthropogenic baseline (IPCC 2006; WRI 2014b). If the use of such baselines is appropriate for *attributional* national greenhouse gas inventories, it can be inferred that they are similarly appropriate for *attributional* LCA.

Soimakallio et al. are also entirely correct to conclude that, because of the use of baselines, attributional LCAs are *not* inventories of absolute (observable) emissions and removals, contrary to previous statements made by the present author (Brander et al. 2009; Brander & Wylie 2012; Brander 2015b). Soimakallio et al. also correctly diagnose one of the reasons for conceptualising attributional methods as inventories of absolute impacts, i.e. natural baseline emissions from fossil fuel combustion are generally zero, and therefore anthropogenic and absolute emissions will tend to be identical. The correct conceptualisation of attributional methods should be as inventories of *anthropogenic* environmental impacts relative to a natural baseline, rather than *absolute* environmental impacts.

3. Scope of Renouncement

Although the characterisation of attributional methods as inventories of absolute environmental impacts should be renounced, it is important to note that a number of previous statements on the conceptual nature of attributional and consequential approaches require only slight restatement, or remain unaffected.

Firstly, the idea that '[attributional] LCA results of all the products in the world should add up to the total environmental impact in the world' (Tillman 2000, pp.116–117)) remains true, though it must be clarified that the total is for total *anthropogenic* impacts and not total *absolute* impacts. The truth of this idea can also be seen by considering the aggregation of all national greenhouse gas inventories, which, in principle, approximate to total global anthropogenic emissions and removals (excepting ad hoc exclusions such as international aviation and shipping, military activities etc.). This conception of attributional methods is also consistent with the idea that attributional LCA attributes 'portions of the total pollution and resource consumption flows occurring from the economy as it is at a given point in time to each existing product life cycle' (Soimakallio et al. 2015).

Secondly, the *inappropriateness* of substitution as a methodological technique within attributional LCA remains true (Brander & Wylie 2012). The baseline used in attributional LCA is for *non-anthropogenic* environmental impacts, for the purpose of separating out anthropogenic from non-anthropogenic impacts. In contrast, the baseline implicit within the technique of substitution is for alternative *product systems* (i.e. anthropogenic systems) which are displaced by the co-products or multiple functions of the system studied. If substitution is used in attributional LCA it will not provide an inventory of *anthropogenic* environmental impacts relative to a natural baseline (and in addition, the sum of all attributional LCAs will not approximate total anthropogenic impacts).

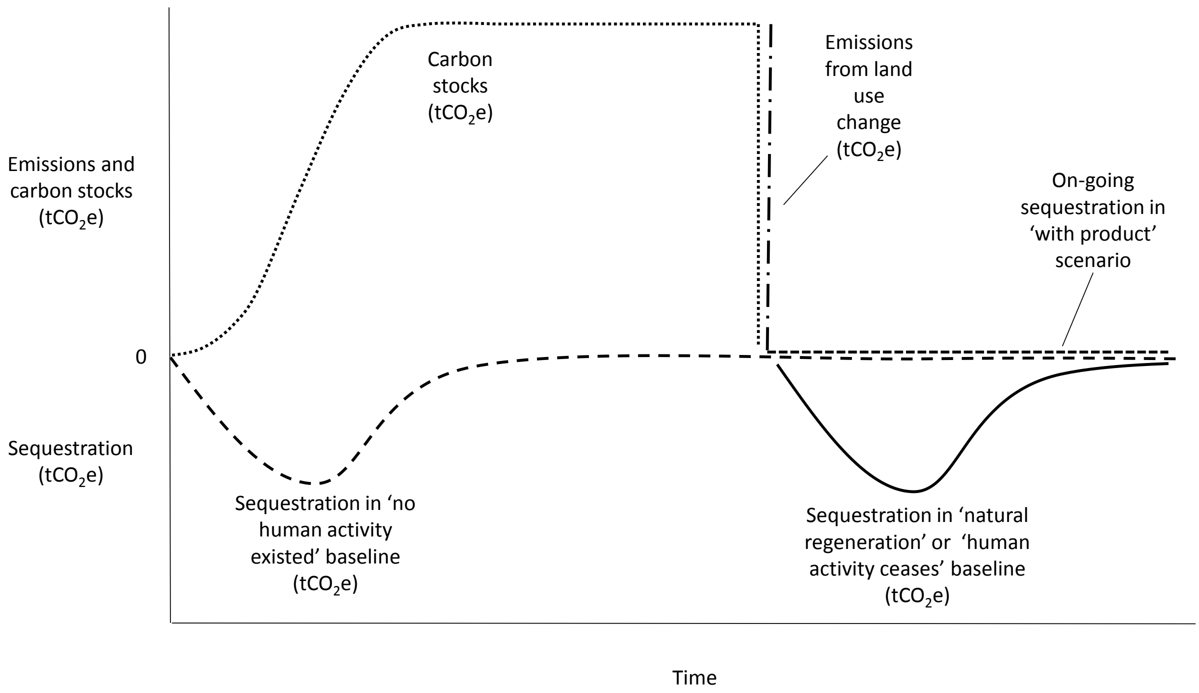
Thirdly, the categorisation of all existing greenhouse gas accounting methods as either attributional or consequential in Brander (2015b) remains unchanged. National greenhouse gas inventories (IPCC 2006), corporate inventories (WBCSD/WRI 2004; ISO 2006c), and community inventories (Schultz et al. 2014; British Standards Institute 2013) are all inventories of anthropogenic emissions and removals (i.e. they are attributional), and project (WBCSD/WRI 2005; ISO 2006d) and policy-level (WRI 2014c) accounting both aim to quantify the total system-wide change in emissions caused by a given decision or action (i.e. they are consequential). Recognition of these ‘families’ of methods, with shared conceptual underpinnings, is useful for transposing methodological innovations and practices between methods. For example, the use of non-anthropogenic baselines within national greenhouse gas inventories under the UNFCCC appears to be relatively limited, i.e. to the treatment of emissions/removals from *unmanaged* land (though guidance for Kyoto Protocol reporting contains other instances of baselines, e.g. natural disturbance baselines for forest land (IPCC 2013b)); innovations within ALCA, such as accounting for foregone sequestration on *managed* land, may also be relevant to national inventories.

4. Appropriateness of Natural Regeneration

Soimakallio et al. (2015) suggest that natural regeneration is the most coherent baseline for attributional LCA, however, the following illustrative example and discussion raise a number of potential problems.

Firstly, consider two of the possible options for the natural baseline for carbon sequestration: a. the level of sequestration that would have occurred in the absence of all human activity; and b. the level of sequestration that would occur if all human activity ceased (i.e. natural regeneration). Figure 15 provides a schematic diagram showing these alternative possible natural baselines (with positive sequestration shown as a negative number), as well as terrestrial carbon stocks and emissions from land use change.

Figure 15. Two possible options for the natural baseline for carbon sequestration



To provide a brief description of this illustrative example: At some point in the past terrestrial carbon stocks accumulated as the ecosystem sequestered CO_2 from the atmosphere; 2. An equilibrium carbon stock was reached and the rate of on-going natural sequestration declined to zero; 3. Following this, anthropogenic land use change occurred, e.g. forest land was converted to agricultural use; 4. The continued cultivation of the land means that it does not revert to a natural state, and the on-going rate of sequestration remains at zero.

If the 'no human activity existed' baseline is chosen, this has the same level of on-going sequestration as the 'with product' scenario (i.e. zero), and so there would be no foregone sequestration from continued land occupation. In contrast, if the 'natural regeneration' baseline is chosen, this indicates an amount of on-going baseline sequestration, and therefore there will be some foregone sequestration from continued anthropogenic land use. The key question is which is the most appropriate baseline for attributional LCA?

One argument against the ‘natural regeneration’ baseline is that it is, in fact, an artifice of human activity, i.e. the potential for natural regeneration and on-going sequestration only exists because *anthropogenic* land use change has reduced terrestrial carbon stocks below their equilibrium level. If attributional LCA is to separate out anthropogenic impacts from background natural flows, then the natural baseline should not itself include anthropogenic activities (such as land use change), or as Soimakallio et al. put it ‘including parts of the technosphere in the baseline is against the fundamental purpose of ALCA’ (2015, p.1371). Essentially, the ‘natural regeneration’ baseline is not *natural*, as it is created by anthropogenic activities. Such a baseline may be appropriate for consequential LCA, where the baseline may include anthropogenic activities (as is the case with the technique of substitution).

A further problem with the natural regeneration baseline is the issue of counting the same foregone sequestration in perpetuity. If the land is maintained in agricultural use, the question of foregone sequestration is ever-present for each successive product-system utilising the land, although the amount of actual foregone sequestration is bounded by the equilibrium carbon stock. Unless there is a way of allocating the foregone sequestration across all future production from the land, the same foregone sequestration may be double-counted *ad infinitum*. For example, supposing that natural regeneration on an area of land would sequester an average of 1tCO_2 per year for 20 years (at which point an equilibrium carbon stock is reached), then products produced from the continued anthropogenic use of the land during those 20 years will incur a total of 20tCO_2 of foregone sequestration. If the products from the next 20 years of continued land use are also allocated 20tCO_2 of foregone sequestration, it will be the same 20tCO_2 allocated twice, and so on.

One of the arguments cited in favour of a natural regeneration baseline, drawn from Milà i Canals et al. (2007), is that ‘land occupation postpones natural regeneration of the land, which is an impact that needs to be accounted for' (Soimakallio et al. 2015, p.1369). Although the postponement of natural regeneration is certainly a *consequence* of continued land occupation, if it is not an anthropogenic impact *relative to a natural baseline* (as argued above), then it simply does not belong in an ALCA. There appears to be an impulse to make ALCA capture the total consequences of an activity, possibly in recognition of the principle that decision-making should be based on an understanding of the total consequences of the decision at hand. However, this is properly the purpose of consequential LCA (which captures all impacts by effectively using a baseline that reflects what would happen in the absence of the decision, whether anthropogenic or non-anthropogenic), and not attributional LCA (which only inventories anthropogenic impacts relative to a non-anthropogenic baseline).

A similar misconception may be present in Milà i Canals et al. (2013), where there is concern that attributional LCA results may lead to perverse outcomes, such as incentivising deforestation rather than continued land occupation. However, perverse outcomes are to be wholly expected if attributional LCA is used (on its own) for decision-making, precisely because the method does not necessarily capture the total impacts of the decision at hand (R. J. Plevin et al. 2014b). The solution is to recognise that attributional methods, by their very nature, are not *sufficient* for decision-making (on mitigation actions). In contrast, if a consequential method were used to inform the decision between deforestation and continued land occupation, it would, in theory, identify the option with the lower overall impacts – precisely because the method is intended to capture the total consequences of the decision at hand.

Arguably, the correct use of attributional LCA is for applications such as: assigning responsibility for the on-going management of a set of impacts (e.g. as is the case with national greenhouse gas inventories under the UNFCCC); target setting (e.g. setting percentage reductions relative to a base year inventory); and budgeting for total anthropogenic impacts (e.g. setting carbon budgets to ensure total anthropogenic emissions do not exceed an aggregate threshold (and for this purpose the *additivity* of attributional inventories, described in Tillman (2000), is essential)). However, it is important to reiterate that any *decisions* or *actions* aimed at mitigating inventory impacts or meeting reduction targets should be assessed using *consequential* methods to avoid unintended consequences which are not captured in the attributional inventory.

5. Conclusions

Soimakallio et al. (2015) make an extremely useful contribution to the evolving conceptualisation of attributional and consequential methods, both in terms of what they get right and what may not be entirely right. The proper conceptualisation of attributional and consequential methods is highly important for ensuring that the appropriate method is used for a given purpose. Attributional methods are inventories of anthropogenic impacts relative to a natural baseline, and should be used for assigning responsibility for managing those impacts, target setting, and environmental budgeting. However, any mitigation actions must be informed by consequential methods, which seek to describe the total system-wide consequences of the decision at hand. Although this conceptual debate has occurred largely within the field of life cycle assessment, the conceptual apparatus of the attribution-consequential distinction is highly applicable to other forms of environmental accounting, such as national greenhouse gas inventories – which extends the importance of this debate still further.

Conclusions

This concluding chapter is structured in the following way: Section 1 reflects on the overarching themes of the research, the evolving understanding of the attributional-consequential distinction, and provides a summary of the outputs from the research in relation to the research questions; Section 2 discusses a number of limitations and potential criticisms of the research, and possible responses; Section 3 presents a number of ideas for further research, to illustrate the wide range of research opportunities that might be enabled or motivated by using the attributional-consequential distinction as a categorical framework; Section 4 outlines some of the routes to broader societal impact, in fulfilment of the practical motivations for the research; and finally, Section 5 provides some brief concluding remarks

1. Overarching Themes, Findings, and Evolving Understanding

As discussed in the Introduction, the individual papers within this portfolio were written at different points in time, and therefore represent an evolving understanding of the attributional-consequential distinction. Rather than edit the statements in the earlier papers to align with the current position, those statements have been purposefully maintained on the basis that more can be learned from reflecting on those mistakes than by providing a cleansed final product. This section therefore provides a reflective discussion on this evolved understanding, and also attempts to adopt an overarching perspective on the portfolio to summarise the outputs and contribution of the research, and to identify a number of findings that are otherwise dispersed across the papers. This discussion is structured around the research questions, which also affords the opportunity to explicitly revisit those questions and to discuss the answers that the research has been able to provide.

Question 1 asks ‘What is the attributional-consequential distinction and what is its applicability to other forms of GHG accounting?’. The first part of this question, ‘What is the attributional-consequential distinction?’ is initially addressed Paper 1 (Section 2.2), in which, based on the review of the definitions in the LCA literature, the following two key aspects of attributional and consequential methods are proposed: firstly, consequential assessments are concerned with change, while attributional assessments are for absolute quantities of environmental impacts for a static state; and secondly, consequential assessments are concerned with system-wide change, whereas attributional assessments are only concerned with impacts occurring within a defined inventory boundary.

There are a number of elements to this initial answer that could now be restated. Firstly, Paper 4 accepts the argument made by Soimakallio et al. (2015) that attributional LCAs generally aim to provide an inventory of anthropogenic impacts, and are therefore not necessarily inventories of *absolute* (observable) impacts, contrary to the suggestion in Paper 1. Paper 4 goes on to suggest that attributional inventories are ‘inventories of *anthropogenic* environmental impacts relative to a natural baseline, rather than *absolute* environmental impacts’ (Brander 2015a, p.1608). However, on reflection (and with full acknowledgement given to comments made by Francisco Ascui), this suggestion appears to be overly restrictive, and should itself be subject to revision. Although most, if not all, existing attributional methods do aim to provide an inventory anthropogenic impacts it is entirely possible to create an inventory of either total impacts, or only non-anthropogenic impacts, and therefore a more extensive definition of attributional methods is that they are inventories of impacts within a defined inventory boundary.

A further revision to the defining characteristics identified in Paper 1 is based on a more refined account of the notion of ‘change’, and the form and role it can play in attributional accounting. Paper 1 implies that attributional inventories represent a static state and are not concerned with change, whereas attributional methods may in fact measure change in the inventory relative to a base year (WRI 2014b), or relative to a business-as-usual baseline, which represents what the inventory would have been in the absence of some specified action or intervention (WRI 2014b), or relative to a natural baseline, in cases where only anthropogenic impacts are intended for inclusion in the inventory (Soimakallio et al. 2015). The overly restrictive statement in Paper 1 was motivated by the fact that it is possible to produce a single standalone attributional inventory, e.g. a single year inventory of absolute impacts, without any reference to change. In contrast, it is not possible to undertake a consequential assessment without reference to some specified change, i.e. decision or intervention, as the change in question determines the sources/sinks included in the assessment boundary. A more refined articulation of the point made in Paper 1 is that change is an essential characteristic of consequential methods, whereas it is a non-essential characteristic of attributional methods. In answer to the first part of Question 1, the proposed definitions and the key defining characteristics of the attributional-consequential distinction are restated in Table 12 below.

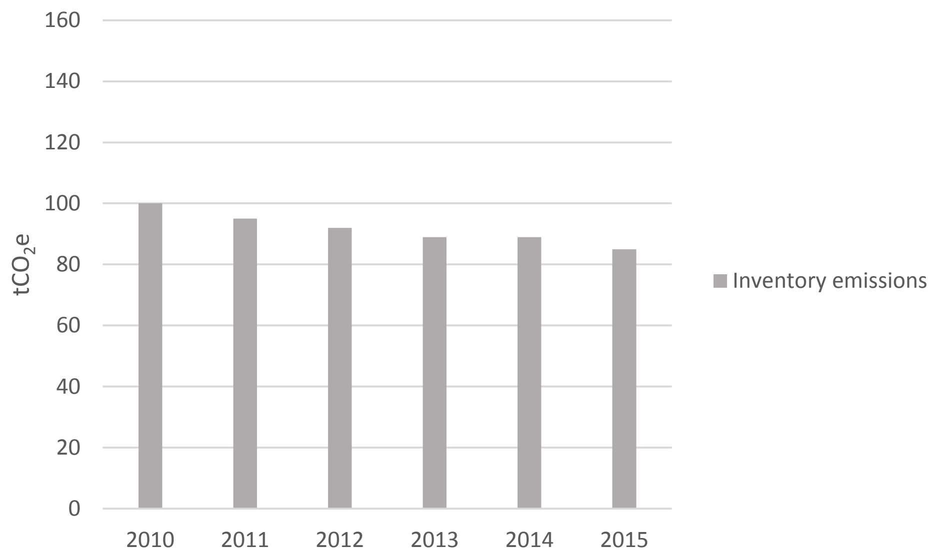
Table 12. The defining characteristics of attributional and consequential approaches

	Attributional	**Consequential**
Proposed definitions	An assessment of impacts within a defined inventory boundary.	An assessment of the system-wide change in impacts caused by a specified decision or action.
Relationship to change	Change is a non-essential element of an attributional approach, but can be included in a number of different forms: • Change in the inventory relative to a base year. • Change in the inventory relative to a business-as-usual baseline. • Change in impacts relative to a natural baseline, in order to separate anthropogenic from non-anthropogenic impacts.	Change is a necessary element in the method as the subject of a consequential assessment is a specified decision or intervention, and change defines which sources/sinks are included in the assessment boundary.
Assessment boundary	• The inventory boundary is determined by a rule or convention (often based on	• The assessment boundary is determined by the sources/sinks that change as a result of the

	some form of physical connectivity or some notion of responsibility).	decision or action in question.

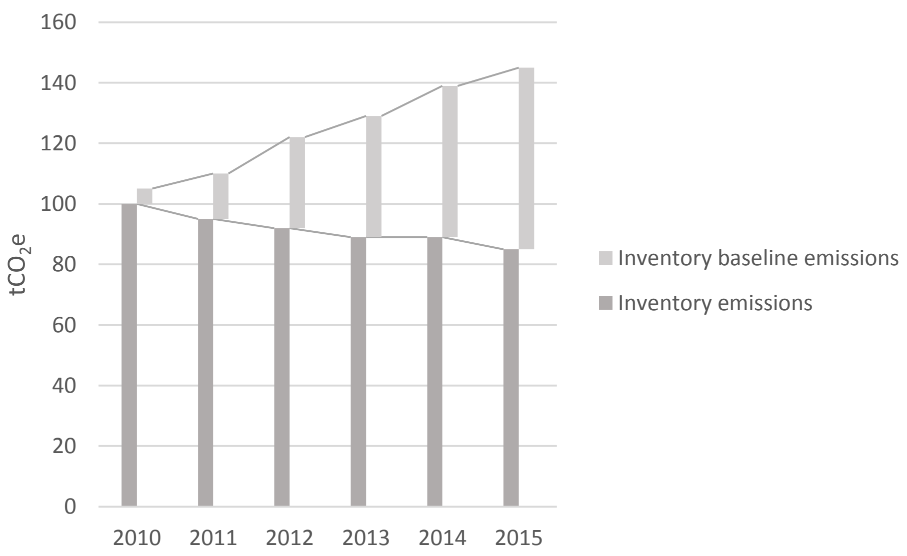
In order to provide further clarification of the attributional-consequential distinction, in answer to Question 1, it may be helpful to describe the necessary steps for transitioning from an attributional inventory to a consequential assessment of change. This transition can be shown graphically, and involves two distinct steps, based on the defining characteristics of the methods. Figure 16 provides an illustration of a generic attributional inventory (representing the general structure of any national, community, corporate or attributional product inventory).

Figure 16. Illustration of attributional inventory



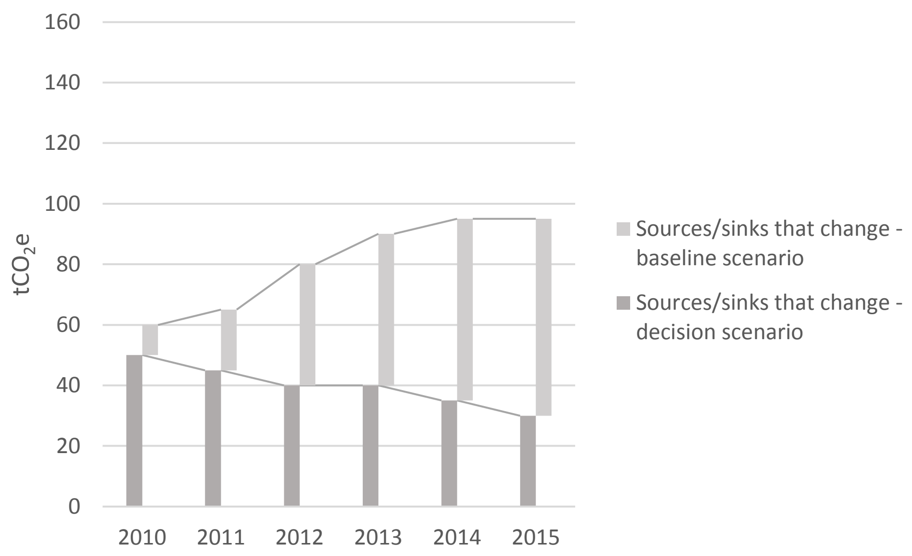
The first necessary step for transitioning from an attributional inventory to a consequential assessment is to identify a decision or action, and to determine what the level of inventory emissions/removals would be in the absence of that decision, i.e. the baseline (as was done in Paper 3 for the corporate inventory without the bioheat plant). Figure 17 illustrates this first step, and indicates that in this example the baseline emissions would have been higher than those in the extant inventory. As an aside, Figure 17 is also useful for illustrating two of the distinct forms of change that can be expressed using an attributional inventory. One form of change is the comparison with a base year, or the change in the inventory over time, e.g. between 2010 and 2015. However, as noted above, it is possible to undertake a single stand-alone attributional inventory, e.g. just for 2010, in which case this form of change would not be present. A second form of change is that between the decision-scenario and the business-as-usual baseline scenario, as illustrated in Figure 17.

Figure 17. Illustration of an attributional inventory with a business-as-usual baseline.



The second necessary step for transitioning from an attributional inventory to a consequential assessment is that the sources and sinks included in the decision-scenario and baseline inventories must include all those that change as a result of the decision in question, in order to ensure that total system-wide change is captured. Figure 18 illustrates the way in which a large proportion of the initial attributional inventory sources may be excluded from the assessment if they do not change as a result of the decision in question, and also that the business-as-usual baseline emissions may increase in size, if there are additional baseline sources that change, but were not included in the initial attributional inventory. The key point is that the sources/sinks included in the assessment are determined by whether they change as a result of the decision studied, and not because they are stipulated by the attributional inventory boundary-setting rule. The overall structure of the decision-scenario and business-as-usual baseline emissions/removals shown in Figure 18 mirrors the generic structure for the project/policy level method presented in Figure 6 in Paper 2, and Figure 8 in Paper 3.

Figure 18. Illustration of decision-scenario and baseline inventories for sources that change



Moving on to the second part of Question 1, this asks ‘what is its [the attributional-consequential distinction’s] applicability to other forms of GHG accounting?’. This is partly addressed in Paper 1 with respect to corporate greenhouse gas inventories, and is answered more comprehensively in Paper 2 through the document analysis exercise, and categorisation of existing greenhouse gas accounting standards and guidance as being either attributional or consequential. Attributional greenhouse gas accounting methods include national, community/city, and corporate inventories, and attributional LCA, and consequential forms of greenhouse gas accounting include the project-level and policy-level methods, and consequential LCA. It is worth noting that even with the updated characteristics of the attributional-consequential distinction, discussed above, the categorisation in Paper 2 remains unchanged.

Question 2 asks 'What are the different forms of *consequential* greenhouse gas accounting method, and what methodological lessons might be shared between them?'. In answer to the first part of this question Paper 2 identifies the different forms of consequential greenhouse gas accounting as consequential LCA, project-level assessment, and policy-level assessment. In answer to the second part, the main methodological lessons that can be shared are the time series and transparent baseline-decision scenario structure from the project/policy method to consequential LCA. In terms of the novelty of these outputs, it is worth highlighting that although the idea of a family of consequential methods has been suggested previously (Brander & Wylie 2012), it has not been demonstrated using a systematic review of existing greenhouse gas accounting standards and guidance documents. Moreover, the comparison and transposition of lessons between different forms of consequential method appears to be wholly novel, and has not been undertaken elsewhere.

Question 3 asks 'Do attributional inventories and the different consequential methods provide different results, and what are the implications for decision-making?'. Paper 3 answers this question by providing an empirical illustration of the conceptual or methodological differences discussed in Papers 1 and 2. This largely follows the research model commonly used within the life cycle assessment literature, i.e. the use of an empirical case to demonstrate a methodological idea or principle (Finnveden & Ekvall 1998; Ekvall & Andræ 2006; Thomassen et al. 2008; Dalgaard et al. 2008; Chalmers et al. 2015). Although the empirical results for the difference between the corporate inventory and the consequential comparators are largely to be expected given conceptual/methodological differences between the methods, the empirical outputs are important for countering the view that attributional methods are 'good enough' for decision-making, or provide a reasonable proxy for system-wide impacts. This view is still evident in existing greenhouse gas accounting standards, e.g. *ISO 14044* (ISO 2006b), *ISO 14067* (ISO

2013b), the GHG Protocol's *Value Chain (Scope 3) Standard* (WBCSD/WRI 2011b) etc., all of which imply that attributional inventories are sufficient for decision-making.

As discussed in the Introduction, and Paper 3 itself, the empirical case study does not provide information on the *probability* of a material difference between attributional and consequential methods, and the case is not intended to be representative of all mitigation decision scenarios. Nevertheless, the empirical case is sufficient for showing that the difference between the methods is not always trivial or immaterial, and it is therefore necessary to use a consequential method to ensure there are no unintended system-wide consequences, and to ensure that the most cost-effective mitigation measures are selected.

Following the novelty of the conceptual comparison between the different forms of consequential method in Paper 2, Paper 3 also appears to be the first study to demonstrate empirically the important benefits of the project/policy method compared to consequential LCA, particularly the benefits of the baseline-decision scenario structure and the temporal distribution of impacts.

A further novel contribution from Paper 3, which does not appear to be present in comparable LCA studies that contrast attributional and consequential methods, is the exploration and interpretation of uncertainty. Existing studies tend to present single output results for attributional and consequential methods (e.g. Ekvall & Andræ (2006)), or include consideration of alternative co-product allocation rules for attributional LCA (e.g. Thomassen et al. (2008)), but either do not show the range of possible consequential results from alternative scenarios, or do not interpret the range as a decision-relevant finding in its own right. Paper 3 suggests that the range of possible outcomes revealed by a consequential method, assuming scenario analysis is used, reflects our state of knowledge about the outcomes of the decision in question, and therefore our ability to justify the decision.

This interpretation of uncertainty stands in contrast to that in Buchholtz et al. (2014), where it is asked ‘at what point does a level of uncertainty rule out a baseline’s usefulness?’. The answer is that the level of uncertainty *is* the useful information, and is highly relevant for informing decision-making. Similarly, Herrmann et al. (2014) develop a classificatory matrix for identifying the trade-offs between the inherent uncertainty of consequential assessments and the limited scope of attributional alternatives, but again, the uncertainty revealed by consequential assessments is not recognized as decision-relevant information in its own right.

The discussion on decision theory in Paper 3 is necessarily brief given the range of other issues addressed in the paper, but is nevertheless sufficient for making the key point that uncertainty is decision-relevant information. Decision theory also offers a number of further concepts and strategies for responding to conditions of uncertainty or ignorance, and exploring their applicability to bioenergy or other climate change mitigation options could be a subject for further research. For example, the ‘mini-max’ strategy (distinct from the maxi-min strategy mentioned in Paper 3) aims to minimize the maximum regret from a decision (Hansson 2005), which may be applicable to bioenergy if it offered the possibility of very large mitigation opportunities that would be foregone due to an overly precautionary attitude to possible increases in emissions. The distinction between unknown *probabilities* and unknown *possibilities* in decision theory (Hansson 2005), also appears to be a useful one, as it helps to characterize the scenario analysis in Paper 3, in which the *probability* of each scenario occurring is not known, and neither are the scenarios modelled exhaustive of all the *possible* scenarios that could occur.

In addition to decision theory, there is also a significant body of work on risk and uncertainty within the social research literature. For example, Beck’s seminal *The Risk Society: Towards a New Modernity* argues that the scientific and technological advances that characterize modern society create new forms of ‘manufactured’ risk, such as the risks created by industrial-scale pollution (Beck 1992). The same scientific and technological advances also enable the quantification, and to some extent the management, of risk, in ways wholly distinct from pre-modern society (Spira & Page 2003). In turn, the resulting increase in attention given to risk in modern society creates opportunities for the appropriation of risk management by specific interest groups, and so becomes a source of power (Power 2004; Spira & Page 2003). This literature also suggests that the growing prominence of risk and risk management may diminish the notion of responsibility, either through the use of risk management as a way of deflecting blame (Power 2004), or because the ‘consequences of risk are likely to result from a complex chain of events and circumstances’ and blame-placing ‘in this context becomes problematic’ (Spira & Page 2003, p.644).

Many of these themes and ideas appear to be highly applicable to the bioheat case study, or for explaining the varying use of attributional and consequential methods more generally. For instance, the globalization of trade, enabled by modern technology, helps to create the complex system of market-mediated effects explored in the bioheat case study, that would not have existed to the same extent in the pre-modern era. Relatedly, the complexity of the ‘chain of events’, e.g. those created by biomass demand, may well undermine the ascription of responsibility for the consequences of decisions, and may partly explain the lack of awareness of possible system-wide impacts by the organization commissioning the bioheat plant. It may also possible to suggest that attributional methods are something of a hangover from the pre-modern era, in which such methods might have adequately captured the consequences of decisions in simpler, more localized, systems.

Generally the themes and ideas explored in the social research literature are used to provide explanations or social critiques of existing accounting practice, rather than to engage in normative method development, which is the primary focus of

Paper 3. The use of social theory for *explaining* the accounting practices of the organization commissioning the bioplant are discussed further below, in Section 3 on ideas for further research. However, it is important to note that there are also instances within the social research literature that come closer to normative method development, such as the observation that if 'modern scientific methods cannot prove that something is absolutely true, the absence of proof can be used to deny the existence of an issue' (Bebbington & Thomson 2007, p.48). This observation can be used to explain the view illustrated by Buchholtz et al. (2014) above, i.e. that uncertainty is a valid reason for discounting an issue, but the observation also appears to carry the implicit normative judgement that the absence of absolute proof is not a good justification for ignoring a problem.

It is perhaps worth emphasizing that generating uncertain results is not in itself desirable, in the sense that our methods should aim to be uncertain, but rather that if our best efforts to understand the outcomes of a decision reveal our ignorance of those outcomes, then *that* finding should be recognized as one that is highly decision-relevant. It is perhaps also worth noting that attributional methods can incorporate some forms of uncertainty, such as parameter uncertainty for the input values used, but cannot reflect the scenario or model uncertainty associated with alternative marginal systems, as illustrated by the different marginal systems modelled in Paper 3, as attributional methods only model the prescribed sources/sinks included within the attributional inventory boundary.

An important corollary or sub-plot to Questions 1, 2 and 3 is the exploration of the potential *benefits* of developing the attributional-consequential distinction as a categorical framework. The discussion and findings related to this theme are dispersed across the papers, for example Paper 1 discusses the potential benefits for corporate greenhouse gas accounting, and Paper 2 identifies lessons between the different consequential methods, but there is not, in the papers themselves, a systematic overview of these potential benefits. Furthermore, the development of the attributional-consequential distinction as a categorical framework, following Denzin's (1970) account of forms of theory, is the primary theoretical contribution of the thesis, and it is therefore worth reflecting on the value or practical relevance of this output.

At a very general level, categories or categorical frameworks are necessary for conceptualising discrete particulars, or dividing phenomena into thinkable content (Kant 1996), i.e. they are prerequisites for cognition:

> Thoughts without intuitions are empty, intuitions without concepts are blind (Kant 1996, sec.B 75)

At a less generalised level, and as suggested in the Introduction, categorisation may enable a number of specific benefits, such as ensuring that individual methods are conceptually coherent, transposing lessons between methods of the same type, and identifying the appropriate uses for different methods. It is worth mentioning that these three forms of benefit were not initially obvious or readily identifiable at the inception of the research project, but have become apparent after reflecting on the findings from the papers. Table 13 summarizes these forms of benefit, and their instances of each in the papers.

Table 13. Summary of the beneficial outputs from using the attributional-consequential categorical framework

Type of benefit	Instances in Papers
Ensuring individual methods are conceptually coherent	1. Paper 1. The European Commission's Organisation Environmental Footprint method includes credits for avoided anthropogenic emissions within what would otherwise be an attributional inventory, and is therefore conceptually incoherent. 2. Paper 2. *ISO 14044* (ISO 2006b) and the GHG Protocol *Product Life Cycle Standard* (WBCSD/WRI 2011c) both combine elements of consequential assessment within what would otherwise be an attributional account, and so the results are neither an assessment of change nor an inventory of impacts. 3. Paper 4. Natural regeneration baselines are an artifice of anthropogenic activities and are therefore not a non-anthropogenic baseline, and are not appropriate for attributional inventories (contrary to Soimakallio et al. (2015))
Transposing lessons between methods of the same type	1. Papers 2 and 3 identify the possibility of transposing the time-series baseline and decision scenario structure from the project/policy method to consequential LCA. 2. Paper 4 infers that natural baselines are appropriate for attributional LCA as they are appropriate for national greenhouse gas inventories. 3. Paper 4 proposes that the inclusion of foregone sequestration in attributional LCA suggested by Soimakallio et al. (2015) can be transposed to all other forms of attributional accounting.
Identifying the appropriate use for different methods	1. Paper 1 suggests that because attributional LCA is insufficient for supporting decision-making then corporate inventories will also be insufficient. 2. Paper 3 illustrates the way in which attributional accounts are not sufficient for supporting decision-making on mitigation actions, as they do not capture the total consequences of the decision. 3. Paper 4 suggests that national inventories are appropriate for assigning responsibility, target-setting, and budgeting, and that these are the appropriate uses for attributional accounts more generally.

Whilst acknowledging the possibility of becoming overly fixated with the development of categorical frameworks, the categorisation of the potential benefits from using the attributional-consequential distinction is, in a sense, a further form of categorical framework, which may be useful for identifying other opportunities for deriving benefits from the distinction. This categorisation of benefits is therefore used in the discussion of potential areas for further research, in Section 3, below.

The third form of benefit listed above is the identification of the appropriate use for a given method, and, as noted, the discussion on this issue is somewhat dispersed across the papers and is worth synthesising here. The key point with this benefit is that once a method is categorised as being attributional or consequential it is then possible to infer the appropriate uses for that method, e.g. national inventories are attributional in nature and therefore are not sufficient for informing mitigation decisions. Table 14 summarises the appropriate uses for attributional and consequential methods identified in Papers 1 to 4.

Table 14. Appropriate uses for attributional and consequential methods

	Purpose	**Limitations**
Attributional methods	1. Assignment of ownership or reasonability for a set of sources/sinks (this could include assignment of responsibility for tax liabilities, or obligation to surrender emission allowances in an emissions trading scheme). 2. Setting reduction targets (e.g. relative to an inventory base year or inventory business-as-usual baseline). 3. Carbon budgeting (e.g. assignment of budgets to ensure total aggregate emissions do not exceed a predetermined level). 4. Hot spot identification (i.e. identifying the sources within the inventory with the highest emissions). 5. Identifying regulatory risk if regulations are imposed on the basis of the attributional inventory (i.e. this use is effectively a subset or corollary of point 1. above).	1. Attributional inventories do not necessarily capture the total system-wide change in emissions and so may result in unintended consequences if used for mitigation decision making.

Consequential methods	1. Quantification of the system-wide change in emissions/removals (or other impacts) caused by a specified decision or intervention (i.e. consequential methods can be used to appraise (ex ante) or evaluate (ex post) any climate change mitigation decision, and can be used to quantify tradable credits from emission reduction projects).	1. Consequential methods cannot be used to assign unique responsibility for a set of sources/sinks as the sources/sinks included in any assessment relate only to the decision in question. 2. Consequential results are not additive, and do not sum to total global emissions, and so consequential methods cannot be used for setting carbon budgets.

Paper 1 discussed Wenzel's (1998) assertion that all life cycle assessment is ultimately concerned with decision-making, and therefore the only appropriate method is consequential LCA. Table 14 clearly suggests that this view is overly simplistic, and that there are a number of appropriate uses for attributional methods. However, on the other side of the debate, there are still many commentators who argue that attributional methods are sufficient for decision-making on mitigation actions (Brandão et al. 2014; Hertwich 2014; Suh & Yang 2014; Dale & Kim 2014; Anex & Lifset 2014), and the present research clearly finds that this is not the case. What should be emphasized instead is that attributional and consequential methods are complementary to one another, with attributional inventories providing a means of assigning responsibility for managing emissions, and consequential methods informing the decisions aimed at achieving reductions.

Although the creation of a categorical framework can enable a number of benefits, such as those identified above, it can also create limitations by privileging the phenomena that are amenable to the categorical scheme, whilst making other phenomena or alternative categorisations cognitively unavailable. This limitation, and others, are discussed in the following section.

2. Limitations with the Research

2.1. Bioenergy case study

One limitation with the research is the use of a bioenergy case study, as this may be perceived as an over studied area, given the number of bioenergy studies that either contrast attributional and consequential life cycle assessment (Searchinger et al. 2008; Hertel et al. 2010), or apply some form of consequential method to bioenergy (Bernier & Paré 2013; Buchholz, Friedland, et al. 2014; Haberl et al. 2012; Haberl et al. 2013; Holtsmark 2012; Holtsmark 2013; Walker et al. 2010; McKechnie et al. 2011; Schlesinger 2014; Schulze et al. 2012; Wilnhammer et al. 2015). In addition, it may be argued that bioenergy is a unique and anomalous case which happens to create large differences between attributional and consequential studies, and that for most other cases the different methods would broadly support the same decision-making outcome.

One response to this latter criticism, which is also presented in the Introduction, Paper 3, and above, is that a single case showing the insufficiency of attributional methods is enough to make the reliability of that approach uncertain. In any decision-making context the only means of ensuring that an attributional method is sufficient is to undertake a consequential assessment, which would then make the attributional account redundant. However, one reason why bioenergy may be viewed as an anomaly is that it often involves indirect land use change, which tends to involve very high levels of emissions. It therefore creates the kind of extreme case where an attributional account, which does not include indirect effects, will support the decision in question, while a consequential assessment will suggest the opposite. That is, indirect land use change may be one of the few emission effects that is large enough to switch the sign of the results from positive to negative.

While it is true that land use change emissions do tend to create extreme cases, which is also the reason that bioenergy has become such a focal issue for illustrating the difference between attributional and consequential methods, there are many other decision scenarios that involve land use change, and therefore such extreme cases are not limited to bioenergy. For example, Chalmers et al. (2015) explore the possible outcomes of a 1% tax on whole milk, and suggest that there may be a large reduction in emissions caused by the increased availability of milk fat co-products from the increased production of lower fat milk. The milk fat co-product may displace the production of palm oil, and therefore reduce the emissions from land use change associated with palm cultivation, and this large reduction in emissions would not be captured in an attributional study. The key point is that land use change is not a unique issue for bioenergy, and other decision scenarios will also give rise to large differences in the results from attributional and consequential methods, and an attributional approach will not necessarily reflect the total system-wide change caused by the decision.

Furthermore, it is important to note that land use change is not the only potentially large source of emissions that can create extreme differences between attributional and consequential results. With the bioheat case study in Paper 3, scenario 3.3 involves the displacement of cement render and the consequential methods estimate a reduction in total emissions of ~830,000 tCO_2e, while the attributional method estimates a reduction of ~177,000 tCO_2e. This very large difference in emissions is predominantly due to material displacement effects rather than indirect land use change.

Although the example in scenario 3.3 does not show a change in the sign of the results between the attributional and consequential approaches, it is important to note that large differences in results are still material, in the sense that they may lead to different decision-making outcomes. For instance, if mitigation decisions are made based on the relative abatement costs of different options, then large under-estimations in the reductions caused by specific options will alter their ranking, and therefore affect whether they are implemented or not. Large differences in the results between attributional and consequential methods, even when the sign of the result is the same, may lead to the prioritisation of actions which are less effective for mitigating climate change.

Notwithstanding the above points, it would be interesting to undertake a specifically non-bioenergy-related case study, such as the aluminium material substitution case outlined in Table 2 in the Introduction.

2.2. Decision-making, critical theory, and other considerations

One presupposition within this thesis is that information on the outcomes of decisions is important for decision-making, and that 'good' methods for decision-making are those that most accurately describe the outcomes of the decision in question. For instance, Question 3 presupposes that selecting the appropriate accounting method matters, as decision-makers will base their decisions on the information provided. Such presuppositions are aligned, at least to some extent, with rational-choice theory, which assumes that decision-makers have preferences, and that decisions are taken by weighing-up different options based the expected fulfilment of those preferences (Burns 2015; Scott 2000). Rational choice theory has been criticised on a number of fronts, which may, by association, be seen as a source of weakness for the present research as well. For this reason, it is worth exploring what those weaknesses might be, and whether they apply to the present research. This exercise may also be useful for teasing out the linkages between this thesis and a number of other conceptual or philosophical issues.

Firstly, rational choice theory has been criticised for failing to reflect the way in which decisions are actually made, i.e. decision-makers do not generally weigh-up the outcomes of different options, but rather decisions are shaped by values, emotions, norms, institutionalised practices, and social context, which may, or more likely may not, be consciously reflected on by the decision-maker (Hoffman &

Jennings 2012; Corner et al. 2014; Kouchaki et al. 2014). However, this criticism is not directly applicable to the present research, which is not concerned with describing how decisions *are* made, but is instead focused on normative method development, for how decisions *should be* made. This distinction is the same as that between *descriptive* and *normative* decision theory (Hansson 2005).

However, a related weakness which may be more applicable is that if decision-makers do not actually use information on the outcomes of their decisions, then there is little point in developing methods that provide such information. It is worth noting that this criticism appears to work equally well on both attributional and consequential methods, as both provide information, which, following the above argument, is irrelevant to how decisions are actually made. One response is to argue that the role of values and emotions is simply overstated in the anti-rationalist picture of decision-making, and that everyday life is replete with instances of actors seeking information on the outcomes of their actions. A slightly different response is to accept that decision-making is determined by institutionalised norms and practices, but that one such practice is the diligent consideration of the outcomes of the decision, i.e. rational deliberation is embedded within the social norms and practices associated with decision-making, and that the two, seemingly opposing views, are not really opposed at all.

A further argument from the critical theory literature, that can be interpreted as a potential problem with the current thesis, is that accounting methods and power are inextricably linked, and that accounting rules and practices systematically favour the interests of certain groups (Lohmann 2009). However, it is not clear to what extent this issue applies to consequential methods, which appear to be agnostic in terms of the decisions and actions to which they are applied, and may be used to challenge rather than support powerful economic interests. An example is the Searchinger et al. (2008) study, which challenged US and EU biofuel policy, and the vested interests of the farming lobby. Another example of a disruptive study is the

Stockholm Environment Institute's use of the GHG Protocol *Policy and Action Standard* to estimate the increase in global emissions caused by the Keystone XL pipeline for delivering oil from Canadian tar sands to the Gulf of Mexico (WRI 2014c).

A further criticism, operating at a deeper level, is that our calculative practices, such as greenhouse gas accounting, give the false impression that addressing climate change can be sub-divided into discrete decisions and actions (Gray 2010), and that the problem of climate change is solvable within the existing social paradigm. Such calculative practices are a constitutive part of the modernist perspective (Foley 2015), that separates humanity from nature, and which is ultimately the cause of environmental problems such as climate change (Gray 2010). These practices also obscure from view the inherently unsustainable nature of capitalism (Gray & Bebbington 2000; Bebbington & Thomson 2007), with its drive for ever increasing levels of wealth and consumption. This critical perspective tends to emphasize the social construction and contingency of the modernist viewpoint, the need for a more holistic or systems-oriented perspective, and the possibility of resolving social and environmental problems through embracing alternative constructions of reality (Gray 2010; Lohmann 2009). A potential problem with the current thesis is that it remains entrenched within a paradigm that perpetuates an ultimately unsustainable system.

This is a challenging problem to address, as any response may also be limited by the conceptual constraints of the inhabited paradigm, and therefore be blind to its own limitations. However, one can only proceed from the conceptual starting point one finds oneself in, and with this caveat acknowledged, a number of concerns with the critical perspective can be expressed. Firstly, although the idea of resolving the climate change problem via a reconceptualization of the world is intellectually appealing, it is not clear to what extent our incumbent conceptual paradigm is open to revision. Quine's revisibility thesis (1951) suggests that our beliefs form a network, with readily revisable beliefs near the periphery, and intractable beliefs at the centre. A similar idea is present in Wittgenstein's discussion on epistemology, which uses the analogy of a river bank that 'consists partly of hard rock, subject to no alteration or only to an imperceptible one, partly of sand, which now in one place now in another gets washed away' (Wittgenstein 1975, sec.99). Or as Neurath puts it, 'We are like sailors who have to rebuild their ship on the open sea, without ever being able to dismantle it in dry-dock and reconstruct it from its best components.' (Neurath 1973). An example of the sheer difficulty of disconnecting from the status quo is provided by Lohmann (2009), in which an attempt is made to move beyond the debate between neoliberalists and their opponents by adopting an alternative framing of the climate problem. What is striking is that many of the substantive points in the paper appear to fall back immediately into the terms of the debate that is (supposedly) rejected. For example, the suggestion that a more fruitful approach would be to ask 'how disruptive and damaging are the practical consequences of carbon accounting's attempts to frame a new omnibus category of "emission reductions"?' appears to be very close to the kind of incommensurability arguments raised by opponents to the monetary valuation of the environment. A further limitation to reconceptualization is that any truly radical reframing may not be intelligible to anyone not already inhabiting the alternative conceptual scheme, given that our beliefs underpin our meanings and language (Wittgenstein 1997; Kuhn 1962). The key point for the present discussion is that the kind of radical reconceptualization envisaged by critical theory, although enticing, may not be achievable.

A different tension within critical theory is that although the categories and calculative practices of modernity, including the natural sciences, may be seen as the ultimate cause of problems such as climate change, it is only through natural science that we are aware of climate change as a problem, i.e. the critical perspective depends on the every sources of knowledge that it rejects. This tension is recognised and grappled with to some extent in Gray (2010, pp.58 – 59):

> …however conditional our truths might be, there are categories of experience that we eschew at our peril. As humans, we embrace the hubris of our febrile and facile intelligence when we deny – or even fail to embrace – our grounding in a physicality and an inextricable entwining with what we call “Nature”. That science and modernity may be conditional, partial, and deserving of challenge in no sense tells us that all conclusions through the lens of science are essential untruths.

Notwithstanding these tensions within critical theory, this perspective also provides a number of highly useful insights. Firstly, the case is made within the critical literature for a holistic or systems-perspective for addressing issues of sustainability, i.e. sustainability is a system-level attribute, and if the system as a whole is unsustainable it is meaningless to claim that individual components, such as corporations or products, are sustainable (Gray 2010). Arguably, consequential methods are aligned with this view, in that they recognise the futility of reducing emissions within an inventory boundary whilst increasing emissions elsewhere in the system. However, it should still be recognised that even when individual actions reduce emissions at the system level, this does not entail that sustainability is achieved, if total global (system-wide) emissions continue to accumulate to unsafe levels in the atmosphere. In order to address this issue, attributional methods are likely to be needed to ensure total aggregate emissions do not exceed what is considered a ‘safe’ threshold level, as suggested in Paper 4.

A second highly useful point which is emphasized in the critical theory literature is the importance of the social processes in which decisions are made, i.e. it is not sufficient to simply produce information if the information is not accessible, meaningful, or recognised by the actors involved. Indeed, what counts as information is itself actively constructed within the social field (Morgan 1988; Ekvall et al. 2005). With similar considerations in mind, a useful distinction is made in the literature on environmental impact assessment between the *political* and *technical* aspects of environmental assessment (Wallington et al. 2007), which to some extent also parallels the distinction between the social and technical literature for greenhouse gas accounting mentioned in the Introduction. The political view emphasizes the social context in which decisions are made, and the need to reflect on the pre-conditions for decision-making such as the relationship between actors and the institutional culture. This perspective tends to view environmental impact assessment as a learning process, in which participants can interact and reflect on their values, and the technical quantification of impacts plays a subordinate role in facilitating that process. This reorientation of the role of quantification methods is an important one, and addresses the simplistic decision-making model suggested by rational choice theory, which ignores the complex role of norms, values, emotions, institutional context etc. in the process of decision-making.

However, this social/political perspective can also be taken to an extreme position in which the quantification of impacts is seen as a purely discursive exercise, in which the accuracy of the numerical values drops out of consideration. Although extreme, this view is not uncommon, and appears attractive as it deflects concerns about modelling errors, accuracy, or completeness, and is particularly prevalent in the interpretation of complex modelling exercises, where it can be claimed that models explore relationships, but should not be viewed as representations of reality. This view appears to be problematic however, as it does not resolve why we should be interested in modelled relationships and impacts, unless they tell us something about real relationships and impacts. Turning again to the example of Searchinger et al. (2008), it is not clear why the results should be of interest unless they are intended to represent the actual greenhouse gas impacts of US biofuel policy.

As Wallington et al. (2007) suggest, the technical and the political should be viewed as complementary approaches, both being highly useful for facilitating decision-making. In terms of the present thesis, the focus is very much on the technical side, through the development of methods for quantifying the outcomes of decisions, and identifying the appropriate use of different methods. But it is essential to acknowledge that this is only part of the picture, and that the generation, recognition, and use of technical information will be shaped by the broader social context. Some ideas for further research addressing these issues are discussed in section 3.1 below.

2.3. Categorical effects

The creation of accounts, and indeed categories more generally, gives emphasis to the phenomena that fall within those accounts/categories, while phenomena that fall outside are made invisible, cognitively unavailable, or unthinkable (Neyland 2007; Strathern 2000; Brown 2014; Davies 2013). This is potentially problematic as we may ignore issues and influences which should be taken into account, and alternative narratives or forms of behaviour will not be considered. This issue is pertinent to the present thesis on at least two levels, firstly the problem applies generally to greenhouse gas accounting, as a form of accounting, and secondly the problem applies to the use of the attributional-consequential distinction as a categorical framework. Both provide frameworks for categorising phenomena, and both may therefore diminish certain phenomena and privilege others, or as Suh and Yang (2014, p.1179) put it:

> How we classify things often helps us see what we couldn't see before, but it may also make us unable to see what should be otherwise obvious. That is because subscribing to a classification, consciously or subconsciously, leads or misleads our minds toward the frame that is created by the way it is done. Sometimes, the influence a classification has on one's mind can be so powerful that it makes the person completely blind to the things that do not follow the order created by the classification.

A number of theorists and commentators warn of the effects of categorization, due to the way they skew or distort discourse, cognition and behaviour (Latour 1993; Everett 2004). However, categorisation is arguably an inescapable prerequisite for cognition (Kant 1996), and the rejection of one conceptual scheme simply entails the adoption of another. In recognition of this, the solution is not to reject categorisation *en masse*, but rather to create forms of categorisation that appear helpful or progressive (Gray 2010).

The development of the attributional-consequential distinction can itself be viewed in this light, i.e. it provides a categorical framework for interpreting different greenhouse gas accounting methods, which appears useful for a number of purposes, e.g. method development, selecting the most appropriate method etc. Similarly, the development of consequential methods can also be viewed as a useful categorical framework, introducing concepts such as 'marginal systems', 'baselines', and 'indirect effects', without which it would be difficult to articulate or conceptualise many of the system-wide effects of decisions or actions. Moreover, there is an interesting parallel between the categorisation problem, which creates both visibility and invisibility, and the difference between attributional and consequential methods themselves, i.e. one of the main limitations with attributional inventories is that they make impacts that occur outside the inventory boundary invisible. In contrast, with consequential methods, the whole purpose is to ensure that all relevant effects are included within the boundary of the assessment.

However, it must be acknowledged that even with consequential methods the category problem still arises, for instance, by focusing exclusively on greenhouse gas emissions all other forms of environmental or social impact are made invisible. The category problem is also likely to arise with the development of the attributional-consequential distinction as a categorical framework, i.e. alternative forms of greenhouse gas accounting that fall outside this framework may be overlooked or dismissed. The only solution appears to be to acknowledge the problem, and maintain an open-mind to alternative categorisations.

3. Ideas for Further Research

Following on from the discussion in the preceding section, any new categorical framework can be expected to bring new issues into focus, or create a novel way of conceptualising or analysing existing problems. This section illustrates this point by setting out a number of ideas for further research which are either motivated by the present thesis, or are enabled by the attributional-consequential categorical framework.

3.1. Why are attributional methods used to inform decision-making?

As discussed in the Introduction, a distinction can be drawn between the *technical* greenhouse gas accounting literature, which is concerned with implementing and developing methods, and the *social* research literature, which focuses on greenhouse gas accounting as a social phenomenon. The latter is often concerned with providing explanatory answers to questions such as ‘Why do companies disclose their greenhouse gas emissions?’ or ‘Why do investment analysts use greenhouse gas data?’. One such question, motivated by the bioheat case study in Paper 3, is ‘Why are apparently inappropriate greenhouse gas accounting methods used to inform mitigation decisions?’. This question also illustrates the point made in the Introduction, and in section 2.2 above, that the technical and social research approaches should be seen as complementary to one another, with each answering different kinds of question.

In the case of the bioheat plant, the commissioning organisation had the explicit intention of reducing its greenhouse gas emissions, and used its attributional corporate inventory to inform the decision to build a bioheat plant. However, the analysis presented in Paper 3 suggests that the outcomes from the bioheat plant are highly uncertain, and may increase rather than reduce emissions, i.e. the opposite outcome to that intended by the organisation. There appear to be a number of interrelated questions around why an attributional method was used, why alternative consequential methods were not considered, and why the possibility of potentially large increases in emissions was not visible to those involved.

The use of the concept of framing, in Paper 1, goes some way to addressing these questions, particularly in terms of the cognitive availability of the consequential approach to academics and practitioners embedded within an attributional frame. An alternative theoretical lens which offers a range of relevant explanatory concepts for answering these questions is institutional theory. Institutional theory seeks to explain social phenomena with reference to the established social structures, schemes, rules, norms and routines that guide and constrain social behaviour, and it is also concerned with how such social structures are constructed or established (Hodgson 2006). The term 'institution' is used in a broad sense to refer to any 'systems of established and prevalent social rules that structure social interactions', and examples of institutions include 'language, money, law, systems of weights and measures, table manners, and firms (and other organizations)' (Hodgson 2006, p.2). Given this broad sense of the term, forms of greenhouse gas accounting may also be described as institutions, and institutional theory may be useful for explaining why one form of institutionalised practice, i.e. attributional accounting, has become established or dominant to the exclusion of others, i.e. consequential accounting.

Institutional theory, in its cognitive turn, also appears to be useful for explaining why the potentially large increases in emissions from the bioheat plant were not visible to the commissioning organisation. As with the category problem discussed above, the accounting practice itself makes some emission impacts visible and others invisible (Larrinaga-Gonzalez & Bebbington 2001; Potter 2005; MacKenzie 2009; Scott 1995), and individuals embedded within a particular accounting practice will not be aware of consequences or impacts beyond those represented within the accounting scheme. As an aside, the lineage of this aspect of institutional theory, i.e. that our conceptual framework both enables and constrains our awareness of the world, can be traced back through earlier formulations in the philosophy of science (Fleck 1979; Kuhn 1962) and epistemology, at least as far as Kant's *Critique of Pure Reason* (Kant 1996).

Institutional theory also offers an explanation for how and why certain institutions emerge, i.e. why attributional corporate accounting appears to have emerged as a dominant institution, to the exclusion of consequential approaches. Individual institutions and their emergence are themselves shaped and explained by the broader institutionalised structures and practices in the wider field or environment (Powell & Colyvas 2008). In the case of organisational/corporate greenhouse gas accounting, the wider field includes national greenhouse gas inventories, and also conventional management and financial accounting, all of which could be described as broadly attributional in nature. The wider institutional environment may also include deontological conceptions of responsibility, such as the fiduciary principle of maximising shareholder value, which lend themselves to an attributional perspective, as opposed to a consequential or whole-systems approach. A supplementary explanation for the apparent dominance of attributional greenhouse gas accounting may also be provided via the notion of 'path dependency' in historical institutional theory (Thelen 1999). The development path for organisational/corporate greenhouse gas accounting has been informed by national greenhouse gas accounting, which is attributional in nature, and therefore the attributional approach has also come to dominate organisational/corporate practice, with the ubiquity of that practice creating a further barrier to the recognition of alternative approaches.

In addition to using institutional theory to provide an explanation of the bioheat case study, the process of developing the explanation can also be used, reciprocally, to appraise institutional theory itself, in terms of its ability to provide an explanation of the empirical case. Institutional theory provides a large number of conceptual elements for building an explanatory narrative, such as institutional logics, satisficing, institutional complexity, isomorphism, mimetic herding, legitimacy, decoupling, coercion, institutional orders, discursive legitimacy, disruptive events, distributed agency, theorisation, translation, sense-making, categorical imperatives, embedding, institutional ascription, and many more (Guerreiro et al. 2012; Scapens 1994; Greenwood et al. 2011; Tuttle & Dillard 2007; Gabbioneta et al. 2013; Richardson 1987; Carruthers 1995; Carpenter & Feroz 2001; Burns 2000; Misangyi et al. 2008; Seidl et al. 2013; Hoffman 1999; Lawrence et al. 2011; Dacin et al. 2002; Modell 2009; Dillard et al. 2004; Suddaby 2010; Adams & Larrinaga-González 2007; Gabbioneta et al. 2014). It may be interesting to reflect on whether there are elements of the case that remain unexplained, and whether there is duplication in the explanatory work done by this extensive array of theoretical concepts.

3.2. Utilitarian and deontological ethics

A different area for further exploration, though one that is linked in some ways to the discussion on rational choice theory above, is the relationship between consequential and attributional greenhouse gas accounting methods and the ethical theories of consequentialism and deontology. Ethical consequentialism holds that the value of an action is determined by its outcomes, and so is closely aligned with the justificatory underpinnings for consequential greenhouse gas accounting methods, and also rational choice theory, normative decision theory, and much of neo-classical welfare economics. Utilitarianism is an archetype of ethical consequentialism, advocating that the 'greatest happiness for the greatest number is the measure of right and wrong' (Bentham 1776). In contrast, deontological ethics proposes that the morality of an action is determined by whether it is made in accordance with an ethical rule or code, rather than by the consequences of the action. This aligns with attributional methods, where the sources/sinks included within the inventory boundary, which in turn determines the scope of responsibility for the reporting entity, are determined by boundary-setting rules, such as operational control or physical connectivity within a value chain. The normative nature of inventory boundary-setting is recognised in the UNEP/SETAC guidance for life cycle assessment, which describes attributional LCA as an approach 'in which inputs and outputs are attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system according to a normative rule' (UNEP & SETAC 2011, p.132). An archetypal deontological theory is Kant's categorical imperative which states that one should act 'only according to that maxim whereby you can, at the same time, will that it should become a universal law' (Kant 1991).

The parallels between attributional and consequential LCA, and deontological and consequentialist ethics, respectively, have been explored previously by Ekvall et al. (2005). However, that analysis does not appear to be have been revisited since its publication, and there appear to be a number of points which could be extended and also challenged through further research. For example, Ekvall et al. (2005) suggest that attributional LCA can be used for decision-making if the decision relates to whether 'we want to become associated with the system' (2005, p.1228). However, such considerations may only apply if the characteristic in question is of a moral hue, such as the use of child labour, as moral responsibilities have a uniquely non-transferrable nature. In contrast, the totalising or system-wide perspective present in consequentialism may be appropriate so long as the value in question is of a non-moral nature, of which, arguably, greenhouse gas emissions are an example.

Notwithstanding the point above, an apparent strength of deontological ethics, and by association attributional methods, is the focus on the notions of responsibility and duty, which therefore provides a basis for assigning responsibility or ownership for certain greenhouse gas sources/sinks. This links to the point made in Paper 4 that attributional inventories create a sense of ownership, and provide a starting point for the management of emissions/removals. What is significant is that consequentialism, and consequential greenhouse gas accounting methods, are not suitable for this kind of allocation of responsibility, and this creates a potentially important complementarity between the two approaches. Attributional methods can be used for the initial allocation of responsibility for emissions, and consequential methods can be used to inform any subsequent mitigation actions aimed at reducing emissions within that sphere of responsibility.

3.3. Extending the distinction to other forms of accounting

This thesis has focused on extending the attributional-consequential distinction beyond the field of life cycle assessment and has applied it to other forms of physical greenhouse gas accounting. One potentially interesting area for further research is to explore whether the distinction can be extended further still, to other forms of environmental or sustainability accounting, financial accounting for carbon-based assets and liabilities, or to other forms of financial and management accounting more generally.

From an initial survey of existing methods and practices, it does appear that the attributional-consequential distinction could be applied to other sustainability accounting practices. For instance, the reporting guidance from the Global Reporting Initiative (GRI 2015) is largely based on the direct physical impacts of the reporting organisation or its value chain, and could therefore be categorised as an attributional inventory of impacts. Based on the limitations and appropriate uses of attributional and consequential methods, as outlined Paper 4, it can be inferred that such inventories may not show the total system-wide impact of the reporting company's decision-making. For example, a company's sustainability report may show that its infrastructure investments have positive impacts on local communities, but the company's policies or decision-making may conceivably have impacts elsewhere in the system (e.g. through taking market-share from companies working elsewhere with more vulnerable communities). Similar limitations are also likely to apply to natural capital accounting, which takes a number of forms but essentially creates an inventory of stocks of natural capital within an inventory boundary (Guerry et al. 2015), and therefore appears to be attributional in nature.

Other broader environmental or sustainability accounting methods, such as environmental impact assessment (EIA) or strategic environmental assessment

(SEA), appear to be more consequential in nature, in that they aim to capture the change in environmental consequences of a specific decision or strategy (Glasson et al. 2005). This would then create the potential for transposing lessons between these methods and those identified as consequential forms of greenhouse gas accounting. For instance, if not already present within EIA and SEA, the baseline and decision-scenario structure could be transposed, as well as presenting the distribution of impacts over time.

The attributional-consequential distinction could also potentially lead to some interesting insights into the nature of financial and management accounting. For example, the income statement in financial accounts is similar in many ways to the inventory of emissions/removals within corporate greenhouse gas accounting, with both showing a flow of values during the reporting period for a defined inventory boundary (i.e. both are attributional in nature). In contrast, a method such as calculating the internal rate of return for investment appraisal may be characterised as having consequential elements, in that it aims to quantify the consequences of an investment decision. However, an interesting difference is that investment appraisal does not attempt to capture the total system-wide change in value caused by the investment decision, but only the value accruing to the entity making the investment. Cost-benefit analysis or full-cost accounting may be more thorough-going consequential methods, as they seek to estimate the total system-wide change in value caused by the decision in question. An interesting feature of the climate change problem is that the distinction between private and public mitigation benefits does not arise, in contrast to private and total economic welfare. This also underlines the futility of actions which only serve to reduce emissions within a specific inventory boundary whilst increasing emissions elsewhere, as, for example, may be the case with biofuels and bioenergy more generally.

3.4. Uncertainty variance between different mitigation options

Paper 3 makes the suggestion that the range of possible outcomes from different mitigation options should itself be viewed as highly decision-relevant information, and that preference should be given to options that do not include potentially large negative outcomes. It would therefore be interesting to undertake further research to explore the range of possible outcomes from non-bioenergy options, such as wind power, heat-pumps etc., and to compare the variance in the modelled outcomes.

There are a number of options for how this study could be undertaken. One option would be to conduct a meta-analysis of the existing literature for other mitigation options, as, in the case of wind power, there is already an extensive number of studies in this area (Arvesen et al. 2014; Arvesen & Hertwich 2012; Dolan & Heath 2012; Ardente et al. 2008; Tremeac & Meunier 2009; Wiedmann et al. 2011; Guezuraga et al. 2012). However, very few existing studies appear to provide a full consequential assessment of the total system-wide change in emissions caused by wind power (Dolan & Heath 2012), or show the temporal distribution of emissions/removals, and it may therefore be necessary to supplement the existing literature with an explicitly consequential study.

If a supplementary modelling exercise is undertaken, this could also offer an opportunity to compare different modelling options, such as Group Model Building (Laurenti et al. 2014), Input-Output-based Hybrid LCA (Wiedmann et al. 2011), or an integrated assessment model such as the Global Change Assessment Model (Joint Global Change Research Institute 2014). Such models, and others, have been used in individual consequential studies, but their relative merits and limitations have not been explored in an empirical comparative analysis. Such a study would therefore provide a multi-dimensioned comparison, between the different mitigation options, and between the different modelling approaches.

3.5. Understanding contractual emission factors for scope 2 emissions

A highly contentious issue within corporate greenhouse gas accounting, mentioned briefly in Paper 1, is the use of contractual emission factors for reporting scope 2 emissions, i.e. point-of-generation emissions from purchased electricity (Brander et al. 2015). Contractual emission factors allow companies to buy certificates, or enter into other contractual arrangements, which convey the exclusive right to claim the emissions associated with renewable electricity (i.e. normally zero, at the point-of-generation). This issue is highly topical as the Greenhouse Gas Protocol has published guidance on scope 2 accounting in early 2015 (WRI 2015), supporting the use of contractual factors. In addition, ISO is considering its own requirements within a revised version of its organisational greenhouse gas accounting standard (ISO 2006c), due for publication in 2017.

One concern with the use of contractual factors is that this practice does not increase the amount of renewable generation, and therefore does not reduce emissions (Gillenwater 2013; Gillenwater et al. 2014). This problem can be described as one of additionality, i.e. the level of system-wide emissions is identical with or without the purchase of contractual factors. A second concern relates to the impact the practice has on the accuracy and relevance of greenhouse gas accounts (Brander 2013). For instance, when a company switches to using contractual emission factors, the accounts will suggest the company has reduced its emissions, whereas the company's contribution to aggregate electricity demand, and therefore emissions, remains unchanged. As a result, the accounts are not an accurate reflection of the company's emissions. Similarly, if a company purchases contractual emission factors for renewable energy then its scope 2 emissions will be shown as zero. As a result the accounts will not help to identify real emission reduction opportunities, such as improved energy efficiency, and will not be relevant for decision-making.

This issue is of interest as, in addition to its importance in terms of managing and reducing emissions, it raises a number of conceptual questions about the underlying rationale for allocating emissions within attributional inventories. Furthermore, the very fact that articulating the appropriateness or inappropriateness of contractual factors is challenging, and that there are divergent opinions, suggests that there may be fundamental issues with the conceptualisation of attributional inventories that remain unresolved.

The connection between this issue and the present thesis is that an improved understanding of the attributional-consequential distinction may, potentially, help to address these conceptual questions. Moreover, use of the distinction is already evident to some extent within the debate over contractual factors. For example, appeal is made to the distinction between corporate (attributional) and project (consequential) accounting to explain why contractual emission factors do not represent avoided emissions (WRI 2015, p.28), and to suggest that the concept of additionality is not necessary for corporate accounting (2015, p.90).

Insights may be gained by reviewing the way in which other attributional methods determine the emission sources that are included within the inventory boundary, i.e. drawing inferences from methods of the same categorical type. For example, within attributional LCA it is generally assumed that the emission sources/sinks included are those physically connected with the life cycle of the product, i.e. physical connectivity is a determining criterion for inclusion. Similarly, with both production-based (IPCC 2006) and consumption-based national inventories (Barrett et al. 2013), the implicit criterion is a physical connection between the sources/sinks and the reporting country in question. Contractual emission factors represent a novel departure from this approach, in that the renewable attributes conveyed by the contract do not have to be from generation facilities physically connected to the distribution grid from which the reporting company consumes its electricity. For example, it is possible for a UK-based company to use attributes from electricity generated in Iceland, although no interconnector currently exists.

The implicit criterion underpinning the use of contractual factors appears to be some form of economic, rather than physical relationship, i.e. purchasing contractual factors may, through aggregate demand effects, increase the amount of renewable generation, and this is the relationship that links the emissions rate to the reporting company. But this is then closer to a market-mediated (consequential) rationale, which leaves contractual factors in the conceptually awkward middle ground between attributional and consequential approaches. The present discussion is intended to provide some initial reflections on this topic, to illustrate a further potentially helpful application of the attributional-consequential distinction, which can be developed in future research.

3.6. The omission of foregone sequestration from national inventories

A final illustration of the research opportunities opened-up by the attributional-consequential categorical framework is the possibility of quantifying the level of foregone sequestration omitted from national inventories. Soimakallio et al. (2015) argue that foregone sequestration relative to a natural baseline should be included in attributional LCA, and Paper 4 suggests that it is possible to extend this requirement to all other forms of attributional inventory, such as national inventories (this being an instance of transferring lessons between methods of the same categorical type). Further research could explore this possibility, and the significance of the results, as the level of foregone sequestration currently omitted from national inventories could be substantial.

Recognising that attributional inventories should include values for foregone sequestration has implications for the allocation or assignment of responsibility for managing emissions/removals, target setting and carbon budgeting. For example, countries with high levels of land appropriation will be held accountable for a greater share of net emissions to the atmosphere. As noted earlier, the omission of items from an account will tend to make that item invisible to decision-makers, and the inclusion of foregone sequestration may have the effect of focusing greater attention on this cause of increased greenhouse gas concentrations in the atmosphere.

This section has outlined a number of ideas for further research in order to illustrate the way the development of the attributional-consequential categorical framework creates new opportunities for exploring and conceptualising accounting practices. The following section provides a number of specific examples of how the research contained in this thesis can be used to develop real-world practice.

4. Relevance to Practice and Routes to Impact

As discussed in the Introduction, one of the main motivations for this thesis is to contribute to the development of greenhouse gas accounting practice, and to improve decision-making for climate change mitigation. There appears to be considerable opportunity for doing so, given the widespread use of greenhouse gas accounting, from individual consumer decision-making to national-level mitigation planning (UNFCCC 2015). This section provides a number of examples of how an improved understanding of the attributional-consequential distinction may be relevant to different forms of greenhouse gas accounting practice, and some of the steps for facilitating that understanding.

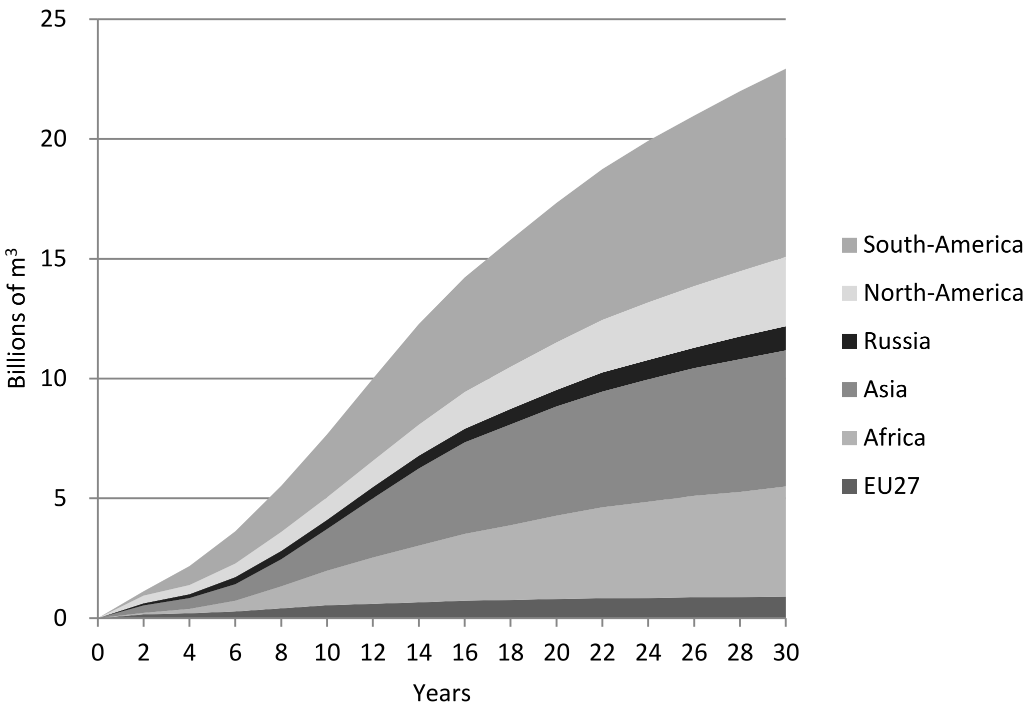
An example at the product level is the ISO technical specification for carbon footprinting, (identified as an attributional method in Paper 2), which lists 'providing information to consumers and others for decision‑making purposes' as one of its contexts of use (ISO 2013b, p.vi). The specification does not include any discussion on the potential limitations of using an attributional method for decision-making, or the necessity of using a consequential method to ensure that any decision does not have unintended consequences. The limitations with using attributional LCA for decision-making have been articulated as far back as Weidema (1993), yet there is still a need for disseminating this understanding to improve accounting methods, such as ISO 14067, and the real-world decisions they inform. ISO 14067 currently has the status of a 'technical specification', and it is expected to be revised and upgraded to the status of an 'international standard' in 2016. In terms of achieving impact from the present research, Paper 2 has been circulated to the ISO working group for this revision process, and I am participating as a technical expert.

An example of relevance to practice at the corporate/organisational level is the Scottish Government's draft Reporting on Climate Change Duties Order (Scottish Government 2015), under the Climate Change (Scotland) Act (Scottish Government 2009). This draft regulation requires public bodies in Scotland to submit annual reports on their climate change planning, governance, and inventory of emissions, with the inventory largely based on the requirements of the GHG Protocol *Corporate Standard* (WBCSD/WRI 2004). The draft regulations require the reporting of emission reductions achieved from mitigation projects, but there is no recommendation on the use of consequential methods to ensure that system-wide impacts are captured. In addition, information is only required on the emission reductions occurring during the reporting year, and so no consideration is given to the broader temporal distribution of emissions/removals resulting from the mitigation action. A further area of improvement would be to require an estimation of 'influenced emissions', i.e. other changes in emissions caused by the reporting organisation which are not captured in either the organisational inventory or in the reductions caused by mitigation projects. For a university this may include the emissions from international students travelling to study, or possibly the reduction in emissions caused by research into improved greenhouse gas accounting methods. For a local authority it could include planning or transportation policies which may either increase or decrease emissions outside the inventory boundary. The present research partly informed the University of Edinburgh's response to the Scottish Government's consultation on these Regulations in 2015, and a copy of Paper 3 has also been submitted to the Scottish Government team managing the development of the regulations.

An example at the policy-level is the European Commission's current study into the environmental implications of EU demand for biomass from North America (Kittler et al. 2015). This study focuses specifically on the impact of increased EU demand on forests in the US, as this is where the majority of imported biomass for energy to the EU physically comes from. However, as suggested above, focusing on physical connectedness is a feature of attributional approaches, and may not represent the marginal system that is affected by the increase in demand. It is possible that the marginal system is the same as the direct product system, i.e. the US may coincidentally happen to be the world marginal supplier of biomass, although this would need to be supported with evidence, e.g. showing that US production is unconstrained and it is the region with the lowest costs of production (as per Ekvall & Weidema (2004)). The European Commission study briefly considers the possibility that increased demand for US biomass will have knock-on effects to other markets (Kittler et al. 2015, sec.5.3), but the study does not explicitly discuss the aim of identifying the marginal system. A similar focus on North America is also present in the UK Department for Energy and Climate Change's Biomass Emissions and Counterfactual Model (DECC 2014), and in the accompanying technical report (Stephenson & MacKay 2014), again based on this being the world region physically supplying the UK.

In contrast to these studies which maintain a largely attributional outlook, modelled data from Lauri et al. (2014) suggest that North America will only contribute a relatively small proportion of marginal biomass supply over the next 30 years, as illustrated in Figure 19 below. In addition, and as noted in Paper 3, the marginal system is more likely to be a combination of systems, and will rarely be the single system that physically supplies the product consumed.

Figure 19. Projected increase in woody biomass supply by world region (based on Lauri et al. (2014))



This adherence to physical connectedness, i.e. the system supplying the physical biomass consumed, is also an underlying assumption for the greenhouse gas reporting requirements under the EU Renewable Energy Directive (European Parliament and Council of the European Union 2009), which are therefore also largely irrelevant in determining the system-wide change in emissions caused by the policy. Increased awareness of consequential modelling approaches could greatly improve the types of greenhouse gas accounting that are used in practice, and the climate change mitigation actions that are undertaken. Paper 3 has been shared with the European Commission study team, and feedback has also been provided on the interim issues paper for the study.

In addition to direct engagement in the development of individual standards or regulations, as illustrated above, a further route to potentially wider impact is through the development of framework standards, which aim to establish the broader methodological and conceptual principles underpinning more specific applications. In this regard, much of the analysis in Paper 2 is being used to initiate the development of a generic consequential framework standard, which incorporates best practice from consequential LCA, project, and policy-level methods. Any sector-specific or action-specific consequential method can then use the framework to ensure it includes the essential elements of a complete consequential approach, such as a transparent baseline and decision scenario, and the distribution of emissions/removals over time. The development of this standard is taking place on Collaborase.com, an online platform for standards development, which allows greater levels of participation and transparency compared with conventional committee-based standards development. One option for publication is to propose the draft framework as a prototype ISO standard, which could then be balloted as an international standard by ISO member bodies. Although an ISO standard has the benefit of recognition and perceived legitimacy, there is a cost-barrier to the use ISO standards which may restrict its application. An alternative option is to publish the framework standard on the Collaborase platform and allow on-going commenting and posting of cases studies etc., with the aim of maintaining the standard as a living document.

The above framework standard is specifically focused on providing a generic consequential method, and an additional initiative is the development ISO 14080, which aims to provide a good practice framework for greenhouse gas methodologies more generally. This initiative is at an early stage within the ISO standards development process, but offers a further opportunity to introduce generic guidance on attributional and consequential methods, and their appropriate uses, as outlined in Paper 4, and in Table 14 above.

5. Concluding Remarks

The research in this thesis is largely based on, and owes a debt of gratitude to, the insights and methodological developments originally developed within the life cycle assessment community, particularly the early articulations of the attributional-consequential distinction (Weidema 1993; Weidema 1998; Weidema et al. 1999; Ekvall & Weidema 2004). Where this thesis makes a novel contribution is in looking over the disciplinary fences between the life cycle assessment community and other forms of physical greenhouse gas accounting, and developing the attributional-consequential distinction as a broader categorical framework. There appears to be considerable benefit from taking this ecumenical perspective to greenhouse gas accounting, primarily because categorisation allows inferences between methods of the same categorical type. The benefits include ensuring that individual methods are conceptually coherent, sharing lessons between methods of the same kind, and, not least, ensuring that the correct method is used for a given purpose. This last point is particularly important if we are to make effective decisions to mitigate dangerous climate change.

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Appendix A - Supporting Material for Paper 3

1. Activity Data and Emission Factors

Table 15. Activity Data and Emission Factors Used in the Corporate Inventory Method

Activity Data					
	Residences	Schools	Units		Year
Natural gas consumption:	25,150,717	39,294,114	9,691,106	kWh/yr	R. Yarr 2014, pers. comm., 28 Oct
Electricity consumption:	5,958,715	13,205,470	4,844,666	kWh/yr	R. Yarr 2014, pers. comm., 28 Oct
Diesel consumption:	59,105	litres/yr			R. Yarr 2014, pers. comm., 28 Oct
Biodiesel consumption:	8,000	litres/yr			R. Yarr 2014, pers. comm., 28 Oct
Petrol consumption:	22,075	litres/yr			R. Yarr 2014, pers. comm., 28 Oct
Water consumption:	298,124	m3			R. Yarr 2014, pers. comm., 28 Oct
Business travel - flights:	14,581,645	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - rail:	1,862,139	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - bus:	189,810	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - ferry:	28,674	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - taxi:	168,189	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - hire cars:	439,407	miles/yr			R. Yarr 2014, pers. comm., 28 Oct

Business travel - coach:	312,101	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - employee owned vehicles:	351,491	miles/yr			R. Yarr 2014, pers. comm., 28 Oct
Business travel - hotel night stays:	1,908,184	£/yr			R. Yarr 2014, pers. comm., 28 Oct
Landfill - paper:	132	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Landfill - card:	185	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Landfill - food:	132	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Landfill - plastics:	187	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Landfill - metals:	14	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Landfill - asbestos:	0	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Recycled waste:	1,654	tonnes/yr			R. Yarr 2014, pers. comm., 28 Oct
Woody biomass:	43,080	MWh/yr			R. Yarr 2014, pers. comm., 28 Oct
Expected increase in energy consumption:	2%	percentage increase per year			R. Yarr 2014, pers. comm., 28 Oct
Expected decarbonisation of the electricity grid:	4.3%	percentage decrease per year			derived from Keep Scotland Beautiful (2011)
Embodied emissions of the Guardbridge Energy Centre:	8,819	tCO_2e in 2015/2016			derived from Cullinan Studio et al. (2014)
Energy content of woody biomass:	43,080	MWh/yr			Cullinan Studio et al. (2014)
Gas consumption replaced by GEC:	32,590	MWh/yr			Cullinan Studio et al. (2014)

Conversions					
Kilometres per mile:	2	km/mile			
Number of passengers on a coach:	30	passengers	working estimate		
Emission Factors					
	$kgCO_2/kWh$	**$kgCH_4/kWh$**	**kgN_2O/kWh**		
Natural gas combustion (net):	0.2064	0.00049	0.00019	in CO_2e	Defra/DECC (2015)
Natural gas combustion (net):	0.20643	0.00002	0.00000	by gas	Derived from Defra/DECC (2015)
	$kgCO_2e/kWh$				
Upstream emission factors for natural gas:	0.02759				Defra/DECC (2015)
Electricity generation - UK grid in 2013:	0.45844	0.00035	0.00334	in CO_2e	Defra/DECC (2015)
Electricity generation - UK grid in 2013:	0.45844	0.000017	0.000011	by gas	Derived from Defra/DECC (2015)
	$kgCO_2/kWh$	**$kgCH_4/kWh$**	**kgN_2O/kWh**		
Emission factors for T&D losses:	0.03785	0.00003	0.00028	in CO_2e	Defra/DECC (2015)
Emission factors for T&D losses:	0.037850	0.000001	0.000001	by gas	Derived from Defra/DECC (2015)
	$kgCO_2e/kWh$				
Upstream emission factors for electricity consumption -	0.04902				Defra/DECC (2015)

generation:					
Upstream emission factors for electricity consumption - T&D losses:	0.00405				Defra/DECC (2015)
Assumed rate of decarbonisation of the electricity grid:	4%	percent/yr			derived from Keep Scotland Beautiful (2011)
	$kgCO_2$/litre	**$kgCH_4$/litre**	**kgN_2O/litre**		
Emission factors for diesel:	2.67942	0.00072	0.03692	in CO_2e	Defra/DECC (2015)
Emission factors for diesel:	2.679420	0.000034	0.000119	by gas	Derived from Defra/DECC (2015)
Emission factors for biodiesel blend:	2.5863	0.0007	0.0368	in CO_2e	Defra/DECC (2015)
Emission factors for biodiesel blend:	2.586300	0.000033	0.000119	by gas	Derived from Defra/DECC (2015)
Biogenic CO2 factor for biodiesel blend:	0.093120			in CO_2e	derived from above
Emission factors for petrol:	2.33171	0.00363	0.00812	in CO_2e	Defra/DECC (2015)
Emission factors for petrol:	2.331710	0.000173	0.000026	by gas	Derived from Defra/DECC (2015)
	$kgCO_2e$/litre				

Upstream emission factors for diesel:	0.5796				Defra/DECC (2015)
Upstream emission factors for biodiesel:	0.62154				Defra/DECC (2015)
Upstream emission factors for petrol:	0.4504				Defra/DECC (2015)
	$kgCO_2$/unit	**$kgCH_4$/unit**	**kgN_2O/unit**		
Emission factors for flights (short-haul average):	0.08887	0.0	0.00087	$kgCO_2e$/pass.km	Defra/DECC (2015)
Emission factors for flights (short-haul average):	0.088870	0.000000	0.000003	kg gas/pass.km	Derived from Defra/DECC (2015)
Emission factors for rail:	0.0448	0.00004	0.00022	$kgCO_2e$/pass.km	Defra/DECC (2015)
Emission factors for rail:	0.044800	0.000002	0.000001	kg gas/pass.km	Derived from Defra/DECC (2015)
Emission factors for bus:	0.10042	0.00008	0.00076	$kgCO_2e$/pass.km	Defra/DECC (2015)
Emission factors for bus:	0.100420	0.000004	0.000002	kg gas/pass.km	Derived from Defra/DECC (2015)
Emission factors for ferry:	0.11516	0.000038	0.000891	$kgCO_2e$/pass.km	Defra/DECC (2015)
Emission factors for ferry:	0.115160	0.000002	0.000003	kg gas/pass.km	Derived from Defra/DECC (2015)
Emission factors for taxi:	0.24298	0.00008	0.00167	$kgCO_2e$/km	Defra/DECC (2015)
Emission factors for taxi:	0.242980	0.000004	0.000005	kg gas/km	Derived from Defra/DECC (2015)
Emission factors for hire cars:	0.298195	0.000161	0.001545	$kgCO_2e$/mile	Defra/DECC (2015)
Emission factors for hire cars:	0.298195	0.000008	0.000005	kg gas/mile	Derived from Defra/DECC (2015)
Emission factors for coaches:	0.0287	0.00005	0.00055	$kgCO_2e$/pass.km	Defra/DECC (2015)
Emission factors for coaches:	0.028700	0.000002	0.000002	kg gas/pass.km	Derived from Defra/DECC (2015)

Emission factors for employee owned vehicles:	0.298195	0.000161	0.001545	$kgCO_2e$/mile	Defra/DECC (2015)
Emission factors for employee owned vehicles:	0.298195	0.000008	0.000005	kg gas/mile	Derived from Defra/DECC (2015)
	$kgCO_2e$/£				
Emission factors for hotel night stays:	0.490605				R. Yarr 2014, pers. comm., 28 Oct
	$kgCO_2e$/unit				
Upstream emission factors for flights:	0.01834	$kgCO_2e$/pass.km			Defra/DECC (2015)
Upstream emission factors for rail:	0.00811	$kgCO_2e$/pass.km			Defra/DECC (2015)
Upstream emission factors for bus:	0.02172	$kgCO_2e$/pass.km			Defra/DECC (2015)
Upstream emission factors for ferry:	0.021591	$kgCO_2e$/pass.km			Defra/DECC (2015)
Upstream emission factors for taxi:	0.052569	$kgCO_2e$/km			Defra/DECC (2015)
Upstream emission factors for hire cars:	0.060898	$kgCO_2e$/mile			Defra/DECC (2015)
Upstream emission factors for coach:	0.00621	$kgCO_2e$/pass.km			Defra/DECC (2015)
Upstream emission factors for employee owned vehicles:	0.060898	$kgCO_2e$/mile			Defra/DECC (2015)
Emission factors for landfill - paper:	490.0	$kgCO_2e$/tonne			Defra/DECC (2015)
Emission factors for landfill - card:	490.0	$kgCO_2e$/tonne			Defra/DECC (2015)

Emission factors for landfill - food:	723.0	$kgCO_2e$/tonne			Defra/DECC (2015)
Emission factors for landfill - plastics:	34.1	$kgCO_2e$/tonne			Defra/DECC (2015)
Emission factors for landfill - metals:	21.3	$kgCO_2e$/tonne			Defra/DECC (2015)
Emission factors for landfill - asbestos:	2.0	$kgCO_2e$/tonne			Defra/DECC (2015)
Emission factor for recycled waste:	21.0	$kgCO_2e$/tonne			Defra/DECC (2015)
Emission factor for woody biomass (biogenic CO_2):	0.354	$kgCO_2$/kWh			Defra/DECC (2015)
Emission factor for woody biomass (non-CO_2 gases):	0.0132	$kgCO_2e$/kWh			Defra/DECC (2015)
Upstream emission factor for woody biomass:	0.01662	$kgCO_2e$/kWh			Defra/DECC (2015)
Emission factor for water consumption:	0.344	$kgCO_2e/m^3$			Defra/DECC (2015)
Emission factor for water treatment:	0.708	$kgCO_2e/m^3$			Defra/DECC (2015)

Table 16. Activity data used in the consequential methods

Description	Value	Units	Reference
Biomass Plant Data			
Estimated total heat demand per year:	34,305	MWh/yr	Cullinan Studio et al. (2014)
Estimated proportion of total heat load from biomass boiler:	95%	percentage	Cullinan Studio et al. (2014)
Estimated proportion of total heat load from gas boilers:	5%	percentage	Cullinan Studio et al. (2014)
Estimated amount of heat delivered from biomass boiler:	32,590	MWh/yr	derived from above
Efficiency of biomass boiler:	85%	percentage	R. Yarr 2014, pers. comm., 28 Oct
Transmission losses to/from use site - lower estimate:	7%	percentage	R. Yarr 2014, pers. comm., 28 Oct
Transmission losses to/from use site - upper estimate:	15%	percentage	R. Yarr 2014, pers. comm., 28 Oct
Efficiency of baseline gas boilers:	90%	percentage	R. Yarr 2014, pers. comm., 28 Oct
Efficiency of decision scenario gas boilers:	75%	percentage	Cullinan Studio et al. (2014)
Estimated emissions from electricity usage for pumping hot water:	119	tCO_2/yr	Cullinan Studio et al. (2014)
Assumed emissions factor for electricity emissions calculation:	0.4621	$kgCO_2/kWh$	Defra/DECC (2013)
Estimated electricity consumption associated with pumping hot water:	257,503	kWh/yr	derived from above
Estimated total bioenergy input value:	43,080	MWh/yr	derived from above

Lifetime of biomass boiler:	25	years	Cullinan Studio et al. (2014)
Total estimated spend on construction of biomass plant:	£17,869,664	£	Rider Levett Bucknall (2014)
Area of woodland in locality			
The total area of woodland in local area:	16,508	hectares	Fife Council (2013)
Energy Characteristics for Biomass			
Energy content of wood pellets:	4.72	kWh (net)/kg	Defra/DECC (2015)
Energy content of wood chips:	3.89	kWh (net)/kg	Defra/DECC (2015)
Energy content of wood chips:	4.09	kWh (gross)/kg	Defra/DECC (2015)
Energy content of logs:	4.08	kWh (net)/kg	Defra/DECC (2015)
Energy content of MDF:	17.20	MJ/kg	Günther et al. (2012)
Energy content of MDF:	4.78	kWh/kg	derived from above
Energy content of wooden pallets:	4.20	kWh (net)/kg	derived from above
Energy content of wood chips:	13.80	MJ/kg	Biomass Energy Centre (2008)
Conversion from MJ to kWh:	0.28	MJ/kWh	Defra/DECC (2015)
Energy content of wood chips:	3.83	kWh/kg	derived from above
Moisture content of fresh logs:	50%	percentage	DECC (2014)
Moisture content of air dried logs:	25%	percentage	DECC (2014)
Moisture content of air dried logs:	30%	percentage	Whittaker et al. (2011)
Moisture content of wood chips:	25%	percentage	Biomass Energy Centre (2008)
Moisture content of wood pellets:	7%	percentage	DECC (2014)
Moisture content of virgin wood used in particle board:	7%	percentage	WPIF (n.d.)

Moisture content of sawn timber:	7%	percentage	working estimate
Moisture content of fence posts:	10%	percentage	working estimate
Moisture content of pallet wood:	18%	percentage	DECC (2014)
Carbon content of dry woody biomass:	47%	percentage	DECC (2014)
Carbon content of debris and leaf litter:	37%	percentage	DECC (2014)
Density of logs (25% moisture):	425	kg/m3	Defra/DECC (2015)
Density of logs (50% moisture):	567	kg/m3	derived from above
Density of wood pellets:	650	kg/m3	Defra/DECC (2015)
Density of wood chips:	250	kg/m3	Defra/DECC (2015)
Density of sawn timber (7% moisture):	349	kg/m3	derived from above
Soil Carbon Stock Data			
Assumed soil carbon stock at time of harvest:	100	tC /hectare	working estimate
Assumed maximum incremental increase in soil carbon stocks (sequestration shown as negative number):	- 0.80	tC/hectare.year	based on Johnston et al. (1996)
Assumed incremental increase at equilibrium:	0.0	tC/hectare.year	working estimate
Assumed time period for reaching equilibrium:	100	years	working estimate
Intrinsic growth rate (based on assumed 50t C of accumulation before equilibrium):	0.068	intrinsic growth rate	working estimate
Forestry Road Building			
Assumed ratio of road length to harvested area - heavy duty:	0.006	km/hectare	Morison et al. (2012) (based on Whittaker et al. (2010))

Assumed ratio of road length to harvested area - light duty:	0.01	km/hectare	Morison et al. (2012) (based on Whittaker et al. (2010))
Forestry Yields			
Assumed yield of stem and branch wood - local production:	120	t dry mass/hectare	Penman et al. (2006)
Yield of forest residues from harvesting:	23%	percentage	Jurevics (2010)
Assumed yield of residues and leaf litter from harvesting - local production:	28	t dry mass/hectare	derived from above
Assumed yield of stem and branch wood - marginal supply system:	120	t dry mass/hectare	Penman et al. (2006)
Assumed yield of residues and leaf litter from harvesting - marginal supply system:	28	t dry mass/hectare	derived from above
Yield of SRW from under-managed local woodland - low quality:	100%	percentage	D. Leslie 2014, pers. comms., 18 November
Yield of pallet wood from under-managed local woodland - low quality:	0%	percentage	D. Leslie 2014, pers. comms., 18 November
Yield of saw logs from under-managed local woodland - low quality:	0%	percentage	D. Leslie 2014, pers. comms., 18 November
Yield of SRW from under-managed local woodland - higher quality:	20%	percentage	D. Leslie 2014, pers. comms., 18 November
Yield of pallet wood from under-managed local woodland - higher quality:	20%	percentage	D. Leslie 2014, pers. comms., 18 November

Yield of saw logs from under-managed local woodland - higher quality:	60%	percentage	D. Leslie 2014, pers. comms., 18 November
Yield of SRW from world marginal timber supply:	20%	percentage	working estimate
Yield of pallet wood from world marginal timber supply:	20%	percentage	working estimate
Yield of saw logs from world marginal timber supply:	60%	percentage	working estimate
Yield of SRW from un-thinned woodland:	60%	percentage	working estimate
Yield of pallet wood from un-thinned woodland:	20%	percentage	working estimate
Yield of saw logs from un-thinned woodland:	20%	percentage	working estimate
Yield of SRW from thinned woodland:	20%	percentage	working estimate
Yield of pallet wood from thinned woodland:	20%	percentage	working estimate
Yield of saw logs from thinned woodland:	60%	percentage	working estimate
Assumed yield of stem and branch wood without thinning - at time of harvest:	120	t dry mass/hectare	working estimate
Assumed yield of stem and branch wood with thinning - at time of harvest:	120	t dry mass/hectare	working estimate
Assumed amount of stem and branch wood after harvest:	0.01	tC/hectare	working estimate
Assumed intrinsic growth rate for forest - local:	0.051	intrinsic growth rate	working estimate
Assumed intrinsic growth rate for forest - marginal system:	0.075	intrinsic growth rate	working estimate

Value for asymptote at which maximum growth occurs:	0.19		working estimate
Forestry Operations			
Diesel consumption for felling:	1.55	Litres of diesel/m3 of felled timber	Morison et al. (2012)
Diesel consumption for transportation to roadside:	0.90	Litres of diesel/m3 of felled timber	Morison et al. (2012)
Assumed loss of soil carbon due to harvesting operations:	23.5	tCO_2e/hectare	based on Morison et al. (2012)
Assumed harvesting cycle:	57	years	working estimate
Chipping			
Assumed input/output ratio for chipping:	1.00	t input/t output	working estimate
Emissions from chipping - lower estimate:	0.003	tCO_2e/tonne of output	DECC (2014)
Emissions from chipping - upper estimate:	0.008	tCO_2e/tonne of output	DECC (2014)
Pelletising			
Emissions from pelletising wood:	0.115	tCO_2/oven dried tonne of wood pellets	Morison et al. (2012)
Emissions from drying - lower estimate:	0.127	tCO_2e/tonne of output	DECC (2014)

Emissions from drying - upper estimate:	0.184	tCO_2e/tonne of output	DECC (2014)
Emissions from pelletising - lower estimate:	0.083	tCO_2e/tonne of output	DECC (2014)
Emissions from pelletising - upper estimate:	0.083	tCO_2e/tonne of output	DECC (2014)
Transportation Emissions			
Assumed distance from harvest operations to pellet plant:	50	km	based on DECC (2014)
Assumed mode of transport:	Truck	mode	based on DECC (2014)
Assumed distance from pellet plant to shipping hub:	50	km	based on DECC (2014)
Assumed mode of transport:	Truck	mode	based on DECC (2014)
Assumed distance from shipping hub to consumer country:	7,200	km	based on DECC (2014)
Assumed mode of transport:	Product tanker	mode	based on DECC (2014)
Assumed distance from consumer shipping hub to power plant:	50	km	based on DECC (2014)
Assumed mode of transport:	Train	mode	based on DECC (2014)
Assumed distance from harvest operations to saw mill:	50	km	working estimate
Assumed mode of transport:	Truck	mode	working estimate
Assumed distance from local woodland to sawmill:	50	km	working estimate

Assumed mode of transport:	Truck	mode	working estimate
Load of a standard haulage truck:	25	tonnes/load	D. Leslie 2014, pers. comms., 18 November
Assumed distance from saw mill to shipping hub:	50	km	based on DECC (2014)
Assumed mode of transport:	Truck	mode	based on DECC (2014)
Assumed distance from shipping hub to timber consumer country:	7,200	km	based on DECC (2014)
Assumed mode of transport:	Product tanker	mode	based on DECC (2014)
Assumed distance from consumer shipping hub to timber consumer:	50	km	based on DECC (2014)
Assumed mode of transport:	Train	mode	based on DECC (2014)
Assumed distance from additional local harvesting to biomass plant:	40	km	working estimate
Assumed mode of transport:	Truck	mode	working estimate
Assumed distance from thinning site to biomass plant:	50	km	working estimate
Assumed mode of transport:	Truck	mode	working estimate
Exponential Decay Functions			
Exponential decay function constant for coarse woody debris (>10cm):	0.083	decay constant	Mattson et al. (1987)
Exponential decay function constant for coarse woody debris (>10cm):	0.011	decay constant	Chambers et al. (2000)

Exponential decay function constant for fine woody debris (<10cm):	0.185	decay constant	Mattson et al. (1987)
Exponential decay function constant for fine woody debris (<10cm):	0.097	decay constant	Vávřová et al. (2009)
Assumed exponential decay function for wooden fence posts:	0.2	decay constant	working estimate
Particle Board			
Weight of virgin wood in particle board:	0.21	oven dried tonnes wood/m^3	DECC (2014)
Percentage of particle board made up of virgin wood:	25%	percentage	DECC (2014)
Assumed thickness of particle board:	40	mm	Euroform (2015)
Assumed life span of particle board:	99	years	DECC (2014)
Breeze block			
Quantity of breeze block required per unit of particle board:	41.7	m2/m3 of particle board	DECC (2014)
Density of medium density concrete blocks:	1400	kg/m^3	Gryphonn (2015)
Quantity of block per m2:	202	kg/m^2	Gryphonn (2015)
Assumed thickness of concrete blocks:	140	mm	Gryphonn (2015)
Sawn timber			
Assumed thickness of sawn timber:	40	mm	based on Euroform (2015)

Medium Density Fibre Board			
Weight of wood fibre in MDF:	1.029	oven dried tonnes wood/m3 MDF	DECC (2014)
Assumed life span of MDF:	99	years	DECC (2014)
Plasterboard			
Quantity of plasterboard required per unit of MDF:	1.39	m2/m3 of MDF	DECC (2014)
Assumed thickness of MDF:	0.012	metres	Australian Wood Panels Association Incorporated (2008)
Number of m2 per m3:	83.3	m2/m3	derived from above
Assumed thickness of plasterboard:	0.012	metres	working estimate
Quantity of plasterboard required per unit of MDF:	1.0	m2 plasterboard/m2 of MDF	derived from above
Wooden Pallets			
Weight of wood fibre in wooden pallets:	0.82	oven dried tonnes wood/tonne of pallet	DECC (2014)
Assumed water content of wood fibre used in pallet:	18%	percentage	DECC (2014)
Assumed life span of wooden pallet:	5	years	DECC (2014)
Assumed weight of a wooden pallet:	21	kg/1000 kg capacity pallet	DECC (2014)
Plastic Pallets			
Assumed life span of plastic pallet:	20	years	DECC (2014)
Assumed weight of a plastic pallet:	17	kg/1000 kg capacity pallet	DECC (2014)

Wooden Fencing			
Weight of wood fibre in wooden fence posts:	0.82	oven dried tonnes wood/tonne of wooden fencing	DECC (2014)
Assumed moisture content of wood fibre used in pallet:	18%	percentage	DECC (2014)
Assumed life span of wooden fencing:	20	years	DECC (2014)
Assumed weight to area ratio for wooden fencing:	78	kg wooden fencing/m2 of fencing	DECC (2014)
Concrete Fencing			
Assumed life span of concrete fencing:	60	years	DECC (2014)
Assumed weight to area ratio for concrete fencing:	236	kg concrete fencing/m2 of fencing	DECC (2014)
Replanting			
Number of seedlings per hectare:	2,500	seedlings/hectare	FAO (2015)
Diesel usage for replanting:	31	MJ/hectare	Biomass Energy Centre (2008)
Thinnings			
Difference between thinning and no thinning - total cumulative sequestration (tCO_2) over 60 years:	78.05	tCO_2/hectare	derived from Morison et al. (2012)
Percentage of biomass removed from forest (in 55 year cycle):	10%	percentage of total sequestration	derived from Morison et al. (2012)

Ratio of thinnings to foregone sequestration/stored carbon:	1.3	quantity of foregone sequestration/quantity of thinnings	derived from above
Amount of additional sequestration/stored carbon without thinning:	32%	percentage	derived from above
Assumed first year of thinning:	20	years	derived from Morison et al. (2012)
Assumed year of harvesting:	57	years	working estimate
Assumed period of thinning:	37	years	derived from above
Average distance from thinning site to use site:	50	km	working estimate
Cement Renders			
Assumed thickness of cement render:	16	mm	Armourcoat (2015)
Timber Cladding			
Assumed thickness of timber cladding:	19	mm	Russwood (2015)
Chapman-Richards Function (for modelling forest growth)			$\boldsymbol{Y(t) = A + \frac{K - A}{(1 + Qe^{-Bt})^{1/v}}}$ A: the lower asymptote K: the upper asymptote B: the growth rate

			V > 0 Q: is related to the value Y(0)

Table 17. Emission factors used in the consequential methods

Description	**Value**	**Units**	**Reference**
Wood pellet transportation - US			
Truck	0.1099525	$kgCO_2/t.km$	DECC (2014)
Shipping - product tanker	0.006077929	$kgCO_2/t.km$	DECC (2014)
Shipping - inland bulk carrier	0.030200889	$kgCO_2/t.km$	DECC (2014)
Rail	0.01748425	$kgCO_2/t.km$	DECC (2014)
Predicted reduction in transport emissions - 2020 - trucks:	12.4%	percentage	DECC (2014)
Predicted reduction in transport emissions - 2030 - trucks:	30.0%	percentage	DECC (2014)
Predicted reduction in transport emissions - 2020 - rail:	15%	percentage	DECC (2014)
Predicted reduction in transport emissions - 2030 - rail:	30%	percentage	DECC (2014)
Predicted reduction in transport emissions - 2020 - other:	20.0%	percentage	DECC (2014)
Predicted reduction in transport emissions - 2030 - other:	30.0%	percentage	DECC (2014)
Natural gas			

Upstream emissions from natural gas:	0.02759	$kgCO_2e$/kWh (net)	Defra/DECC (2015)
Point of combustion emissions from natural gas:	0.20711	$kgCO_2e$/kWh (net)	Defra/DECC (2015)
Point of combustion emissions for natural gas:	360	gCO_2/kWh	Logan et al. (2012)
Whole of life emissions from shale gas:	440	gCO_2/kWh	Logan et al. (2012)
Forestry road construction			
Emissions from road construction:	42	tCO_2e/km	Morison et al. (2012) (based on Whittaker et al. (2010))
Diesel			
Emissions from diesel - point of combustion:	2.717	$kgCO_2e$/litre	Defra/DECC (2015)
Emissions from diesel - upstream emissions:	0.580	$kgCO_2e$/litre	Defra/DECC (2015)
Particle board			
Emissions from processing and transportation of particle board:	403	$kgCO_2e/m^3$	DECC (2014)
Breeze block			
Embodied emissions of breeze blocks:	49	$kgCO_2e/m^2$	DECC (2014)
Embodied emissions of aerated blocks:	0.08	$kgCO_2$/kg	Hammond (2008)
MDF			
Emissions factor for the production of MDF:	790	$kgCO_2e/m^3$ of MDF	DECC (2014)
Plasterboard			

Embodied emissions of plasterboard:	4	$kgCO_2e/m^2$	DECC (2014)
Embodied emissions of plasterboard:	0.38	$kgCO_2/kg$	Hammond (2008)
Wooden pallets			
Embodied emissions of wooden pallets:	62	$kgCO_2e$/tonne of pallet	DECC (2014)
Plastic pallets			
Embodied emissions of plastic pallets:	183	$kgCO_2e$/tonne of plastic pallet	DECC (2014)
Wooden fencing			
Emissions factor for the production of wooden fencing:	76	$kgCO_2e$/tonne of fencing	DECC (2014)
Concrete fencing			
Embodied emissions of concrete fencing:	252	$kgCO_2e/m_2$ of concrete fencing	DECC (2014)
Embodied emissions of concrete fencing:	0.096	$kgCO_2e$/kg of concrete block	Hammond (2008)
Sawn timber			
Emissions from processing and transportation of sawn timber:	71	$kgCO_2e$/tonne of sawn timber	DECC (2014)
Seedlings			
Embodied emissions of seedlings:	0.0567	$kgCO_2$/seedling	Biomass Energy Centre (2008)
Chipping			

Assumed input/output ratio for chipping:	1	t input/t output	DECC (2014)
Emissions from chipping - lower estimate:	0.003	tCO_2e/tonne of output	DECC (2014)
Emissions from chipping - upper estimate:	0.008	tCO_2e/tonne of output	DECC (2014)
Pelletising			
Emissions from pelletising - lower estimate:	0.083	tCO_2e/tonne of output	DECC (2014)
Embodied emissions from construction			
Input-output emission factor for construction:	0.494	$kgCO_2e$/£ spent	Defra/DECC (2012)
Drying wood			
Emissions from drying - lower estimate:	0.127	tCO_2e/tonne of output	DECC (2014)
Emissions from drying - upper estimate:	0.184	tCO_2e/tonne of output	DECC (2014)
Thinning			
Emissions from harvesting whole tree thinnings:	85	$kgCO_2e$/ODT	Whittaker et al. (2011)
Emissions from harvesting brash bales:	52	$kgCO_2e$/ODT	Whittaker et al. (2011)
Cement			
Embodied emissions of cement mortar (1:3 cement and sand mix):	0.213	$kgCO_2$/kg	Hammond (2008)
Transportation emission factors over time			
Trucks			
Emissions efficiency of trucks - 2014 value:	0.110	$kgCO_2e$/t.km	DECC (2014)

Emission reduction by 2020 - trucks	0.124	percentage	DECC (2014)
Emissions efficiency of trucks - 2020 value:	0.096	$kgCO_2e/t.km$	derived from above
Number of years between 2020 and 2014:	6.000	years	
Annual incremental change in emissions factor for trucks - 2014 to 2020:	- 0.002	$kgCO_2e/t.km$	derived from above
Emissions efficiency of trucks - 2020 value:	0.096	$kgCO_2e/t.km$	derived from above
Emission reduction by 2030 - trucks	0.300	percentage	DECC (2014)
Emissions efficiency of trucks - 2030 value:	0.077	$kgCO_2e/t.km$	derived from above
Number of years between 2030 and 2020:	10.000	years	
Annual incremental change in emissions factor for trucks - 2020 to 2030:	- 0.002	$kgCO_2e/t.km$	derived from above
Product tankers			
Emissions efficiency of product tankers - 2014 value:	0.006	$kgCO_2e/t.km$	DECC (2014)
Emission reduction by 2020 - shipping	0.200	percentage	DECC (2014)
Emissions efficiency of product tankers - 2020 value:	0.005	$kgCO_2e/t.km$	derived from above
Number of years between 2020 and 2014:	6.000	years	
Annual incremental change in emissions factor for product tankers - 2014 to 2020:	- 0.0002	$kgCO_2e/t.km$	derived from above
Emissions efficiency of product tankers - 2020 value:	0.005	$kgCO_2e/t.km$	derived from above
Emission reduction by 2030 - shipping	0.300	percentage	DECC (2014)
Emissions efficiency of product tankers - 2030 value:	0.004	$kgCO_2e/t.km$	derived from above

Number of years between 2030 and 2020:	10.000	years	
Annual incremental change in emissions factor for product tankers - 2020 to 2030:	- 0.0001	$kgCO_2e/t.km$	derived from above
Rail			
Emissions efficiency of rail - 2014 value:	0.017	$kgCO_2e/t.km$	DECC (2014)
Emission reduction by 2020 - rail	0.150	percentage	DECC (2014)
Emissions efficiency of rail - 2020 value:	0.015	$kgCO_2e/t.km$	derived from above
Number of years between 2020 and 2014:	6.000	years	
Annual incremental change in emissions factor for rail - 2014 to 2020:	- 0.0004	$kgCO_2e/t.km$	derived from above
Emissions efficiency of rail - 2020 value:	0.015	$kgCO_2e/t.km$	derived from above
Emission reduction by 2030 - rail	0.300	percentage	DECC (2014)
Emissions efficiency of rail - 2030 value:	0.012	$kgCO_2e/t.km$	derived from above
Number of years between 2030 and 2020:	10.000	years	
Annual incremental change in emissions factor for rail - 2020 to 2030:	- 0.0003	$kgCO_2e/t.km$	derived from above
Year	Emissions factor for trucks ($kgCO_2e/t.km$):	Emissions factor for product tankers ($kgCO_2e/t.km$):	Emissions factor for trains ($kgCO_2e/t.km$):
2016	0.105426122	0.005672734	0.016610038
2017	0.103162933	0.005470136	0.016172931

2018	0.100899744	0.005267538	0.015735825
2019	0.098636555	0.005064941	0.015298719
2020	0.096373366	0.004862343	0.014861613
2021	0.094432705	0.004801564	0.014599349
2022	0.092492043	0.004740785	0.014337085
2023	0.090551381	0.004680005	0.014074821
2024	0.08861072	0.004619226	0.013812558
2025	0.086670058	0.004558447	0.013550294
2026	0.084729397	0.004497667	0.01328803
2027	0.082788735	0.004436888	0.013025766
2028	0.080848073	0.004376109	0.012763503
2029	0.078907412	0.00431533	0.012501239
2030	0.07696675	0.00425455	0.012238975
2031	0.07696675	0.00425455	0.012238975
2032	0.07696675	0.00425455	0.012238975
2033	0.07696675	0.00425455	0.012238975
2034	0.07696675	0.00425455	0.012238975
2035	0.07696675	0.00425455	0.012238975
2036	0.07696675	0.00425455	0.012238975
2037	0.07696675	0.00425455	0.012238975
2038	0.07696675	0.00425455	0.012238975
2039	0.07696675	0.00425455	0.012238975
2040	0.07696675	0.00425455	0.012238975
2041	0.07696675	0.00425455	0.012238975
2042	0.07696675	0.00425455	0.012238975
2043	0.07696675	0.00425455	0.012238975

UK electricity grid emission factors over time			
Emissions from UK grid electricity - generation (2013):	0.46213	$kgCO_2e/kWh$	Defra/DECC (2015)
Emissions from UK grid electricity - T&D losses (2013):	0.03816	$kgCO_2e/kWh$	Defra/DECC (2015)
Emissions from UK grid electricity - upstream emissions - generation (2013):	0.04902	$kgCO_2e/kWh$	Defra/DECC (2015)
Total emissions from UK grid electricity consumption:	0.54931	$kgCO_2e/kWh$	derived from above
Assumed decarbonisation of the electricity grid:	4%	percentage decrease/yr	derived from Keep Scotland Beautiful (2011)
Year	Emissions from UK grid electricity - $kgCO_2e/kWh$:		
2016	0.484362984		derived from above
2017	0.463615387		derived from above
2018	0.44375651		derived from above
2019	0.424748284		derived from above
2020	0.406554271		derived from above
2021	0.389139596		derived from above
2022	0.372470875		derived from above
2023	0.356516156		derived from above
2024	0.341244854		derived from above
2025	0.326627695		derived from above
2026	0.312636659		derived from above
2027	0.299244927		derived from above
2028	0.286426827		derived from above

2029	0.274157789		derived from above
2030	0.262414292		derived from above
2031	0.251173825		derived from above
2032	0.240414842		derived from above
2033	0.230116718		derived from above
2034	0.220259712		derived from above
2035	0.210824929		derived from above
2036	0.201794283		derived from above
2037	0.193150463		derived from above
2038	0.184876899		derived from above
2039	0.176957732		derived from above
2040	0.169377781		derived from above
2041	0.162122516		derived from above
2042	0.155178029		derived from above
2043	0.148531007		derived from above
2044	0.14216871		derived from above
2045	0.13607894		derived from above
2046	0.130250024		derived from above
2047	0.124670789		derived from above
2048	0.11933054		derived from above
2049	0.114219038		derived from above
2050	0.109326488		derived from above

2. Causal Chain Maps Used for the Consequential Methods

Figure 20. Causal-chain map for Scenario 1.1 (overseas production – sustainable forest management)

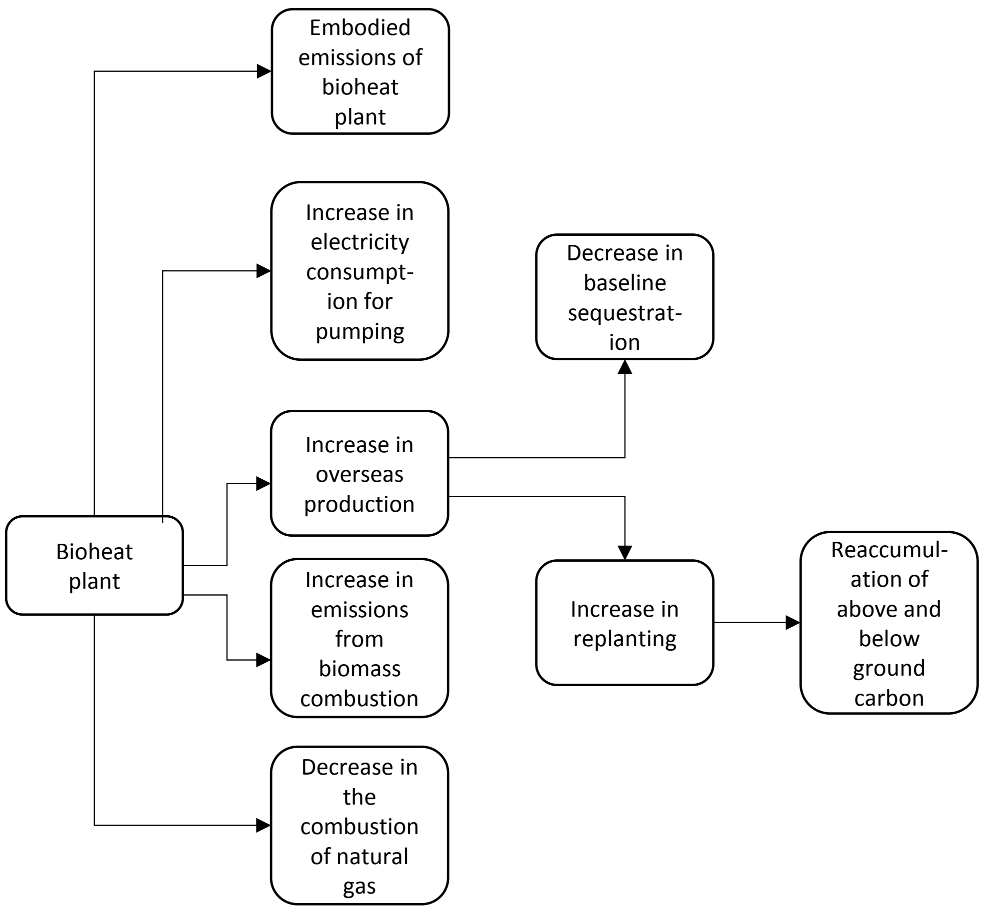


Figure 21. Causal-chain map for Scenario 1.2 (overseas production – unsustainable forest management)

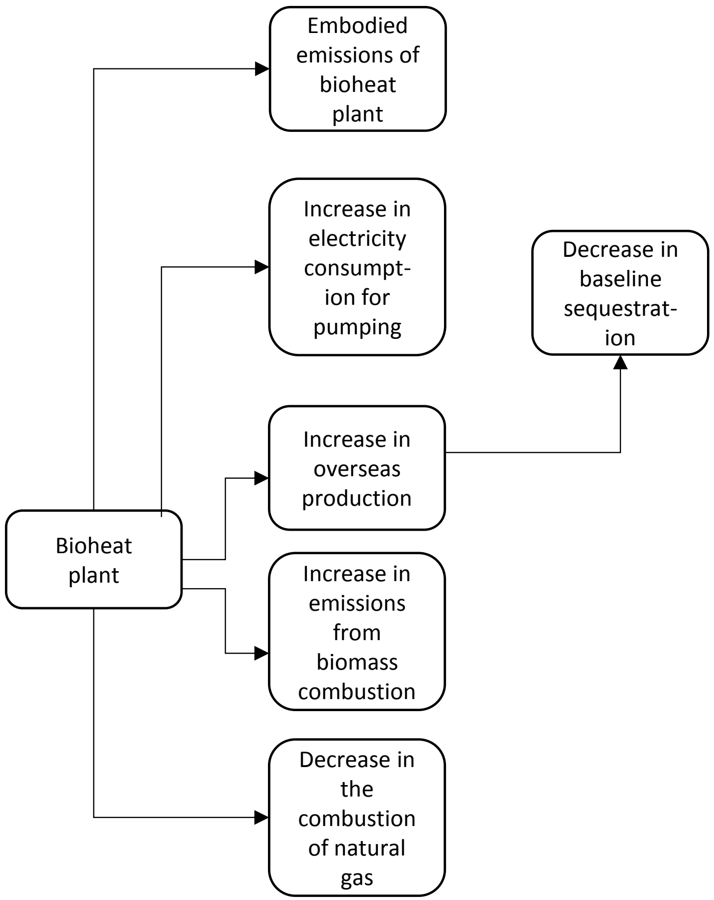


Figure 22. Causal-chain map for Scenario 2.1 (local production without co-products)

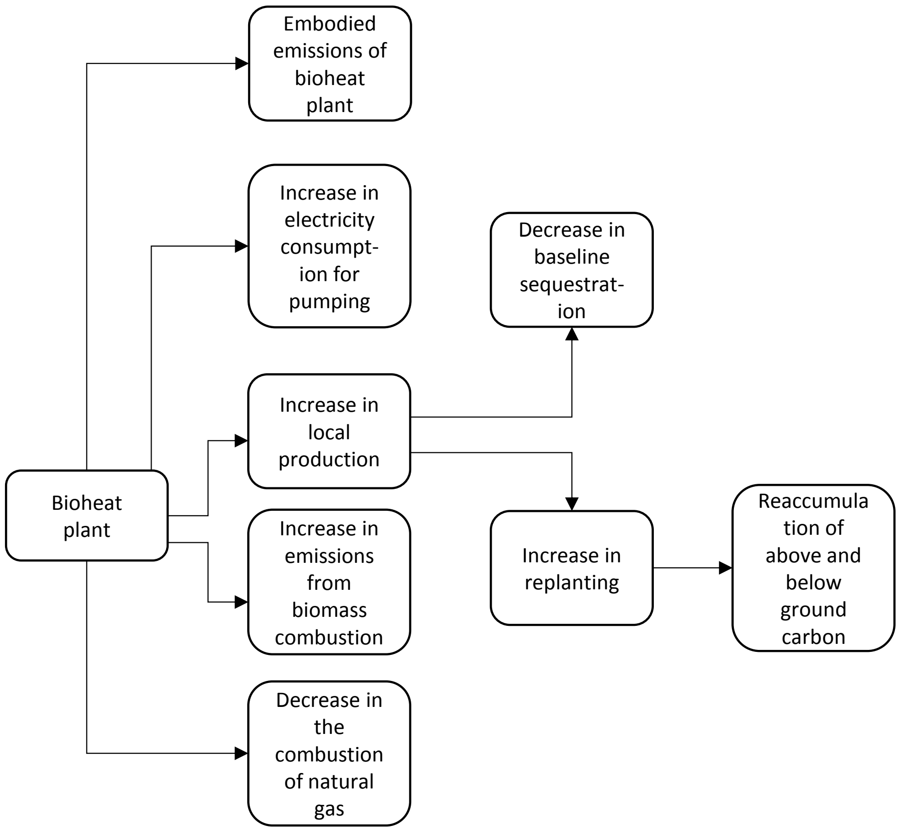


Figure 23. Causal-chain map for Scenario 2.2 (local production with co-products)

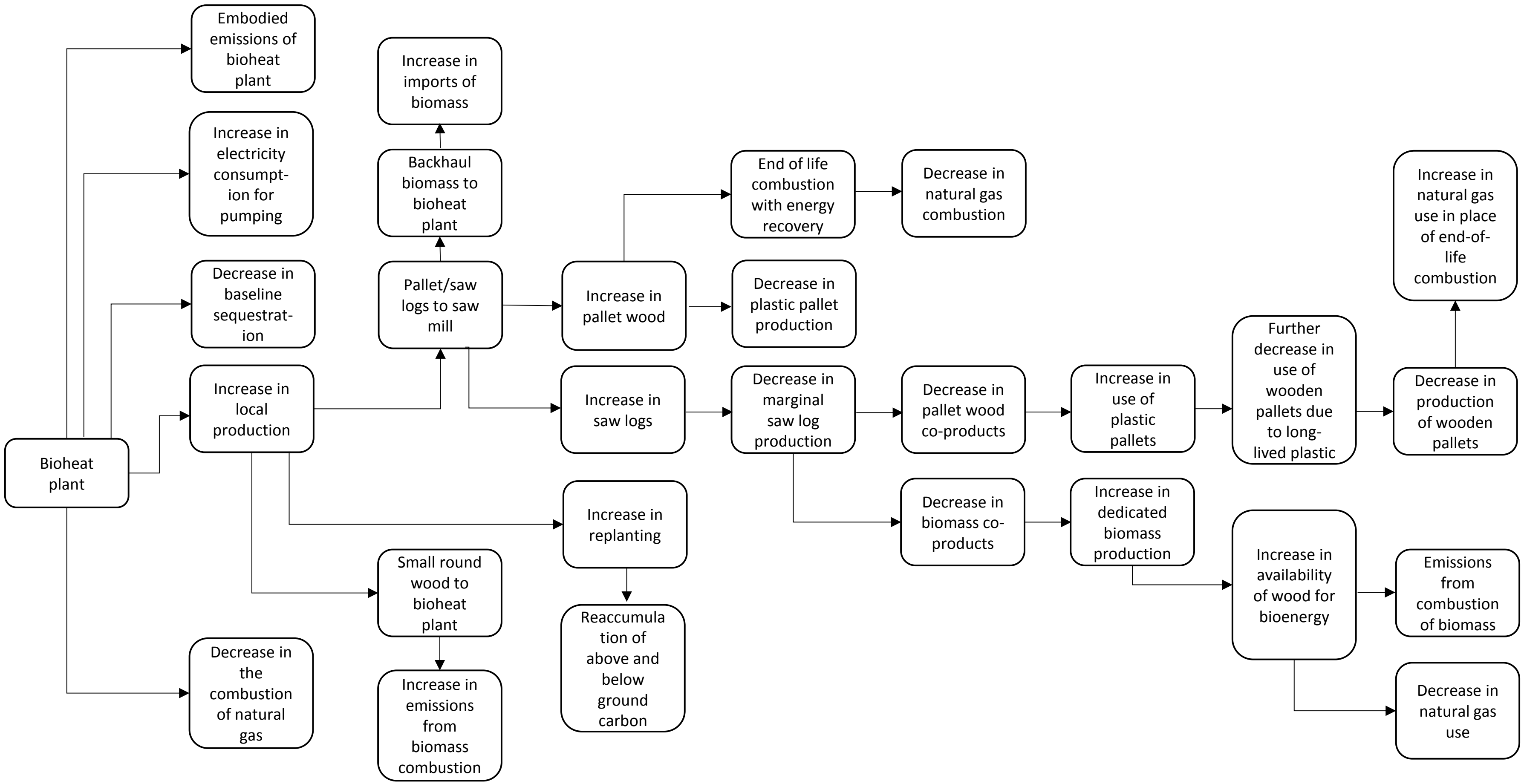


Figure 24. Causal-chain map for Scenario 3.1 (thinnings without co-products)

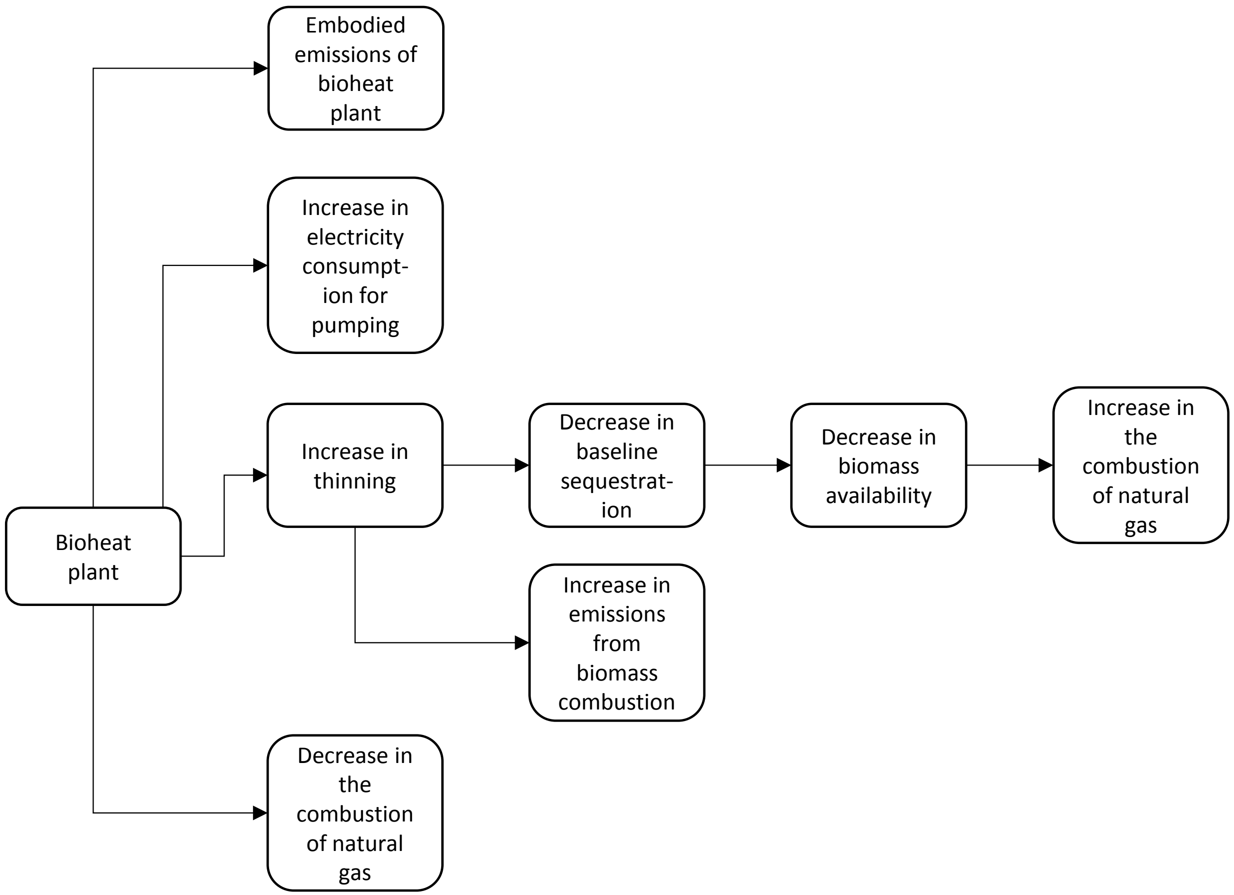


Figure 25. Causal-chain map for Scenario 3.2 (thinnings with co-products – sawlog displacement)

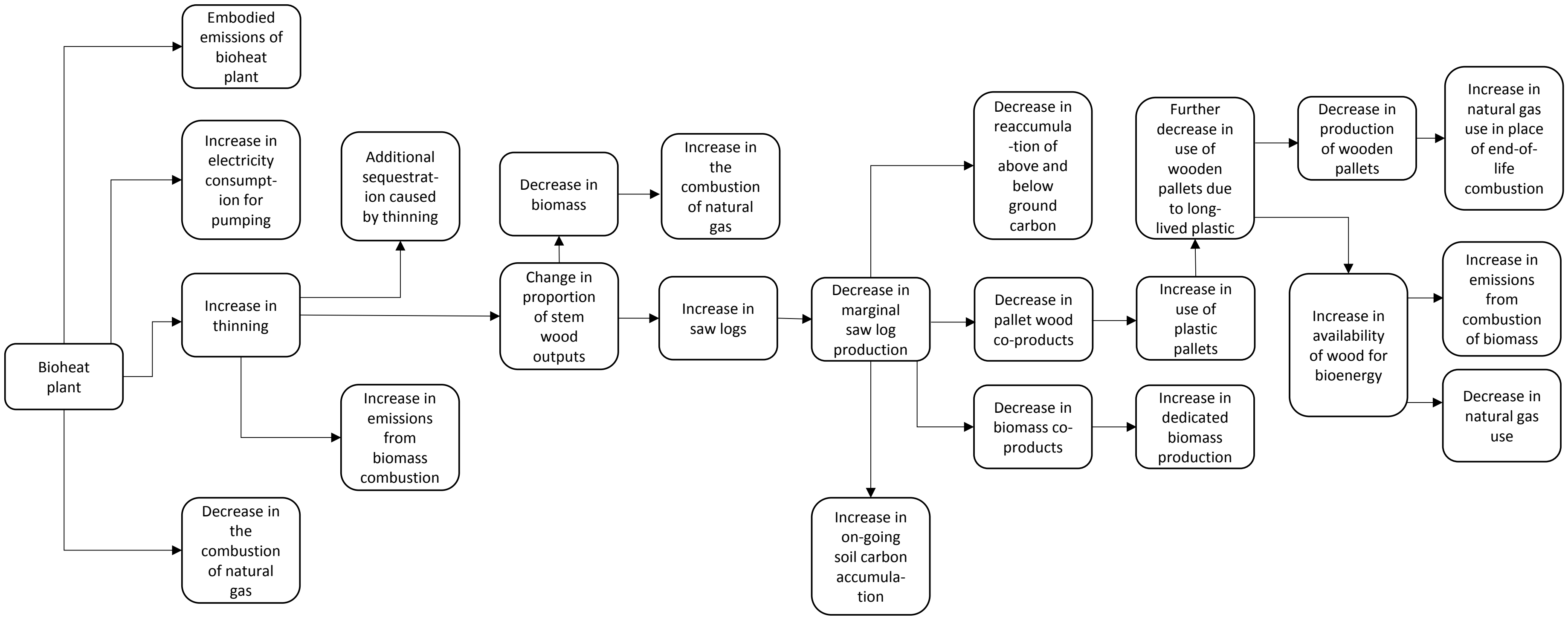


Figure 26. Causal-chain map for Scenario 3.3 (thinnings with co-products – cement render displacement)

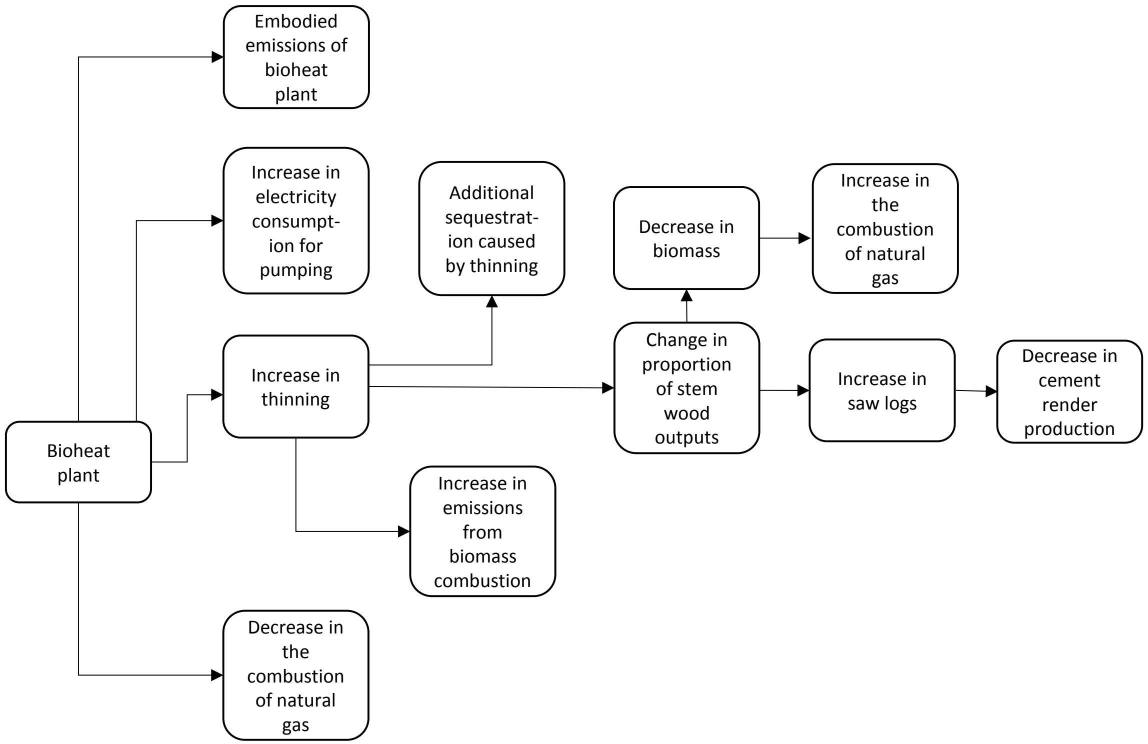


Figure 27. Causal-chain map for Scenario 4.1 (fencing with end-of-life combustion (natural gas displacement))

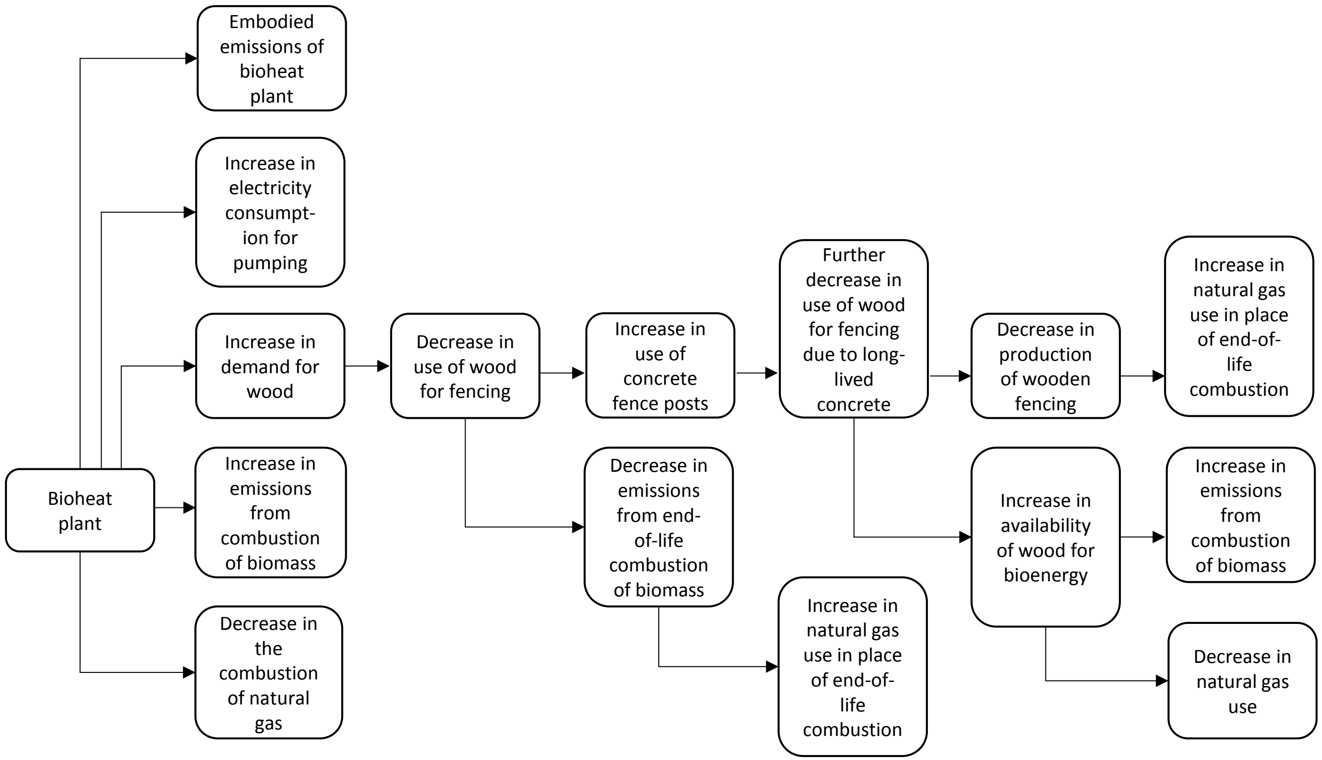


Figure 28. Causal-chain map for Scenario 4.2 (fencing with end-of-life decay)

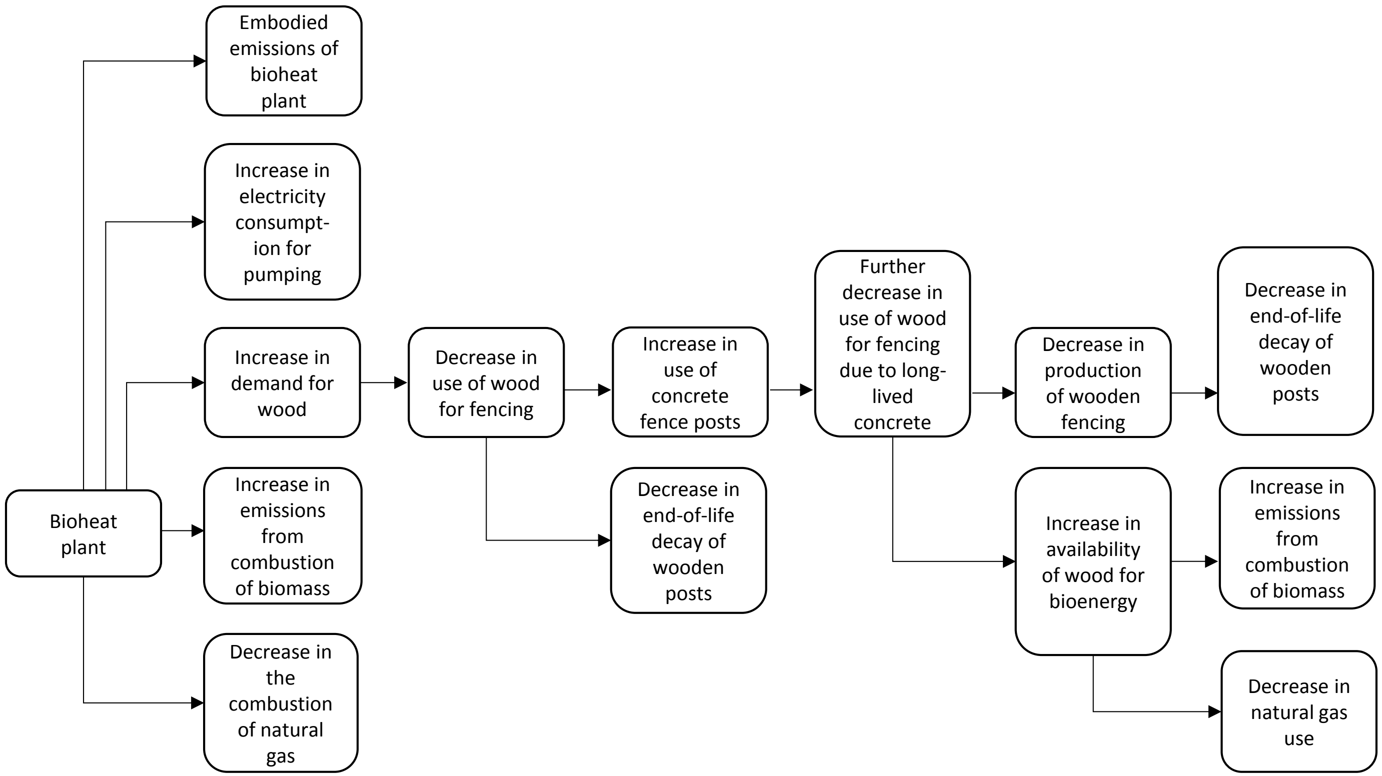


Figure 29. Causal-chain map for Scenario 5.1 (displaced wooden pallets with freed-up biomass displacement)

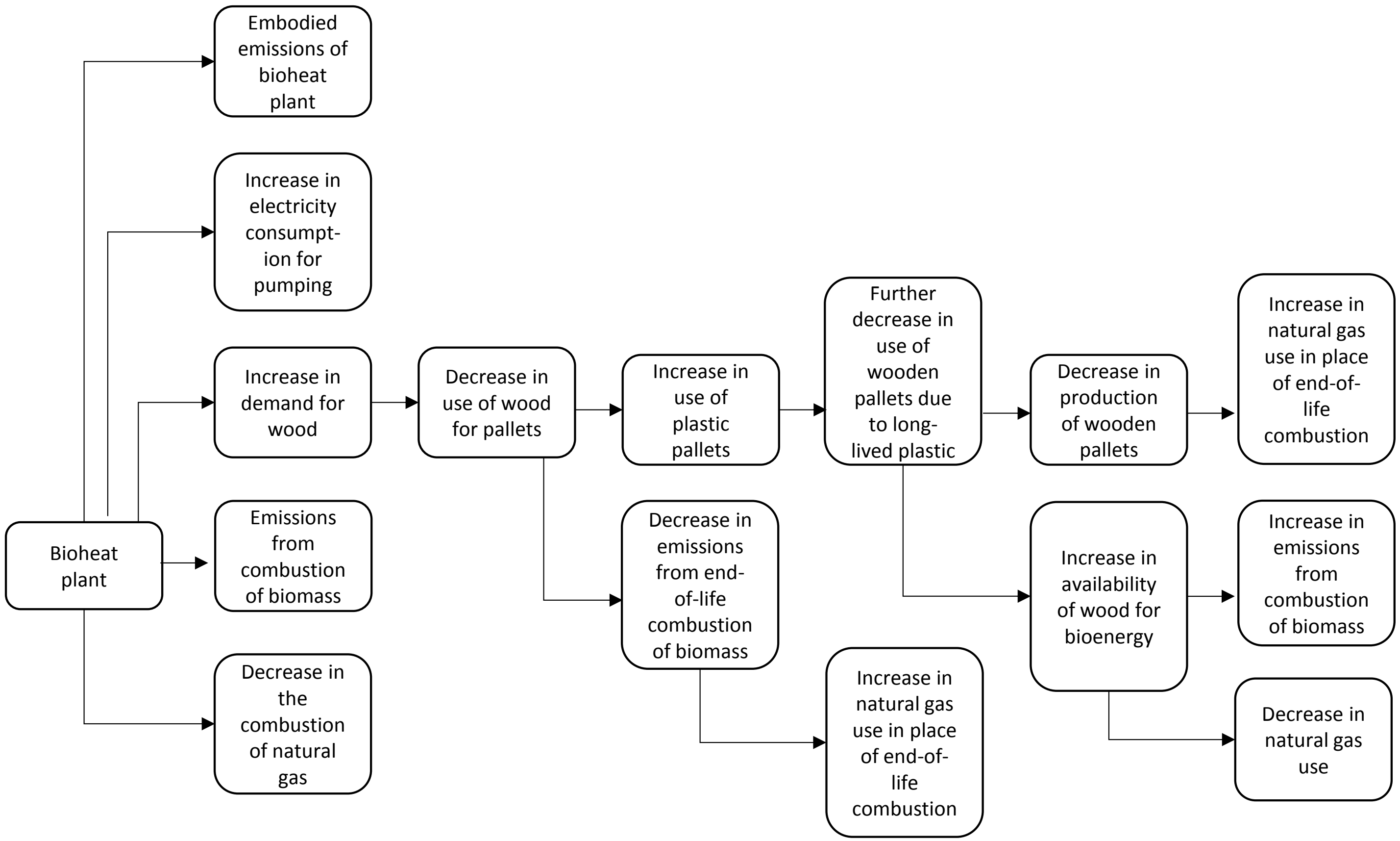


Figure 30. Causal-chain map for Scenario 6 (MDF displacement)

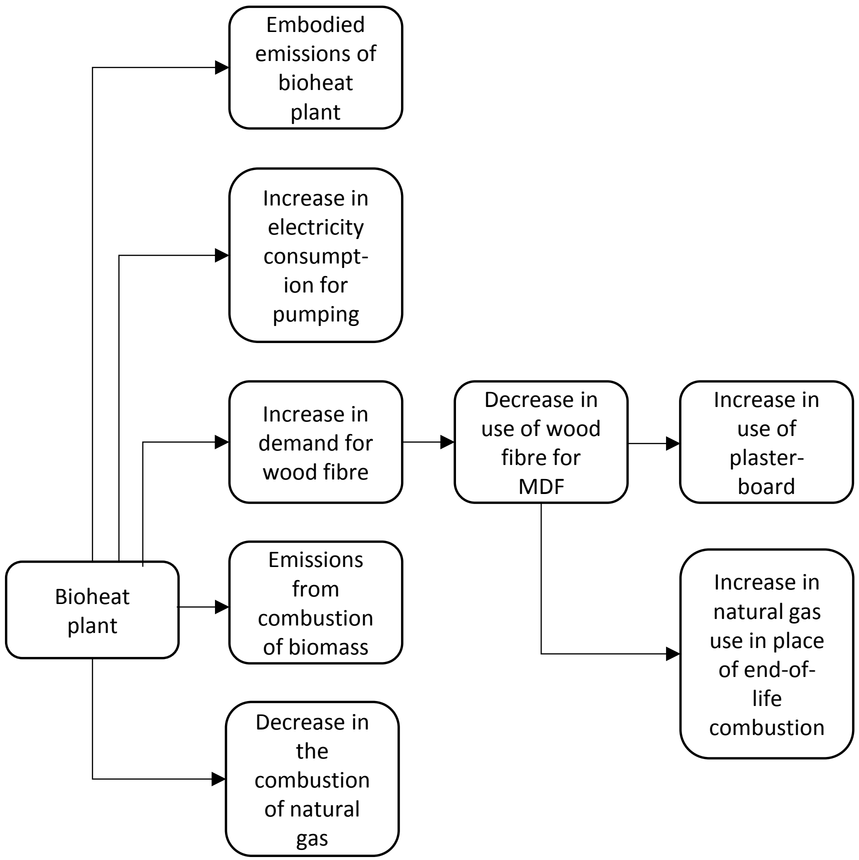
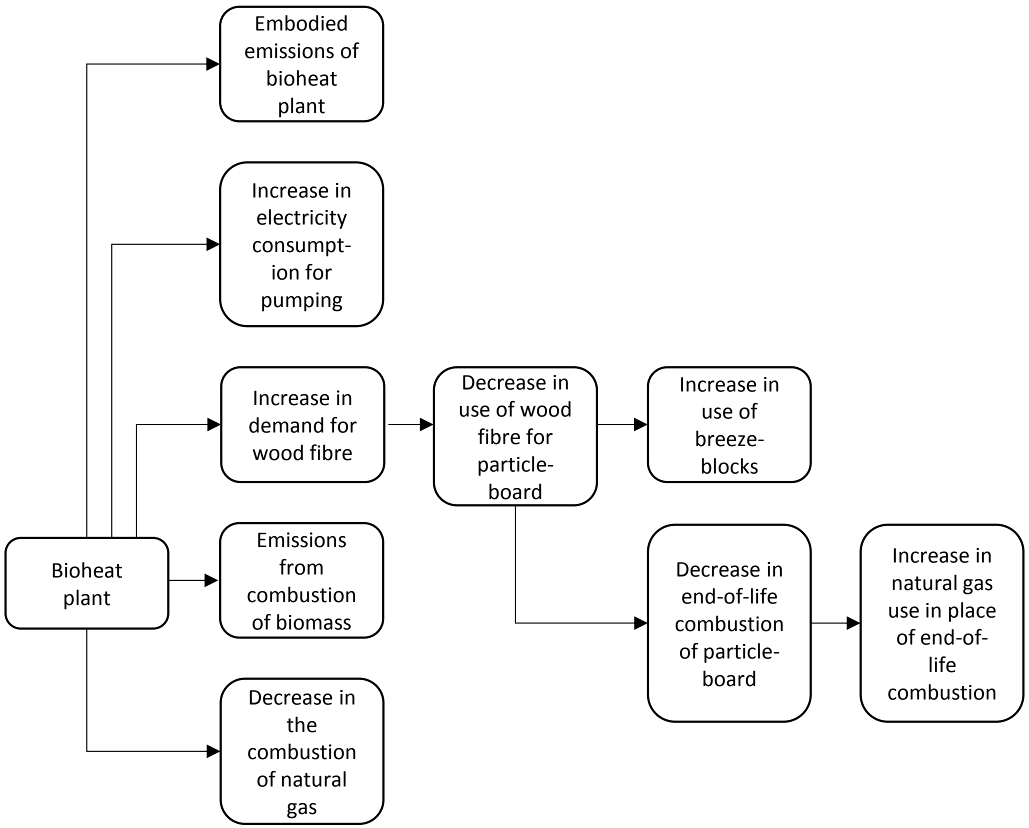


Figure 31. Causal-chain map for Scenarios 7.1 and 7.1 (particleboard displacement – with higher and lower emission factors for breeze blocks)



3. Results – Times-Series Outputs from the Project/Policy Method

Figure 32. Project/policy method time-series results for Scenario 1.1 (overseas production – sustainable forest management)

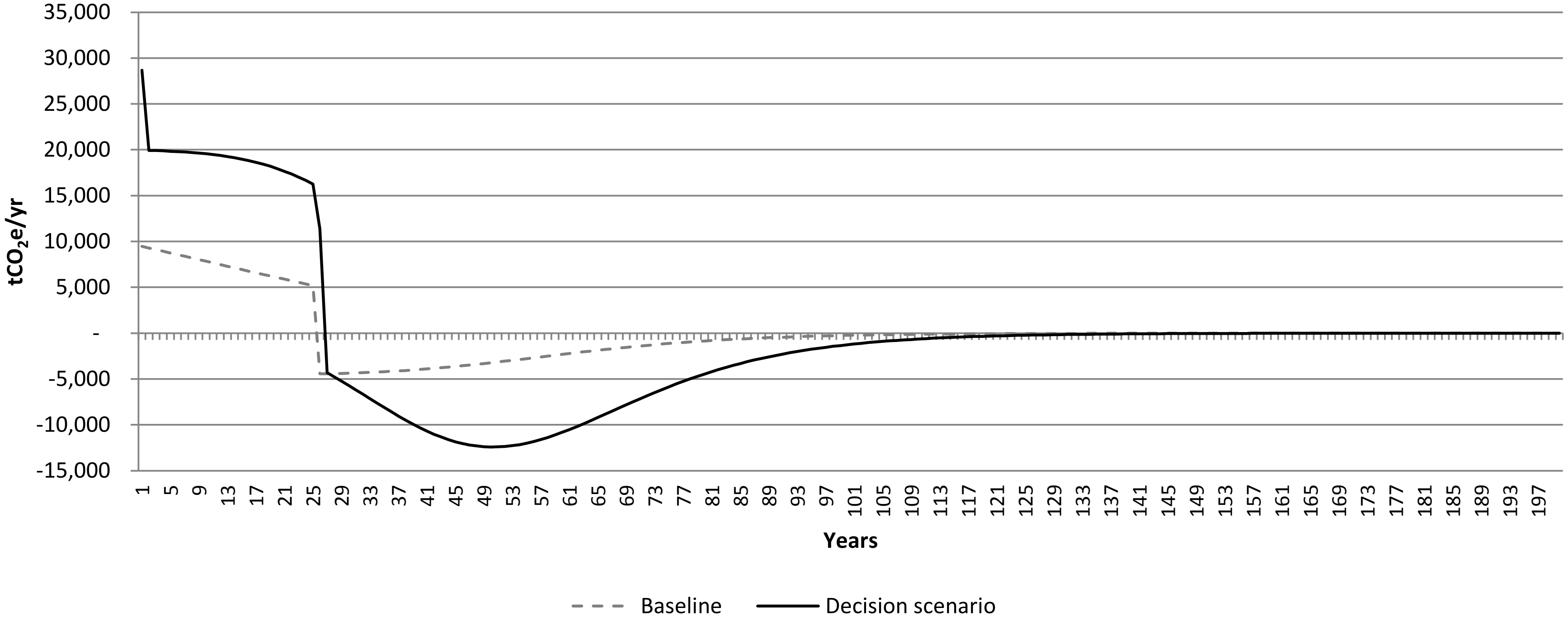


Figure 33. Project/policy method time-series results for Scenario 1.2 (overseas production – unsustainable forest management)

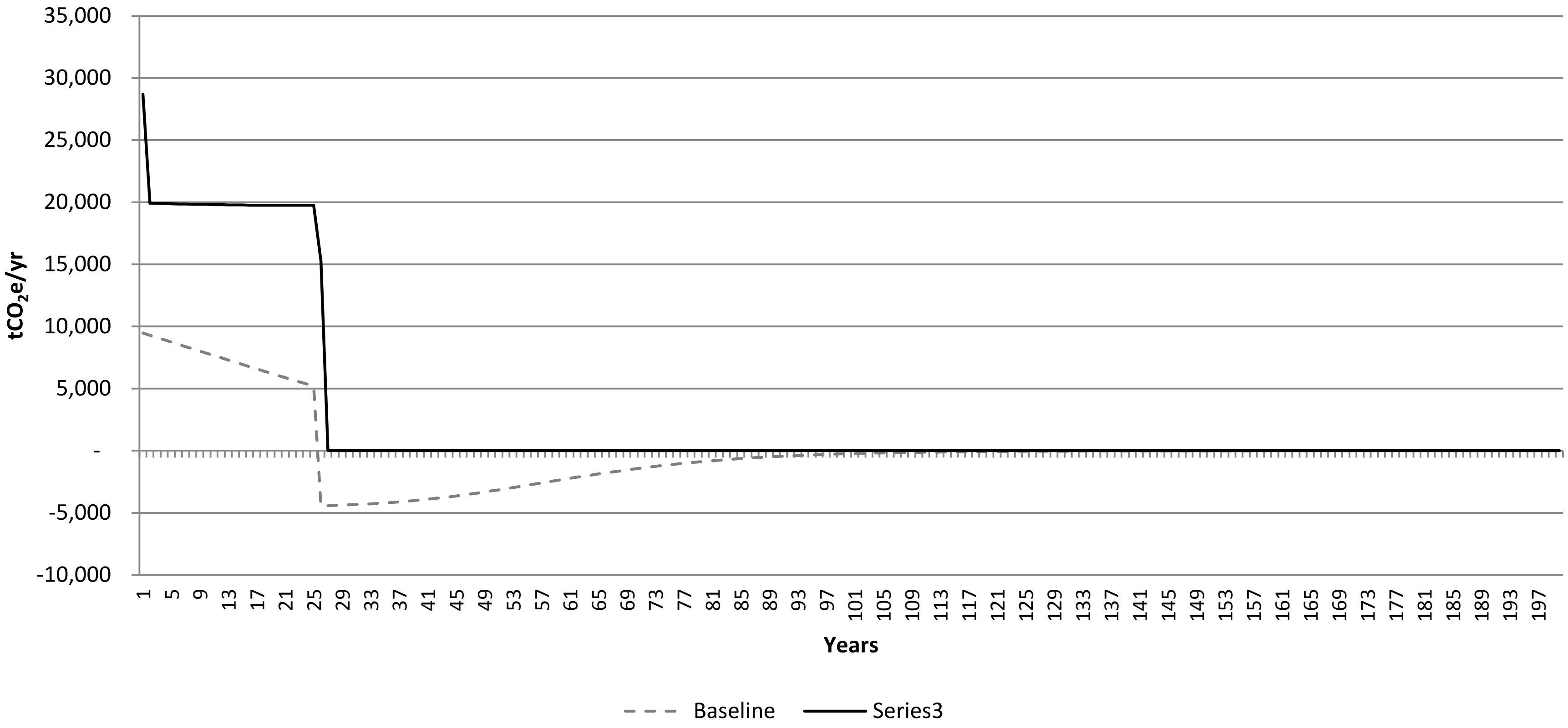


Figure 34. Project/policy method time-series results for Scenario 2.1 (local production without co-products)

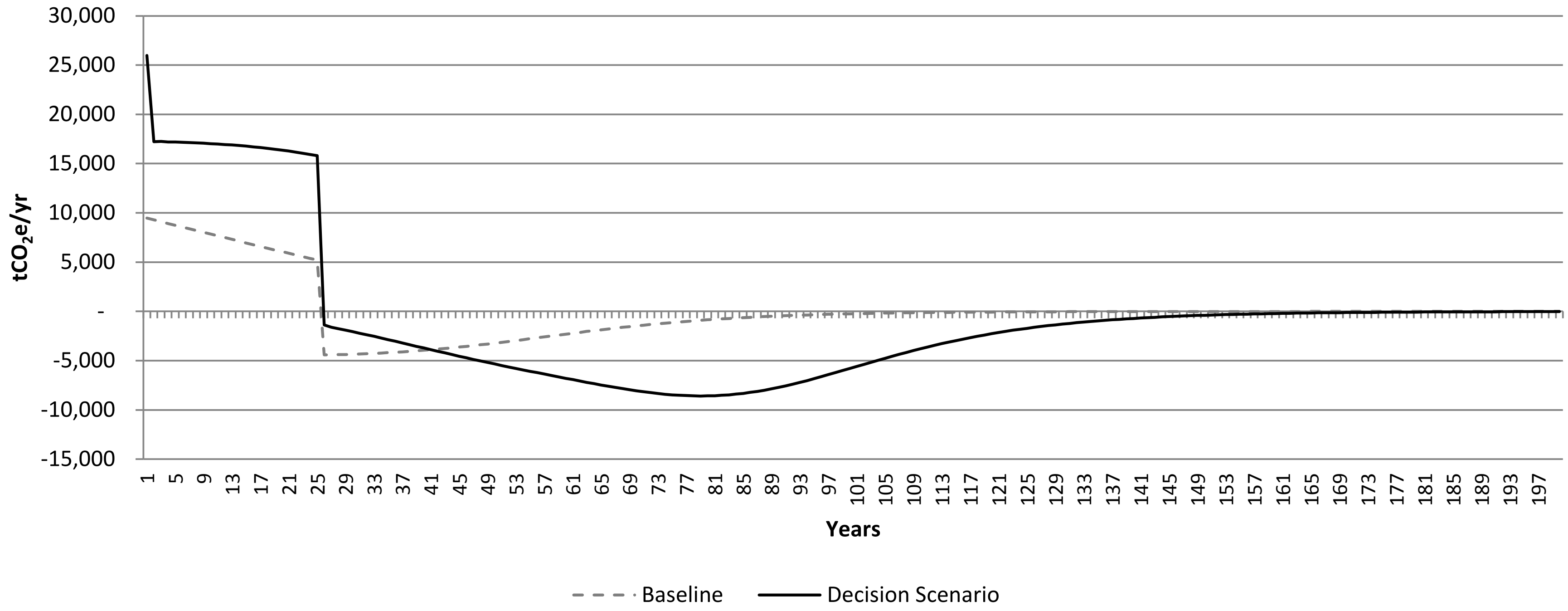


Figure 35. Project/policy method time-series results for Scenario 2.2 (local production with co-products)

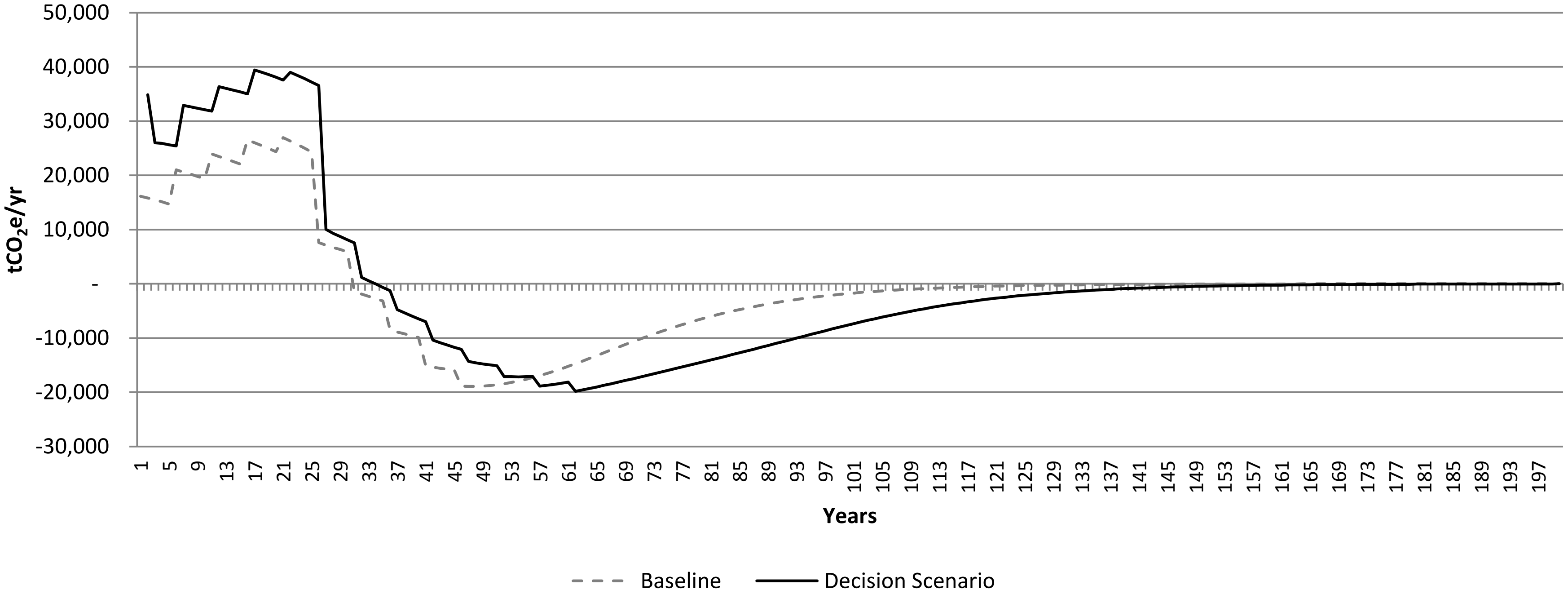


Figure 36. Project/policy method time-series results for Scenario 3.1 (thinnings without co-products)

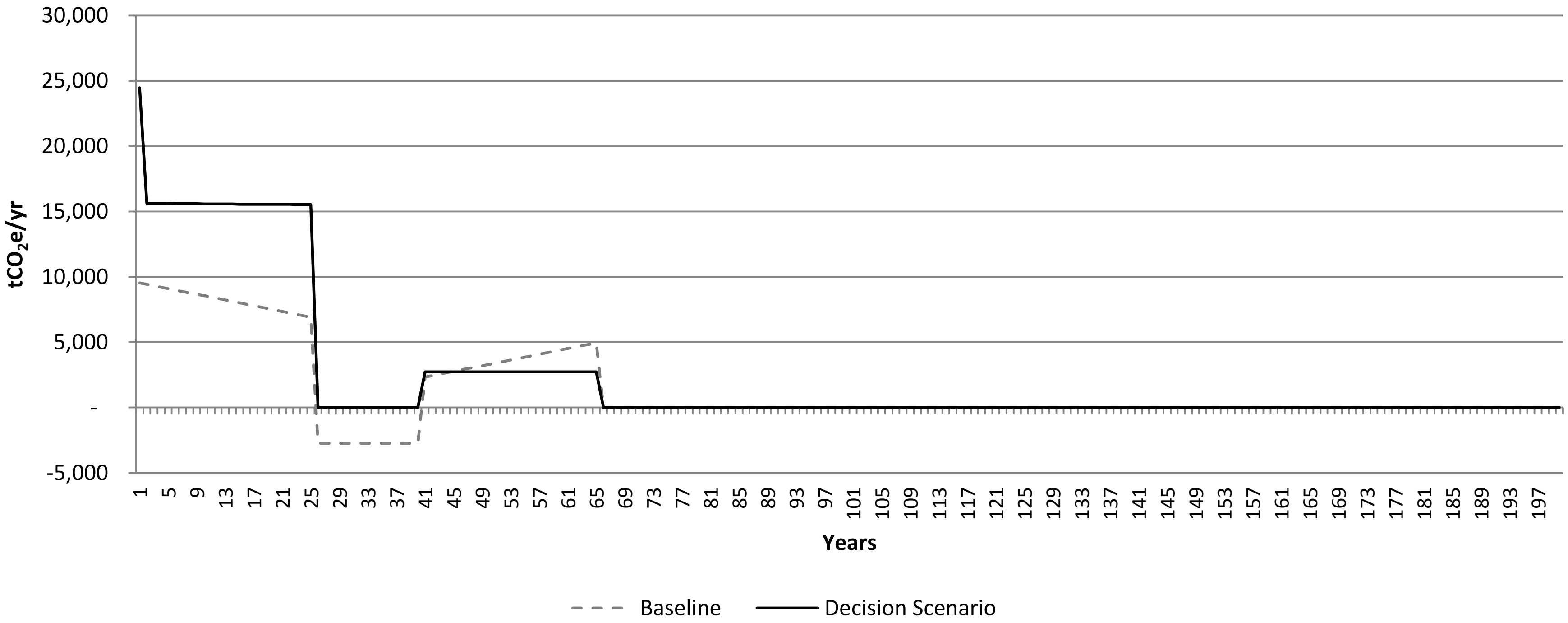


Figure 37. Project/policy method time-series results for Scenario 3.2 (thinnings with co-products – sawlog displacement)

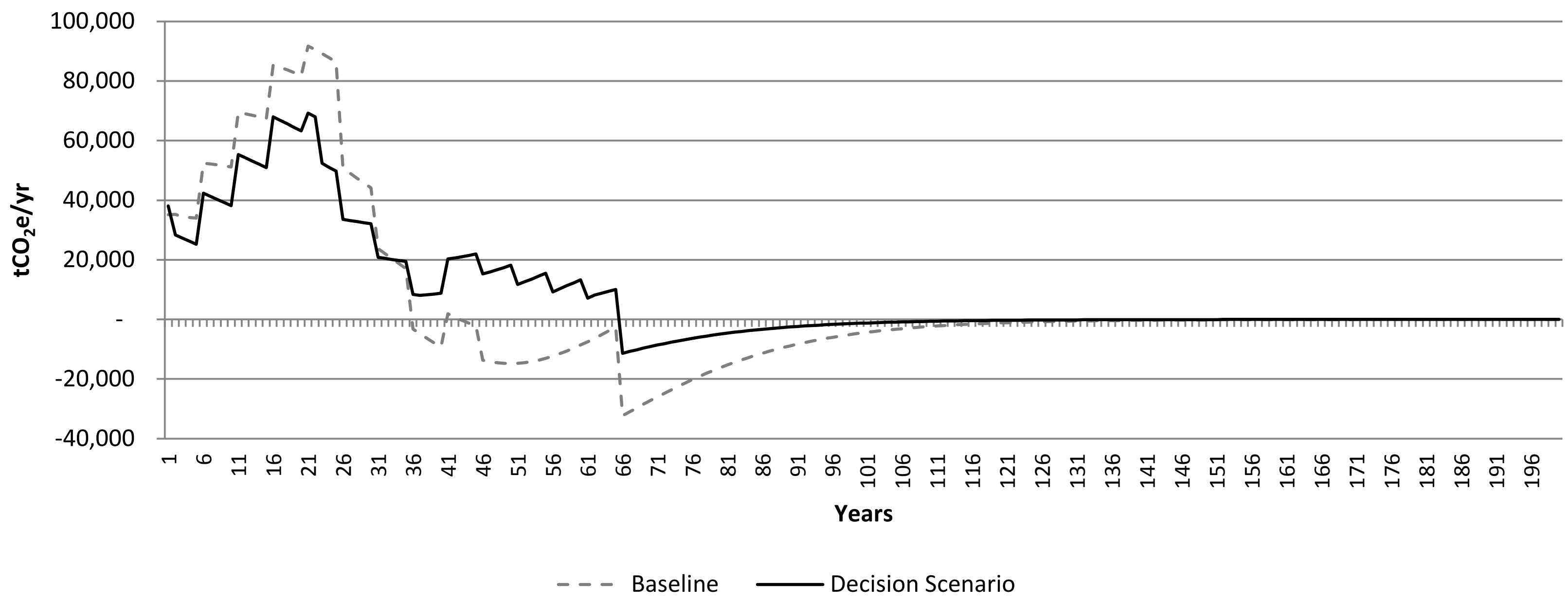


Figure 38. Project/policy method time-series results for Scenario 3.3 (thinnings with co-products – cement render displacement)

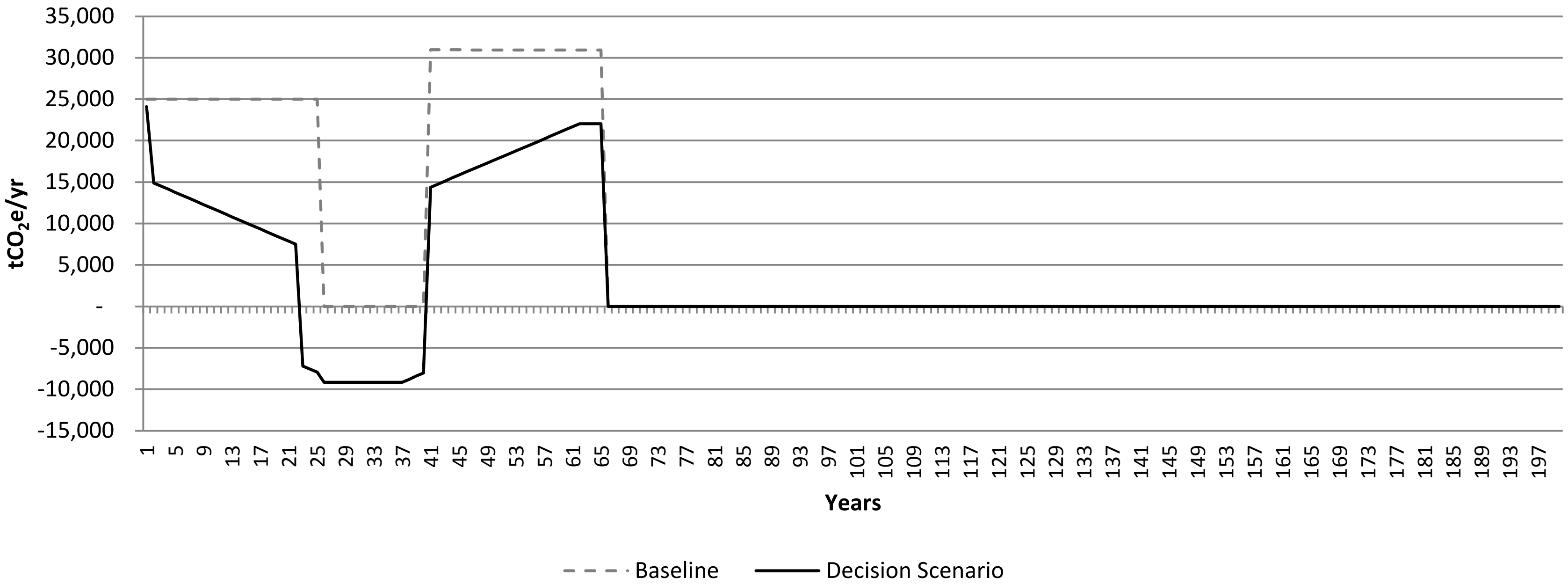


Figure 39. Project/policy method time-series results for Scenario 4.1 (fencing with end-of-life combustion (natural gas displacement))

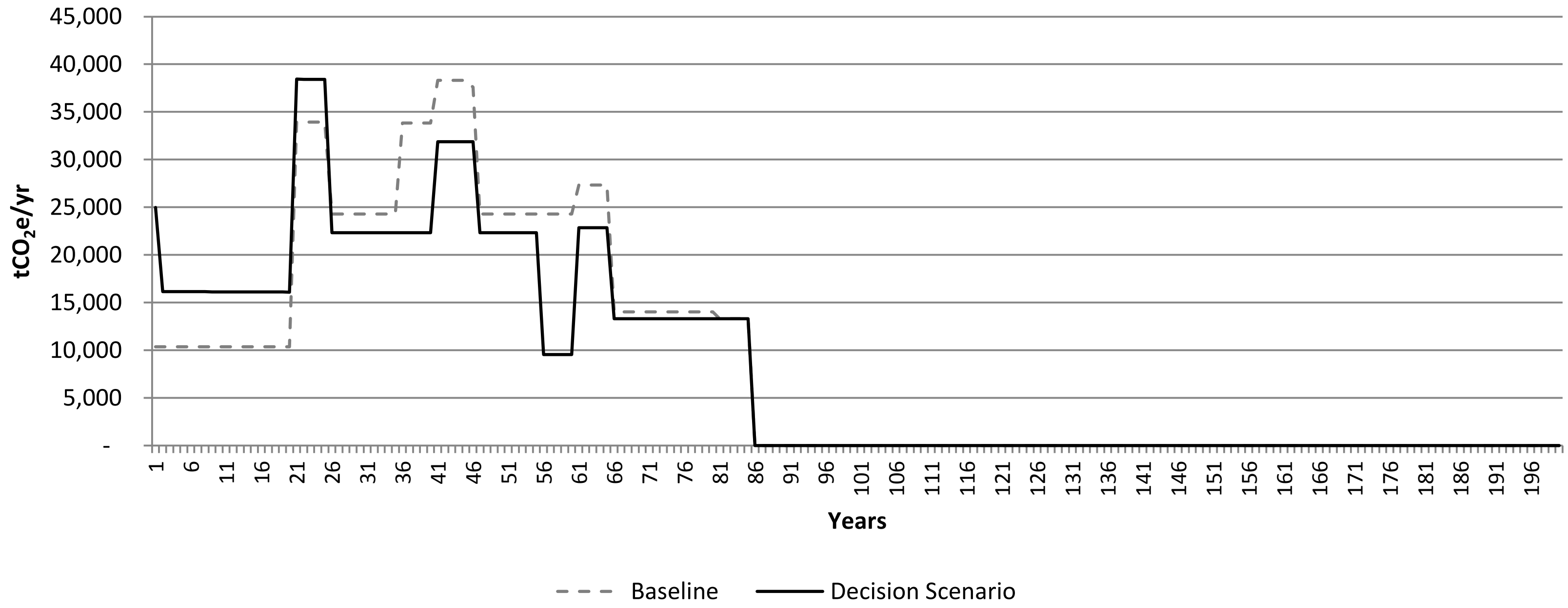


Figure 40. Project/policy method time-series results for Scenario 4.2 (fencing with end-of-life decay)

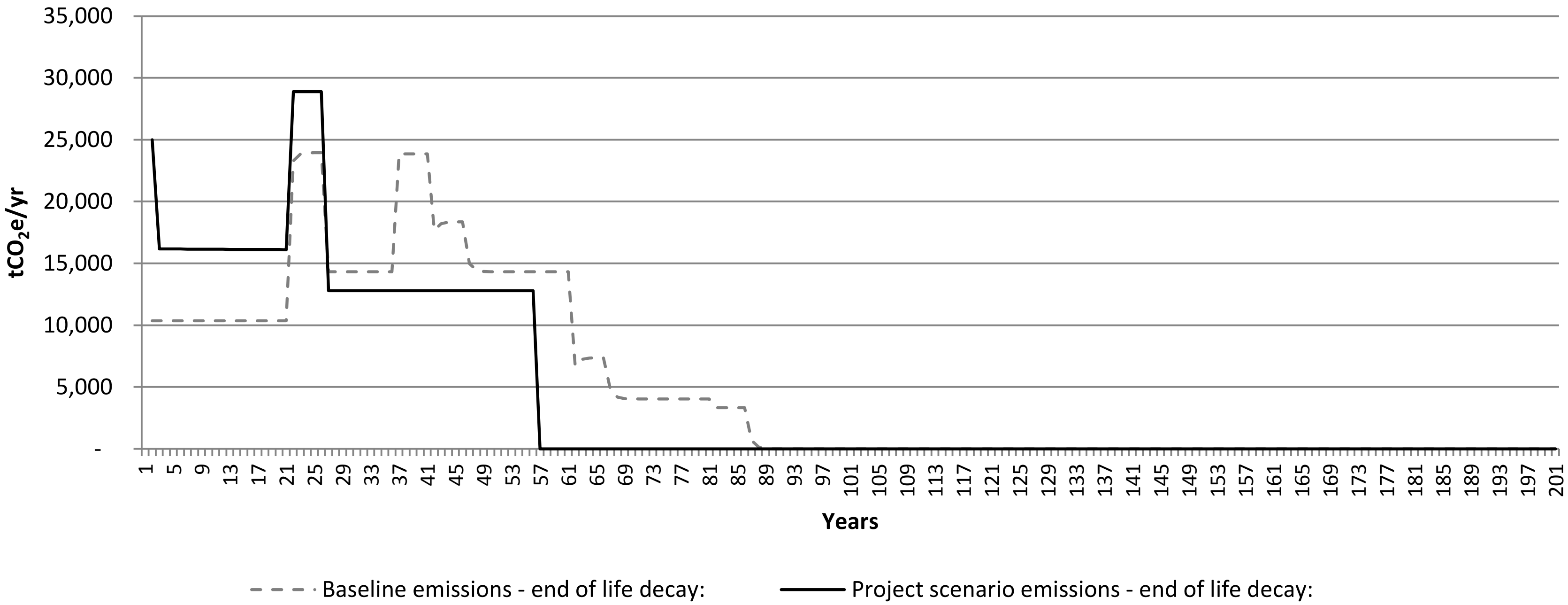


Figure 41. Project/policy method time-series results for Scenario 5.1 (displaced wooden pallets with freed-up biomass displacement)

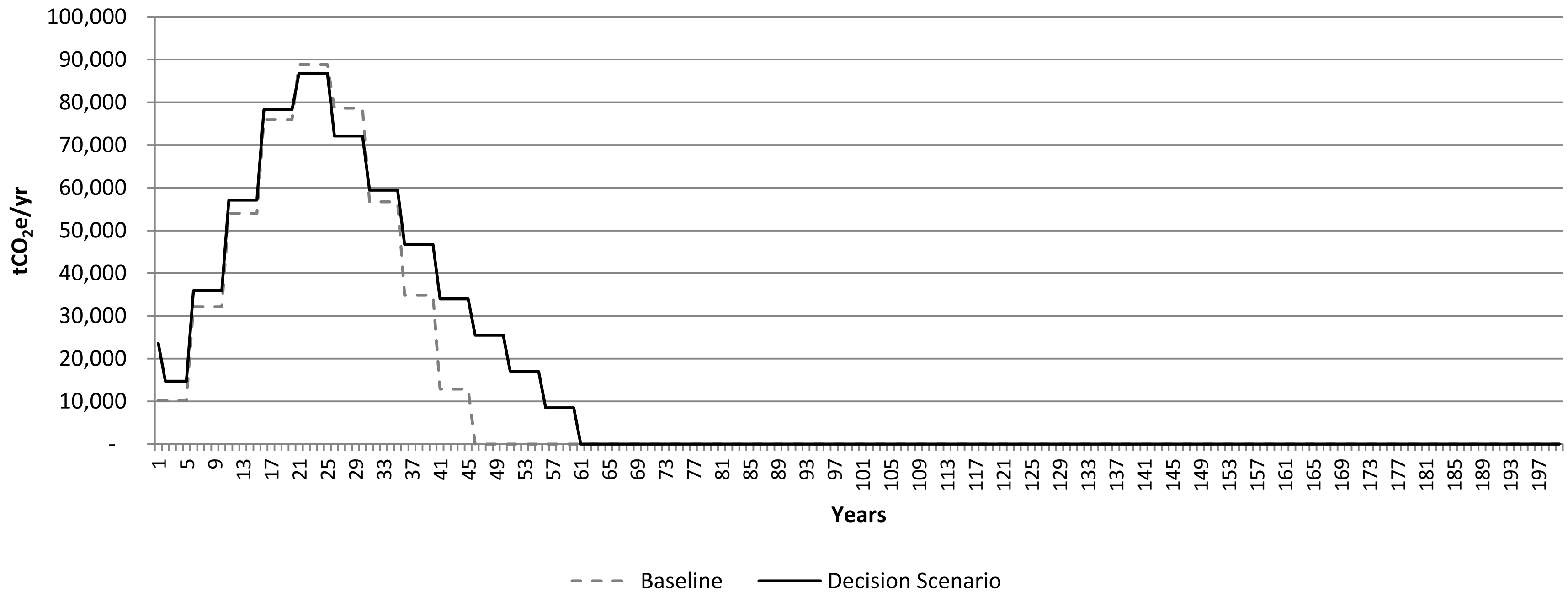


Figure 42. Project/policy method time-series results for Scenario 6.1 (MDF displacement)

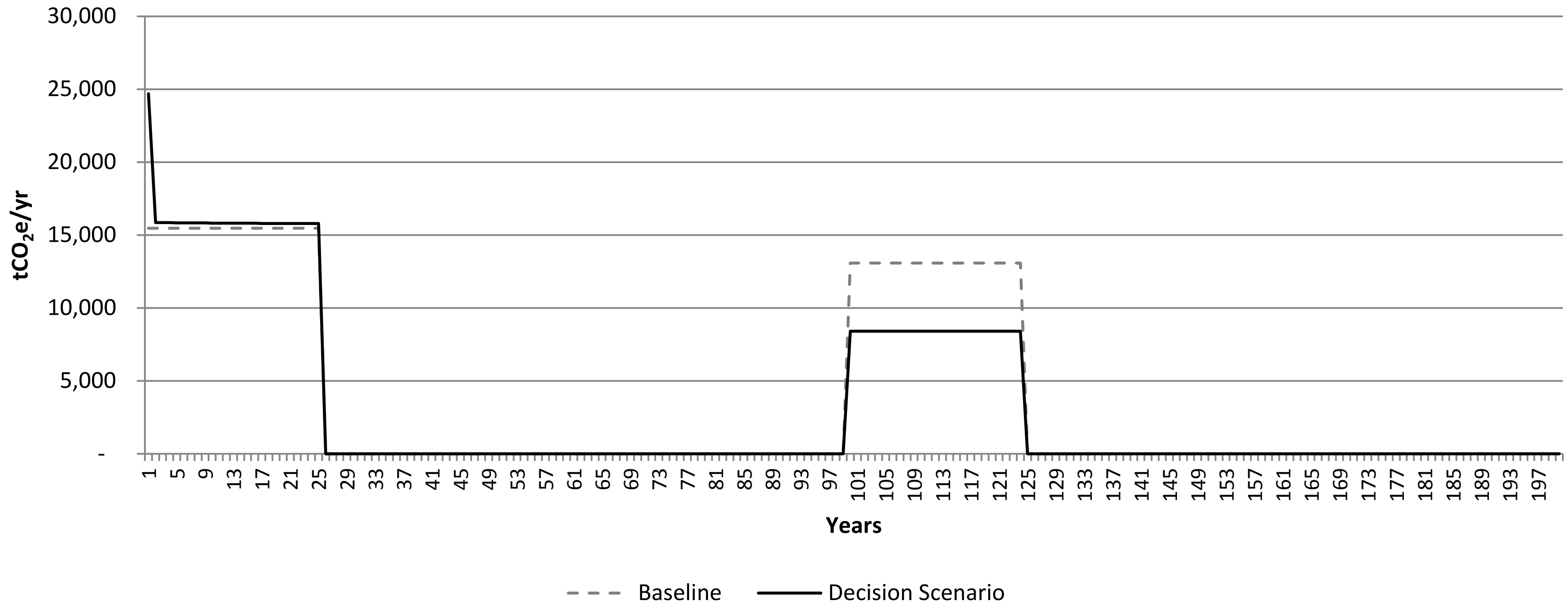


Figure 43. Project/policy method time-series results for Scenarios 7.1 (particleboard displacement – with lower emission factors for breeze blocks)

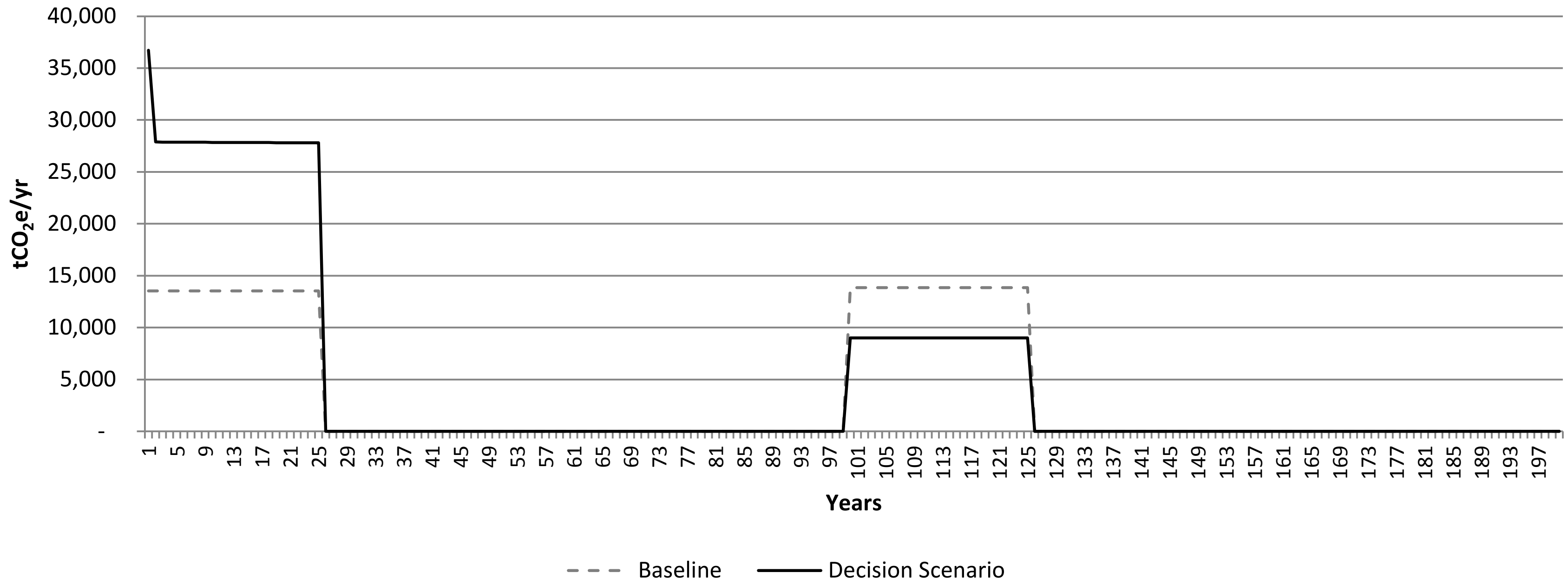
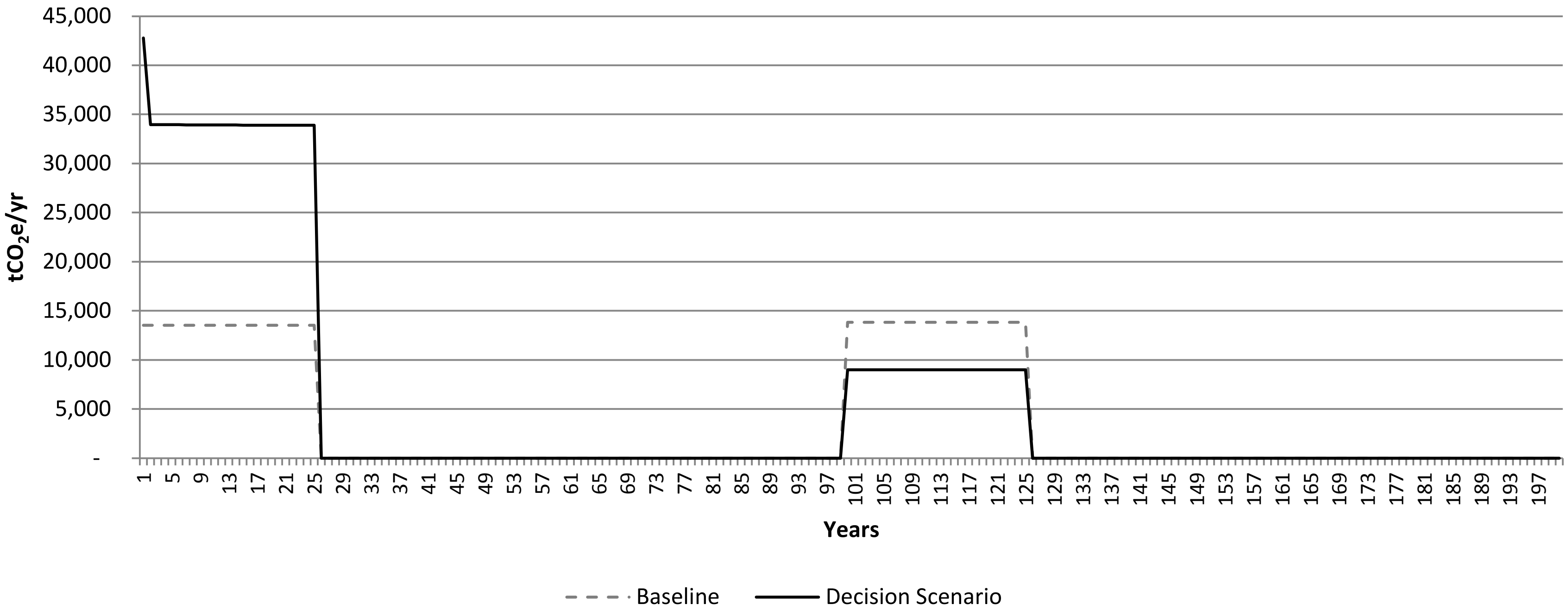


Figure 44. Project/policy method time-series results for Scenarios 7.2 (particleboard displacement – with higher emission factors for breeze blocks)



4. Sensitivity Analysis

Table 18. Sensitivity of lifetime change in emissions/removals to one-at-a-time adjustment of parameter input values

Scenario	Sub-scenario	Stem and branch wood/hectare		Forest growth rate		Carbon content of wood		Rate of soil carbon accumulation		Substitution ratio between wooden and concrete posts		Decay rate for wooden posts	
		-50%	**+50%**	**-50%**	**+50%**	**-10%**	**+10%**	**-50%**	**+50%**	**-50%**	**+50%**	**-50%**	**+50%**
1. Imports	1.1. Imports - sustainable forest management	-2%	1%	-6%	1%	10%	-10%	0%	0%	0%	0%	0%	0%
	1.2. Imports - unsustainable forest management	52%	-17%	0%	0%	-8%	8%	-23%	23%	0%	0%	0%	0%
2. Local production	2.1. Local production without co-products	0%	0%	-13%	1%	4%	-4%	0%	0%	0%	0%	0%	0%
	2.2. Local production with co-products	23%	-7%	228%	-15%	-63%	63%	-8%	8%	0%	0%	0%	0%
3. Thinnings	3.1. Thinning - without co-products	0%	0%	0%	0%	-22%	22%	0%	0%	0%	0%	0%	0%
	3.2. Thinning - with co-products (saw log displacement)	-1%	0%	-2%	0%	10%	-10%	0%	0%	0%	0%	0%	0%
	3.3 Thinning - with co-products (cement render displacement)	0%	0%	0%	0%	0%	-2%	0%	0%	0%	0%	0%	0%
4. Fencing	4.1. Fencing - end of life combustion	0%	0%	0%	0%	9%	-9%	0%	0%	-74%	25%	0%	0%
	4.2. Fencing - end of life decay	0%	0%	0%	0%	34%	-34%	0%	0%	-45%	15%	-92%	118%
5. Pallets	5.1 Pallets - displacing plastic pallets	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%

Scenario	Sub-scenario	Stem and branch wood/hectare		Forest growth rate		Carbon content of wood		Rate of soil carbon accumulation		Substitution ratio between wooden and concrete posts		Decay rate for wooden posts	
		-50%	**+50%**	**-50%**	**+50%**	**-10%**	**+10%**	**-50%**	**+50%**	**-50%**	**+50%**	**-50%**	**+50%**
6. MDF	6.1 MDF - displacing plasterboard	0%	0%	0%	0%	-1%	1%	0%	0%	0%	0%	0%	0%
7. Particle board	7.1. Particle board - breeze block lower estimate	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
	7.2. Particle board - breeze block upper estimate	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%

Table 19. Sensitivity of payback period to one-at-a-time adjustment of parameter input values

Scenario	Sub-scenario	Stem and branch wood/hectare		Forest growth rate		Carbon content of wood		Rate of soil carbon accumulation		Substitution ratio between wooden and concrete posts		Decay rate for wooden posts	
		-50%	+50%	-50%	+50%	-10%	+10%	-50%	+50%	-50%	+50%	-50%	+50%
1. Imports	1.1. Imports - sustainable forest management	9%	-4%	40%	-12%	-4%	3%	-5%	4%	0%	0%	0%	0%
	1.2. Imports - unsustainable forest management	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2. Local production	2.1. Local production without co-products	12%	-6%	25%	-8%	-2%	1%	-10%	5%	0%	0%	0%	0%
	2.2. Local production with co-products	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
3. Thinnings	3.1. Thinning - without co-products	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	3.2. Thinning - with co-products (saw log displacement)	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	3.3 Thinning - with co-products (cement render displacement)	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%	0%
4. Fencing	4.1. Fencing - end of life combustion	0%	0%	0%	0%	-9%	2%	0%	0%	11%	-9%	0%	0%
	4.2. Fencing - end of life decay	0%	0%	0%	0%	-22%	9%	0%	0%	12%	-3%	34%	-26%
5. Pallets	5.1 Pallets - displacing plastic pallets	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Scenario	Sub-scenario	Stem and branch wood/hectare		Forest growth rate		Carbon content of wood		Rate of soil carbon accumulation		Substitution ratio between wooden and concrete posts		Decay rate for wooden posts	
		-50%	**+50%**	**-50%**	**+50%**	**-10%**	**+10%**	**-50%**	**+50%**	**-50%**	**+50%**	**-50%**	**+50%**
6. MDF	6.1 MDF - displacing plasterboard	0%	0%	0%	0%	-90%	4%	0%	0%	0%	0%	0%	0%
7. Particle board	7.1. Particle board - breeze block lower estimate	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	7.2. Particle board - breeze block upper estimate	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA