Carbon and Energy Payback of Variable Renewable Generation

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Lay Summary

Governments around the world have set ambitious targets to reduce Greenhouse Gas (GHG) emissions from electricity generation, and this has led to the development of renewable generators to harness the available energy in the wind, waves and tides. These generators, however, cannot act as a direct replacement for conventional coal and gas-fired power stations, as there is limited control over their constantly fluctuating variable power output. This has led to concerns that they do not actually perform as expected, with some reports claiming that they are responsible for an increase in GHG emissions and that they consume more energy than they ever generate.

Energy is consumed and GHGs are emitted during the life cycle of the renewable generators themselves - from manufacture and construction, through maintenance and operation to final dismantling and disposal. Furthermore, their presence on the electricity network may force coal and gas-fired generators to operate less efficiently, increasing their GHG emissions per unit of energy. In order to confirm that renewable generators actually reduce emissions and also generate useful energy, it is necessary to show that their life cycle GHG emissions (or carbon footprint) and energy consumption (or embodied energy) are lower than their lifetime carbon saving and energy output, respectively. One way to achieve this is to show that the carbon and energy payback periods - the length of time for enough carbon to be saved (or energy produced) to offset that emitted (or consumed) over the whole life cycle - are significantly shorter than the design life of the generator. Using Great Britain as a case study, the work presented in this thesis examines these issues in order to provide more robust and reliable estimates of the carbon and energy payback of variable renewable generation.

This thesis includes a review of existing literature, which shows that estimates of carbon footprint and embodied energy vary widely, due to differences in assumptions and calculation methods. A detailed analysis of the life cycle of the Pelamis wave energy converter investigates these methodological differences, and the impact of the resulting variations on estimates of payback periods.

There is also no reliable estimate for the carbon savings of renewable power in Great Britain. Carbon savings arise due to renewable power displacing other types of generation, but the actual savings (or emissions displacement) will depend upon the mix of generation that they replace, and whether these generators are operating at reduced efficiency. A detailed analysis of real historical data from the National Grid is presented in this thesis to demonstrate that the actual emissions displacement of variable renewable power is slightly higher than the value currently used to calculate payback. This work demonstrates that variable renewable generation in Great Britain should achieve both carbon and energy payback within a few years.

Abstract

The continued drive to reduce Greenhouse Gas (GHG) emissions in order to mitigate climate change has led to an increase in demand for low-carbon energy sources, and the development of new technologies to harness the available energy in the wind, waves and tides. Many controversies surround these technologies, however, particularly with regards to their economic cost, environmental impacts and the implications of the variability of their output for security of the electricity supply. In order to make informed policy decisions on future developments of the electricity system, it is necessary to address these controversies and confirm the environmental, economic and social sustainability of these new renewable generators.

This thesis specifically examines two key issues: whether new variable-output renewable energy generators actually deliver a net reduction in greenhouse gas emissions over their lifetimes, and whether they produce a viable energy return on energy investment. Although renewable energy sources are themselves 'carbon free', GHG emissions (and energy consumption) occur during the construction, maintenance and decommissioning of the generator infrastructure required to convert this energy into electricity. Furthermore, the variability of the output power from such generators has implications for the operation of the grid - there may be a requirement for additional reserve capacity and the increased part-loading of conventional plant is likely to reduce its operating efficiency. Carbon and energy paybacks are measures of the time required for a new renewable installation to offset these life cycle impacts. The work presented in this thesis examines both the life cycle impacts and the GHG emissions displacement of variable renewable generation, using Great Britain as a case study, in order to provide a basis for significantly more robust and reliable estimates of carbon and energy paybacks.

The extensive literature survey concentrates on two key areas: current calculation methodologies and estimates for life cycle carbon and energy consumption of power generators; and the marginal emissions displacement of variable renewable generation. A detailed life cycle assessment of the Pelamis wave energy converter is presented, which sets the embodied carbon and energy in the context of the wider environmental impacts and includes an examination of the effect of different assumptions on the analysis results. In order to investigate the true emissions displacement of renewable generation, a historical analysis of real data from the National Grid was carried out, identifying the marginal displacement factor of wind power and taking into account the effect of the efficiency penalties of conventional plant. The findings of the analyses presented in this thesis are combined with information from the literature to examine the actual carbon and energy payback of existing renewable generation infrastructure on the British grid, and to provide detailed recommendations for future carbon and energy payback calculations.

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Declaration

I declare that this thesis was composed by myself, that the work contained herein is my own except where explicitly stated otherwise in the text, and that this work has not been submitted for any other degree or professional qualification except as specified.

R Camilla Thomson

Abbreviations

AD	Anaerobic Digestion
AEF	Average Emissions Factor
ASA	Advertising Standards Authority
BETTA	British Electricity Transmission and Trading Arrangements
BM	Balancing Mechanism
BMRA	Balancing Mechanism Reporting Agent
BOAL	Bid-Offer Acceptance Level
BSC	Balancing and Settlement Codes
BSI	British Standards Institute
CAP	Committees of Advertising Practice
CCGT	Combined Cycle Gas Turbine
CED	Cumulative Energy Demand
CEF	Combustion Carbon Dioxide Emissions Factor
CF	Carbon Footprint
CFC	Chlorofluorocarbon
CIGS	Copper Indium Gallium Diselenide thin-film PV
CL	Closed Loop
CO_2 eq	Carbon Dioxide equivalent
CSP	Concentrating Solar Power
DALY	Disability Adjusted Life Years
DECC	Department for Energy and Climate Change
DEFRA	Department for Environment, Farming and Rural Affairs
DUKES	Digest of UK Energy Statistics
EDIP	Environmental Design of Industrial Products
EE	Embodied Energy
ELCD	European Life Cycle Database
EMEC	European Marine Energy Centre
EP	Eco-points
EPA	Environmental Protection Agency
EPD	Environmental Product Declaration
EROI	Energy Return on Investment
FPN	Final Physical Notification
GB	Great Britain
GHG	Greenhouse Gas

GRP	Glass Reinforced Plastic
GWP	Global Warming Potential
HFC	Hydrofluorocarbon
ICE	Inventory of Carbon and Energy
IO	Input-Output
IPCC	Intergovernmental Panel on Climate Change
IQR	Interquartile Range
ISO	International Organization for Standardisation
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LWR	Light water nuclear reactor
MDF	Marginal Displacement Factor
MEF	Marginal Emissions Factor
MEL	Maximum Export Level
MIL	Maximum Import Level
MOX	Mixed oxide fuel
NG	National Grid
NGO	Non-governmental Organisation
NREL	National Renewable Energy Laboratory
OCGT	Open Cycle Gas Turbine
PAS	Publicly Available Specification
PCM	Power Conversion Modules
PCR	Product Category Rules
PFC	Perfluorocarbon
PV	Photovoltaic
PVC	Polyvinylchloride
PWP	Pelamis Wave Power Ltd
RC	Recycled Content
SETAC	Society for Environmental Toxicology and Chemistry
SO	System Operator
UK	United Kingdom
UKERC	United Kingdom Energy research Council
USA	United States of America
WEC	Wave Energy Converter

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Chapter 1

Introduction

1.1 Thesis Background

There is a continued drive to decarbonise electricity supplies around the world in an ongoing effort to mitigate climate change by reducing Greenhouse Gas (GHG) emissions. In the UK, ambitious targets have been set by the Government to reduce the average emissions from power generation to around 50 g CO_2/kWh by 2030, from current levels of approximately 500 g CO_2/kWh (Committee on Climate Change, 2010). Furthermore, expected increases in demand for electricity as it replaces current fossil fuel consumption in other sectors, such as domestic heating and transport, is likely to mean that a virtually complete decarbonisation of the electricity sector will be required by 2050 to meet national emissions reduction targets (Wiedmann *et al.*, 2011). This has encouraged an increase in capacity of renewable energy generation, particularly wind, wave and tidal power that are seen to have the greatest potential in the UK.

Controversies surround these new technologies, however, with questions being raised over their economic cost, environmental impacts, and the effect of their variable power output on grid operation and security of supply. It is necessary to address these questions and, in particular, to demonstrate that such technologies will contribute towards the decarbonisation of power generation by achieving a viable energy return on energy investment while also delivering a net reduction in greenhouse gas emissions over their lifetimes.

Although renewable energy sources are inherently low-impact, energy is consumed and pollutants are emitted during the construction, operation and decommissioning of the electricity generators themselves. With significant investment expected in new infrastructure, particularly in the UK where most of the existing power stations are due to be replaced within the next 20 years, neglecting GHG emissions from the generator life cycle could significantly underestimate the ability of a country to meet emissions reduction targets (Wiedmann *et al.*, 2011). Furthermore, if the driver for installing such technologies is to reduce GHG emissions of power generation, these life cycle emissions must be lower than that of the existing generating infrastructure. This may be a challenge on networks where much of the electricity comes from nuclear or hydro power stations (Tremeac and Meunier, 2009), but is also a concern on networks with a high proportion of fossil-fuelled power generation - where the response of such plant to the variable output of renewable generators may cause an increase in their own GHG emissions.

Alongside concerns about greenhouse gases and climate change, it is thought that worldwide energy demand may soon begin to outstrip production. Easily accessible oil and gas reserves will be replaced by oil sands and shale gas, which require more energy to extract and process (Roberts, 2010). In order to remain viable for power generation it will be necessary for renewable generators to have a good energy return on energy investment, and a short energy payback period.

The concept of carbon and energy payback periods is taken from economics, where a payback period is defined as the length of time required for an investment to recover its initial outlay in terms of costs or savings. In the case of carbon payback, it is the time for displaced emissions to match the life cycle carbon footprint, while energy payback is achieved when the total output equals the lifetime energy consumption. Provided that these payback periods are significantly shorter than the lifetime of the renewable power installation, then it will achieve a net reduction in emissions and a positive energy return on investment.

The carbon and energy payback periods of a renewable generator are calculated from estimates of the life cycle GHG emissions and energy consumption, and the annual energy production and emissions displacement. The reliability of these estimates has a significant impact on the reliability of the calculated payback periods. However, existing estimates of the carbon footprint and embodied energy of power generation technologies vary widely, and there is some debate over the actual emissions displacement of variable-output renewable generation. Currently the latter is typically estimated to be the average annual emissions of the entire network (Wagner *et al.*, 2011; Douglas *et al.*, 2008; Parker *et al.*, 2007), but this is known to be inaccurate (ASA, 2007b): variable-output renewable energy does not displace all types of generation equally, and the use of an average factor neglects the impacts of any change in efficiency of plants that do respond. These uncertainties raise doubts over the accuracy of existing carbon and energy payback estimates.

In order to make informed decisions for future developments of the energy system, and support the transition to non-conventional energy sources, more reliable values are required. This thesis investigates the uncertainties that arise on both sides of the payback equations and, using Great Britain as a case study, develops more robust and reliable estimates of the carbon and energy payback of variable-output renewable generation technologies.

1.2 Research Objectives and Scope

This research had several objectives:

1. to determine the current state of research into carbon footprint, embodied energy, and GHG emissions displacement of variable-output renewable generation;
2. to gain an understanding of current carbon footprinting and embodied energy calculation methodologies and identify their limitations;
3. to quantify the effect of methodological choices and assumptions, and set embodied carbon and energy in the context of wider environmental impacts, through a detailed life cycle assessment of the Pelamis wave energy converter;
4. to devise a methodology to identify the historical marginal GHG emissions displacement of variable-output renewable generation, and apply this to data from the National Grid to quantify the marginal emissions displacement of wind power in Great Britain;
5. to investigate the effect of variable-output renewable generation on the efficiency, and therefore GHG emissions, of conventional plant;
6. to apply the resulting marginal emissions displacement factor in calculating more robust and reliable estimates of the carbon and energy payback of variable-output renewable generation infrastructure on the British network.

1.3 Thesis and Contribution to Knowledge

Overall, this thesis will test the hypothesis that:

> *variable-output renewable energy generators in Great Britain do deliver a net reduction in greenhouse gas emissions over their lifetimes, and also produce a viable energy return on energy investment.*

While several attempts have been made to identify the limitations of existing life cycle carbon footprint and embodied energy calculation methodologies, the work presented here is the first to quantify the impacts of these limitations, and identify those that have the most significant effect on the results. Furthermore, the full life cycle assessment of the Pelamis wave energy converter presented in this thesis is the most comprehensive study for a marine energy converter published to date, examining a much broader set of environmental impacts than previous studies.

There is currently no reliable and robust estimate for the GHG emissions displacement of variable-output renewable power on the National Grid in Great Britain. A novel methodology is developed in this thesis that enables the marginal emissions displacement of wind power to be calculated from historical operational data. The calculation is further developed to consider the impact of variable-output renewable generation on the efficiency of conventional power stations. The methodology is then applied to real data from the National Grid, to present more robust and reliable estimates of the emissions displacement of wind power than any previously published for the UK.

The impact of the uncertainty in embodied energy, carbon footprint and emissions displacement estimates is then reviewed in detail to provide a novel insight into the accuracy and reliability of estimates of carbon and energy payback periods.

It is expected that this work will be relevant to anyone with an interest in the carbon footprint, embodied energy, and carbon and energy paybacks of renewable generation, such as: policy-makers, planners, landowners, network operators, power station operators and generator manufacturers. It may also be of interest to members of the public affected by new renewable energy developments. This work should help to improve the reliability and comparability of future estimates, allow wind, wave and tidal farm developers to more accurately estimate the carbon payback of proposed installations, and allow the true carbon savings of variable renewable power to be estimated with improved confidence.

1.4 Thesis Outline

This thesis consists of nine chapters, with necessary appendices. It is loosely divided into two parts, with the first part (Chapters 2 to 4) concentrating on the calculation of embodied energy and life cycle GHG emissions, and the second (Chapters 5 to 7) examining the emissions displacement of variable renewable energy. These two are then brought together at the end to examine the carbon and energy payback.

Chapter 2 introduces the calculation of embodied energy and life cycle GHG emissions for electricity generation. The terms 'carbon footprint' and 'embodied energy' are defined, along with the accepted calculation methodology. The limitations of this methodology are then examined, particularly with reference to the introduction of variability in reported results; relevant recommendations to reduce this variability are identified from existing standards and guidance.

Chapter 3 sets the work in context by presenting a detailed review of existing estimates of life cycle carbon and embodied energy of a range power generation technologies. The methodological choices are examined with reference to the limitations identified in the previous chapter, and the variability of the resulting estimates is explored. Gaps in the existing body of work are also highlighted.

Chapter 4 quantifies the effect of differing methodological choices and assumptions by presenting a detailed Life Cycle Assessment of the Pelamis wave energy converter, with the results compared to those from an earlier carbon and energy audit (Parker *et al.*, 2007). This includes an examination of the carbon footprint and embodied energy in the context of the broader environmental impacts of the life cycle of the device, and also a comprehensive sensitivity analysis to highlight the choices that introduce the greatest variation in results.

Chapter 5 moves on to another aspect of calculating carbon payback, examining the complexities of identifying the emissions displacement of variable-output renewable generation, with particular reference to Great Britain. The values currently used for carbon payback calculations are explored, along with the limitations of these, and the concept of marginal emissions is introduced. In order to set in context the challenge of determining emissions displacement, the operation of the liberalised British electricity trading market, BETTA, is then explained. Finally, existing research into the marginal emissions of networks, both in Britain and around the world, is reviewed.

Chapter 6 develops a methodology for calculating the marginal emissions displacement of variable-output renewable generation from empirical data. Using wind power on the British grid as a case study, the differences between the marginal displacement of variable-output generation, the marginal emissions of changes in demand and the average emissions of the network are examined. Furthermore, any trends in marginal emissions displacement over time, season, time-of-day or power output level are examined.

Chapter 7 expands the work presented in the previous chapter to consider the effect of the variable power output of renewable generators on the efficiency of conventional plant. The effect of any efficiency penalties on the GHG emissions intensity of energy from coal and CCGT power stations is explored, and typical emissions intensity curves are developed. As metered data is not publicly available for individual generators, detailed power output curves are derived from information published through the balancing mechanism. The methodology developed in the previous chapter is then applied to the new data to provide more robust and reliable estimates of the marginal emissions displacement of variable-output renewable generation.

Chapter 8 brings together everything presented in the previous chapters to explore the reliability of carbon and energy payback estimates for variable-output renewable generation. The significance of uncertainties in calculated carbon footprint, embodied energy, emissions displacement and energy generation is examined.

Finally, *Chapter 9* presents a summary of the findings of this research, and draws general conclusions about the carbon and energy payback of existing variable-output renewable generation technologies that might be connected to the British grid. The limitations of these conclusions are also discussed, and suggestions are made for possible further work in this area.

1.5 Publications

The work presented in this thesis has been published as follows, with copies included in Appendix B:

Thomson, R. C., Harrison, G. P. and Chick, J. P., How eco-friendly is wave power? A full life cycle assessment of a wave energy converter, Poster presented at *UKERC Annual Assembly* (First prize), 2011

Thomson, R. C., Harrison, G. P. and Chick, J. P., Full life cycle assessment of a wave energy converter. In *IET Conference on Renewable Power Generation (RPG 2011), Edinburgh, UK*, 6-8 Sept, 2011. doi: 10.1049/cp.2011.0124

Thomson, C., Harrison, G. and Chick, J., Life cycle assessment in the marine renewable energy sector, In *The LCA XI International Conference, Chicago, IL, United States*, 4-6 October, 2011.

Thomson, R. C., Harrison, G. P. and Chick, J. P., Marginal greenhouse gas offset for renewable energy in the UK, In *The LCA XII International Conference, Tacoma, WA, United States*, 25-27 September, 2012.

Thomson, C., Greenhouse gas emissions savings from wind power, Poster presented at *SET for BRITAIN 2013, Houses of Parliament, London, UK*, 18 March, 2013.

Thomson, R. C., Harrison, G. P. and Chick, J. P., Greenhouse gas emissions savings from wind power, Poster presented at *Global Energy Systems conference, Edinburgh, UK*, 26-28 June, 2013.

Chapter 2

Carbon Footprint and Embodied Energy

2.1 Introduction

The first step in examining the carbon or energy payback period of a generating technology is to identify the energy consumption or Greenhouse Gas (GHG) emissions from the whole life cycle of the generator and fuel - from material extraction, through construction and operation, to decommissioning and disposal. This chapter examines the existing calculation methodologies, with particular reference to power generation technologies. It includes definitions of the common terminology, a detailed review of existing standards and guidance, and highlights the significant methodological limitations that can introduce uncertainty to the reported results.

2.2 Carbon Footprinting of Products

2.2.1 Definition

Growing national and international interest in 'carbon footprinting' has led to the term appearing widely in both the literature and media, but, despite this, it has yet to be precisely defined. It emerged from the language of ecological footprinting and is generally accepted to be a measure of the gaseous emissions relevant to climate change from consumption or production activities (Wiedmann and Minx, 2008). This definition does not, however, identify the calculation method, or specify whether it is limited to only direct CO_2 emissions, or includes the full life-cycle emissions of all greenhouse gases. Furthermore, there is some debate over the unit of measurement, as the term 'footprint' implies a spatial measure, as per ecological footprinting, but the vast majority of published carbon footprints are reported in units of mass. Concerns have been raised that this imprecision may introduce confusion and that 'carbon weight' might be a more appropriate term (Hammond, 2007).

A range of different definitions for carbon footprinting exist in the standards and literature. In 2008, Wiedmann and Minx reviewed these and suggested that the use of the word 'carbon' could imply the inclusion of all carbon-based gaseous emissions, even those with no global warming potential - one example of this would be carbon monoxide (CO), which has human health and environmental impacts and may also be converted to CO_2 in the atmosphere. However, as carbon footprinting is a term widely used in public debate concerning the threat of climate change, the review concluded that it should be limited to greenhouse gases. The paper also suggested a 'carbon' footprint should include only carbon-based GHG emissions, but as a partial inventory of such greenhouse gases would not be useful, the final recommended definition includes only CO_2:

> The carbon footprint is a measure of the exclusive total amount of carbon dioxide emissions that is directly and indirectly caused by an activity or is accumulated over the life stages of a product.

There was no consensus, however, to accept the definition of Wiedmann and Minx, and the term continued to be applied to a range of different values. In a more recent publication by The Carbon Trust (2012), the definition of carbon footprint clearly advocates the inclusion of all greenhouse gases defined by the Kyoto Protocol. This was possibly influenced by British government policy, where commitment to Kyoto is a key driver for the quantification of GHG emissions. Their definition reads:

> A carbon footprint is the total greenhouse gas (GHG) emissions caused directly and indirectly by an individual, organisation, event or product, and is expressed as a carbon dioxide equivalent (CO_2 eq). A carbon footprint accounts for all six Kyoto GHG emissions:
>
> - carbon dioxide (CO_2)
> - methane (CH_4)
> - nitrous oxide (N_2O)
> - hydrofluorocarbons (HFCs)
> - perfluorocarbons (PFCs)
> - sulphur hexafluoride (SF_6).

This is further clarified:

> A product carbon footprint measures the GHG emissions over the whole life of a product (goods or services), from the extraction of raw materials and manufacturing right through to its use and final re-use, recycling or disposal.

Very recently the International Organization for Standardization published a technical specification on carbon footprinting: PD ISO/TS 14067:2013 (ISO, 2013). This recommends that the carbon footprint of a product should include all the greenhouse gases identified by the Intergovernmental Panel on Climate Change (IPCC, 2007). The precise definition is that a carbon footprint of a product is the:

> sum of greenhouse gas emissions and removals in a product system, expressed as CO_2 equivalent and based on a life cycle assessment using the single impact category of climate change.

It is possible that the recent ISO publication will provide a consensus on the precise definition of a carbon footprint, but currently the term continues to be applied to values that include a range of different gases. The existing literature does, however, agree that a carbon footprint is a measure of the whole life-cycle emissions of one or many greenhouse gases, and is not limited to only direct emissions. This is often equated to the life cycle impact category of climate change, which is also referred to as a Global Warming Potential (GWP) (ISO, 2013). The impact of the ongoing disagreement over which gases to include is examined further in Section 2.5 and Chapter 4.

An agreement has also emerged in recent years that the unit of measurement is mass of carbon dioxide equivalent, as the use of units of mass avoids the assumptions and uncertainties associated with a conversion to units of area (Wiedmann and Minx, 2008; The Carbon Trust, 2012; ISO, 2013; WRI and WBCSD, 2011a). While it may be more accurate to apply the term 'carbon weight', the common usage of the term 'carbon footprint' is considered to render such precision unnecessary.

2.2.2 Standards and guidance

The accepted methodology for calculating the life-cycle carbon emissions (i.e. carbon footprint) of a product is based upon Life Cycle Assessment (LCA), as defined by ISO 14040 and 14044, and detailed in Section 2.4.2 (ISO, 2006a,b). The methodological framework for LCA, however, is very generic, and decisions made by the practitioner on methodology, assumptions and result interpretation can introduce a wide margin of flexibility to the results. The ISO standards explicitly state that the results of different studies will only be comparable if the assumptions and context of each are equivalent. In order to promote some methodological consistency for carbon footprinting, additional guidance is provided by PAS 2050 (BSI, 2011), the GHG Protocol Product Standard (WRI and WBCSD, 2011a) and the new ISO Technical Specification 14067 (ISO, 2013). Although each of these standards has been developed separately, with different purposes, there has been considerable cross-collaboration in their development in order to ensure that key methodological rules remain consistent. Detailed recommendations for avoiding the methodological limitations of LCA, including sector specific guidance emerging from existing studies of power generation technologies, are examined in Section 2.5.

PAS 2050

PAS 2050 is one of the principal guidance documents for carbon footprinting, and was originally developed in 2008 in response to a broad community and industry desire for a consistent method for assessing the life cycle GHG emissions of goods and services (BSI, 2011). As the first consensus-based and internationally-applicable standard on product carbon footprinting, the original version was used as the basis for the development of other standards around the world. The later revision, published in 2011, was developed through extensive consultation with international stakeholders, and drew upon the lessons learned in the development of the GHG Protocol Product Standard (WRI and WBCSD, 2011b).

The primary objective of this specification is to provide a common basis for GHG emission quantification in order to inform and enable meaningful emissions reduction programmes. In contrast to the GHG Protocol Product Standard, this specification does not provide recommendations for the reporting of carbon footprints, instead giving emphasis to proper recording of processes and outcomes. The recommended methodology is based upon LCA, as detailed in ISO 14040 and 14044 (ISO, 2006a,b), but addressing the single impact category of climate change.

The Greenhouse Gas Protocol

The Greenhouse Gas Protocol is a collaboration of the World Resources Institute and the World Business Council for Sustainable Development, aiming to help achieve a low-emissions economy worldwide. Through a multi-stakeholder process they have developed a set of separate but complementary standards, protocols and guidelines for assisting the understanding, quantification and management of greenhouse gas emissions (WRI and WBCSD, 2011b):

- GHG Protocol Corporate Accounting and Reporting Standard (Corporate Standard)
- GHG Protocol for Project Accounting (Project Protocol)
- GHG Protocol Corporate Value Chain (Scope 3) Accounting and Reporting Standard (Scope 3 Standard)
- GHG Protocol Product Life Cycle Accounting and Reporting Standard (Product Standard)
- GHG Protocol for the U.S. Public Sector
- GHG Protocol Guidelines for Quantifying GHG Reductions from Grid-Connected Electricity Projects
- GHG Protocol Land Use, Land-Use Change, and Forestry Guidance for GHG Project Accounting
- Measuring to Manage: A Guide to Designing GHG Accounting and Reporting Programs

The Corporate Standard was the first to be published (in 2001) and was accepted by the International Organization for Standardization in 2006 as the basis of ISO 14064-1: Specification with Guidance at the Organization Level for the Quantification and Reporting of Greenhouse Gas Emissions and Removals (WRI and WBCSD, 2004). It has also been used by many industry, non-governmental, and government GHG programs as a basis for their accounting and reporting systems. The standard was developed with the aim of assisting businesses to manage GHG risks, identify reduction opportunities, provide information for reporting programs and participate in carbon trading markets. It contains guidelines for business-scale accounting and reporting of the six greenhouse gases covered by the Kyoto Protocol - carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulphur hexafluoride (SF_6).

Following publication of the Corporate Standard, further guidelines were developed for a range of other applications, and of these, the Product Standard is the most applicable to carbon footprinting power generation technologies (WRI and WBCSD, 2011a). This provides guidance for quantification and reporting of life cycle GHG emissions at the individual product level and is based upon attributional LCA as defined by ISO 14040 (ISO, 2006a). It contains detailed recommendations for calculating a carbon footprint that meets the five overarching guidelines of the GHG Protocol (relevance, accuracy, completeness, consistency and transparency), and is examined in more detail in Section 2.5. This standard has some overlap with the guidance provided in PAS 2050, so harmonisation of the methodologies was sought where possible, although minor differences remain (WRI and WBCSD, 2011b). It also expands upon the guidance contained within PAS 2050 by providing detailed requirements for public reporting of carbon footprints.

The GHG Protocol also provides guidelines for calculating the reduction in GHG emissions from climate change mitigation projects, such as renewable energy installations. The Project Protocol and associated 'Guidelines for Quantifying GHG reductions from Grid-Connected Electricity Projects' (WRI and WBCSD, 2005, 2007), however, do not contain significant recommendations for estimating the carbon footprint of these technologies. The analysis method is instead based upon consequential LCA, with the primary impacts being the displacement of power generation from conventional plant, while the life cycle emissions of the generating infrastructure are considered to be secondary impacts.

ISO/TS 14067

The Technical Specification 14067 on the carbon footprint of products was published by the International Organization for Standardization on 30th June 2013 (ISO, 2013). It was considered appropriate for publication only as a technical specification because an agreement to publish an international standard could not be reached - reflecting the developmental nature of this subject. The ISO/TS builds upon existing international standards, including ISO 14040 and 14044 on life cycle assessment (ISO, 2006a,b), to set specific requirements for the quantification and communication of a carbon footprint.

One issue that is specifically addressed in this technical specification is that of the comparability of different carbon footprints. This is acknowledged to be a limitation of the LCA methodology, so the ISO/TS introduces the concept of product category rules (PCRs) for carbon footprinting: a set of specific rules, requirements and guidelines for equivalent products, originally developed for preparing environmental declarations. Carbon footprints can be considered to be comparable if they are carried out with equivalent PCRs and meet further detailed recommendations provided in Annex D. A PCR exists for preparing Environmental Product Declarations for power generation within Europe (The International EPD System, 2013), and contains detailed guidance on specifying the goal, scope and required data quality for analyses that can be applied to carbon footprints.

2.2.3 Carbon payback

In order for renewable energy converters to be effective at decarbonising the electricity network, they should be responsible for reducing the GHG emissions of power generation by much more than the emissions from their own life cycle. The 'carbon payback' period is a measure of this reduction; it is defined as the time for the carbon footprint of a renewable energy converter to be offset by the GHG emissions displacement, and should be significantly shorter than the design life.

There is continued debate over the GHG emissions displacement of renewable power generation, particularly for variable-output technologies that might have unpredictable impacts on generator dispatch. This is examined in much greater detail in Chapters 5, 6 and 7.

2.3 Embodied Energy

2.3.1 Definition

The embodied energy of a product is defined as the total primary energy consumed over its life cycle, from cradle to grave (Hammond and Jones, 2011; Mortimer *et al.*, 2003). This includes the total energy extracted from the natural system: the energy consumed as fuels or electricity by the processes that make up the product system; energy consumed or lost during the extraction, processing, generation and transport of these energy carriers; and also any energy sources that are irretrievably removed from the natural system (Baumann and Tillman, 2004). Embodied energy may also be referred to as the gross energy requirement, cumulative energy demand or cumulative energy requirement.

The basic unit of measurement for embodied energy is the Joule, as this is the SI unit for energy. As electrical energy is conventionally measured in kilowatt-hours, this results in most embodied energy values for power generating technologies being reported in kJ/kWh. It is, however, fairly straightforward to convert this unit into an energy ratio or an energy return on investment.

2.3.2 Standards and guidance

The concept of embodied energy emerged in the late 1970s as energy analysis methods were developed in response to concerns over energy availability (Hammond and Jones, 2008a). Energy analysis was actually one of the earliest methodologies to encompass the whole life-cycle concept (Baumann and Tillman, 2004), and ISO 14044 explicitly includes the quantification of energy inputs and outputs in the life cycle inventory stage of an LCA (ISO, 2006b).

The process of calculating the embodied energy of a product is, therefore, also based on LCA methodology as defined by ISO 14040 and 14044 (ISO, 2006a,b). It therefore suffers from the same limitations as carbon footprinting, where methodological inconsistencies can limit the comparability of embodied energy calculated from different studies. Unlike carbon footprinting, however, no additional standards and guidance have been identified for calculating the embodied energy of products.

2.3.3 Energy payback and return on investment

If renewable power generation is to form a sustainable contribution to the future energy mix, the energy extracted from the renewable energy source must be greater than the energy required to construct, maintain and decommission the energy converter. As with carbon payback period, the energy payback period of a power generation technology can be defined as the time for the embodied energy of a generator to be offset by the energy it produces, and should be significantly shorter than the design life.

Another metric often used to examine the embodied energy of power generation and energy sources is the Energy Return on Investment (EROI). This is the ratio of energy output to embodied energy, and must be greater than 1 for the generation or energy source to be useful (Gupta and Hall, 2011). It is currently of growing interest due to the development of fossil fuel extraction from poorer quality sources, such as tar sands and shale gas: extraction of fuels from these sources requires more energy than extraction from conventional oil and gas wells, and they therefore have a lower EROI. It can also be argued that for true economic viability the EROI of renewable energy converters should be similar to that for fossil-fuelled generation. As renewable generation technologies mature and conventional fossil fuel sources diminish, it is likely that the EROI of the technologies will converge.

2.4 Life Cycle Assessment

Life Cycle Assessment (LCA) is an established technique for identifying and evaluating the inputs, outputs and potential environmental impacts of products or services. It began to emerge as a coherent methodology in the 1990s, during a series of workshops organised by SETAC (the Society for Environmental Toxicology and Chemistry), which led to the development of a technically rigorous framework formalised in the ISO 14040 series of standards (UNEP, 2011a; ISO, 2006a,b). While the principles of LCA can be applied to evaluate carbon footprints and embodied energy, the methodology has the potential to investigate a much broader range of environmental impacts on resources, ecology and human health. The results of such comprehensive analyses can highlight the components, materials or stages of the life cycle with the largest environmental impacts, and can also be used to promote environmental credentials in product marketing. LCA is, therefore, increasingly used to inform strategic decision making within organisations, and assist with the development of environmental policy by governments (UNEP, 2011a).

The results of LCAs are also often used to examine the relative environmental benefits of different products; for example, the carbon footprint of renewable generation is frequently compared to that of conventional generation. However, the methodological framework for LCA is very broad, to enable it to be applied to a wide range of goods and services (ISO, 2006a,b), which introduces considerable scope for variation between studies, limiting their comparability.

It is also widely recognised that the existing LCA framework is not perfect, and new guidance continues to emerge as the methodology develops. In particular, the methods for assessing land-use and water-use impacts, uncertainty assessment methods and toxicity impact potentials have all been identified as requiring improvement (UNEP, 2011a). A recent publication by UNEP (2011b) also recognised that improved consistency in life cycle datasets would improve the quality of LCA worldwide. Furthermore, LCA is limited to examining only quantifiable environmental impacts; factors such as visual impact are neglected, and there is a movement to develop a full life cycle sustainability assessment method that will incorporate economic and social impacts alongside the environmental considerations (UNEP, 2011a). Recent developments also include the emergence of consequential LCA, and hybrid approaches that combine conventional process-based LCA with input-output models to avoid any truncation errors.

This section examines different types of LCA and introduces the process-based methodological framework. The limitations of LCA, particularly for comparability, are examined in greater detail in Section 2.5.

2.4.1 Types of LCA

The most established and commonly-used methodology for assessing the life cycle environmental impacts of products is process-based attributional LCA. This method, described in detail in Section 2.4.2, involves systematically analysing each process within the product life cycle in order to build up an estimate of the environmental impacts attributable to that product. However, the processes within a product life cycle form part of a highly complex network, so the methodology requires a system boundary to be defined to facilitate the calculation. This system boundary introduces truncation errors that might underestimate environmental impacts by as much as 80 % (Crawford, 2005).

An alternative to the process-based method is economic input-output analysis. This was developed to quickly estimate the impacts associated with a product according to monetary flows, using data gathered to create national input-output tables. This method has the benefit that it takes into account the higher order impacts excluded in the process method, thus avoiding truncation errors. However, as the input-output data is based on economic activities at sector level, this methodology is less suited to detailed calculations concerning individual products, such as power generators, and research has indicated that it often underestimates life cycle impacts (Lenzen, 2008). It is therefore considered to be an unreliable method for these assessments.

More recently, hybrid approaches have emerged that combine both methods. These are considered to be the most robust, state-of-the-art methodologies by many LCA and carbon footprinting practitioners (Crawford, 2005; Lenzen and Munksgaard, 2002; Wiedmann *et al.*, 2011). Hybrid LCA involves combining national data from input-output tables (or supply-and-use tables) with detailed process information, to achieve the level of detail provided by the process-based analysis whilst also avoiding truncation errors. Studies have found that such hybrid approaches produce significantly higher estimates for the environmental impacts of products (Wiedmann *et al.*, 2011). However, other research suggests that these impacts may be exaggerated by the input-output data (Davidsson *et al.*, 2012). This thesis concentrates on process-based LCA as it is currently the most commonly-applied methodology for estimating the carbon footprint and embodied energy of power generating technologies, and therefore provides the most opportunities for finding comparative studies (see Chapter 3).

Another recent development in LCA is the emergence of consequential analyses. These concentrate on quantifying the changes in environmental impacts as a result of a given process; such analyses do not produce results for the whole-life carbon footprint or embodied energy of a product, instead directly examining factors such as carbon payback or energy return on investment.

2.4.2 Methodology

The LCA process, illustrated in Figure 2.1, involves systematically analysing resource use and pollutant emissions at each stage of the product life cycle; from extraction of raw materials, through manufacture and operation, to decommissioning and disposal. The detailed results are then described as a set of identifiable consequences or 'impact potentials', such as the Global Warming Potential (GWP). A full assessment involves examining much more than the greenhouse gas emissions and energy consumption, instead attempting to quantify all potential environmental impacts - this includes resource use, impact on human health and any ecological impacts.

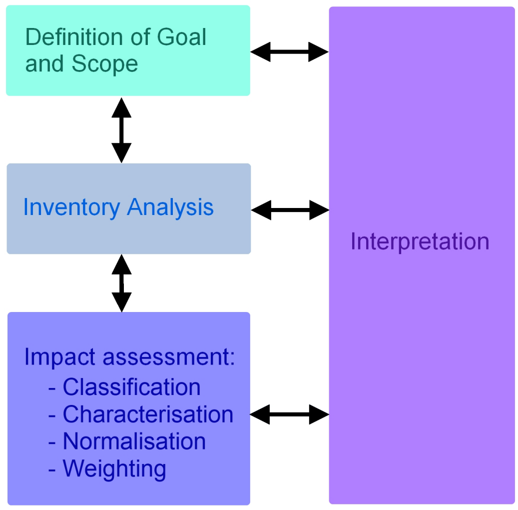


Figure 2.1: Life cycle assessment framework (after Baumann and Tillman (2004))

Every LCA must begin by clearly defining the goal and scope, to enable appropriate methodological decisions to be made (Baumann and Tillman, 2004). The first analysis step is then to create a Life Cycle Inventory (LCI) of all resource use and pollutant emissions - this can provide valuable detailed information, such as the lifetime emissions of the six Kyoto GHGs. However, in order to understand the environmental impacts of the product, a Life Cycle Impact Assessment (LCIA) should be carried out. This involves classifying all of the resources and emissions in the LCI, and applying characterisation factors to convert these into a set of impact potentials; for example, the global warming potency of each GHG in comparison with carbon dioxide is used to calculate the GWP in kilograms of CO_2 equivalent. The uncertainty of the results of an LCA, and sensitivity to practitioner assumptions, should be presented alongside the interpreted findings.

Goal and scope definition

The clear definition of a goal and scope is an integral part of any LCA, providing details of the context and purpose of the study, the system boundary and the functional unit (ISO, 2006a).

Goal and context

The goal definition must unambiguously state the intended application of the study, the reasons for carrying it out and the intended audience, as well as making it clear whether the results will be used in comparative assertions for public disclosure (ISO, 2006b). These are all closely tied to one another; for example, an LCA being carried out to demonstrate the low environmental impacts of a product for marketing information will be published to consumers, and will also be intended for use in making comparisons with other products. Most existing LCAs or carbon and energy studies of power generation technologies have been carried out to examine opportunities for reducing environmental impacts or for promoting the environmental benefits of a given technology, such as wind or nuclear power, over its competitors.

Project scope

The scope should be sufficiently well defined to ensure that the breadth, depth and detail of the study will meet the requirements of the stated goal (ISO, 2006a). It should include information about the particular product and scenario being studied, the functional unit, the choice of impact categories and assessment method, the system boundary, the principles for allocation and the data quality requirements (Baumann and Tillman, 2004). Allocation and data quality are examined in greater detail later in this section.

The project scope of LCAs that have been published for existing conventional generation technologies have either considered a typical technology for a given region (such as the analysis by Spath *et al.* (1999) on coal-fired plant in the USA) or a particular power plant (such as the Environmental Product Declarations (EPDs) published for a number of nuclear power stations (Axpo, 2011; AEA Energy and Environment, 2009, 2008b)). However, for emerging technologies like marine power, a hypothetical scenario must be defined; for example, in Parker *et al.* (2007) the scenario specified the number of wave energy converters in the completed wave farm, the location of the farm, and the location of the port from which the wave farm was accessed.

Functional unit and reference flow

The functional unit for a life cycle assessment is the reference unit to which the input and output data are normalised - in the Environmental Product Declaration methodology this is equivalent to the 'declared unit'. In many cases the device being studied will have multiple functions, even if the primary function is clearly defined, resulting in a variety of outputs. The functional unit for the analysis should, therefore, be selected according to its goal (Bousquin *et al.*, 2012; Reap *et al.*, 2008).

Once the project scope and functional unit have been defined, the reference flow can be identified. A full product life cycle involves the flows of many different materials and processes, and these must be measured relative to the reference flow of the product under study. The reference flow is defined as the "measure of the outputs from processes in a given product system required to fulfil the function expressed by the functional unit" (ISO, 2006b). In the case of power generation, this is typically a single power plant or renewable energy converter.

System boundary

The definition of a system boundary is a key step in any life cycle assessment, as this defines which processes will be included in the analysis and the level of detail to which these will be studied (ISO, 2006b). Assumptions about geographical and time constraints, technical limits (including recycling) and ecological boundaries are defined.

One such boundary is which life cycle stages to include in the analysis. While it might be appropriate to consider only part of the life cycle for some products, for power generation the analyses usually consider the impacts of the whole life of the plant from cradle to grave, thus including every stage of manufacture, operation and decommissioning. A decision must also be made whether to include the impacts of capital goods. These are items that might be used during life cycle stages, such as equipment in manufacturing plant or lorries for transport. Within LCA three orders of analysis can be defined (Goedkoop *et al.*, 2008):

1. First order: Only the production of materials and transport processes are included
2. Second order: All life cycle processes are included, excluding capital goods
3. Third order: All life cycle processes and capital goods are included. Usually the latter are only modelled in first order mode, so only the materials within the capital goods are considered.

Historically, most LCAs are second order analyses, but this may result in up to 30 % of the environmental impacts being neglected, and is one of the most significant limitations of process-based LCA (Goedkoop *et al.*, 2008). One example of this would be in including the impacts of hydroelectricity consumption: in a second order analysis only the operational impacts of generating that electricity would be considered, and these are very low; however, in a third order analysis the impacts of the large-scale infrastructure required to generate the hydroelectricity, such as the dam and all machinery, would be included. In order to avoid these limitations, detailed LCI databases, such as Ecoinvent, attempt to include full third-order impacts (Ecoinvent, 2010). Input-output data also inherently includes all capital goods used by the given process (Crawford, 2005).

Some carbon footprinting studies do not follow the requirements of first, second or third order analyses. One example of this is studies of fossil-fuelled electricity generation, where the impacts of manufacture and decommissioning may be considered negligible so only operational impacts are reported. The included life cycle stages must be clearly stated, and limitations of such assumptions are examined in Section 2.5 and Chapter 4.

The definition of the system boundary should also detail the physical, geographical and temporal limits of the analysis. This should include which components or items of equipment fall within the system boundary (for power generation the physical boundary is usually the point of connection with the grid), the location of the system being studied, the lifetime of the equipment, and the date of manufacture. Two details to consider in the definition of the physical boundary are whether the impacts of a change in land-use will be considered, and whether any emissions reductions should be included, as a full LCA should take all aspects of the environment into account. This is of particular consideration for wind farms - many of the existing studies assume a typical installation location, and therefore do not consider the wider implications of the effect on the land, which may be significant for rural sites (Vestas, 2006b). One example of this would be the installation of a wind farm on peat land: peat plays a significant role in the carbon cycle, absorbing and releasing carbon dioxide and methane dependent upon its moisture content; the installation of foundations and access roads for a wind farm in such an environment could lead to the peat bog drying out, and studies have shown that this will lead to a net decrease in the embodied carbon of the peat (Nayak *et al.*, 2008); this impact should be allocated to the wind farm. Similarly, any GHG emissions due to forestry clearance should also be quantified and included in a complete study. Where carbon capture and storage is to be considered, this must fall within the physical boundary of the analysis, and be clearly defined.

Cut-off criteria

The ISO (2006a,b) methodology allows cut-off criteria, or insignificance thresholds, to be specified to exclude some inputs and outputs from an analysis. This is intended to avoid unnecessary effort calculating or estimating resource use and emissions data for processes that will not significantly change the overall conclusions. However, cut-off criteria are based on mass, energy and environmental significance, and it is argued that an assessment of these requires enough information to simply include the processes in the full analysis (Reap *et al.*, 2008). In practice, however, it is unlikely for a process-based LCA to include the impacts of every nut and bolt in a large power generator, and such cut-off decisions may be made without being explicitly acknowledged. ISO 14044 requires that cut-off criteria should be well documented and the effect on the outcome of the study assessed (ISO, 2006b).

Data quality

Data quality can significantly affect the reliability of the results of an LCA, so data quality requirements should be specified at this stage, particularly for analyses that are intended to be used in comparative assertions for public disclosure. Where data is applied that does not meet these requirements, it should be clearly documented.

The quality requirements for data fall into three categories: relevance, reliability and accessibility (Baumann and Tillman, 2004). Data used in the LCA should be relevant to the goal and scope of the study, but it will not always be possible to find an exact match for each process. In order to ensure consistency throughout the analysis, the required time-related coverage, geographical coverage, technology mix, completeness and representativeness of any source data should be specified (ISO, 2006b). The reliability of this data, particularly its precision, uncertainty and consistency, will influence the uncertainty of the final results of the analysis, so these factors should also be included in the data quality requirements. Finally, any data should be reasonably accessible, and requirements should be specified at this stage for a qualitative estimate of the reproducibility of any data sourced for the study.

Life cycle inventory analysis (LCI)

The Life Cycle Inventory (LCI) details all resource consumption and pollutant emissions for every stage of the product life cycle. It is generally compiled from detailed information about the materials and processes involved in each life cycle stage, with the corresponding inventory information extracted from published databases, such as the Inventory of Carbon and Energy (ICE (Hammond and Jones, 2008a)) or Ecoinvent (Ecoinvent, 2010). Where existing LCI data is unavailable for a given process or material, it should be calculated from upstream life cycle information, or appropriate estimates should be made. All of the collected data should be subjected to a validity check to ensure consistency (ISO, 2006b).

The process of creating an LCI normally begins with a detailed flow chart for the product life cycle, as shown in Figure 2.2. This allows the materials and processes to be systematically analysed for each life cycle stage. The next step is to collect and aggregate data for each process and material, ensuring that it meets the data quality requirements set out in the goal and scope. In many cases this process is straightforward, but problems may emerge where processes have multiple inputs or outputs, or where materials are recycled. In reality the life cycle of a product is part of a complex network of processes, rather than a simple unidirectional flow. One of the strengths of using LCA software is that it has been developed to deal with these complex networks.

Resource allocation

Where there are multiple inputs and co-products sharing a given process, it is necessary to partition the resource consumption and pollutant emissions between them (ISO, 2006b). The principles for this allocation should be defined in the goal and scope, but are applied during the LCI data collection stage. Note that a co-product that has no economic value is considered waste, and is, therefore, not included in the allocation process (WRI and WBCSD, 2011a).

The types of process that raise allocation problems can be divided into three basic categories (Baumann and Tillman, 2004): multi-output - where several co-products are produced; multi-input, such as waste disposal to landfill - where different types of input may not all be equally responsible for pollutant emissions; and open-loop recycling - where the resource extraction,

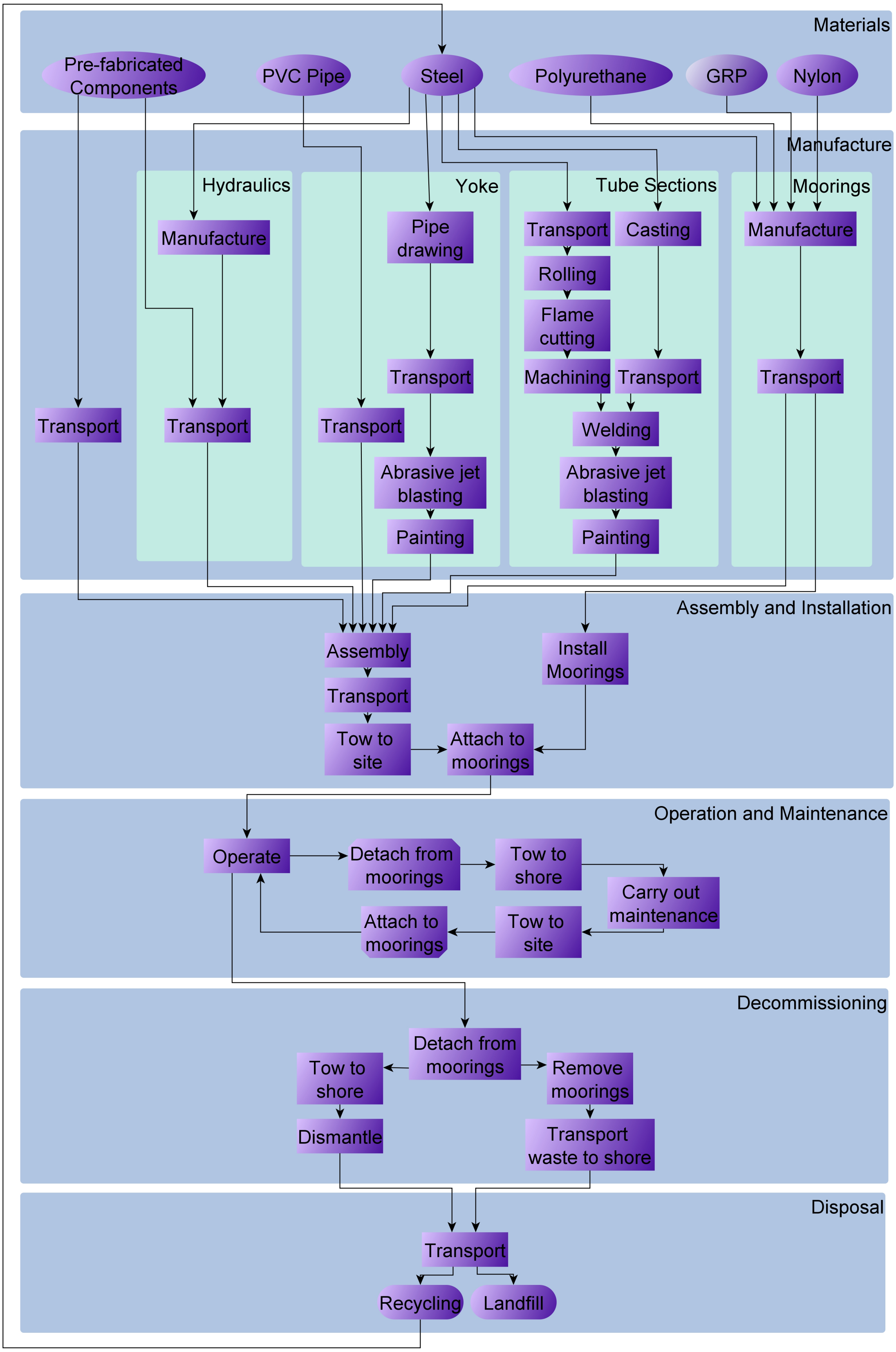


Figure 2.2: Detailed flow chart for the Pelamis wave energy converter

recycling process, and end-of-life disposal impacts need to be divided between the primary product and the recycled product.

Standard methods have been developed for dealing with allocation problems, either by avoiding allocation entirely or developing rules for partitioning the resource consumption and pollutant emissions. These must all meet the fundamental requirement of ISO 14044 that the “sum of the allocated inputs and outputs of a unit process shall be equal to the inputs and outputs of the unit process before allocation” (ISO, 2006b). Where possible, allocation should be avoided by using process subdivision or system expansion. The alternative is to allocate emissions and consumption by considering the underlying physical, economic or other relationships.

Process subdivision can be applied where it is possible to divide the common process into multiple sub-processes; the resource consumption and pollutant emissions are disaggregated by sub-metering specific process lines, or developing more detailed process models. The common process only needs to be sub-divided to the point where the studied product and its function are isolated (WRI and WBCSD, 2011a). Process subdivision involves examining the common process at a high level of detail, and this might not always be possible; for example, the separate processes in an oil refinery can be sub-divided according to the different types of fuel output, but the impacts of the initial extraction of crude oil cannot be partitioned in the same way.

An alternative method for avoiding an allocation problem is to apply system expansion. This involves expanding the system boundary to include relevant parts of the life-cycle of the co-products, and using this to estimate the resource consumption or pollutant emissions attributable to that co-product. One example of this would be a situation where the co-product is solely used for electricity generation: it would be reasonable to assume that the impacts of electricity generation from the co-product are equal to the average impacts of electricity generation, and thus the impacts attributable to the product of interest are the remainder once these have been removed. The significant limitation of this method is that it requires the resource consumption or pollutant emissions of the co-product, or a similar product, to be known (WRI and WBCSD, 2011a; Baumann and Tillman, 2004).

If allocation cannot be avoided, the resource consumption and pollutant emissions should be partitioned between the co-products according to the underlying physical relationships. These allocation rules should reflect the “way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system” (ISO, 2006b). This means that they will not necessarily be in proportion to simple measurements like mass. One example of this would be in transportation of goods: if the loading of the vehicle is limited by mass then the allocation should be by mass, but if it is limited by volume then the impacts should be allocated according to the volume of the different goods being transported (WRI and WBCSD, 2011a; Baumann and Tillman, 2004). The factor that is chosen for performing the physical allocation should be that which most accurately reflects the underlying relationship between the studied product, co-product and process. Other potential physical allocation factors could be (WRI and WBCSD, 2011a):

- Energy content of heat and electricity co-products;
- Number of units;
- Chemical composition.

Only in situations where allocation is unavoidable and suitable physical relationships cannot be identified should other relationships be used to partition the process impacts. This is most commonly economic allocation: where the resource consumption and pollutant emissions are divided according to the economic values of the products at the point of leaving the common process. Other allocation methods may be applied where there are conventions for a particular sector.

Recycling

Allocation principles also apply to reuse and recycling situations (ISO, 2006b). The potential to recycle materials can have a significant effect on the environmental impact of a product, as recycling provides the opportunity for avoiding both the environmental impacts of primary material production and of waste treatment. However, recycling must be treated with care to avoid the double-counting that can arise when any reduction in resource consumption or pollutant emissions is assigned to both the recyclable scrap and the resulting product. This problem arises in both open and closed-loop recycling situations - whether the material is recycled into the same production line, or goes on to become a different product. With closed-loop recycling it may be most straightforward to apply system expansion to include the recycling process in the product life cycle, but this is not possible for open-loop recycling, which is much more widespread.

Currently there is no consensus on the most appropriate methodology for allocating the benefits of recycling - whether to consistently allocate it to the product that uses the recycled material, the product that produces the recyclable scrap, or to partition it between both products (Jones, 2009; Hammond and Jones, 2010). There are three principal methods for dealing with recycling allocation problems in LCA: the recycled content method, the closed loop approximation method and the 50:50 method (Hammond and Jones, 2010). Each of these methods involves making an assumption about which product is responsible for the recycling credit, and therefore there is guidance about which to use for each situation. This is discussed further in Section 2.5.

The recycled content approach is one of the most commonly applied allocation methods, as it is used in the assessment of cradle-to-gate impacts of materials for LCI datasets. The basis of this method is that recycling is of no benefit unless the waste material is consumed, so the benefit should be allocated solely to the product that consumes the recycled material. It is described in Equation 2.1. Any recycling credit (the reduction in resource consumption or pollutant emissions) is allocated to the product that uses the recycled material, as recycling is of no benefit without the resulting material being consumed. In order to avoid double-counting, no credit can be taken for recycling materials at their end-of-life. However, the principle of this approach is that the product is only responsible for impacts directly caused by that product, so any end-of-life recycling does reduce the amount of material going to waste disposal, and therefore the end-of-life impacts are reduced. This method is also referred to as the 100:0 method or the cut-off method (Hammond and Jones, 2011; Baumann and Tillman, 2004).

$$E = (1-R)E_v + RE_R + (1-r)E_d \tag{2.1}$$

where:

E	=	Embodied impact per unit of material
E_v	=	Embodied impact of virgin material
E_R	=	Embodied impact of recycled material
E_d	=	Embodied impact of waste disposal
R	=	Recycled content
r	=	Recycling rate at end-of-life

Recycled materials could not exist without a primary product to generate them, and therefore it could be argued that the recycling credit should be allocated to the product that generates the recyclable scrap. In this case, the recycling credit can be calculated with the closed-loop approximation method, also known as the 0:100 method or the substitution method, which is described by Equation 2.2. This method is based on the assumption that open-loop recycling can be approximated by a closed-loop model, applying system expansion to calculate the recycling credit. It is, therefore, only valid for materials like metals that do not suffer a significant loss of quality during the recycling process. All of the credit from the recycling of scrap materials is assigned to the product, and any recycled content of the input materials is disregarded by assuming that all input material is primary material. This neglects any potential benefit of using a high proportion of recycled material in the manufacturing stage.

$$E = (1-r)E_v + rE_R + (1-r)E_d \tag{2.2}$$

The 50:50 method is a compromise that recognises that both the upstream and downstream products are necessary for recycling (Jones, 2009). It also promotes sustainable design by allocating credit for both minimising primary material consumption and maximising the recyclability of materials at the end-of-life. The simplest implementation of this method, advocated by Hammond and Jones (2011) and described in Equation 2.3, is to allocate 50 % of the credit from the recycled content and 50 % of the credit from scrap recycling to the studied product. While the 50 % figure is fairly arbitrary, it is no more arbitrary than the 100:0 or 0:100 ratios applied in the other methods.

$$E = \frac{1}{2}(1-R)E_v + \frac{1}{2}RE_R + \frac{1}{2}rE_R + \frac{1}{2}(1-r)E_v + (1-r)E_d \tag{2.3}$$

Alternative approaches do exist for dealing with recycling allocation problems, but these all require an understanding of the whole recycling chain, and therefore the collection of information outside the system boundary. Such methods, particularly the 'relative loss of quality' method (described in greater detail in Baumann and Tillman (2004)), may be more appropriate for degradable materials, such as paper and plastic, that lose quality during the recycling process.

Accounting for energy use

All product life cycles include energy consumption processes. Energy may enter and leave the system boundary as an energy source (raw material) or as an energy carrier (fuel or electricity). ISO 14044 requires these energy inputs and outputs to be treated similarly to any other process input or output to an LCA (ISO, 2006b). A Cumulative Energy Requirements Analysis should be carried out to estimate both the direct and indirect life cycle energy consumption - including both upstream energy use and feedstock energy, in addition to energy directly consumed in the life cycle processes of the product system itself. This means that full LCAs should include an estimate of the embodied energy of the product.

Feedstock energy is defined as the heat content of any raw material that is not used as an energy source within the product system, such as oil products used as the raw material for plastics (ISO, 2006b). Care must be taken to avoid double-counting when calculating the embodied feedstock energy of the product. Published LCAs often consider only the feedstock energy that represents a permanent loss of valuable resources, such as fossil fuels, and therefore do not include the heat content of raw materials that are more easily renewed, like wood (Hammond and Jones, 2008b; Mortimer *et al.*, 2003).

Care must also be taken to include the full life cycle energy consumption of fuels and electricity. These energy carriers are often quantified in terms of their available energy, but energy will have been consumed in their production and transmission. The total energy requirement or embodied energy of the product will be the total consumption of primary energy involved in its life cycle (Mortimer *et al.*, 2003), with primary energy defined as the energy extracted from the natural system to produce the fuel or electricity (Baumann and Tillman, 2004). In order to estimate the primary energy for common energy carriers, primary energy multipliers can be defined that indicate the amount of primary energy required to produce one unit of delivered energy. Different methods exist for determining the primary energy requirement of energy carriers, each with different approaches over: whether to consider the lower or higher heating value; whether to distinguish between renewable and non-renewable resources; or how to handle nuclear and hydroelectricity. The Cumulative Energy Demand (CED) method is one option for calculating the energy requirement or embodied energy of a product (Hischier *et al.*, 2010). It divides primary energy carriers into eight distinct categories, and each is treated separately. This method has its limitations, however, and considerable variation remains in the calculation of total primary energy consumption within LCA.

Life cycle impact assessment (LCIA)

The final stage of an LCA is to interpret the results of the inventory analysis in a life cycle impact assessment. This aims to describe the results as a set of environmental consequences, or potential impacts; an LCI may include hundreds of different resources and pollutants, and the LCIA stage makes these more understandable and environmentally relevant by classifying and characterising them according to their impact categories.

The first step in the LCIA is to identify and select impact categories and endpoints that meet the requirements of the goal and scope. Impact categories represent environmental issues of concern, such as global warming potential. The category endpoint is the actual issue of concern or damage potential, such as the environmental effects of climate change. This may be difficult to objectively quantify, so impact category indicators are defined to be the quantifiable representation of the impact category, also known as the midpoint, such as mass of carbon dioxide equivalent. Impact categories should be selected to cover all environmental problems of relevance within the bounds of the goal and scope, should be mutually independent to avoid double-counting, and should be scientifically verifiable (ISO, 2006b).

In theory, an LCA can examine any environmental impact that it is possible to objectively quantify, but three general categories should be considered - resource use, human health, and ecological consequences. These are usually further sub-divided into more specific impacts, which may include:

- Climate change or global warming potential - The greenhouse gas emissions or the carbon footprint of a product, which is always reported in kg CO_2 eq
- Ozone depletion potential - The emission of gaseous compounds that contribute to the depletion of stratospheric ozone
- Ozone formation potential - The emission of nitrogen oxides and volatile organic compounds which can react, in the presence of sunlight, to form ozone in the lower layers of the atmosphere (also known as smog)
- Acidification potential - The emission of gases that dissolve in atmospheric water to form acid, and fall as acid rain
- Eutrophication potential - The emission of nutrients to an ecosystem in large enough quantities to cause a change in species composition or limit biological growth
- Toxicity potential - The emission of substances toxic to humans and ecosystems
- Radiation potential - The emission of substances that can increase the level of ionising radiation in the environment
- Carcinogenic potential - The emission of substances that may cause cancer in humans

- Particulate emissions - The emission of particulates that may be hazardous to humans and cause respiratory problems
- Land-use change - The area of land that is changed as a result of the product life cycle
- Resource use - Depletion of resources
- Waste - Total bulk, radioactive, or hazardous waste

Once the impact categories have been selected the LCI results can be sorted and assigned to their relevant impact category(ies) in the classification stage. Some environmental loads might be assigned to more than one category. These must be independent of each other to avoid double-counting; for example, NO_x can take part in chemical reactions leading to photochemical ozone formation, then cause the release of an acidifying hydrogen ion before contributing to eutrophication (Baumann and Tillman, 2004). Similarly chlorofluorocarbons (CFCs) are known to contribute to ozone layer depletion, while also having a global warming potential.

The third mandatory step in an LCIA is characterisation - the calculation of the magnitude of the environmental impacts per category, reported in terms of the category indicators. Characterisation models are used to estimate characterisation factors to describe each pollutant emission in terms of the category indicator. However, the environmental consequences of different pollutant emissions are complex and often poorly understood, leading to the development of a range of different characterisation models (Baumann and Tillman, 2004). While the underlying physical mechanisms for some impact categories are relatively simple and well known, or have been studied in detail, such as climate change (IPCC, 2007), considerable debate remains over the units of measurement and characterisation factors for others, particularly the toxicity categories.

Standard impact assessment methods

Although ISO 14044 allows proprietary characterisation models and impact assessment methods to be developed (ISO, 2006b), many standard methods exist. Several of these are examined in detail in Chapter 4, which also examines the implications of LCIA method selection on the results of an LCA. The selection of a standard impact assessment method should be based upon two key criteria: that it includes all relevant impact categories, and that the number of mismatches between the inventory results and characterisation factors is minimised. Mismatches are defined as pollutants or resources listed in the LCI for which no characterisation factor is provided.

Standard impact assessment methods fall into two categories: those that present the results as a set of midpoint impact potentials, and those that present the results as endpoint damage potentials. The limitation of the latter is that some level of weighting is required, which adds uncertainty and makes the results invalid for comparison (ISO, 2006b). However, midpoint impact potentials can be considered to be more abstract and difficult to interpret. Carbon footprints and embodied energy are both midpoints, as they do not provide information about the damaging effects of increased levels of greenhouse gas emissions or energy consumption.

The four most widely used midpoint impact assessment methods are EDIP (Environmental Design of Industrial Products, Hauschild and Potting (2005)), EPD (Environmental Product Declaration, The International EPD Cooperation (2008)), CML 2 (Institute of Environmental Sciences, 2013) and ReCiPe (ReCiPe, 2013). These all use similar factors to calculate the global warming and ozone depletion potentials, presenting the results with the same units. There is also some consensus in the calculation of acidification and photochemical oxidation potentials, but other impact categories, such as toxicity, resource consumption and land-use change are dealt with very differently.

There are also three endpoint impact assessment methods that are commonly used in published LCAs: Eco-indicator 99 (Goedkoop and Spriensma, 2001), Ecological Scarcity or Eco-points method (Frischknecht *et al.*, 2009) and ReCiPe (which includes both midpoint and endpoint factors) (ReCiPe, 2013). These mostly apply very different factors for calculating the damage potentials, although the method used in ReCiPe was based upon that developed for Eco-indicator.

Normalisation and weighting

Three further optional stages can be applied during the LCIA to refine the results and present them in a format that can be easily interpreted. The first of these is normalisation. This allows impact potentials to be compared across impact categories, by relating the characterisation results to a reference value, such as the total quantity of pollutants emitted in a region. Normalisation should always be carried out according to a clearly defined set of factors based on a scientific measure; for example, the EDIP impact assessment methodology contains normalisation factors that compare the impact potentials to those of a typical person over the course of a year (Hauschild and Potting, 2005). Normalisation may provide additional information for examining where the greatest reduction in environmental impact may be made.

The second optional stage is grouping, which involves sorting and possibly ranking the category indicator results. Grouping might be carried out according to geographical coverage (global/regional/local impacts) or impact priority. Grouping is also often carried out on LCI results according to physical properties, such as emission to air, water or soil.

The final optional stage of an LCIA is the aggregation of characterisation results into a single value, through weighting. This is achieved by applying weighting factors to the data. These weighting factors are normally subjective and not based on measurable scientific data, so ISO 14044 prohibits the use of weighting in LCA studies that are to be used to make comparative assertions for public disclosure (ISO, 2006b).

Interpretation and presentation of results

The final stage of an LCA is to interpret and present the results, drawing conclusions and making recommendations. The results, data, methods, assumptions and limitations should be presented in enough detail to allow the reader to understand the inherent complexities and trade-offs of the LCA. It may not be practical or useful, however, to present all results; for example, the inventory for the study presented in Chapter 4 contains almost 1000 different substances, and it is unlikely that the presentation of this full inventory would be of use to the reader. It is, therefore, necessary to refine the results so that meaningful conclusions can be drawn, without introducing any form of bias.

The presentation of the results should reflect the aims of the study outlined in the goal and scope; therefore, if the LCA has been carried out to only assess the carbon footprint and embodied energy of a product, it is acceptable to simply present the results of the energy analysis and the global warming impact category. In such a case, it may also be of interest to present the inventory values for carbon dioxide and other significant greenhouse gases. Alternatively, where the goal of the study is to examine the potential to reduce overall environmental impacts, the results would need to be presented for each life cycle stage or process. In this situation, the normalised results would also be of interest to provide information on the relative severity of the different calculated impact potentials. The conclusions reached in the interpretation stage will also have some bearing on the presentation of the results.

Sensitivity and uncertainty analysis

The international standards also require the robustness and reliability of the final results and conclusions to be tested by examining their sensitivity to data uncertainty, practitioner assumptions and methodological choices (ISO, 2006a,b). The following checks are recommended (Baumann and Tillman, 2004):

1. Completeness Check - identifies any data gaps in the inventory, and mismatches with the impact assessment method;
2. Consistency Check - assesses the appropriateness of the model and applied method to meet the requirements of the goal and scope;
3. Uncertainty Analysis - identifies the effect of data uncertainties and approximations;
4. Sensitivity Analysis - identifies and tests the effect of variations in critical data assumptions, such as design life;
5. Variation Analysis - examines the effect of alternative scenarios and models;
6. Data Quality Assessment - assesses the compliance of the collected data with the data quality requirements set out in the goal and scope, and the degree of any data gaps;

A detailed example of a sensitivity and uncertainty analysis is given in Section 4.4.

Completeness and consistency

Completeness and consistency checks should be carried out to verify that all relevant information and data is available and complete, and confirm that the analysis follows the guidelines set out in the goal and scope. Specifically, this includes: identifying that data quality requirements have been met and allocation rules have been followed; that the analysis includes all processes within the system boundary; and that data coverage meets the right geographical and temporal constraints (ISO, 2006b). Where any data gaps or inconsistencies are found, further investigations should be made at this stage to confirm that they do not affect the ability of the analysis to meet the requirements of the goal and scope.

Uncertainty analysis

The uncertainty analysis examines how uncertainties and variability in the input data affect the reliability of the results. Often the materials and process data gathered to create the life cycle inventory is generic data, but the environmental performance of different suppliers can vary, and production processes can operate under different conditions, so this generic data has an uncertainty range (Baumann and Tillman, 2004). Furthermore, there may be uncertainties or variability in the primary data collected for the product being studied, such as material quantities. Statistical analysis of the uncertainty and variability of the input data, often using Monte Carlo simulations, allows the uncertainty of the results to be characterised by ranges and/or probability distributions (ISO, 2006b).

Sensitivity analysis

This analysis tests the sensitivity of the results to practitioner estimates of critical data, such as design life. Also, where there is insufficient information to include the uncertainty of primary data in the statistical uncertainty analysis, it can be included here. Typically, a sensitivity analysis is carried out by varying assumptions and data by a given range and examining the effect on the results. The findings are normally expressed as the percentage change or absolute deviation from the reported value.

Variation analysis

The variation analysis provides the opportunity to examine how methodological choices affect the results. Within the LCA methodology outlined in the International Standards, many choices are made by the practitioner, such as: allocation rules, cut-off criteria, boundary setting and system definition, selection of impact category, assignment of inventory results (classification), calculation of category indicator results (characterisation), normalisation, weighting method and data quality (ISO, 2006b). In contrast to the factors considered in the sensitivity analysis, these methodological choices are not numerical, so instead the effect is tested by applying alternative options and reporting the results.

LCA tools

Full life cycle assessment involves gathering and processing a large quantity of data. While it is fairly straightforward to carry out a partial LCI, such as that presented in Parker *et al.* (2007), with a simple spreadsheet, LCA software packages facilitate data management within much more complex analyses. Furthermore, comprehensive LCI databases have been developed that collate and verify data for a wide range of materials and processes, simplifying the LCA calculation process and improving comparability between published studies.

LCA software packages

Many different LCA calculation tools exist, each offering different features and databases. Two of the most popular, which are frequently used for the analysis of power generation, are SimaPro and GaBi (PRe Consultants, 2010; PE International AG, 2013). These allow simple construction of a life cycle network, with materials data available from in-built LCI databases. A range of LCIA characterisation methods are also readily available within the software, simplifying the calculation process. The danger of such tools is that the user may use default values, thus making methodological assumptions that they are unaware of, but their power lies in making large quantities of data readily available, and in speeding up the construction of the analysis. Furthermore, the leading tools are highly flexible and allow results to be extracted at all stages of the LCA, as well as detailed examination of the process network.

LCI databases

A number of LCI databases are available that provide comprehensive information for a range of materials and processes. The choice of data from a particular database depends upon its compliance with the data quality requirements and system boundary defined in the goal and scope. Some LCA software, such as GaBi, contain proprietary databases, but others, such as SimaPro, build in publicly-available databases such as Ecoinvent, the European Life Cycle Database (ELCD) and the US LCI database (Ecoinvent, 2010; European Commission, 2013; NREL, 2012).

Only one UK-specific LCI database has been identified - the Inventory of Carbon and Energy (ICE) (Hammond and Jones, 2011). This was developed by researchers at the University of Bath to provide comprehensive and verified cradle-to-gate carbon footprint and embodied energy data for a range of different construction materials. This database was formed by selecting data from published literature according to a specific set of criteria, and then calculating average values and ranges to be used in partial LCAs concentrating on energy and global warming potential.

One of the most comprehensive databases for full resource consumption and pollutant emissions of materials and processes in Europe is the Ecoinvent database (Ecoinvent, 2010). This was developed by the Swiss Centre for Life Cycle Inventories and contains process-based LCI data for a number of materials and processes, with corresponding information about process networks and flows. It also presents cradle-to-gate, gate-to-gate and gate-to-grave impact assessment results using a number of leading characterisation methods.

Alternative databases are also available that contain information for other geographical regions, other LCA methodologies (such as input-output data), and a range of different data quality criteria.

2.5 Methodological limitations

The generic methodological framework for LCA allows it to be applied to a wide range of different products and services, but introduces considerable scope for inconsistencies and reduces the comparability of the results. Meta-analyses of published LCA data, such as the development of the Inventory of Carbon and Energy (Hammond and Jones, 2011) or the LCA Harmonization Project (Warner *et al.*, 2010) have observed that there is a variation in published data stemming from differences in boundary definitions, age of data sources, and rigour of the analyses. Several papers have identified specific limitations of LCA and called for further methodological guidance and better, more consistent reporting to improve reliability and comparability (Price and Kendall, 2012; Davidsson *et al.*, 2012; Finkbeiner, 2009). Although these publications often provide recommendations, few have attempted to quantify the effects of individual practitioner choices.

This section examines these limitations, with particular reference to applying LCA to estimate carbon footprints and embodied energy of power generation technologies. It also includes details of specific additional guidance provided for carbon and energy audits, and sector specific recommendations for power generation to further refine the methodology (WRI and WBCSD, 2011a; BSI, 2011; ISO, 2013). The impact of these limitations is examined in greater detail in the analysis presented in Chapter 4.

2.5.1 Type of analysis

Although process-based LCA is the most commonly-used method for carbon footprinting, and that defined by the guides and standards, it has its limitations, particularly in that the definition of a system boundary can introduce significant truncation errors (Crawford, 2005). There is growing support for hybrid input-output methodologies that are thought to avoid these errors (Crawford, 2005; Lenzen and Munksgaard, 2002; Wiedmann *et al.*, 2011), but it is unclear whether these actually result in an overestimate of environmental impacts. Given that the guidance for carbon footprinting and the majority of existing studies are based on process-based LCA, this remains the recommended method for calculating carbon footprints and embodied energy, and some rules have been developed with regards to the system boundary and cut-off criteria to maximise comparability between studies.

2.5.2 System boundary

Defining the system boundary for an analysis includes specifying the life cycle stages to consider, the time-frame and the physical boundary of the analysis subject. There is broad agreement in the literature that carbon and energy audits should include every stage of the generator and fuel life cycles, from extraction and processing of raw materials to decommissioning and disposal (ISO, 2013; The International EPD System, 2013; Gaines and Stodolsky, 1997). Many existing studies, however, are not this comprehensive; with conventional generation it is typical to consider only the life cycle of the fuel (Whitaker *et al.*, 2012; Price and Kendall, 2012). In these cases the exclusion of other life cycle stages could be a function of the specified cut-off criteria, as the impacts associated with the extraction and combustion of fuel are much more significant than those of the generator itself; however, the truncation errors from such assumptions affect the comparability of studies across the sector, so should be avoided.

The decommissioning and disposal stages provide a particular problem for power generating technologies, as they are often poorly understood. In order to determine the impacts of disposal of any product at its end-of-life, the guidance requires that a waste disposal profile is developed for each component or material. This should be based upon published international, national or industry guidelines and standards but, where such information is not available, a profile may be developed by the practitioner carrying out the carbon footprinting analysis (BSI, 2011). With emerging technologies, such as wave power, realistic assumptions need to be made, usually with advice from the device developer.

The temporal boundary, or time-frame, for an LCA of a power generating technology is generally taken to be its design life. For some types of installation such as hydroelectric dams, however, this is difficult to identify - the oldest grid-connected generators in the UK were commissioned in 1926 and there appear to be no plans to decommission them (Scottish Power, 2012). Published guidance suggests that, where the lifetime of a product is unknown, it should be taken to be 100 years (BSI, 2011; WRI and WBCSD, 2011a). This may be too long for renewable energy converters, and in their recent harmonisation project NREL concluded that the design life of wind turbines should be harmonised to 20 years (Dolan and Heath, 2012). Counter to this approach, in a comparative analysis of four very different power technologies, Rule *et al.* (2009) set the temporal boundaries to be 100 years, in order to maximise comparability across technologies. This study, however, included allowances for maintenance and replacement of generators, and therefore still required design life estimates. In order to produce comparable carbon footprints for power generation, there is an argument that typical design lives should be agreed for each type of technology; at the very least an analysis of the sensitivity of the results to the design life should be reported, alongside a clearly defined temporal boundary. The existing Product Category Rules for carrying out Environmental Product Declarations (EPDs) of electricity generation include a list of the typical technical service lives for a range of different generating technologies (The International EPD System, 2013).

The physical system boundary of a carbon footprinting study or LCA of power generation systems should be the point of connection with the grid. Some studies, particularly EPDs, do include the distribution system, but this is usually reported separately (The International EPD System, 2013; Vattenfall, 2013). The transport of employees is typically excluded, but all manufacturing, transport, storage and energy are included, such as all emissions from fuel inputs and other upstream processes (BSI, 2011). Capital goods, such as lorries and machinery, are used throughout the system life cycle, but the existing guidance requires only the impacts of their operational emissions to be included (WRI and WBCSD, 2011a). However, as discussed in Section 2.4.2, this may result in up to 30 % of the impacts being neglected, so the international standards specify that the manufacture, maintenance and decommissioning of capital equipment should be included (Goedkoop *et al.*, 2008; ISO, 2006a). This disagreement in existing guidance means that the choice of whether to include capital goods is left to practitioner discretion. Comprehensive LCI databases, such as Ecoinvent, typically do include capital goods in their data, although rarely in the impacts of an electricity supply (Ecoinvent, 2010).

Where any process results in a direct land-use change the emissions and removals associated with it should be included in the system boundary according to recognised guidance, such as the GHG Protocol Land Use, Land-Use Change, and Forestry Guidance (WRI, 2006; ISO, 2013). Data sources for such information include other studies, such as that carried out by Nayak *et al.* (2008) for wind farms on peat lands, or standard factors, such as those included in Annex C of PAS 2050 (BSI, 2011).

The impact of reductions and GHG removals associated with carbon storage can be included in an LCA or carbon footprinting study, but should be reported separately. Stored carbon is any carbon that is retained in a form other than atmospheric carbon for the entire 100-year assessment period. This includes carbon capture and sequestration when carried out as part of the life cycle of the given power generation technology (BSI, 2011). Offsetting, where the GHG emissions of a product are compensated for by reducing emissions or removing GHGs in another process, should not form part of an LCA, as the emissions savings occur outside the system boundary (ISO, 2012; BSI, 2011; WRI and WBCSD, 2011a).

The location of all manufacturing plant and the final installation of any power generating technology can also affect the results, so should be clearly reported (Lenzen and Munksgaard, 2002). This is difficult to normalise as it is generally specific to each analysis scenario.

The review of existing published studies, presented in Chapter 3, found that there is considerable variation in the physical boundaries applied for carbon and energy audits and LCAs of power generating technologies, with analyses of wind power ranging from including only a single wind turbine through to an entire farm and local transmission. Most studies do not clearly report whether the impacts associated with capital goods and direct land-use change have been included. The effect of such variation in assumptions is difficult to assess, but must be considered when comparing the results of different studies.

2.5.3 Cut-off criteria

As discussed in Section 2.4.2, the international standards allow cut-off criteria to be specified to exclude some inputs and outputs from an analysis. Guidance for carbon footprinting recommends that any process likely to contribute less than 1 % to the life cycle GHG emissions may be excluded, provided that at least 95 % of the total emissions are accounted for (BSI, 2011). The PCR for carrying out EPDs of electricity generation are more onerous, requiring that all processes contributing more than 1 % to the overall environmental impact should be included (The International EPD System, 2013). A sensitivity analysis to the selected cut-off criteria should be presented.

2.5.4 Functional unit

The definition of the functional unit can be another potential source of error in LCA. The device being studied may have multiple functions, so the functional unit must be carefully selected to meet the goal and scope (Bousquin *et al.*, 2012; Reap *et al.*, 2008). Strict adherence to this functional unit, however, can skew the results when there are multiple functions, and introduce difficulties in allocation. In analysing power generation technologies these problems are much less significant, except in the case of combined heat and power generation, and the functional unit is consistently defined as a unit of electricity generated, normally 1 kWh (The International EPD System, 2013).

2.5.5 Scope of analysis

Life cycle assessment is inherently limited in scope by the focus on environmental impacts, and there are concerns that the exclusion of social and economic effects may fail to address potential trade-offs (Reap *et al.*, 2008; UNEP, 2011a). Carbon footprint and embodied energy studies are even more limited, as these typically only consider one or two impacts. It is, therefore, important to present the results of such studies alongside their goal. Where the motivation is to specifically address questions of carbon payback, for example, the exclusions of other impact categories may be justified.

Within the limitation of considering only environmental impacts in LCA, the standards allow for significant variation in the range of impacts that might be considered. As discussed in Section 2.4.2, many different standard impact assessment methods have been developed, each with different impact categories and characterisation factors. Practitioners may also choose to apply their own factors, and the lack of consensus on the unit of measurement for some impact categories precludes comparison of different studies.

The inconsistency of characterisation factors leads to considerable variation in results, which is a particular issue with embodied energy analysis; in one instance a 45 % discrepancy in embodied energy was observed when different impact assessment methods were applied to the same system (Davidsson *et al.*, 2012). Estimates of the primary energy or energy density of energy carriers, such as fuels and electricity, can be very variable (Hischier *et al.*, 2010) - with the estimated energy density of uranium found to vary by 250 % in the analyses presented in Section 4.4.3. Electricity generation can introduce particular problems, as the primary energy consumption associated with a unit of electricity depends upon the power generating technologies and generation mix (Davidsson *et al.*, 2012). It is therefore important to detail assumptions made about primary energy multipliers and characterisation factors.

In carbon footprinting analyses there are also inconsistencies in the scope of emissions that are included. Many of the existing studies for power generation technologies only consider life-cycle CO_2 emissions. As a carbon footprint is a measure of climate change impact, however, other greenhouse gases should be included, particularly where emissions are significant. Some standards recommend including only the six gases specified by the Kyoto Protocol (WRI and WBCSD, 2011a; The Carbon Trust, 2012), perhaps due to much of the political motivation for carbon footprinting being driven by a commitment to this agreement. However, the inclusion of all gases listed by the IPCC will provide a more complete result, and is favoured in other standards (ISO, 2013; BSI, 2011). Furthermore, the ISO technical specification also recommends that fossil and biogenic GHG emissions should be documented separately.

Another discrepancy between studies is in the inclusion of feedstock energy in the embodied energy calculation. This is the heat content of energy sources that are used as raw materials rather than fuel within the product life cycle. ISO 14044 recommends the inclusion of feedstock energy (ISO, 2006b), but it is common practice to only include it if it represents a permanent loss of valuable resources - excluding the feedstock energy of timber but including that of oil products used to make plastics (Hammond and Jones, 2011). Decisions on whether to include feedstock energy should be reported.

2.5.6 Data quality

The quality of input data can significantly affect the reliability of results of any LCA. Primary data should be collected from the product manufacturer for all processes under their control, and it may be necessary to collect primary data from multiple manufacturers (WRI and WBCSD, 2011a; ISO, 2013). Secondary data is used for all other processes, including production of materials and manufacture of generic components, preferably from well documented studies or other competent sources (BSI, 2011). When impacts are dominated by upstream (e.g. fuel supply) or downstream (e.g. disposal) processes, poor data quality can be a significant problem (Schreiber *et al.*, 2012). Furthermore, the repeated re-use of secondary data from earlier published studies may lead to errors propagating through the literature undetected (Teehan and Kandlikar, 2012). In order to avoid some of these limitations in preparing an EPD, the PCR for electricity generation specifies specific source datasets (The International EPD System, 2013).

One specific area of uncertainty in data quality is in the GHG emissions associated with consumption of electricity. Current best practice assumes that the carbon intensity for electricity is the average carbon intensity of the local grid, specified for the UK as the published system average (Ricardo-AEA, 2012). Where the electricity source can be more precisely identified, however, such as from renewable generators that are not included in the reported average mix, the guidance allows lower emissions factors to be defined (BSI, 2011). The use of such average emissions may not actually reflect the true impacts of electricity consumption, as not all power stations respond equally to changes in demand. There is an argument that the emissions of electricity consumption should reflect the marginal generation mix (Davidsson *et al.*, 2012). This discussion is extended to the carbon displacement factor to use in calculating the carbon paybacks of renewable generators: current government recommendations in the UK require that the average carbon intensity be again applied, assuming that all generation is replaced equally, which is unlikely to be the case (Defra, 2013). Further discussion on the marginal emissions of electricity generation can be found in Chapters 5, 6 and 7.

2.5.7 Allocation and recycling

Allocation rules must be defined to describe how to partition the resource consumption and pollutant emissions between multiple inputs and co-products that share a given process. As detailed in Section 2.4.2, the guidance recommends that process subdivision or system expansion methods should be applied in the first instance and, where this is not possible, allocation should be based on the underlying physical relationships between the co-products. Economic allocation or other allocation rules should only be applied if the other methods cannot be used (ISO, 2006b, 2013; The International EPD System, 2013). Selected allocation methods should be reported alongside the results (WRI and WBCSD, 2011a); however, existing studies rarely state what allocation methods have been applied. Even the leading LCA software rarely details the allocation processes, although it would seem that process subdivision and physical allocation should be relatively straightforward using comprehensive LCI databases coupled with sophisticated computational tools.

The process of dealing with the allocation of recycling credit is also rarely addressed within existing studies. Recycling credit in an LCA is the reduction of the environmental impacts of a product due to the recycling of materials; if the product is manufactured from recycled material that is, in turn, recycled at the end-of-life, assigning credit to both the waste material and the resulting product would be double-counting. With recycling allocation methods rarely reported in published LCAs, double-counting of recycling credit may be common.

Standard recycling allocation methods have been developed to avoid the effects of double-counting, and are described in detail in Section 2.4.2. These meet the requirement of published guidance that the allocation of emissions and removals due to recycling should be carried out at the material level. Two standard recycling allocation methods are preferred: recycled content and closed loop approximation. The latter, which calculates the credit for recycling material at the end-of-life, is recommended for use where the virgin and recycled materials are indistinguishable, the market for recycled material is not saturated, and the time period of the use stage is short or well known (ISO, 2013). In all other cases the guidance recommends that the recycled content method should be applied, so that only the avoided emissions of using recycled materials in the manufacturing stage are considered (WRI and WBCSD, 2011a). In the power sector the recycled content method will be most appropriate for the majority of generating technologies, as the use stage is long and not necessarily clearly defined. Renewable energy converters, however, have relatively short design lives, and are often made of highly recyclable materials like steel - in this instance the practitioner may choose either method. Alternatively, the 50:50 method, which is a compromise between the two, might be the most appropriate, but is not advocated by current guidance documents. Further discussion of this topic can be found in Hammond and Jones (2010).

Chapter 3

Existing Carbon and Energy Audits of Power Generation

3.1 Introduction

Many life cycle assessments, carbon footprinting studies and energy analyses of power generating technologies have already been published, but the quality of these varies widely. In a recent systematic review of published carbon footprints, analysts at the National Renewable Energy Laboratory (NREL) in the USA found that, of over 2100 studies reviewed, fewer than 15 % met basic criteria for quality, relevance and transparency (NREL, 2013b). Furthermore, the majority of these considered only conventional power generation and selected renewable technologies, including coal, gas, nuclear, wind and bio-power; other established and emerging technologies, such as hydropower and marine energy, were found to be relatively poorly represented in the literature (NREL, 2013d). There is, therefore, significant scope for further analyses to be carried out to estimate the carbon footprint and embodied energy of a broader range of generating technologies, and for the quality of these to be improved to maximise comparability.

This chapter presents a detailed review of the published literature on the life cycle carbon and energy consumption of power generators, examining both the applied methodologies and the resulting estimates. The review concentrates on process-based analyses, as this is the most commonly applied methodology for power generation technologies, but hybrid studies are also considered. Setting renewable power generation in the UK context, the review concentrates on the most significant types of renewable and conventional generation in the current UK energy mix. The majority of electricity in the UK currently comes from coal, gas and nuclear power stations: specifically sub-critical pulverised coal plant, combined-cycle gas turbines and advanced gas-cooled nuclear reactors (although one nuclear power station (Sizewell B) is a pressurised water reactor). Wind is the most significant renewable energy source, followed by bio-power, with the installed capacity being 8.8 GW and 2.2 GW respectively at the end of 2012, each supplying around 5 % of total demand (RenewableUK, 2012; Elexon, 2013a; MacLeay *et al.*, 2013). Hydroelectricity and solar power are also significant, with 1.7 GW of installed capacity each, although the latter met only 0.3 % of demand while the former was responsible for 1.4 %.

3.2 Overview and Methodologies of Existing Studies

Existing studies of power generating technologies have applied a variety of different methodologies, possibly introducing significant variation to the results. It was found that the type of analysis was often related to the type of generation, with full life cycle assessments mostly concentrating on wind power (Vestas, 2006a,b; Ardente *et al.*, 2008; Tremeac and Meunier, 2009), and highly detailed Environmental Product Declarations (EPDs) being published for conventional thermal power stations (AEA Energy and Environment, 2009; Axpo, 2011; AEA Energy and Environment, 2008b). The vast majority of the studies reviewed, however, were process-based partial LCAs considering only carbon and energy (Walker and Howell, 2011; Parker *et al.*, 2007; Rule *et al.*, 2009; Douglas *et al.*, 2008), with the remaining analyses generally comprising detailed life cycle inventories (Dones *et al.*, 2005) and hybrid input-output analyses (Lenzen, 2008; Odeh and Cockerill, 2008; Wiedmann *et al.*, 2011).

In many cases the assumptions and methods used in the analyses were not reported in enough detail to address all of the key issues identified in the Section 2.5, or verify compliance with guidance such as BSI (2011) and WRI and WBCSD (2011a). In particular, practitioner methodological decisions on cut-off criteria, allocation rules and recycling methodology were rarely defined, even in reports that otherwise contained a high level of detail (Ardente *et al.*, 2008; AEA Energy and Environment, 2009; Kannan *et al.*, 2005). Other significant omissions were details of the physical and temporal system boundaries, end-of-life conditions and inclusion of capital goods. This highlights the need for improved compliance of published studies with reporting guidelines.

The scope of greenhouse gases (GHGs) included in the carbon footprints was usually reported, and typically only included three significant gases: carbon dioxide, methane and nitrous oxide (Dolan and Heath, 2012; Guezuraga *et al.*, 2012; Kannan *et al.*, 2007; Zhang *et al.*, 2009); although many analyses, particularly those for nuclear power stations, were limited to only carbon dioxide emissions (Warner and Heath, 2012a; Lenzen, 2008). The inclusion of all greenhouse gases identified by the IPCC was found to be relatively rare. The significance of this exclusion is examined in Chapter 4.

It was also observed that many analyses of conventional thermal power stations were limited to consider only the life cycle of the fuel. Where the impacts of construction and decommissioning of the power station were included, they were frequently taken from other studies (Whitaker *et al.*, 2012; Koornneef *et al.*, 2008). Studies carried out on renewable energy converters, however, did typically include the life cycle of the whole device or installation, due to the higher relative impact of the manufacturing and maintenance stages (Vestas, 2006b).

Despite these limitations, the recent LCA Harmonization Project, detailed in Section 3.3, found that the distribution of carbon footprints for wind power generation was very narrow (Dolan and Heath, 2012). This suggests that similar process-based life cycle assessments of similar technologies may not be too severely affected by practitioner assumptions. The much larger distribution of studies of coal-fired power stations can be attributed to variations in technology, fuel quality and fuel source (Whitaker *et al.*, 2012). Figure 3.1 shows a selection of published carbon footprints and embodied energy values for power generation technologies. In general these studies found that the greatest carbon emission and energy consumption for renewable generators arises from the extraction and production of stock materials (Parker *et al.*, 2007; Rankine *et al.*, 2006; Douglas *et al.*, 2008; Vestas, 2006b), while for nuclear and fossil fuel power, the extraction, processing and combustion of fuels has the greatest impact (Lenzen, 2008; Whitaker *et al.*, 2012). Although impacts of the distribution and transmission of electricity are not considered in this review, which concentrates on power generation, it has been found that the greatest impacts of the transmission network are principally due to power losses (Harrison *et al.*, 2010).

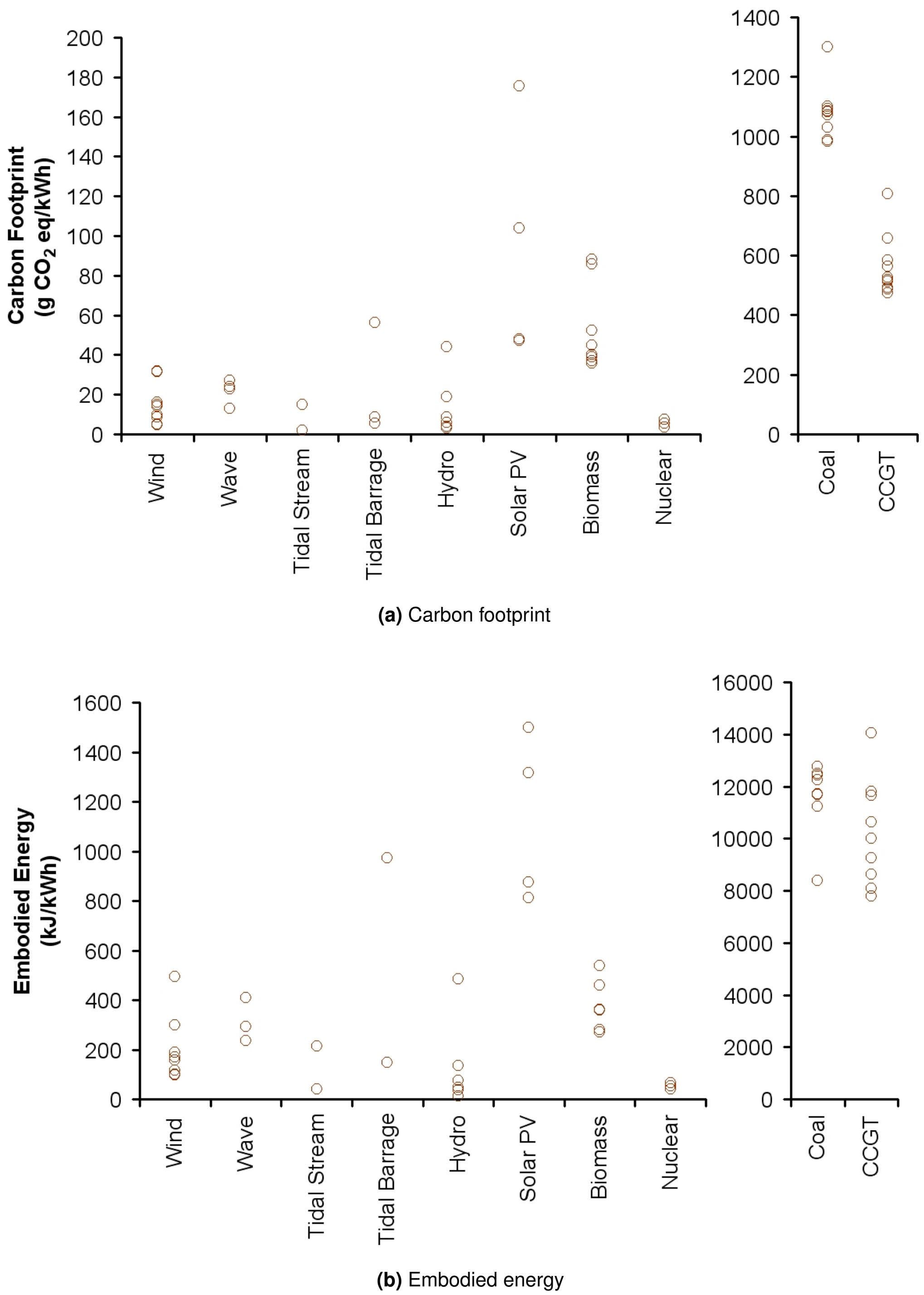
3.3 LCA Harmonization Project

As mentioned in the previous sections, the National Renewable Energy Laboratory in the USA has recently carried out an extensive LCA Harmonization Project concentrating on the life cycle GHG emissions of power generation. The project was initiated in response to concerns over the comparability of existing published LCAs, with the aim of developing guidelines for harmonising environmental impacts and using GHG emissions as a case study. 'Harmonisation' is the process by which methodological inconsistencies between previously published LCAs can be aligned to enable proper comparison and more generalised conclusions (Warner *et al.*, 2010).

Harmonisation can be carried out on studies that have used a similar LCA methodology for a similar technology. The LCA Harmonization Project considers only studies that apply the process-based attributional method, as these are the most common type of LCA in the existing literature. Neither consequential nor economic input-output studies are comparable with process-based analyses, but hybrid LCAs may be included in the harmonisation process following evaluation on a case-by-case basis (Warner *et al.*, 2010).

The aim of harmonisation is to enable comparison between studies while still maintaining variations introduced by each study's unique perspective, detail and insight. It is, therefore, not desirable to eliminate all of the methodological inconsistencies, so key areas have been identified for harmonisation: spatial variables, such as background energy mix or radiation levels; temporal impacts, such as day-to-day impacts or discount rates; system boundaries; functional units; and data/parameter inputs.

The guidance developed by NREL recommends that one of the initial phases of any harmonisation study should be to categorise and filter the papers under consideration. The filtering process



(a) Carbon footprint

(b) Embodied energy

Figure 3.1: Carbon footprints and embodied energy of power generation

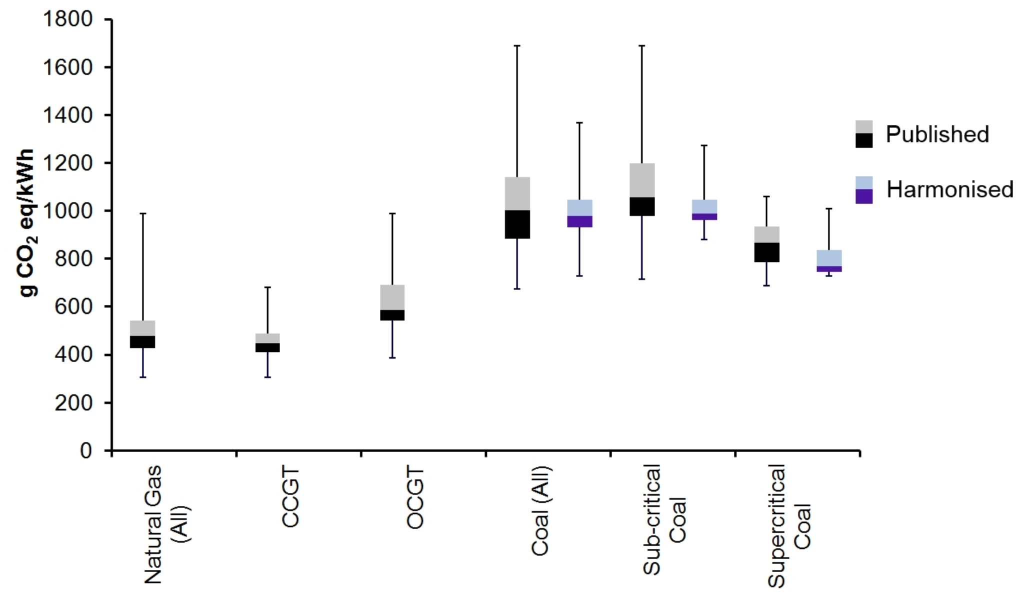
involves screening these analyses to establish whether they meet basic quality criteria on the applied LCA methods, the completeness of the reporting, and the recent or future relevance of the technology and input data. The harmonisation process itself is then divided into two stages: a preliminary stage, where simple steps, such as harmonisation of functional units and lifetime, are implemented; and a meta-modelling stage, which enables more detailed harmonisation of a selection of the literature. This LCA harmonisation process, detailed in Warner *et al.* (2010), was applied to filter and harmonise the GHG emissions of power generation, with the results published in a number of papers (Whitaker *et al.*, 2012; Hsu *et al.*, 2012; Warner and Heath, 2012a; Kim *et al.*, 2012; Burkhardt *et al.*, 2012; Dolan and Heath, 2012) and online (NREL, 2013d). A summary of the current findings is illustrated in Figure 3.2. This provides valuable insight into the quality and comparability of the existing literature on carbon footprints of power generation.

It is important to note, however, that the statistical results of the LCA Harmonization Project are not indicative of the accuracy of existing life cycle GHG emissions estimates - they only show the distributions of current estimates. Both empirical and theoretical data have been included, and equal weight is given to data points from separate studies as multiple data points emerging from the same study, which could skew the findings towards a particular set of practitioner assumptions (Dolan and Heath, 2012). Furthermore, the LCA methodology encourages the re-use of data from earlier analyses, and it is possible that errors could propagate throughout the literature (Teehan and Kandlikar, 2012). It is expected that the accuracy of carbon footprint estimates will improve with time, but the focus of studies such as the LCA Harmonization Project is to improve comparability rather than absolute accuracy.

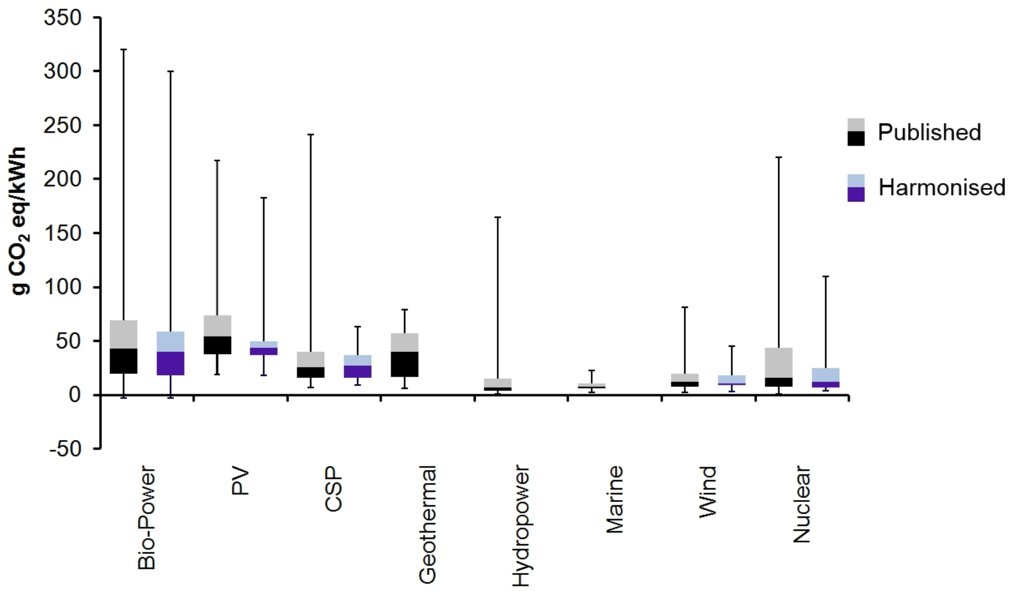
3.4 Renewable Energy Technologies

3.4.1 Wind

The installed capacity of wind power generation (Figure 3.3) continues to grow around the world, and it is currently the largest source of renewable electricity in the UK (MacLeay *et al.*, 2013). Further increases in the installed capacity of transmission-connected wind are expected, as both onshore and offshore farms are currently in construction and planning approval is expected for many more (RenewableUK, 2013). As a leading low-carbon energy source, a number of studies have been carried out to identify the carbon footprint of wind power and support its low-carbon credentials; some of these analyses also consider the embodied energy and other environmental impacts of wind turbines and farms. (Note that small-scale and building-mounted wind turbines are not considered in this section: the embodied carbon and energy of these are highly dependent upon the technology and the wind profile at the installation location, which is unlikely to have been selected for its wind availability. For more information see Phillips *et al.* (2007).)



(a) High-carbon technologies



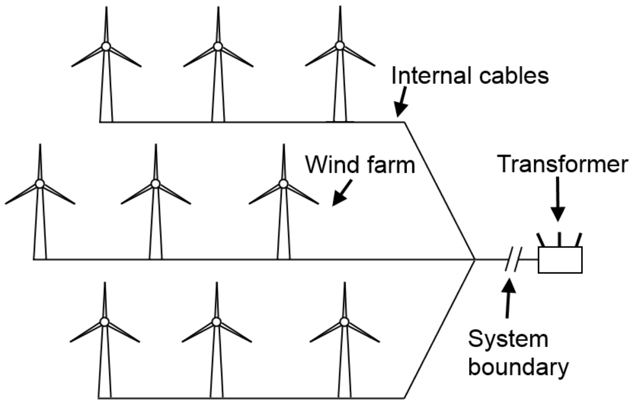
(b) Low-carbon technologies

Figure 3.2: Results of LCA Harmonization Study (from NREL (2013d))

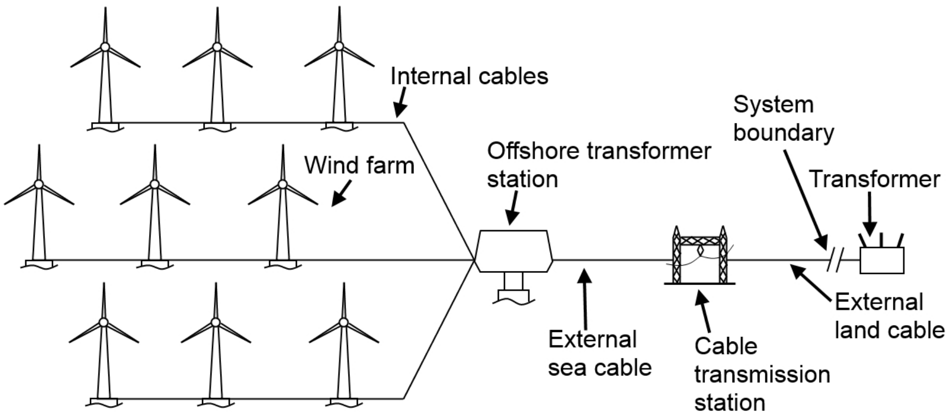
Figure 3.3: Whitelee wind farm

There are significant differences between the life cycle of onshore and offshore wind farms (Figure 3.4). With the typical system boundary set at the point of connection with the grid, normally either before or after the transformer, the equipment included in an onshore wind farm will be limited to turbines and cables; however, an offshore wind farm will also include equipment to collect, transform and export the power to shore. The precise design of such offshore power electronic equipment is still evolving. Furthermore, while the impacts of land-use change must be considered for onshore farms, particularly when constructed on peat lands (Nayak *et al.*, 2008), such as that shown in Figure 3.3, offshore farms involve maintenance by helicopter and boat, and have impacts on marine environments.

The LCA Harmonization Project identified approximately 240 LCA studies of onshore and offshore wind power generation, but found that only 49 of these met their quality criteria for estimating the carbon footprint of utility-scale wind power (Dolan and Heath, 2012). The published estimates ranged from 1.7 to 81 g CO_2 eq/kWh, with the median and interquartile range (IQR) both being 12 g CO_2 eq/kWh. Harmonisation of these studies was carried out at the level of the 'preliminary stage' outlined in Section 3.3: firstly by proportionally adjusting the published estimates to consistent values of capacity factor and system lifetime, and then subtracting or adding values to reach a consistent system boundary in terms of major life cycle stages. (Note that, while the capacity factor and system lifetime is a function of each specific wind farm and installation location, the purpose of the LCA Harmonization Study was to compare findings from different studies assuming a generic wind farm location.) Where possible, the global warming potentials for different gases were also harmonised. This harmonisation adjusted the range of estimated carbon footprint of wind power to 3.0 to 45 g CO_2 eq/kWh, with an IQR of 10 g CO_2 eq/kWh and a median of 11 g CO_2 eq/kWh.



(a) Onshore



(b) Offshore

Figure 3.4: Components of typical wind farms (After Vestas (2006b))

The study found that harmonisation of the capacity factor had the greatest impact on the results. Most significantly, however, it concluded that "the large number of previously published life cycle GHG emission estimates of wind power systems and their tight distribution suggest that new process-based LCAs of similar wind turbine technologies are unlikely to differ greatly" (Dolan and Heath, 2012). This study also found very similar carbon footprints for both onshore and offshore installations, suggesting that the two different types of technology may not have significantly different life cycle GHG emissions.

Published LCAs and carbon and energy audits of wind power have been carried out by a range of stakeholders, including turbine manufacturers (Vestas, 2006a,b), wind farm operators (Vattenfall, 2013; Enel, 2004), and academics (Ardente *et al.*, 2008; Tremeac and Meunier, 2009; Wiedmann *et al.*, 2011; Guezuraga *et al.*, 2012). The majority of the studies considered here are process-based LCAs (Martinez *et al.*, 2009; Tremeac and Meunier, 2009; Vestas, 2006a) and EPDs (Vattenfall, 2013; Enel, 2004) considering a range of different environmental impacts, although the findings of some hybrid studies have also been considered (Wiedmann *et al.*, 2011; Crawford, 2009).

A very comprehensive EPD of wind power was recently published by Vattenfall, one of Europe's largest electricity generators (Vattenfall AB, 2013; Vattenfall, 2013). This estimated the environmental impacts of their entire Nordic wind portfolio, including both onshore and offshore installations in Denmark and Sweden, following the product category rules outlined in UN CPC 171 (The International EPD System, 2013). The analysis was based on a representative selection of farms, using proprietary data on the wind farm installation, operation and decommissioning, coupled with detailed data from turbine manufacturers, to quantify the environmental impacts of the entire life cycles. This study found the portfolio-average carbon footprint to be 14 g CO_2 eq/kWh and the fossil fuel energy consumption to be 169 kJ/kWh, excluding the impacts of transmission and distribution. This carbon footprint estimate is slightly higher than the median suggested by the LCA Harmonization Project, but is well within the expected range. The study appears to apply the recycled content methodology for dealing with recycling credit, and does not include the impacts of land-use change, as these are documented separately.

Many process-based LCAs have been published for wind power generation, and the detailed LCAs carried out by Vestas are often cited as good examples (Elsam, 2004; Vestas, 2006a, 2005, 2006b). As one of the largest wind turbine manufacturers in Europe, Vestas have access to confidential data about the manufacturing and maintenance of their turbines. Their studies have been based upon the installation of these turbines in typical wind farms, both on and offshore, considering the whole life cycle of the turbines and farm, up until the point of connection with the grid.

One study compares onshore and offshore wind farms composed of V90-3.0 MW turbines (Vestas, 2006b). This found that the environmental impacts per unit of output energy were close to being identical for both the installations, within the expected uncertainties, except for resource consumption (Vestas, 2006b). The carbon footprints were found to be 4.6 and 5.2 g CO_2/kWh, and the embodied energy 98 and 102 kJ/kWh, for the onshore and offshore farms respectively. While the impacts of onshore maintenance were lower, the greater energy output of the offshore turbines (due to higher wind speeds) more than offset this. Comparison of the results with that of an earlier study also showed an improvement in the energy payback period and environmental performance of the V90-3.0 MW wind turbines compared to the V80-2.0 MW model.

The carbon footprint estimates from these studies are, however, unusually low for LCAs of wind turbines. This could be attributable to the high assumed capacity factor, or particular methodological choices. One unusual aspect of these analyses is that Vestas supplies a proportion of the electricity to its manufacturing plant from its own wind farm, and this decreases the environmental impacts of electricity consumption during the manufacturing stage. Such an assumption is unusual in an LCA, and must be handled with care, as guidance on carbon footprinting recommends that the lower impacts of renewable electricity sources should only be used if they are not included in national average figures (BSI, 2011). Furthermore, the recycling allocation methods applied in this study are unclear, despite the highly detailed report. Generic materials data is sourced from the GaBi EDIP database, which is likely to include average recycled content data, while end-of-life recycling is modelled using system expansion, which would normally be the closed-loop approximation method. Further clarification is required to confirm that the impacts of recycling have not been double-counted.

Other process-based LCAs produce higher estimates of both the carbon footprint and embodied energy of wind power generation. Ardente *et al.* (2008) carried out a detailed analysis of a real wind farm in Italy and estimated the carbon footprint to be 15 g CO_2 eq/kWh and the embodied energy to be 188 kJ/kWh. Guezuraga *et al.* (2012) compared two 2 MW turbines and found the carbon footprints to be 9.7 and 8.8 g CO_2 eq/kWh, and the embodied energy to be 118 and 116 kJ/kWh, for geared and gearless turbines respectively. Tremeac and Meunier (2009) modelled a 4.5 MW turbine installed in a French wind farm and found the carbon footprint to be 16 g CO_2 eq/kWh, and the embodied energy to be 300 kJ/kWh (the latter value is very high - possibly due to unusual assumptions within the analysis, such as the inclusion of blade replacement in regular maintenance - while the carbon footprint is closer to other estimates because the analysis is based upon a turbine with a concrete tower, rather than one made of steel). These studies are all comprehensive process-based LCAs, based on different technologies and practitioner assumptions, which result in considerable variation in results.

Wiedmann *et al.* (2011) recently carried out a study examining the results of hybrid LCA on wind power generation. This was based upon an inventory of materials, electricity and products for the manufacture, operation and decommissioning of a wind turbine taken from the Ecoinvent database (Ecoinvent, 2010). Using economic input-output (IO) data for the UK, two hybrid methods were applied: IO-based hybrid LCA and integrated hybrid LCA. The results of both methods were found to be very similar, producing carbon footprint estimates of 29.7 and 28.7 g CO_2 eq/kWh, 120 % higher than the estimate produced by the process-based methodology. There is continued debate about which methodology is the most accurate, with reviews suggesting that the process-based method usually underestimates the impacts, while the hybrid method produces overestimates (see Chapter 2).

A selection of results from some well-documented studies is given in Table 3.1. Estimates of the carbon footprint from process-based LCAs were found to range from around 5 to 16 g CO_2 eq/kWh, with the results from EPDs being approximately 15 g CO_2 eq/kWh and hybrid analyses 27 g CO_2 eq/kWh. Similarly the embodied energy estimates were found to vary widely, with the results of process-based LCAs ranging from 105 to 300 kJ/kWh, and the findings of hybrid LCAs and EPDs being 150 to 170 kJ/kWh. The higher results from hybrid LCAs can be expected, due to the use of input-output data to avoid the truncation errors that can be introduced by setting a system boundary; however, one detailed process-based LCA was identified that estimates the carbon footprint and embodied energy to be higher than that estimated by the hybrid studies, at 32 g CO_2 eq/kWh and 493 kJ/kWh respectively (Wagner *et al.*, 2011). This study specifically investigates the first German offshore wind farm located beyond the territorial waters (twelve mile zone), in water depths of 30 m. It is very detailed, and contains highly conservative estimates for annual maintenance and parts replacement, which may have resulted in higher impact estimates than other similar studies.

Device	**Carbon footprint** (g CO_2 eq/kWh)	**Embodied energy** (kJ/kWh)	**Reference**
Harmonised median	11	-	Dolan and Heath (2012)
Nordic turbine (EPD)	14	169	Vattenfall (2013)
Vestas v90 Onshore	4.64	98.2	Vestas (2006b)
Vestas v90 Offshore	5.23	102	Vestas (2006b)
1.8 MW gearless turbine	8.82	116	Guezuraga *et al.* (2012)
2 MW geared turbine	9.73	118	Guezuraga *et al.* (2012)
Italian wind farm	14.8	188	Ardente *et al.* (2008)
4.5 MW turbine	16	300	Tremeac and Meunier (2009)
3.0 MW turbine (Hybrid)	31.6	157	Crawford (2009)
Alpha Ventus wind farm	32	493	Wagner *et al.* (2011)

Table 3.1: A selection of carbon footprint and embodied energy estimates for wind power generation

3.4.2 Marine

Marine energy is an emerging technology that has the potential to supply a significant proportion of future UK electricity demand. Current developments are centred around devices to harness energy from the waves and tides, with several currently in operation, mostly at existing test installations at the European Marine Energy Centre in Orkney and the Wave Hub in South West England (Krohn *et al.*, 2013; EMEC, 2013; Wave Hub, 2013). While the installed capacity at the end of 2012 was only 7 MW, considerable further development is expected, with almost 40 wave and tidal sites licensed between 2010 and 2012, and between 100 and 200 MW of generation expected to be deployed by the end of 2020 (MacLeay *et al.*, 2013; Krohn *et al.*, 2013). Although development in the UK is centred on wave and tidal energy, marine power can also be generated from ocean currents, ocean thermal energy and salinity gradients.

To date, very few LCAs and carbon and energy audits have been carried out on marine energy converters but, as this sector grows, it is becoming increasingly important to identify the carbon footprint and wider environmental impacts of these technologies. Furthermore, with little consensus on the optimum design of marine energy converters, the results of such studies will be able to inform design developments.

The LCA Harmonization Project included marine energy in their review of the GHG emissions of power generation technologies, and published their findings in Chapter 6 of the IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation (IPCC, 2011). This review identified only 5 studies of marine energy converters that passed the quality screens, and none of these examined devices for conversion of tidal and ocean currents, ocean thermal energy or salinity gradients. Of the studies identified in this review, it was found that estimates of the life cycle GHG emissions from wave and tidal energy converters were below 23 g CO_2 eq/kWh, with the median estimate for wave energy being around 8 g CO_2 eq/kWh. This suggests that the carbon footprint of marine energy converters is very low in comparison to fossil-fuelled generators. The review concludes that insufficient studies have been published to corroborate the existing results, or determine whether there are significant differences in carbon footprint between the different types of marine energy converter. The variation in type of energy converter also precludes any attempts at harmonisation.

Wave

Several different types of wave energy converter are currently under development, with no clear design consensus emerging. Existing models can be broadly divided into three categories - oscillating water columns, oscillating body systems and overtopping devices. These may be shore-mounted, floating, or sea-bed mounted, in a range of different water depths.

Oscillating water columns, such as the shore-mounted Limpet on the island of Islay (Figure 3.5), harness the energy from the waves by allowing sea water to rise and fall within a column, which draws air into or out of the chamber at the top via an air turbine, thus generating electricity (IPCC, 2011).

Oscillating body systems include both the Pelamis and the Oyster, shown in Figure 3.6. These extract energy from the oscillation induced by the wave motion on two bodies, and designs vary considerably: the Pelamis, described in IPCC (2011) as “angularly articulating buoyant cylinders”, generates power from the relative motion induced in the tube sections as the wave front passes (PWP, 2011); the Oyster is a hinged flap fixed to the sea bed, with the wave surges inducing horizontal oscillations (Aquamarine Power, 2013); tethered buoys, such as the PowerBuoy (Ocean Power Technologies, 2013), have been developed as uni-directional point absorbers; and fully submerged devices are also being developed to extract energy from the fluctuating hydrodynamic pressure caused by waves (IPCC, 2011). These devices use a variety of different power take-off systems.

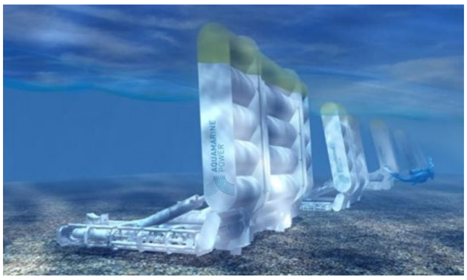
The final category of wave energy converter are overtopping devices, such as the Wave Dragon shown in Figure 3.7 (Wave Dragon, 2013). These capture the wave surges in a reservoir slightly higher than the free water surface, which is then drained through a low-head hydraulic turbine to generate electricity (IPCC, 2011).

Several first-order estimates of the carbon footprint and embodied energy of wave energy converters have been published, but these are based only on the mass of steel within the device (except in one instance where the mass of copper is also included (Banerjee *et al.*, 2006)) and do not, therefore, capture the full impacts (Banerjee *et al.*, 2005; The Carbon Trust, 2006). These produce widely varying estimates of 22 to 40 g CO_2/kWh and 241 to 443 kJ/kWh. Only three published studies have been identified that do consider the whole life cycle of the energy converters and associated infrastructure. All of these are partial life cycle inventories: two only reporting CO_2 emissions and energy consumption (Walker and Howell, 2011; Parker *et al.*, 2007), while the third is more comprehensive and also reports emissions of methane (Soerensen and Naef, 2008).

Despite the three studied devices being very different, the estimates of carbon footprint and embodied energy for the Pelamis and Oyster are similar: 23 g CO_2/kWh and 293 kJ/kWh for the Pelamis (Parker *et al.*, 2007), and 25 g CO_2/kWh and 236 kJ/kWh for the Oyster (Walker and Howell, 2011). These are both oscillating body systems with hydraulic power take-offs, and are largely made of steel. In contrast the Wave Dragon floating overtopping device, which is predominantly concrete, was found to have a carbon footprint of only 13 g CO_2/kWh (Soerensen and Naef, 2008). The variations in these published estimates may be due to the considerable differences in technology, but could also be due to variations in analysis methodology and assumptions.

Figure 3.5: The Limpet oscillating water column (Photo by Peter Church)

(a) Pelamis



(b) Oyster

Figure 3.6: Oscillating body systems

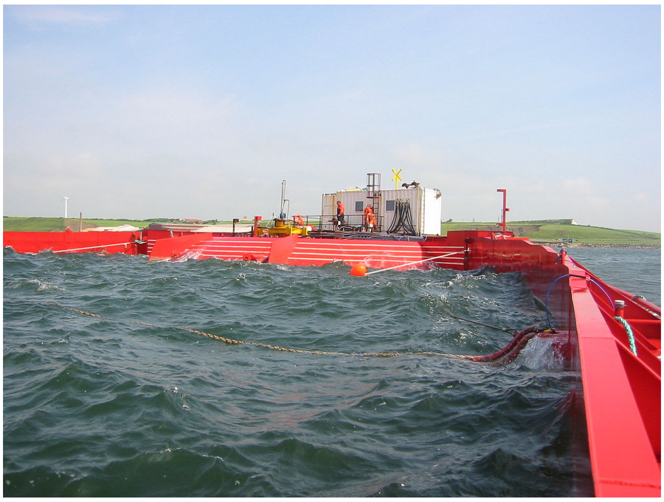


Figure 3.7: The Wave Dragon floating overtopping wave energy converter (Photo by Erik Friis-Madsen at en.wikipedia)

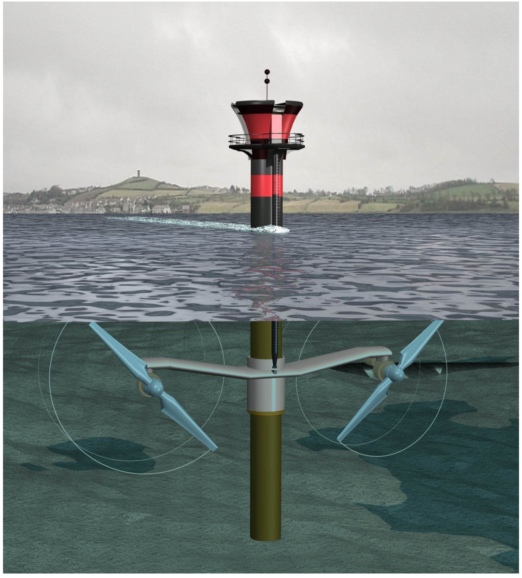
Tidal

Tidal energy converters broadly fall into two categories: tidal stream devices, which extract energy from tidal currents; and tidal range converters, which extract energy from tidal changes in sea level. Existing tidal range power plants, such as that installed at La Rance in France, are based on the tidal barrage design (Figure 3.8): a barrage encloses an estuary that creates a reservoir behind it, and conventional low-head hydropower turbines operate as the tide ebbs and flows. Recent developments in the concept of tidal range power plants have suggested multiple basin designs and the creation of offshore tidal lagoons - the latter may have lower environmental impacts than estuarine installations (IPCC, 2011). In the UK, the Severn Estuary has been identified as an appropriate location for a tidal barrage, but concerns over the cost and environmental impacts have hindered the progress of developing this site (BBC, 2013b,a).

Figure 3.8: La Rance tidal barrage

The majority of early tidal stream devices are based on the design of horizontal-axis wind turbines (see Figure 3.9a); however, these must contend with harsh underwater conditions, reversing flows, and cavitation (where localised areas of low pressure cause gas bubbles to form that subsequently implode, creating shockwaves that can damage the turbine). Alternative designs have been developed, including cross-flow turbines, which are not sensitive to flow direction (such as vertical-axis turbines), and reciprocating devices. Reciprocating devices oscillate in a direction transverse to the tidal flow, and include ‘flutter systems’, such as the hydrofoil shown in Figure 3.9b, and devices based on vortex shedding (IPCC, 2011).

Three partial life cycle inventories have been identified for tidal energy converters: two of these are based on tidal stream turbines, and the third on one of the proposed designs for a tidal barrage across the Severn Estuary. None of these analyses include a broad scope of greenhouse gases (instead concentrating only on carbon dioxide emissions) and only two of them consider embodied energy. In contrast to these, a life cycle assessment of the Severn Barrage


(a) SeaGen turbine

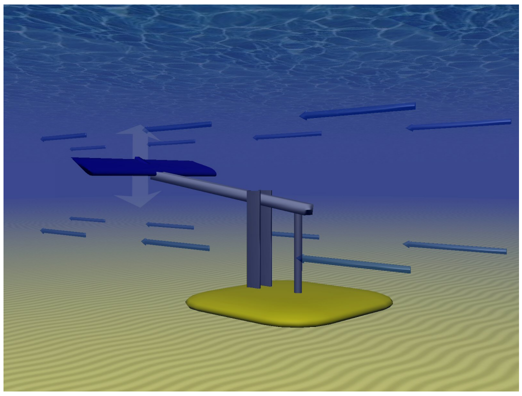

(b) A hydrofoil tidal energy converter (Aquaret, 2013)

Figure 3.9: Tidal current energy converters

has been published by Kelly *et al.* (2012), and presents both the cumulative energy demand and greenhouse gas emissions. All four of these studies produce very different estimates.

Douglas *et al.* (2008) analysed a typical installation of a single SeaGen tidal turbine, and estimated the carbon footprint to be 15 g CO_2/kWh and embodied energy 214 kJ/kWh. In contrast, the analysis published by Rule *et al.* (2009) of a theoretical installation of 200 tidal current turbines installed in New Zealand produced estimates of 1.8 g CO_2/kWh and 42.3 kJ/kWh. Such a wide variation is likely to have been introduced by the different assumptions regarding the manufacture and installation of these devices, which were still under development at the time that the analyses were carried out.

Tidal barrages have very different life cycles in comparison to tidal current turbines, and therefore the results of the analyses by Woollcombe-Adams *et al.* (2009) and Kelly *et al.* (2012) cannot be corroborated by comparison with those by Rule *et al.* (2009) and Douglas *et al.* (2008). The study by Woollcombe-Adams *et al.* (2009) was a detailed analysis of the carbon dioxide emissions of only the materials and manufacturing stages, and estimated this to be 5.7 g CO_2/kWh. At first glance this seems a reasonable estimate, as it is similar to that of large-scale hydropower plants (see Section 3.4.3), which use the same technology; however, Kelly *et al.* (2012) highlights that the impacts of the operational stage of a tidal barrage, which is not considered in the earlier analysis, can be significant. Energy is consumed by auxiliary systems when the barrage isn't generating, and the process of 'flood pumping', used to improve efficiency, is very energy intensive. When these operational impacts are included they dominate the life cycle impacts of the barrage and raise the estimated carbon footprint by a factor of 10 to 56.2 g CO_2/kWh, with the embodied energy calculated to be 973 kJ/kWh. If flood pumping is removed from the design consideration, taking into account the loss of efficiency, the estimates become significantly more favourable at 8.6 g CO_2/kWh and 149 kJ/kWh.

Summary

All but one of the existing published studies identified for marine energy converters are process-based partial life cycle inventories, with only one considering a broad spectrum of greenhouse gas emissions. Similarly to all other studies reviewed in this section, the reports generally do not include details of cut-off criteria, allocation rules and recycling methodology. Their findings are summarised in Table 3.2, and it can be seen that there is considerable variation in the estimated carbon footprints and embodied energy; this may be due to variations in technology, analysis methodology or assumptions. Furthermore, no LCAs, carbon or energy audits were identified for ocean current, ocean thermal energy or salinity gradient converters. There is clearly significant scope for further analyses to be carried out in this area.

Device	**Carbon footprint** (g CO_2 eq/kWh)	**Embodied energy** (kJ/kWh)	**Reference**
Wave energy converters			
Pelamis	23	293	Parker *et al.* (2007)
Oyster	25	236	Walker and Howell (2011)
Wave Dragon	13	-	Soerensen and Naef (2008)
Pelamis	27	411	Chapter 4
Tidal stream converters			
Seagen	15	214	Douglas *et al.* (2008)
Tidal turbines	1.8	42.3	Rule *et al.* (2009)
Severn Barrage			
Partial lifecycle	5.7	-	Woollcombe-Adams *et al.* (2009)
With flood pumping	56.2	973	Kelly *et al.* (2012)
No flood pumping	8.6	149	Kelly *et al.* (2012)

Table 3.2: A selection of carbon footprint and embodied energy estimates for marine energy converters

The study presented in Chapter 4 builds upon that carried out by Parker *et al.* (2007) and is believed to be the first detailed environmental life cycle assessment of a marine energy converter to be published. This analysis is based upon the same manufacturer's data as the earlier study, and applies the same functional unit, system boundaries and waste disposal profiles, enabling an assessment to be made of the effects of practitioner decisions on the results. A broader scope of resource consumption and emissions are included and the results are presented as a wide range of environmental impacts. This study found the carbon footprint to be 27 g CO_2 eq/kWh and the energy intensity 411 kJ/kWh, both higher than the existing estimates for wave energy converters. In order to identify where discrepancies may have arisen, a detailed sensitivity analysis is also presented: specifically this examines the impact of practitioner decisions on cut-off criteria, data quality, recycling allocation and impact assessment method.

3.4.3 Hydro

Hydroelectric power generation is the oldest and most established form of renewable generation. In the UK, it is the third largest source of renewable electricity in terms of both installed capacity and production (MacLeay *et al.*, 2013). Also, the majority of the longest-running operational power stations in the UK are hydroelectric, with the two oldest being a pair of run-of-river plants on the Falls of Clyde, just outside Glasgow, which have been operating since 1926 (Scottish Power, 2012).

Hydroelectric power stations fall into four categories, based on the type of technology: run-of-river, reservoir, pumped storage and in-stream. These range from very small- to very large-scale, depending of the hydrology and topology of the watershed.

Run-of-river hydroelectric plants draw their energy from the natural flow of the river, by diverting a proportion of the flow through a channel or pipeline to the hydraulic turbine and generator (Figure 3.10a). The water is normally diverted by a weir, such as that shown in Figure 3.10b, which has sluice gates that can control the flow of water into the power station and may be able to provide some short-term (hourly or daily) storage capacity. Run-of-river plant have the advantage over similar-sized reservoir power stations of being relatively inexpensive, with generally lower environmental impacts, but are dependent on the variability of the flow of the source river (IPCC, 2011).

(a) Pipeline to power station (Photo by Elliott Simpson)

(b) The weir at Bonnington Linn (Photo by Gordon Brown)

Figure 3.10: Bonnington hydroelectric power station

Reservoir hydroelectric plant, such as that shown in Figure 3.11, avoid the dependence on river flow rates by storing the source water in a lake. This is normally an artificial lake created by building a dam across a river valley, but in some parts of the world, such as Scandinavia, the topology has created natural high-altitude lakes, where the potential energy is exploited by building tunnels under the lake to draw water off to high-pressure generating stations (IPCC, 2011). The construction of artificial lakes can have significant environmental impacts, due to both the inundation of vegetation under the new lake and the change in flow rate of the river downstream of the dam.

Figure 3.11: Revelstoke hydroelectric dam and generating station, BC, Canada (Photo by Kelownian Pilot at en.wikipedia)

The third type of hydroelectric power station in common use around the world is pumped storage. These are constructed as large energy storage facilities that use electricity from the grid at times of low demand to pump water from a low reservoir to a high reservoir, and then release the water from the high reservoir to generate electricity at times of peak demand. While some energy is lost during this process, due to energy conversion inefficiencies, pumped storage remains the most efficient method of storing electrical energy at a large-scale (IPCC, 2011). The largest pumped storage plant in the UK is at Dinorwig in Wales (Figure 3.12a). With a generating capacity of 1728 MW, this power station was built at the site of an old slate mine, with the generating plant concealed within the hillside in a large cavern, and the upper reservoir, pictured in Figure 3.12b, just over the peak of the mountain (MacLeay *et al.*, 2013; First Hydro Company, 2013). Pumped storage hydroelectricity is not strictly speaking, however, a source of renewable energy, instead being an energy storage facility that will be a net consumer of energy over its lifetime.

The newest and least mature type of hydroelectric power generation is based on in-stream technology, which involves the installation of small hydrokinetic turbines to generate electricity from existing weirs, barrages, canals or falls (IPCC, 2011). In the UK there may be considerable potential for these, as existing weirs and mill leats, originally installed for watermills that have fallen into disuse, could be re-used for hydroelectric power generation.

(a) Low reservoir and intake/outlet (Photo by Denis Egan)

(b) Marchlyn Mawr upper reservoir (Photo by Ian Greig)

Figure 3.12: Dinorwig pumped storage station

Life cycle carbon and energy audits of hydropower installations are relatively rare. The LCA Harmonization Project identified only 11 published studies in their review of the GHG emissions of power generation, of which the majority were for reservoir plants. They found that existing estimates of the carbon footprint of hydropower ranged from 1 to 165 g CO_2 eq/kWh, reflecting the variation introduced by differences in climatic conditions, types of land cover before impoundment, and technologies (NREL, 2013d; IPCC, 2011). In particular, the inclusion of GHG emissions from land-use change had a significant effect on the results, as the magnitude of these emissions from the flooding of reservoirs is highly uncertain. The review found that the median estimate of the carbon footprint of reservoir hydropower plants is 7 g CO_2 eq/kWh, with an interquartile range (IQR) of 4 to 40 g CO_2 eq/kWh. The range of estimates for run-of-river plant is much smaller, as only two unique references were identified, giving a median of 7 g CO_2 eq/kWh and an IQR of 4 to 10 g CO_2 eq/kWh. Only one estimate was found for pumped storage: 5.6 g CO_2 eq/kWh (Denholm and Kulcinski, 2004). The review concludes that further LCA studies are required to corroborate existing estimates and increase the breadth of coverage in terms of climatic zones, types of technology, sizes of dam and types of land cover (IPCC, 2011).

One of the most comprehensive recent studies of the impacts of hydropower generation is the EPD published by the operator Vattenfall (2011) on their Nordic plant. This is very similar to that published for wind farms (Vattenfall, 2013), but instead examines the environmental impacts of electricity generated from the Vattenfall portfolio of hydropower installations in Sweden and Finland, including both reservoir and run-of-river plants. The methodology is very robust, as the analysis follows the guidance in the UN CPC 171 product category rules (The International EPD System, 2013), and the carbon footprint and embodied energy were found to be 8.6 g CO_2 eq/kWh and 11.2 kJ/kWh respectively. Most of the greenhouse gas emissions were found to be due to inundation of land, and the materials and construction impacts were also significant in all categories. The operation stage was found to contribute less than 1 % to any impact category.

Another comprehensive study of the life cycle impacts of hydropower was carried out by the Swiss Centre for Life Cycle Inventories (Dones *et al.*, 2007). This analysis was based upon real data from both reservoir and run-of-river plant in Switzerland. In order to develop European average values, modifications were made to represent European averages where possible (such as applying European-average carbon intensities for electricity consumption), and the uncertainty of the estimates was increased. European-average estimates for the carbon footprint and embodied energy for non-alpine reservoir hydropower stations were found to be 4 g CO_2 eq/kWh and 46 kJ/kWh respectively; however, the impacts of inundation may not be truly representative due to limited availability of relevant information. The estimates for European-average run-of-river plant were slightly lower at 3 g CO_2 eq/kWh and 38 kJ/kWh - this is likely to be due to the high impacts of dam construction. Another difficulty that Dones *et al.* (2007) observed in estimating the life cycle impacts of hydropower was that the results are highly dependent upon the assumed lifetime of the plants, but few have yet reached their end-of-life and it is, therefore, difficult to reliably estimate this value. Current estimates range from 40 to 150 years (Oak Ridge National Laboratory, 1994; Dones *et al.*, 2007).

In contrast to the process-based studies published by Vattenfall (2011) and Dones *et al.* (2007), Zhang *et al.* (2007) used a hybrid method to analyse two reservoir hydropower plants in China. This study was very comprehensive, and in one case produced estimates significantly higher than those from the process-based method: a carbon footprint of 44 g CO_2 eq/kWh and embodied energy of 485 kJ/kWh for the 44 MW dam. However, this was based on an existing design with older technology, and when a more modern and larger dam was studied (3600 MW) the estimated carbon footprint was only 6 g CO_2 eq/kWh and embodied energy 74 kJ/kWh. This highlights the significant variation that can be introduced by the different technologies.

A preliminary study has been carried out examining the carbon footprint and embodied energy of a reservoir hydropower plant in Great Britain, considering the life cycle impacts of the 100 MW Glendoe Hydro Scheme (Johnston, 2009). Preliminary results estimated the carbon footprint to be 19 g CO_2/kWh and embodied energy to be 135 kJ/kWh. These are significantly higher than the other estimates that have been identified. The principal contributor to the embodied energy was the concrete used for the dam, and the greatest carbon dioxide emissions arose from the disruption of peat. The latter has not been mentioned in the other studies reviewed, and could account for the greater carbon footprint.

There is clearly scope for further studies of hydropower plant to be carried out, particularly for run-of-river installations, to improve confidence in carbon footprint and embodied energy estimates. Particular issues are the impacts of land inundation and the expected lifetime of the installation; however, in contrast to the studies of marine energy converters and wind turbines, most of the existing published studies of hydropower plant were found to be based on real installations, and could therefore attempt to include land-use change impacts more comprehensively. A summary of some estimates of carbon footprint and embodied energy of hydropower is given in Table 3.3.

Device	**Carbon footprint** (g CO_2 eq/kWh)	**Embodied energy** (kJ/kWh)	**Reference**
Average Nordic	8.6	11.2	Vattenfall (2011)
European non-alpine reservoir	4.1	16.4	Dones *et al.* (2007)
European run-of-river	3.1	37.5	Dones *et al.* (2007)
44 MW	44	485	Zhang *et al.* (2007)
3600 MW	6	74	Zhang *et al.* (2007)
Scottish reservoir	19	135	Johnston (2009)

Table 3.3: A selection of carbon footprint and embodied energy estimates for hydropower plants

3.4.4 Solar

Despite the latitude and prevailing weather systems in the UK limiting the scope for solar power generation, solar photovoltaics (PVs) provide one of the most efficient options for building-mounted renewable electricity generation, particularly in urban locations. In April 2010, the UK government introduced the ‘Feed-In Tariff’ subsidies to encourage the installation of small-scale domestic renewable energy generators, and this resulted in a sharp increase in the installation of solar photovoltaics (PV), with the installed capacity at the end of 2012 equalling that of hydropower at 1.7 GW (MacLeay *et al.*, 2013; Energy Saving Trust, 2013). It is significant to note, however, that the output of this installed capacity was reported to be 1.2 TWh, only 22 % of the energy production of the hydroelectric plant. It is likely that improvements in technology will further increase the installed capacity of solar PV, although it is expected that this will continue to be restricted to small-scale domestic installations rather than large transmission-connected plant.

There are two principal technologies for generating electricity from solar energy: photovoltaic (PV) and concentrating solar power (CSP). PV technology, such as that shown in Figure 3.13, is the only type currently installed in the UK, and exploits a phenomenon where the energy from light can induce an electric current in a doped semiconductor: impurities are introduced into the semiconductor crystal to create a p-type layer that has missing electrons (known as holes), and an n-type layer containing free electrons; when the two sides are connected to a load, the energy from the incident photon allows the free electron to flow across the p-n junction inducing an electric current. Existing PV technologies include wafer-based crystalline silicon (c-Si) cells, and thin-film cells based on copper indium gallium diselenide ($CuInGaSe_2$ - CIGS), cadmium telluride (CdTe), amorphous silicon or microcrystalline silicon. Many new PV technologies, based on cheaper materials and likely to have different operating characteristics, are currently under development (IPCC, 2011).

Figure 3.13: Solar photovoltaics installed at the King's Buildings, University of Edinburgh

Concentrating solar power technologies are based on the same concept as existing coal, gas and nuclear power plants - where the solar irradiance is concentrated to heat a fluid that then passes through a turbine to generate electricity. They depend on direct-beam irradiation, and are therefore best suited to near-equatorial cloud-free regions and deserts, such as the installation shown in Figure 3.14 (IPCC, 2011). There is no scope for CSP installations in the UK, and therefore a review of the life cycle carbon footprint and embodied energy is not included here. More information about the GHG emissions of CSP can be found in Burkhardt *et al.* (2012).

Figure 3.14: Solar Energy Generating Systems' concentrating solar troughs in the Mojave Desert, California (Photo by Alan Radecki)

The life cycle environmental impacts of solar power have been extensively studied, and there are hundreds of published estimates of GHG emissions of both domestic and utility-scale solar installations. These vary widely, however, with inconsistencies attributable to variations in technology (system design assumptions, models based on reality or theoretical concepts, and technology improvements over time) as well as LCA methods and assumptions (NREL, 2013c). The LCA Harmonization Project systematically reviewed and harmonised these existing estimates, screening them for quality, transparency and relevance, before imposing standardised values for several key performance criteria. The results of the screening process highlights the extreme variations in quality of existing carbon footprint estimates, as around 95 % did not meet the basic screening requirements.

In order to further review and harmonise the GHG emissions of solar PV, the published studies were divided according to the type of technology: crystalline silicon PV (Hsu *et al.*, 2012) and thin-film PV (Kim *et al.*, 2012). Crystalline silicon (c-Si) PV cells can be either mono- or multi-crystalline; the former is more efficient than the latter, but has higher manufacturing costs. Estimates of the GHG emissions associated with electricity from both types of c-Si PV vary widely, with some estimates as high as 200 g CO_2 eq/kWh (Hsu *et al.*, 2012). The initial screening process produced a selection of 13 published studies containing 41 estimates of the GHG emissions of electricity from c-Si PV cells, with a median value of 57 g CO_2 eq/kWh and an interquartile range of 44 to 73 g CO_2 eq/kWh. Harmonisation, carried out at the level of the 'preliminary stage' discussed in Section 3.3, concentrated on harmonising the scenario-based assumptions affecting the estimated lifetime generation of the installation, such as solar irradiation, system lifetime, module efficiency and performance ratio. No harmonisation of the lifetime GHG emissions was carried out, with the exception of one study, due to insufficient reporting regarding the practitioner assumptions in the LCA process. Considerable variations remained, therefore, in the harmonised results. Furthermore, a completed PV system normally includes batteries, but none of the harmonised studies considered the impacts of this battery storage (Hsu *et al.*, 2012). The harmonisation process did, however, decrease the interquartile range to 39 to 49 g CO_2 eq/kWh with a median value of 45 g CO_2 eq/kWh (based on a solar irradiation of 1700 kWh/m^2 - corresponding to the average irradiation in southern Europe; if a solar irradiation of 2400 kWh/m^2, typical to southwest USA, is applied, the median becomes 32 g CO_2 eq/kWh). It was found that the adjustment of the solar irradiation estimate to a consistent value had the greatest impact. Differences between mono- and multi-crystalline cells, or ground-mounted versus roof-mounted installations, were not found to be significant (NREL, 2013b). The review concluded that, while this harmonisation reduced the variability of the existing published carbon footprint estimates, the literature doesn't cover all possible installation scenarios or represent the actual distribution of c-Si PV cell manufacture, so may not be truly representative.

A similar harmonisation process was carried out for thin-film PV installations (Kim *et al.*, 2012). This concentrated on amorphous silicon (a-Si), cadmium telluride (CdTe) and copper indium gallium diselenide (CIGS) technologies. The initial screening process eliminated over 100 published studies from the harmonisation process, with only five studies meeting the rigorous criteria for completeness, validity, and relevance. Particular emphasis was placed on the relevance of the study to current thin-film commercially-available technologies, and therefore the screening process considered whether the product is manufactured commercially, the production-line is still in existence, and the data for the manufacturing scenario is unique to the published study (to avoid considering the same data twice). These results were then harmonised by aligning the efficiency, irradiation, performance ratio, balance of system and lifetime to consistent values.

With so few initial estimates, a single harmonised result was provided for each technology, and is summarised in Table 3.4. Note that these values assume a solar irradiation of 2400 kWh/m^2, typical to southwest USA, and much higher than that commonly applied in many existing studies (Hsu *et al.*, 2012). However, all of these estimates suggest that the carbon footprint of thin-film technologies is similar to, or slightly lower than, that of crystalline silicon, although many further studies are required to corroborate the results.

Technology	Ground-mounted	Roof-mounted
a-Si	20	21
CdTe	14	14
CIGS	26	27

Table 3.4: Harmonised GHG emissions of thin-film PV in g CO_2 eq/kWh (Kim *et al.*, 2012)

Most existing studies of solar photovoltaics are based on real or proposed installations. While the most significant GHG emissions and energy consumption arise during the manufacture of the PV modules themselves, the lifetime carbon footprints and embodied energy estimates are very sensitive to assumptions around solar irradiation that are location specific. Therefore, when examining results that have not been harmonised, there is a considerable range. A selection of published carbon footprint and embodied energy estimates of silicon photovoltaics is shown in Table 3.5. It can be seen that these are generally higher than for other renewable technologies.

Device	Carbon footprint (g CO_2 eq/kWh)	Embodied energy (kJ/kWh)	Reference
Harmonised median	45	-	Hsu *et al.* (2012)
Polycrystalline, Catalonia	47.3	814	Sumper *et al.* (2011)
Monocrystalline, Hong Kong	176	1316	Lu and Yang (2010)
Mix, Michigan	48	875	Pacca *et al.* (2007)
Polycrystalline, Germany	104	1500	Pehnt (2006)

Table 3.5: A selection of carbon footprint and embodied energy estimates for silicon photovoltaics

3.4.5 Bio-power

Bio-power already supplies a significant proportion of UK electricity, and met 4 % of total demand in 2012. It includes energy from landfill gas, sewage sludge digestion, waste incineration, anaerobic digestion (AD), animal biomass (non-AD) and plant biomass (such as straw and short-rotation coppice energy crops). The existing installed capacity is 3.2 GW, of which plant biomass makes up 1.2 GW (MacLeay *et al.*, 2013). Much of this is embedded generation, although a new grid-connected biomass-fuelled power station was commissioned at Steven's Croft, in Scotland, in 2007 (E.ON, 2013b; MacLeay *et al.*, 2013), and two coal-fired power stations have been converted to run on plant biomass until their scheduled closure due to the Large Combustion Plant Directive (E.ON, 2013a; RWE npower, 2013). The largest and newest coal-fired power station in the UK, Drax, is also in the process of converting three of its six generators to operate on plant biomass (Drax, 2013).

Hundreds of life cycle assessments have been carried out to examine the environmental impacts of bio-power generation. The LCA Harmonization Project reviewed and analysed a number of these, with a focus on GHG emissions, and found that published estimates (excluding the net changes from land-use and land management impacts) mostly ranged from 16 to 74 g CO_2 eq/kWh, with some as high as 360 g CO_2 eq/kWh, and some below zero (due to avoided emissions or carbon sequestration) (IPCC, 2011). Such a wide variation in carbon footprint estimates was found to be attributable to differences in selected feedstock or technology, as well as differences in analysis methodology, agricultural practice, technology performance and the maturity of the studied development. Several key processes that particularly affected the calculated carbon footprint were estimates of the GHG emissions of biomass production (particularly estimates of land-use change and nitrous oxide emissions), the methodology for allocating impacts of co-products (particularly if the fuel is a waste product), and assumptions about the conversion of the biomass to a usable fuel (IPCC, 2011).

A significant challenge remains in analysing the carbon footprint of bio-power generation: the system boundary can include the absorption of carbon dioxide from the atmosphere and the emissions avoided by not disposing of biomass waste to landfill. It is common to assume that all greenhouse gas emissions from fuel combustion are offset by the GHGs they store during growth (Zhang *et al.*, 2009), despite recommendations in the published guidance for carbon footprinting that recommends that these are reported separately (WRI and WBCSD, 2011a). There seems to be relatively little consensus on how to truly deal with the complex carbon cycle issues of bio-power generation. The LCA Harmonization project identified that further studies are required to examine the impacts of supply chains and land-use change, and also to corroborate existing published estimates, but a harmonisation study on bio-power has yet to be published (NREL, 2013a).

A selection of published estimates of carbon footprint and embodied energy for bio-power is given in Table 3.6 - only values for woody crops are shown, as this thesis concentrates on transmission-connected generation, and currently woody crops are the chosen fuel for all transmission-connected bio-power plants in the UK. While these values vary considerably, they remain considerably lower than those for conventional generation, as illustrated in Figure 3.1.

Device	**Carbon footprint** (g CO_2 eq/kWh)	**Embodied energy** (kJ/kWh)	**Reference**
Forest wood	45	280	Pehnt (2006)
Short rotation forestry	86	460	Pehnt (2006)
Waste wood	37	360	Pehnt (2006)
Wood (min)	36	360	Hennig and Gawor (2012)
Wood (max)	88	540	Hennig and Gawor (2012)
NREL gasifier	39	271	Heller *et al.* (2004)
EPRI gasifier	40	279	Heller *et al.* (2004)
EPRI direct-fired	52	364	Heller *et al.* (2004)

Table 3.6: A selection of carbon footprint and embodied energy estimates for Bio-power fuelled by woody crops

3.5 Conventional Generation Technologies

Despite continued developments of renewable energy technologies, 87 % of electricity supplied to the UK grid in 2012 came from conventional thermal generation (MacLeay *et al.*, 2013). This mostly consists of coal, gas and nuclear power stations, with only 0.8 % of total energy production supplied by oil-fired generators. As all oil-fired power stations are due to be decommissioned by the end of 2015 (National Grid plc, 2011; MacLeay *et al.*, 2013), the life cycle carbon footprints and embodied energy of such generators are not considered in this section.

3.5.1 Coal

Coal-fired power generation currently supplies a large proportion of UK electricity (39 % in 2012), with an installed capacity of approximately 25 GW, 28 % of all generation in the UK (MacLeay *et al.*, 2013). All of the existing plants are at least 20 years old, and use subcritical pulverised coal technology, with the newest and most highly efficient coal-fired power station, Drax, completed in 1986. Up to 8.5 GW of the existing plant is due to be decommissioned by the end of 2015 (some has already closed) due to restrictions imposed by the Large Combustion Plant Directive, a European directive aimed at reducing the emissions of sulphur dioxide, nitrogen oxides and dust from such plant (National Grid plc, 2011). The UK government is also moving away from coal-fired power as an option for future generation, and has recently announced that they will no longer fund construction of coal-fired power stations in the developing world (DECC, 2013). Although no new coal-fired power plants are currently planned, it is likely that any new plant will be significantly more efficient than existing power stations, and will include some level of Carbon Capture and Storage (CCS). As this thesis concentrates on technologies currently connected to the National Grid, this section does not examine the carbon footprint and embodied energy of coal generation with CCS, although many published studies do exist; for example Odeh and Cockerill (2008).

Many published studies examine the life cycle environmental impacts of coal-fired generation, both for comparison with other energy sources and to enable an examination of potential reductions that might be provided by newer technologies. The LCA Harmonization Project carried out a systematic review of existing publications in this area, particularly with reference to GHG emissions, and found 53 references (of 270 identified) that met their basic quality screening criteria (Whitaker *et al.*, 2012). These references yielded 164 estimates of the GHG emissions of coal-fired generation, ranging from 675 to 1689 g CO_2 eq/kWh; this wide range makes it difficult to determine a reasonable estimate of the actual life cycle carbon footprint of electricity from coal.

Whitaker *et al.* (2012) found that these variations could be attributed to several assumptions regarding the scenario and system boundary. In particular, studies varied in the type of technology, technology vintage, location, coal quality, the inclusion of plant construction and decommissioning impacts, and the consideration of methane emissions from coal mines. In order to attempt to reduce the variation in carbon footprint estimates for coal-fired generation, harmonisation was carried out at the level of the ‘preliminary stage’ outlined in Section 3.3. This involved adjusting the published estimates, where possible, to use consistent characterisation factors for the global warming potential of GHG emissions, to include coal mine methane emissions, and to exclude transmission and distribution processes that were considered to be outside the system boundary for electricity generation. Upstream and downstream processes associated with the construction and decommissioning of the power station, waste disposal and coal mine land rehabilitation, were considered to have a negligible impact; the inclusion of these were, therefore, not required for a study to pass the literature screening and were not harmonised. The next step in the harmonisation process was to adjust key operational input parameters, including: thermal efficiency, coal carbon content, coal lower heating value, and combustion carbon dioxide emission factor (CEF).

It was found that the choice of combustion CEF had the greatest impact on the results, and the technology-specific harmonisation process significantly reduced the range of carbon footprint estimates to 729 to 1366 g CO_2 eq/kWh with an interquartile range (IQR) of 110 g CO_2 eq/kWh and a median of 980 g CO_2 eq/kWh (Whitaker *et al.*, 2012, 2013). The results were also reported according to type of technology, with the median estimated carbon footprint for subcritical pulverised coal plant (the technology currently installed in all coal-fired power stations in the UK) found to be 989 g CO_2 eq/kWh, with a range of 879 to 1274 g CO_2 eq/kWh and an IQR of 86 g CO_2 eq/kWh. The lowest estimates were calculated for the latest supercritical pulverised coal technology, with a range of 729 to 1009 g CO_2 eq/kWh, an IQR of 91 g CO_2 eq/kWh and a median of 768 g CO_2 eq/kWh (NREL, 2013d).

The report concluded that the small distribution of the harmonised results across the different combustion technologies implies that first-order estimates of carbon footprint of coal-fired generation could be based on simply on knowledge of the technology type, coal mine emissions, thermal efficiency, and CEF alone (Whitaker *et al.*, 2012).

A summary of published studies is provided in Table 3.7. The embodied energy estimates are particularly high because they include the embodied primary energy in the raw coal.

Location	**Carbon footprint** (g CO_2 eq/kWh)	**Embodied energy** (kJ/kWh)	**Reference**
Harmonised median	989	-	Whitaker *et al.* (2012)
UK (hybrid analysis)	984	8400	Odeh and Cockerill (2008)
Brazil	1300	-	Restrepo *et al.* (2012)
Netherlands	1092	-	Koornneef *et al.* (2008)
Austria	983	11678	Dones *et al.* (2007)
Belgium	1082	12239	Dones *et al.* (2007)
Spain	1102	12486	Dones *et al.* (2007)
France	1074	12430	Dones *et al.* (2007)
Italy	1031	11718	Dones *et al.* (2007)
Netherlands	1084	12428	Dones *et al.* (2007)
Portugal	988	11244	Dones *et al.* (2007)
Germany	1094	12777	Dones *et al.* (2007)

Table 3.7: A selection of carbon footprint and embodied energy estimates for coal-fired generation

3.5.2 Gas

Gas-fired power stations include both open-cycle and combined-cycle gas turbines (OCGT and CCGT). The former has a lower electrical efficiency, but quick response times, so is normally used to meet peak-load demand, while CCGT plants have higher efficiencies (normally over 50 %) and play a significant role in supplying electricity in the UK (UKERC, 2013). At the end of 2012, 40 % of the total installed generating capacity were CCGT power stations, which supplied 28 % of the total energy generated in that year (this is slightly lower than previous years due to fluctuations in the relative prices of coal and gas) (MacLeay *et al.*, 2013). With the carbon emissions of gas-fired generation about half that of coal, the construction of new CCGT plant is expected to continue in order to meet rising demand and replace ageing generators scheduled for decommissioning.

The recent growth in hydraulic fracturing to access shale gas reserves in the USA has led to growing interest in the environmental impacts and GHG emissions of this process, and has subsequently delayed the publication of the findings of the LCA Harmonization Project for gas-fired power generation, according to G. A. Heath (personal communication). However, preliminary findings of the study have been communicated via the project wiki (NREL, 2013d). The initial screening process yielded 62 estimates of the GHG emissions from gas-fired generation, from 38 published studies, with a range of 307 to 988 g CO_2 eq/kWh and a median of 477 g CO_2 eq/kWh. Around two-thirds of these estimates were for natural gas-fired CCGT, the most common gas-fired generation in the UK, yielding a median of 449 g CO_2 eq/kWh and an interquartile range of 76 g CO_2 eq/kWh. In contrast to this, the carbon footprint estimates found by the 5 studies examining natural gas-fired OCGT were higher, with a median of 588 g CO_2 eq/kWh and an IQR of 543 to 692 g CO_2 eq/kWh. The publication of the harmonised results, and details of the harmonisation process, is expected to further refine these estimates.

A summary of published carbon footprint and embodied energy estimates is shown in Table 3.8. These estimates may be optimistic, as they are based on a high capacity factor for gas-fired generation, which may not be the case in a future network with a high proportion of variable-output renewables.

3.5.3 Nuclear

The UK led the worldwide development of harnessing nuclear fission for electricity generation, with the first full-scale nuclear power station opening in 1956 at Calder Hall, in Cumbria. This power station was in operation until 2003, and nuclear power continues to provide a significant proportion of UK electricity, making up 11 % of the total installed capacity and contributing 19 % to total generation in 2012 (MacLeay *et al.*, 2013). Government policy currently supports nuclear power as a significant producer of low-carbon electricity (DECC and DfT, 2013), and in October 2013 a commercial agreement was reached for the first new nuclear power station to be built since 1995, at Hinkley Point in Somerset (BBC, 2013c).

Location	Carbon footprint (g CO_2 eq/kWh)	Embodied energy (kJ/kWh)	Reference
Singapore	474	7790	Kannan *et al.* (2005)
Singapore	493	8100	Kannan *et al.* (2007)
Japan	518.8	-	Hondo (2005)
Austria	807	14059	Dones *et al.* (2007)
Belgium	527	10019	Dones *et al.* (2007)
Spain	513	9256	Dones *et al.* (2007)
France	488	8639	Dones *et al.* (2007)
Italy	658	11656	Dones *et al.* (2007)
Netherlands	586	11816	Dones *et al.* (2007)
Germany	563	10637	Dones *et al.* (2007)

Table 3.8: A selection of carbon footprint and embodied energy estimates for CCGT power stations

Figure 3.15: Torness nuclear power station, near Edinburgh

Most of the existing power stations installed in the UK are based on advanced gas-cooled reactor technology - a British technology that uses graphite as the neutron moderator and carbon dioxide as the coolant. More recent nuclear power stations, however, including one currently operational in the UK (Sizewell B), are pressurised water reactors that use liquid water as both the coolant and moderator. It is likely that all new nuclear plants will be pressurised water reactors, and the current design for Hinkley Point C is based on the UK EPR nuclear reactor, which is a pressurised water reactor design developed by Areva and EDF (EDF Energy, 2013; Areva, 2013).

The LCA Harmonization Project carried out an extensive review and analysis of the existing published estimates of GHG emissions of nuclear power generation. This study was focussed on light water reactors - including both boiling water reactors and pressurised water reactors (Warner and Heath, 2012a). Advanced gas cooled reactors, of which few remain in operation around the world, have been much less widely studied, with only 3 analyses passing the literature screen - these published carbon footprint estimates ranged from 5 to 28 g CO_2 eq/kWh, and no harmonisation was carried out (Warner and Heath, 2012b).

In contrast, 274 studies were identified that examined the GHG emissions of light water reactors, and 27 of these passed the screening process for quality, completeness and relevance. These published estimates had a median value of 13 g CO_2 eq/kWh, but also had a very wide range of 220 g CO_2 eq/kWh (Warner and Heath, 2012a). The differences in methodology and system specification that could introduce these variations are described in greater detail in Warner and Heath (2012b). Specifically, differences in ore quality, extraction and enrichment processes, fuel rod fabrication processes, and the reprocessing and disposal of the spent fuel can all result in variations in estimated GHG emissions. These variations may be significant, as the greatest environmental impacts of nuclear power generation are attributable to the extraction and preparation of the fuel for the operation of the power station (Lenzen, 2008).

The first step in the harmonisation process was to adjust all estimates to reflect consistent gross system boundaries. In order to pass the literature screening, the study system boundary had to include the emissions associated with uranium mining, processing and fuel rod fabrication. During harmonisation, missing data was added for the materials, manufacture and decommissioning of the power station itself, as well as nuclear waste storage.

The next step was to adjust several key system parameters to consistent values. This included adjusting the global warming potentials of the reported GHGs (where possible), and the capacity factor, operational lifetime and thermal efficiency of the plant. The harmonisation process reduced the range of estimates to 110 g CO_2 eq/kWh, with an interquartile range of only 17 g CO_2 eq/kWh and a median of 12 g CO_2 eq/kWh (Warner and Heath, 2012a).

In this study, harmonisation was carried out at the level of the 'preliminary stage' outlined in Section 3.3. This precluded full harmonisation of more complex factors that have been previously identified as contributing to the variability in life cycle impact estimates (Warner and Heath, 2012a) so, in order to examine these, the published results were categorised according to LCA methodology, uranium enrichment method, and source energy mix, and were qualitatively compared. The influence of uranium ore grade was also investigated. This harmonisation found that the median carbon footprint estimate from process LCA was about one-third of that from hybrid methods, and had a much smaller variability in published estimates. The primary source energy mix (used in the mining and enrichment processes) and choice of enrichment method also appeared to have an impact on the variability. No trend was identified between the GHG emissions estimate and ore grade, although this may have been due to insufficient reporting of ore grade in the published studies (Warner and Heath, 2012a). There is scope for further work to examine the influence of ore grade on carbon footprint estimates, particularly as the availability of high quality ore is likely to diminish in the future, as well as further harmonising estimates for consistent values of primary energy mix, enrichment method and LCA method.

One particular limitation with selection of LCA method for assessment of the environmental impacts of nuclear power generation, is that economic input-output analyses may provide significant overestimates (Warner and Heath, 2012b). A high proportion of the costs associated with the construction, maintenance and decommissioning of the power station are due to administrative and safety tasks rather than the components themselves. Economic input-output data based on national production statistics is unlikely to include an allowance for these tasks, and therefore cost-based analyses may overestimate the quantity of materials required, and the corresponding environmental impacts.

Many of the published carbon footprint and embodied energy estimates for nuclear reactors are environmental product declarations (EPDs). These are highly detailed, based on existing power stations, and are often carried out by the operator. A summary of results from some EPDs is given Table 3.9. It can be seen that these are the lowest for any type of conventional thermal generation.

Power Station	**Carbon footprint** (g CO_2 eq/kWh)	**Embodied energy** (kJ/kWh)	**Reference**
Torness, UK	7.35	-	AEA Energy and Environment (2009)
Beznau, Switzerland	3.54	54	Axpo (2011)
Ringhals, Sweden	5.6	64.8	Vattenfall (2010)
Forsmark, Sweden	3.7	39.6	Vattenfall (2007)

Table 3.9: A selection of carbon footprint and embodied energy estimates for nuclear power stations

Chapter 4

The Limitations of Carbon and Energy Footprinting for Power Generation Technologies - A full LCA of a wave energy converter

4.1 Introduction

Despite the growing number of studies examining the carbon footprint and embodied energy of power generation, the review in Chapter 3 highlights that there is still considerable variation in the results and some significant gaps remain. The analysis presented in this chapter has several aims: to contribute to the body of published studies of marine power generation by carrying out a full life cycle assessment (LCA) of the first-generation Pelamis wave energy converter (WEC); to examine any potential trade-offs or co-benefits by setting the carbon footprint and embodied energy in the context of a broader range of environmental impacts; to identify where variations may be introduced into carbon footprint and embodied energy estimates; and to quantify the impacts of a range practitioner assumptions and methodological choices on the resulting values, and highlight those with the most significant effect.

As discussed in Chapter 3, very few carbon footprinting studies have been carried out in the marine energy sector to date. This makes it difficult to draw definitive conclusions about wave and tidal energy from the existing literature, as no clear consensus or trend in the results can be observed. Furthermore, the focus on GHGs and embodied energy in existing studies is in conflict with the principle of comprehensiveness of LCA, and potential trade-offs or co-benefits between environmental impacts might be overlooked (WRI and WBCSD, 2011a). At the time of writing, no full LCA of a wave energy converter has been published, so the results of this analysis will provide data on the broader environmental impacts to inform future developments in this sector.

Section 2.5 identified that the methodology for calculating carbon footprints and embodied energy, based on Life Cycle Assessment (LCA), contains significant scope for variations in practitioner assumptions and choices, which affects the comparability and reliability of results (ISO, 2006a,b). Although there is existing guidance that can mitigate some of these problems, there is still considerable scope for variation. The comprehensive sensitivity analysis presented in this chapter allows the effects of many of these choices and assumptions to be examined. Furthermore, the Pelamis WEC was chosen due to the availability of the original data and calculations used in an earlier carbon and energy audit of the same device by Parker *et al.* (2007), allowing the results of the two studies to be compared.

4.1.1 The Pelamis wave energy converter

Figure 4.1: The Pelamis P1 wave energy converter

Marine energy is likely to form a significant part of the future energy mix in the UK and the Pelamis is emerging as one of the most promising devices in this sector (Figure 4.1). Developed by Pelamis Wave Power Ltd (PWP), it is a semi-submerged snake-like offshore device. The first-generation P1 was successfully installed at the world's first commercial wave farm at Aguaçadoura, off the coast of Portugal, in 2008. The experience gained here was fed directly into the development of the second-generation P2, currently on test at the European Marine Energy Centre (EMEC). Several large projects are in the development stages, with lease agreements having been agreed for two farms off the coast of Scotland comprising around 70 devices (PWP, 2011).

The Pelamis P1 is 120 m long, 3.5 m in diameter and rated at 750 kW. It has four cylindrical sections linked by three power conversion modules (PCMs) at the hinged joints. The compliant moorings allow the Pelamis to face into the oncoming waves, and the joints flex vertically and horizontally as the wave front passes (Figure 4.2). This motion is resisted by hydraulic rams that pump high-pressure oil into banks of accumulators, and these are drained at a constant rate through hydraulic motors to drive the induction generators. The resistance of the rams can be

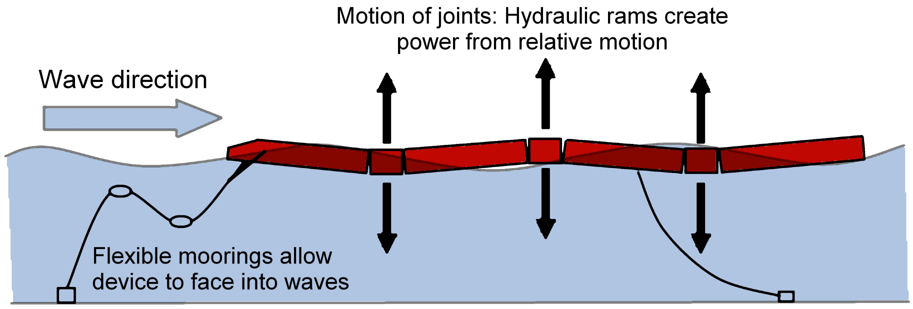


Figure 4.2: Side view of the Pelamis (Parker *et al.*, 2007)

tuned to provide a resonant response in small sea states to maximise power capture, and can also assist in protecting the device from potentially damaging storm waves. Further information on the Pelamis can be found in Henderson (2006).

4.1.2 Carbon and energy audit

Prior to commencing work on the full LCA, a preliminary carbon and energy audit was carried out. This was a partial life cycle inventory (LCI) following the same methodology as that applied by Parker *et al.* (2007). The principal differences were that embodied carbon and energy data for materials was taken from an updated version of the Inventory of Carbon and Energy (Hammond and Jones, 2008a), primary multipliers were applied to take into account the full embodied energy and carbon of fuel and electricity consumption (Jones and McManus, 2008; Mortimer *et al.*, 2003), and the 50:50 recycling allocation method was applied.

The study found that the total embodied energy of the Pelamis was 28000 GJ and the total embodied carbon 1800 tonnes. This corresponds to an energy intensity of 470 kJ/kWh and a carbon intensity of 30 g CO_2/kWh, respectively 60 % and 31 % higher than the values of 293 kJ/kWh and 23 g CO_2/kWh found by the earlier study (Parker *et al.*, 2007).

The discrepancy between the two analyses was investigated and it was found that the most significant variation was introduced by the selection of the allocation method for recycling credit. Parker *et al.* had considered only the credit for recycling at the end of life (closed loop approximation) and the new analysis applied the 50:50 method. (See Section 2.4.2 for further details on allocating recycling credit in LCA.) By applying the same recycling allocation method, the calculated carbon and energy intensities were reduced to 378 kJ/kWh and 22 g CO_2/kWh, 29 % higher and 5 % lower than Parker *et al.*.

The remaining discrepancies were found to be due to different raw materials inventory data, variations in assumptions about transport distances, and the inclusion of primary multipliers for energy sources. In particular it was found that the applied data for the embodied energy of steel was significantly different between the two studies, but the embodied CO_2 was very similar.

The inclusion of primary multipliers, which are correction factors to account for the indirect impacts of fuel extraction and feedstock energy, also had a greater impact on the calculated energy intensity than CO_2. The effects of cumulative changes on the results, from recycling allocation method to the inclusion of primary energy multipliers, is illustrated in Figure 4.3.

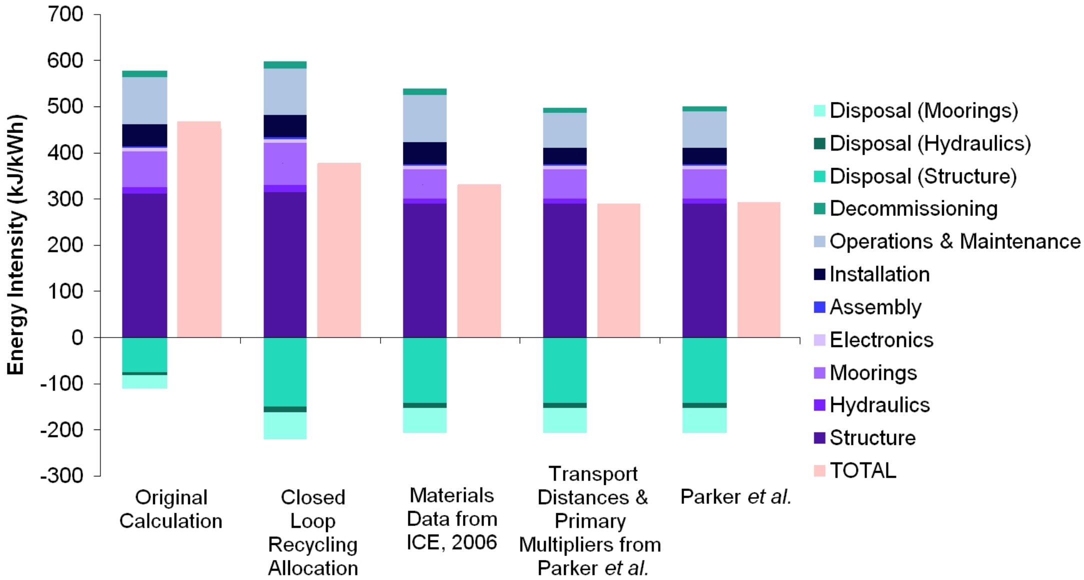


Figure 4.3: Cumulative effect of practitioner assumptions on energy intensity (from left to right)

This preliminary analysis highlighted that the application of different practitioner assumptions can introduce considerable uncertainty, even within the ISO LCA framework. Specific practitioner decisions were identified that most significantly affected the results: the quality of raw data and decisions about data sources were found to be important, as observed in the literature (see Section 2.5 and Schreiber *et al.* (2012)); the inclusion of primary energy multipliers, a function of the system boundary set by the practitioner, was also significant, due to the considerable fuel consumption; however, the greatest impact on the results was made by the choice of recycling allocation method.

4.1.3 Goal and scope of the new LCA

The goal of the analysis presented in this chapter was to carry out a detailed LCA of the first-generation Pelamis, examining a wide range of environmental impacts and highlighting the practitioner decisions that significantly influence the results. An inventory of all environmentally significant resource consumption and pollutant emissions was created for every stage of the device life cycle, and these were characterised according to their impact potentials. This comprehensive analysis contributes to the wider body of research on the environmental impacts of power generation and may also inform future design developments.

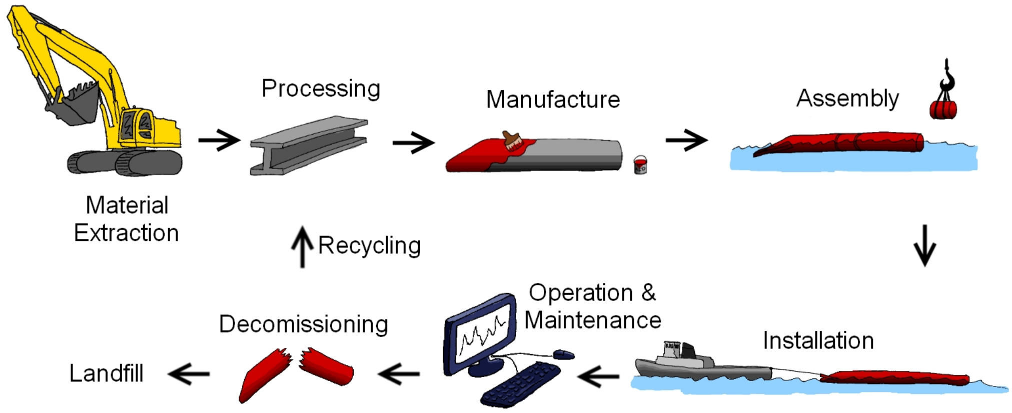


Figure 4.4: Pelamis life cycle

The study system boundary encompasses the full cradle-to-grave life cycle of the Pelamis P1 WEC, including the device, its moorings and sub-sea connecting cable. All downstream electrical components were excluded. Every stage of manufacture, operation and decommissioning was examined, from extraction of raw materials to disposal at the end-of-life, as illustrated in Figure 4.4. The functional unit is one kilowatt-hour of output electrical power (1 kWh), with a calculation reference flow of 1 Pelamis.

The analysis was carried out with SimaPro (version 7.2 PhD), which is leading Life Cycle Assessment software. Life cycle inventory data was mostly sourced from the Ecoinvent database published by the Swiss Centre for Life Cycle Inventories: one of the most comprehensive sources of cradle-to-gate resource and emissions data for materials, transport and other processes in Europe (Ecoinvent, 2010). European average data was applied throughout this study, except where otherwise stated; data not available within Ecoinvent was sourced from alternative datasets or literature, as detailed in later sections. The life cycle impact assessment (LCIA) was carried out with the EDIP 2003 impact assessment method, which includes a broad range of impact categories, and was developed for use with Ecoinvent data - this should minimise inaccuracies caused by mismatches at the final analysis stage.

In order to facilitate the investigation into the effect of methodological choices and LCI data selection, all fundamental assumptions and base data for the Pelamis were taken to be the same as those used by Parker *et al.* (2007). This study, therefore, considers a generic case for the production of a single Pelamis based on manufacturer's data for the first production machines, and a fixed scenario for manufacture, assembly and deployment of the device was defined: the typical wave farm is located off the north-west coast of Scotland; manufacture of the large steel tube sections, as well as final assembly of the Pelamis, takes place at a steel fabrication yard on the nearest coast; the power conversion modules housing the complex power take-off equipment are assembled at the Pelamis plant in Fife, approximately 420 km away; once completed the Pelamis is towed to the installation location, assumed to be within 200 miles

(320 km) of the steel fabrication yard, implying a travel time of 24 hours at 7 knots. Later versions of the device and different installation scenarios will not have the same impacts as those presented here; for example, the current generation P2 devices are manufactured and assembled at the new Pelamis plant in Leith Dock, Edinburgh.

The power output of a single device installed at a typical site off the coast of Scotland is estimated to average 2.97 GWh/year over the 20-year design life, and the successful installation at Aquaçadoura found that the Pelamis performed as expected, so this assumption is still considered valid (Parker *et al.*, 2007; PWP, 2011). Unless otherwise specified it is also assumed that all components are manufactured in the UK and subject to UK energy statistics and transport distances.

4.2 Analysis of the Pelamis Life Cycle

The first stage of this analysis was to inventory all resource consumption and pollutant emissions for the system model, including all environmentally relevant flows throughout the life cycle of the device, as previously illustrated in Figure 4.4 (Baumann and Tillman, 2004). (A more detailed flow chart is given in Figure 2.2 in Chapter 2.) Where data was not readily available, necessary assumptions were made, and these are detailed in the relevant sections of this chapter. Such assumptions ranged from applying a cut-off criteria to approximating oxy-acetylene flame cutting by using data from a similar process. The impacts of assumptions and estimates are reviewed in Section 4.4. Base data for quantities of raw materials, transportation, processing and manufacturing methods were based on figures derived by Parker *et al.* (2007) from PWP's own records.

4.2.1 Materials and manufacture

Figure 4.5 shows the principal components of the Pelamis. The main structure is formed from four cylindrical steel tube sections that increase in length from fore to aft, and sand ballast is placed within these tubes to optimise the buoyancy. The nose tube, which is tapered at one end to allow the device to cut through waves in rough conditions, houses the switchgear and transformer to collect the power for export to shore. Three PCMs sit between the tube sections and house the hydraulic power take-off, generators and control equipment. The Pelamis is connected to the mooring and cabling system via the yoke, a Y-shaped element connected to the nose tube; this has a quick-release tethering system to allow for rapid attachment and detachment.

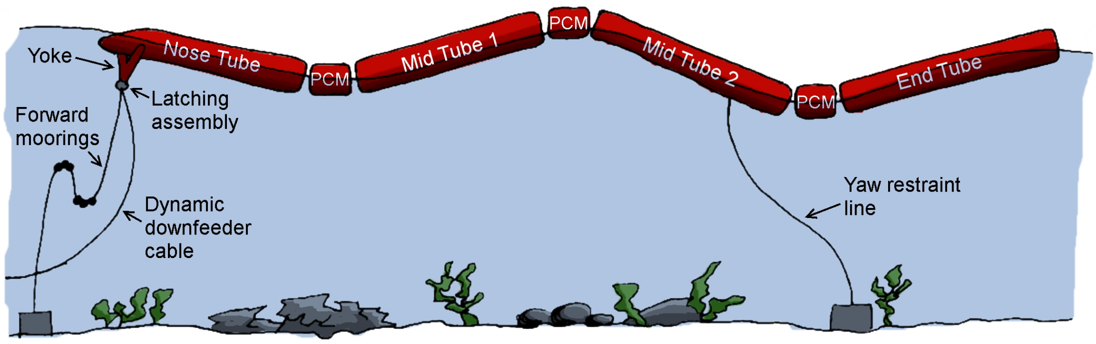


Figure 4.5: Sketch of Pelamis components

Steel production and processing

A mass-based analysis was carried out for the structure, hydraulic system and mooring components, with a full breakdown of the materials shown in Table 4.1. Over 50 % of the total mass of the Pelamis is steel, with the main tubes made from standard steel plates that are cut to length, rolled and welded to form tubular sections (Parker, 2007). The selected Ecoinvent LCI data includes steel produced in both blast and electric arc furnaces, hot rolled into plate sections (Ecoinvent, 2010). As Ecoinvent doesn't include explicit emissions and resource use data for virgin and recycled steel, it is assumed that steel produced in a blast furnace is primarily virgin material and that produced in an electric arc furnace is primarily recycled, which gives an assumed recycled content of 37 % (Classen *et al.*, 2009). Average European data for further steel processing, such as welding and wire drawing, was also sourced from Ecoinvent.

Stock Material	Mass (kg)
Steel	561954
Sand	475722
Stainless Steel	550
Nylon 6	416
Polyurethane	343
Glass Reinforced Plastic (GRP)	90
PVC Pipe	55

Table 4.1: Material quantities in the Pelamis P1

Some of the linking sections of the Pelamis are sand-cast. Ecoinvent does not include data for cast steel, so the base material was assumed to be similar to the European average, but excluding the hot rolling process. Data for resource consumption and emissions for sand-casting was taken from a mass balance analysis of the British foundry manufacturing sector, which was carried out by Donohoe (2001) as part of the Mass Balance Project.

Oxy-acetylene flame cutting is used to cut the steel plate to shape. Similar to gas welding, the manufacturer's data for this process is quantified by the area of material removed. No LCI data was readily available so the process was approximated with available Ecoinvent data for welding, estimating that each square metre of removed material would be equivalent to a 50 m length of weld, and that a typical weld is 20 mm wide. The weld material included in the Ecoinvent data is minimal, so this was disregarded.

The data provided by Pelamis Wave Power for machining includes all small-scale precision removal of material, such as milling, grinding and drilling. This was approximated by selecting Ecoinvent data for milling steel, where the documentation provides an estimate that each square millimetre of material removed is equivalent to 7.8 x 10^{-6} kg (Classen *et al.*, 2009). This was verified by considering the energy consumption within the Ecoinvent data of 13.3 J/mm^3 and comparing it to figures quoted by Kalpakjian *et al.* (2008) who suggest it should be 2.7 - 9.3 J/mm^3: the Ecoinvent data is a conservative estimate.

The finished tube sections are blasted with abrasives to remove any oxidation or impurities from the surface before being painted with a corrosion-resistant paint. As these processes are not detailed within Ecoinvent, they were approximated from published information. An estimate of the required quantity of compressed air was calculated to be 5.8 m^3 from manufacturer's data for abrasive blasting equipment, as shown in Table 4.2 (Axxiom, 2008); this assumes that abrasive jet blasting requires a compressed air supply at a pressure of around 850 kPa (Kalpakjian *et al.*, 2008), and 10 kg of abrasive is required to clean each square metre of steel (Jiven *et al.*, 2004).

Nozzle Size (inches)	Abrasive (lb/hr)	Air (cfm)	Time (min)	Air (m^3)
1/8	165	26	8	5.9
3/16	375	58	3.5	5.7
1/4	660	105	2	5.9
5/16	1050	160	1.3	5.9
3/8	1475	235	0.90	5.9
7/16	2050	315	0.65	5.8
1/2	2650	410	0.50	5.8
5/8	4100	640	0.32	5.8
3/4	5950	925	0.22	5.8
Mean				5.8

Table 4.2: Estimating the compressed air requirement for jet blasting with 10 kg of abrasive (Axxiom, 2008)

The Ecoinvent report on metals processing contains some information on the abrasive blasting included in standard steel production processes - in particular data for the associated particulate emissions (Classen *et al.*, 2009). The abrasive blasting process was, therefore, approximated by applying sand and compressed air data from Ecoinvent and including additional particulate emissions to the air.

The corrosion-resistant 'glass-flake' paint used on the Pelamis is comprised of a primer, paint and protective polymer top coat. This was approximated from manufacturer's data for a typical high-solids epoxy paint, pigmented with glass flakes and cured with polyamide/amine (Hempel, 2007). This is applied with an airless spray at 250 bar, providing a coverage of 3.9 m^2/l with a thickness of 200 μm. Parker *et al.* (2007) estimated the paint thickness as 1 mm, equivalent to five layers. It has, therefore, been assumed that the coating will comprise a base coat primer, three layers of glass-flake paint, and a polymer topcoat. This was approximated using Ecoinvent data for glass flakes, epoxy resin, and a curing agent made up of a combination of different chemicals, selected to provide a close approximation to the Hempel (2007) data. The composition of the primer and topcoat were taken to be the same as the paint layer without the glass flakes.

An estimate of the energy consumption for the process of applying the paint was based upon manufacturer's data for an airless spray pump (Graco, 2010). This is powered by compressed air at 200 m^3/min to provide paint coverage at 12 l/min; combining this with the coverage information for the glass flake paint suggests that 21.4 m^3 of compressed air is required for each square metre of coating.

Sand

The second most massive material in the Pelamis is the sand ballast, which is placed in each tube section to optimise the mass and buoyancy. There is little processing involved, and therefore it was considered acceptable to use the data provided by Ecoinvent for sand extraction in Switzerland. As this data is given for sand at the quarry, transport of the sand from a quarry in the UK to the steel fabrication yard was also included. Further information on transport is given in section 4.2.2.

Plastics

The Pelamis also has a number of plastic components, mostly within the moorings. The Plastics Europe database, which is based on manufacturer's information, has much more comprehensive data for a wide range of different plastics than Ecoinvent, so was used for the PVC pipe and polypropylene.

The data provided by PWP for the polyurethane foam mooring buoy was volumetric, so the mass was estimated by applying a typical density of 110 kg/m^3 - a figure obtained from data published by a European manufacturer of mooring buoys (Trelleborg, 2009).

Other components

The Pelamis also contains many pre-fabricated components, including fixings and hydraulic and electrical equipment. Published guidance allows for cut-off criteria to be defined to exclude inputs that do not have a significant environmental impact, provided that at least 95 % of the life cycle impacts are included (ISO, 2006b; BSI, 2011). A cut-off was applied to the pre-fabricated components by examining the results of the preliminary carbon and energy audit detailed in Section 4.1.2, which included a cost-based analysis of the pre-fabricated components based on information from PWP: all items contributing less than 10 % to the total cost, embodied carbon or embodied energy of the pre-fabricated components were excluded. These are estimated to contribute less than 1 % to the total impacts of the entire device, in accordance with existing guidance. The application of this cut-off criterion found that only the transformer, main generators and switchboard were to be included in the analysis, and LCI data for these were sourced from Environmental Impact Assessments (ABB, 2007, 2011, 2010).

4.2.2 Assembly and installation

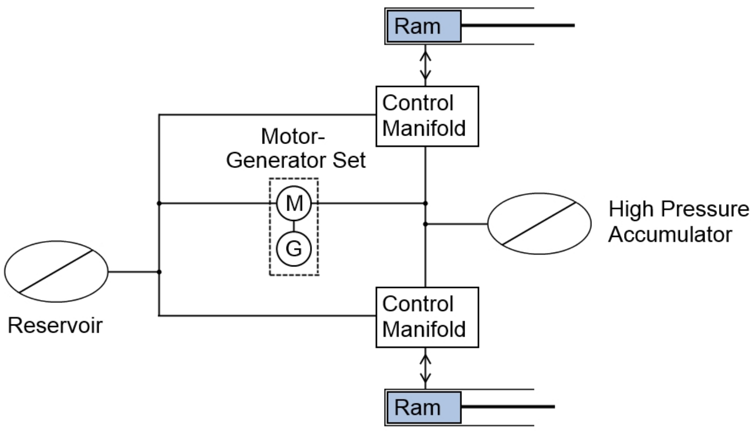


Figure 4.6: Schematic of power conversion module

The most complex part of the assembly stage is building the power conversion modules that house most of the hydraulic equipment (Figure 4.6) and are assembled at the PWP plant in Fife. The first step is to transport the separate components to the plant. Typical mass-distance data for freight transport was applied from Ecoinvent, based on information provided by PWP, as shown in Table 4.3; where data was not available, estimates were made and are indicated by *italics*. Components with no specific source location were assumed to come from the UK, and this was taken to be the centre of population, which is about 540 km by road from Fife (Dorling and Atkins, 1995).

Assembly of the PCMs requires fork-lift trucks and 60-tonne overhead cranes. The required hours of operation were provided by PWP (4.7 hours of fork-lift operation and 40 hours of crane

Component	Total Mass (kg)	Source Location	Distance (km)	Transport Method
Panels	*20*	Scotland	100	Road haulage
MG Set	60	Scotland	106	Road haulage
Structural shell	23207	Scotland	130	Road haulage
Hydraulic rams	5800	England	510	Road haulage
Reservoirs & oil	*2620*	UK	*540*	Road haulage
Manifold & hoses	450	UK	*540*	Road haulage
Misc items	*160*	UK	*540*	Road haulage
Heat exchanger	*100*	Holland	*750*	Cargo ship
Accumulator pack	3000	Wales	722	Road haulage
Bellows	*100*	China	*18 000*	Cargo ship

Table 4.3: Transport data for PCM components

operation per Pelamis), and the impacts were approximated by considering the corresponding fuel and energy consumption. Information from equipment manufacturers suggest that a 60-tonne overhead crane is likely to be a double girder crane (SWF, 2011), such as that shown in Figure 4.7. This would be powered by three separate electric motors for positioning and operation. The manufacturer's data suggests that the nominal rating of the hoist motor would be approximately 18 kW. Assuming that the average power consumption across the three motors is equivalent to one motor operating at full power, the hourly energy consumption of the crane is estimated to be 18 kWh of electricity from the UK grid. This is a very rough approximation that could be improved by obtaining detailed electricity consumption data from PWP, but the impacts of crane operation are expected to be relatively small.

Figure 4.7: Overhead crane for PCM assembly (Photo by Ronald Parker)

A typical fork-lift truck is estimated to consume 2.55 l/hr of diesel fuel (based on the DP20N model (Caterpillar, 2011)). Ecoinvent data for the combustion of diesel in equipment is given per unit of energy consumed, so the density of diesel was taken to be 0.8325 kg/l and the specific energy to be 43 MJ/kg to convert this volumetric data to a fuel consumption of 91.3 MJ/hr (BSI, 2010; EC, 2009).

The three completed PCMs are transported to the steel fabrication plant for final assembly and installation of the Pelamis. A range of specialist sea vessels are required at the installation stage to install moorings and power cabling, carry out sea trials, tow the device to site and latch it to the moorings. Fuel consumption data for these processes was provided by PWP and approximated by appropriate scaling of the Ecoinvent operational data for a barge (see Table 4.4).

Sea Vessel	Fuel Consumption (l/day)	Total days of operation
Barge	290	11.8
Multicat	1710	23.8
Tug	1490	11.8

Table 4.4: Sea vessel operations for Pelamis installation

4.2.3 Operations and maintenance

During operation the Pelamis is remotely monitored and controlled by an onshore computer, which is likely to have relatively small environmental impacts; therefore, none were considered for the operational stage. In contrast, the impacts from maintenance processes are significant. Annual maintenance requirements were estimated by PWP and are understood to be conservative, with the key aim of confirming and ensuring survivability (Table 4.5). These involve sea vessel operations, which are again approximated by scaling the Ecoinvent operational data for a barge to reflect PWP's own fuel consumption data. Due to uncertainty over the likely maintenance requirements, and following the same assumptions made by Parker *et al.* (2007), no allowance was made for repairs and replacement of parts, which may underestimate impacts from this stage.

Sea Vessel	Fuel Consumption (l/day)	Total annual days of operation
Tug	1490	4
Inspection Vessel	500	26.5

Table 4.5: Sea vessel operations for Pelamis maintenance

4.2.4 Decommissioning and disposal

It is expected that decommissioning will require the operation of sea vessels to unlatch the Pelamis, tow it back to shore and recover all mooring hardware. The impacts of this were again estimated from Ecoinvent data for a barge, based on fuel consumption data provided by PWP (see Table 4.6). Due to uncertainty over the processes involved, no allowances were made for the impacts of dismantling the device.

Sea Vessel	Fuel Consumption (l/day)	Total days of operation
Barge	290	2.5
Multicat	1710	8.5
Tug	1490	2.5

Table 4.6: Sea vessel operations for Pelamis decommissioning

Waste is expected to be divided into two streams, with the majority of metals being recycled and the remainder of the waste going to landfill. The Ecoinvent database does not include European average data for landfilling waste, so this was taken from the European Life Cycle Database (v2.0) where available, and was otherwise approximated with Ecoinvent data for Switzerland.

Recycling is more complicated to analyse within life cycle assessment because it provides the opportunity for avoiding both the impacts of waste treatment and those of primary material extraction. Care must be taken to avoid assigning recycling credit to both the waste material and the resulting product, as this would be double-counting. Several different methods have been developed for allocating recycling credit within LCA, and are discussed in Section 2.4.2. In this analysis the 50:50 method was selected, as it is the only method that can fully reflect the benefits of truly sustainable design: in order to design for minimum environmental impact, both the use of recycled materials and end-of-life recyclability should be maximised (Hammond and Jones, 2010). This method considers credit for both the recycled content of the raw materials and recycling at the end-of-life, but avoids double-counting by only taking 50 % of the credit for each. The implementation of this method within SimaPro and an examination of the sensitivity of the results to the chosen recycling allocation method are described in detail in Section 4.4.4.

It is expected that most, if not all, waste metal from the Pelamis could be recycled, but for this analysis the recycling rate is taken to be 90 % for only steel, copper and aluminium. These materials make up 54 % of the total mass of the Pelamis, and are estimated to account for over 99 % of the total recyclable content. No end-of-life recycling is considered for other metals and recyclable materials, such as plastics, although the selected materials data may contain recycled content credit.

The estimated recycled content of the three metals is shown in Table 4.7, taken from the Ecoinvent data. As Ecoinvent doesn't include explicit emissions and resource use data for virgin and recycled steel, it has been assumed that steel produced in a blast furnace is primarily virgin material and that produced in an electric arc furnace is primarily recycled. As previously noted, the data for the average European mix contains 37 % steel from an electric arc furnace, so this is the assumed recycled content applied throughout the calculations (Classen *et al.*, 2009).

Material		**Recycled Content**
Steel		37 %
Copper		44 %
Aluminium	From new scrap	21.6 %
	From old scrap	10.4 %

Table 4.7: Recycled content of metals

4.3 Results

4.3.1 Life cycle inventory

The life cycle inventory analysis produced a list of almost 1000 substances consumed or emitted during the Pelamis life cycle. Although further calculation is required to quantify the associated environmental impacts, the LCI contains useful information about raw material consumption and GHG emissions. Table 4.8 shows the total life cycle emissions of the six Kyoto greenhouse gases for the Pelamis. It can be seen that CO_2 emissions are significant; these mostly arise during steel manufacture and sea vessel operations. More information on the calculation of the global warming potential (GWP), or carbon footprint, is given in Section 4.3.2.

Gas		**Emissions (g/kWh)**	**GWP (kg CO_2 eq/kWh)**
Carbon Dioxide	CO_2	25	25
Methane	CH_4	0.042	0.96
Nitrous Oxide	N_2O	0.0013	0.38
Sulphur Hexafluoride	SF_6	$5.6\text{x}10^{-7}$	0.013
Hydrofluorocarbons	HFC	$2.8\text{x}10^{-6}$	0.0039
Perfluorocarbons	PFC	$2.0\text{x}10^{-6}$	0.013

Table 4.8: Emissions of the GHGs identified in the Kyoto Protocol

The LCI data also provides detailed information on the consumption of raw materials, with the most significant shown in Figure 4.8. Gravel is the raw material for the sand ballast, which makes up a large proportion of the final mass of the Pelamis, and it therefore appears in considerable quantity in the raw material consumption, although it has low environmental impacts. Coal, iron ore and calcite are all consumed in the manufacture of steel, which makes up over 50 % of the mass of the finished device (Table 4.1). Crude oil consumption, which is also significant, is mostly refined into the fuel used in sea vessels during installation, maintenance and decommissioning.

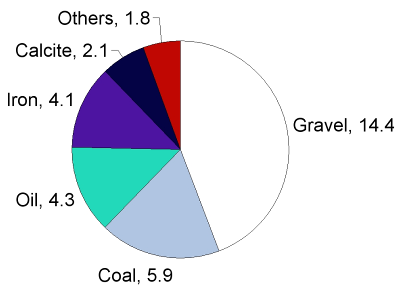


Figure 4.8: Summary of raw materials (g/kWh)

4.3.2 Life cycle impact assessment

The EDIP 2003 impact assessment method was applied to classify and characterise the results of the LCI and find the environmental impacts, summarised in Table 4.9 (Hauschild and Potting, 2005). The selection of this method, which was designed for use with the Ecoinvent database, was intended to minimise mismatches between the substances listed in the LCI and those with specified characterisation factors. Some mismatches, however, are inevitable: of the substances identified in the inventory, 12 % by mass were not included in the impact assessment method.

Upon further examination, most of the mismatches were found to be substances with insignificant environmental impacts, such as water and air. Biogenic carbon dioxide, however, was identified as a substance that could have a global warming potential. It is convention to exclude this from standard impact assessment methods, as it is the carbon dioxide emissions of living organisms or biological processes. The sensitivity of the results to the inclusion of this was tested by assuming a GWP of 1 kg CO_2 eq, which resulted in an increase of only 0.6 % to the total impact potential of the Pelamis. The emission of 'unspecified oils to soil' was also identified as a potentially significant mismatch, as it accounts for 78 % of the total emissions to soil by mass and may affect the toxicity categories. The actual impacts of this are, however, difficult to assess, and therefore it is recommended that the toxicity results are treated with caution.

Global warming potential

The carbon footprint, or global warming potential, was found to be 27 g CO_2 eq/kWh, corresponding to a carbon payback period of only 14 months if the grid carbon intensity is taken to be 460 g CO_2/kWh (see Sections 4.1.3 and 5.2). Note that this will be even shorter if the device offsets only marginal carbon intensive generation, as discussed in Chapters 5, 6, 7 and 8. The contribution of each of the six Kyoto gases to this total is given in Table 4.8.

Impact Potential		
Global warming	27	g CO_2 eq/kWh
Ozone depletion	2.3	μg CFC-11 eq/kWh
Ozone formation (Vegetation)	0.39	m^2.ppm.h/kWh
Ozone formation (Human)	2.6 x10^{-5}	pers.ppm.h/kWh
Acidification	0.0027	m^2/kWh
Terrestrial Eutrophication	0.0051	m^2/kWh
Aquatic Eutrophication (N)	20	mg N/kWh
Aquatic Eutrophication (P)	8.3	mg P/kWh
Human toxicity (Air)	558	m^3/kWh
Human toxicity (Water)	1.0	m^3/kWh
Human toxicity (Soil)	0.0049	m^3/kWh
Ecotoxicity (Water, chronic)	9.2	m^3/kWh
Ecotoxicity (Water, acute)	1.8	m^3/kWh
Ecotoxicity (Soil, chronic)	0.0025	m^3/kWh
Hazardous waste	1.6	mg/kWh
Slag/ashes	3.4	mg/kWh
Bulk waste	17	g/kWh
Radioactive waste	429	μg/kWh
Resources (all)	78	mg/kWh
Energy	411	kJ/kWh

Table 4.9: Results of LCIA and cumulative energy demand calculation

Figure 4.9 illustrates the contribution of different processes and life cycle stages to the GWP. It can be seen that steel production and sea vessel operations for maintenance have the greatest impacts, contributing 62 % and 26 % to the total respectively. In contrast, the impacts of freight transport, which includes all conventional transport such as road haulage and container shipping, contribute only 0.6 % to the total, and assembly processes are only 0.1 %. It can also be seen that waste disposal results in a significantly negative impact due to the recycling credit.

The relative contributions of the impacts from refining steel in a blast furnace compared to an electric arc furnace are also significant. As described in Section 4.2.1, steel from an electric arc furnace is taken to be recycled, and the assumed recycled content of the steel mix used in the Pelamis is 37 %. However, the application of the 50:50 method for assigning recycling credit reduces this figure to 18.5 % in the modelled life cycle (see Sections 4.2.4 and 4.4.4). Significantly, although 18.5 % of the total steel is produced in an electric arc furnace, this accounts for only 5 % of the total global warming potential of steel production. As can be seen in the following sections, a similar trend can be observed in all impact categories.

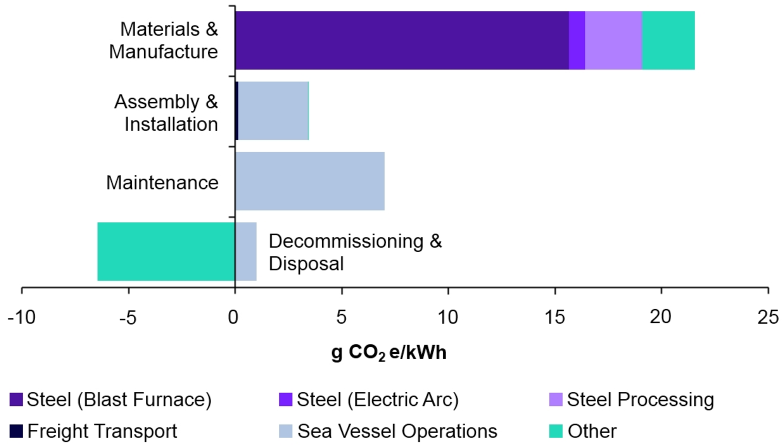


Figure 4.9: Global warming potential

Ozone depletion potential

Depletion of stratospheric ozone is caused by man-made emissions of gaseous compounds containing chlorine or bromine, such as CFCs, and the unit of measurement is mass of CFC-11 equivalent. The total impact for the Pelamis was found to be 2.3 μg CFC-11 eq/kWh, dominated by the processes associated with material extraction, manufacturing and sea vessel operations, as shown in Figure 4.10. In particular, the operation of sea vessels for maintenance procedures contributes 32 % to the total impact, while freight transport only accounts for 1.15 %, and assembly processes are only 0.04 %. In contrast to the figures for the GWP, the recycling credit is relatively small in this impact category.

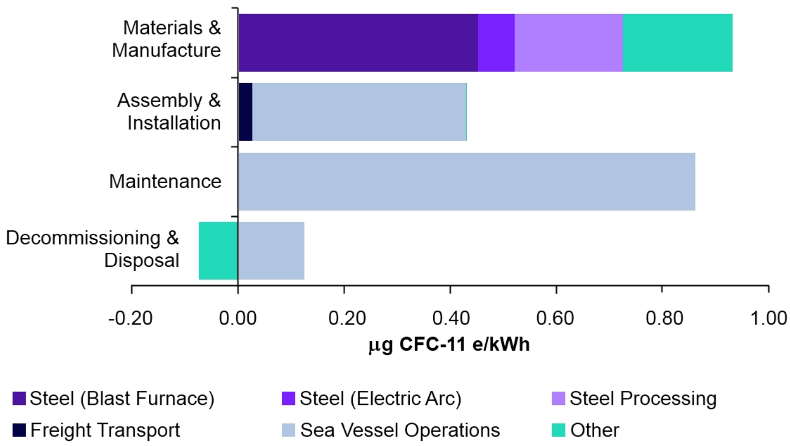


Figure 4.10: Ozone depletion potential

Ozone formation potential

Photochemical ozone formation occurs in the lower layers of the atmosphere where nitrogen oxides and volatile organic compounds react, in the presence of sunlight, to form ozone. The presence of this highly reactive compound in the troposphere can lead to respiratory problems in humans and reduce agricultural yield. The impacts are therefore divided into their impact on human health and ecosystem health, and were found to be 2.6×10^{-5} pers.ppm.h/kWh and 0.39 m^2.ppm.h/kWh, respectively, for the Pelamis. The units of measurement are defined as follows (Hauschild and Potting, 2005):

> For humans the impact is expressed as the AOT60, the accumulated exposure above the threshold of 60 ppb times [sic] the number of persons which are exposed as a consequence of the emission...
>
> For vegetation, the impact is expressed as the AOT40, the accumulated exposure ... above the threshold of 40 ppb times [sic] the area that is exposed as a consequence of the emission. The threshold of 40 ppb is chosen as an exposure level below which no or only small effects occur.

As can be seen in Figure 4.11, the relative contribution of the different processes and life cycle stages is very similar for both impact categories, with sea vessel operations accounting for 74 % of the total. As the blast furnace steel contributes a further 25 %, these impacts combine to account for 99 % of the total ozone formation potential of the Pelamis; the recycling credit offsets all impacts from the other materials and processes.

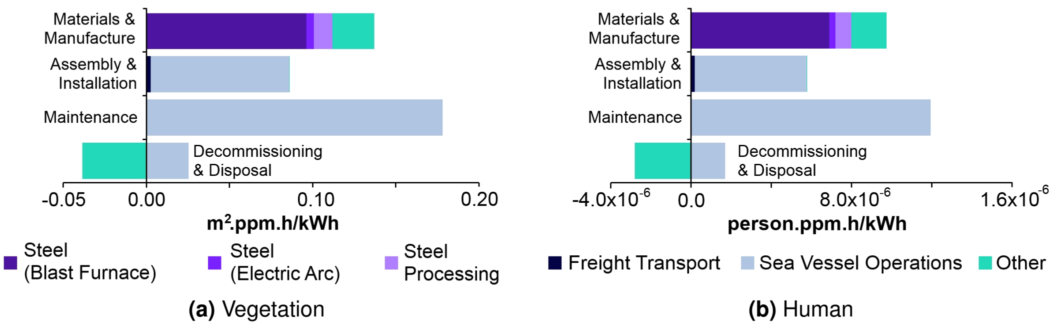


Figure 4.11: Ozone formation potentials

Acidification potential

The acidification of water in the atmosphere ultimately increases the acidity of terrestrial water or the soil matrix. When this reaches a critical level, toxic aluminium ions are released in harmful quantities; this is used as the basis for the unit of measurement in the EDIP 2003 methodology (Hauschild and Potting, 2005):

> ... the area of ecosystem within the full deposition area which is brought to exceed the critical load of acidification as a consequence of the emission...

The acidification potential of the Pelamis life cycle was found to be 0.0027 m^2/kWh, with 41 % of this from the production of steel, as shown in Figure 4.12. Sea vessel operations are also significant, particularly those for maintenance.

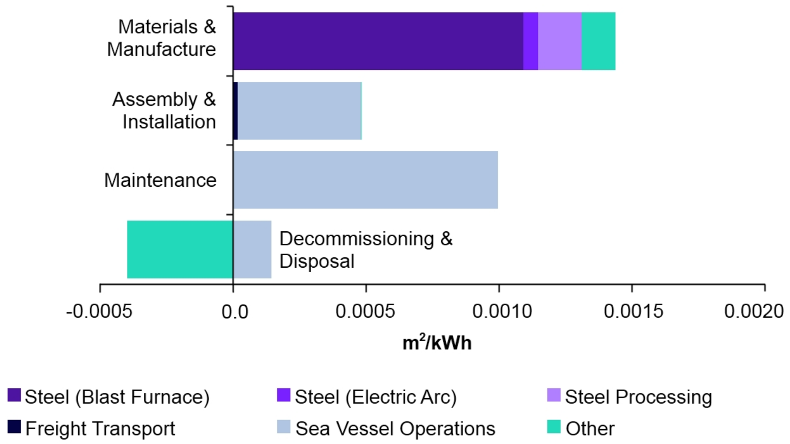


Figure 4.12: Acidification potential

Eutrophication potential

The enrichment of an ecosystem with nutrients is only harmful when a critical level is reached that leads to a change in species composition. In natural soils, atmospheric deposition is the principal man-made source of nutrients. Similarly to the acidification potential, the unit of measurement for the terrestrial eutrophication potential is the area of the terrestrial ecosystem, within the full deposition area, where the emission of pollutants raises the level of nutrients in excess of this critical value (Hauschild and Potting, 2005).

The terrestrial eutrophication potential of the Pelamis was found to be 0.0051 m^2/kWh, with sea vessel operations accounting for 78 % of this, as shown in Figure 4.13. Steel manufacturing is also significant, and there is some credit due to recycling at the disposal stage.

Eutrophication in aquatic ecosystems can also limit biological growth and is caused by emissions of compounds to the air, water and soil that contain biologically-available nitrogen and phosphorous. It is expressed as the maximum potential exposure of the ecosystem to these nutrients caused by the pollutant emission (Hauschild and Potting, 2005).

The aquatic eutrophication potentials of the Pelamis were found to be 20 mg N/kWh and 8.3 mg P/kWh. The relative contribution of different processes to the aquatic eutrophication potential due to nitrogen was found to be very similar to those for terrestrial eutrophication, but for phosphorous-containing compounds the production of steel is dominant (Figure 4.14). Blast-furnace production of steel accounts for 109 % of the total aquatic eutrophication potential due to phosphorous, offset by a recycling credit of (-)39 %.

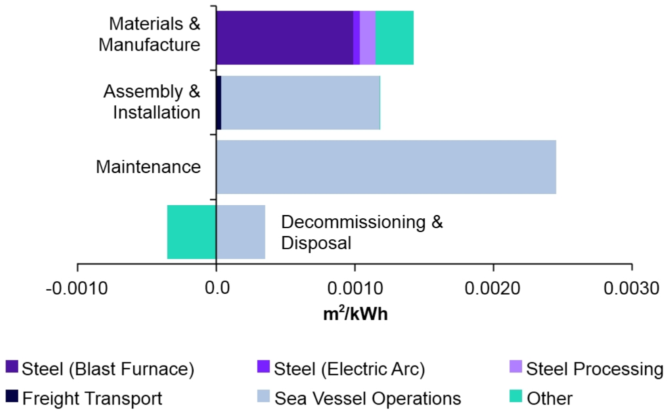
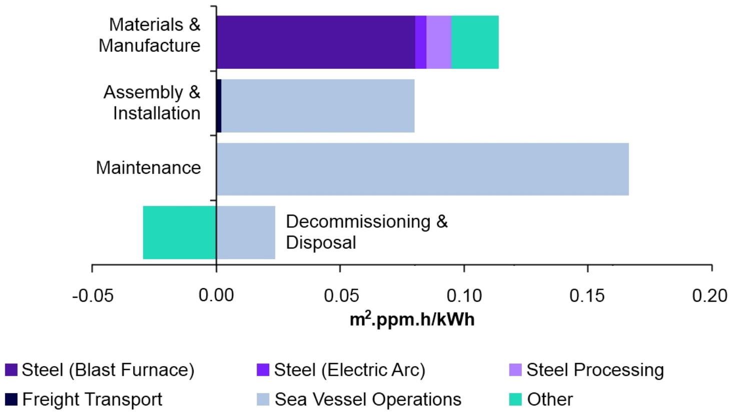
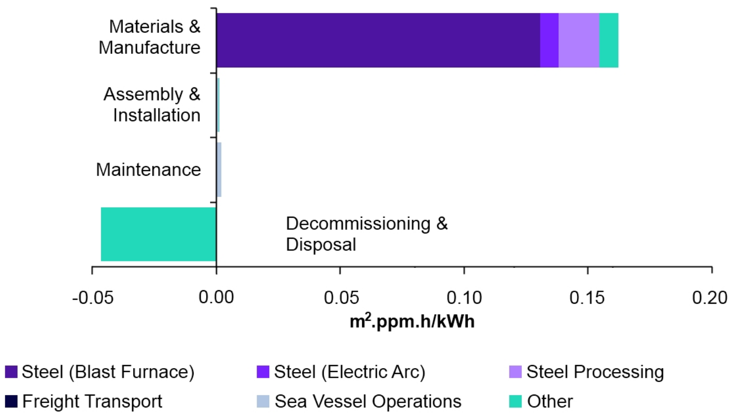


Figure 4.13: Terrestrial eutrophication potential



(a) Nitrogen



(b) Phosphorous

Figure 4.14: Aquatic eutrophication potentials

Human toxicity potential

Determining the impact of life cycle processes on human toxicity is complicated by the potential of nearly every substance to be toxic to human beings. This is only the case, however, when the dose, as determined by the exposure, exceeds a critical limit. Humans are exposed to environmental pollutants through three principal mechanisms: inhalation with air, ingestion with food and water, and penetration of the skin. The characterisation factors used in EDIP 2003 are exposure factors to evaluate the variations in human exposure through inhalation at different locations. It is therefore considered that these do not replace the earlier EDIP 97 characterisation factors, but are to be used in combination with them (Hischier *et al.*, 2010). The EDIP 97 methodology divides the human toxicity potential into three environmental compartments (air, water and soil), measured as the (Hauschild and Potting, 2005):

> ...volume of environmental compartment... which can be polluted up to the common reference concentration or -dose, [sic] the level not expected to cause effects on lifelong exposure...

The human toxicity potentials for the Pelamis life cycle were found to be 558 m^3/kWh in air; 1.0 m^3/kWh in water; and 0.0049 m^3/kWh in soil. It can be seen in Figure 4.15 that material extraction and sea vessel operations contribute significantly to the air and soil categories. Freight transport and assembly processes each contribute less than 1 % to the total impacts.

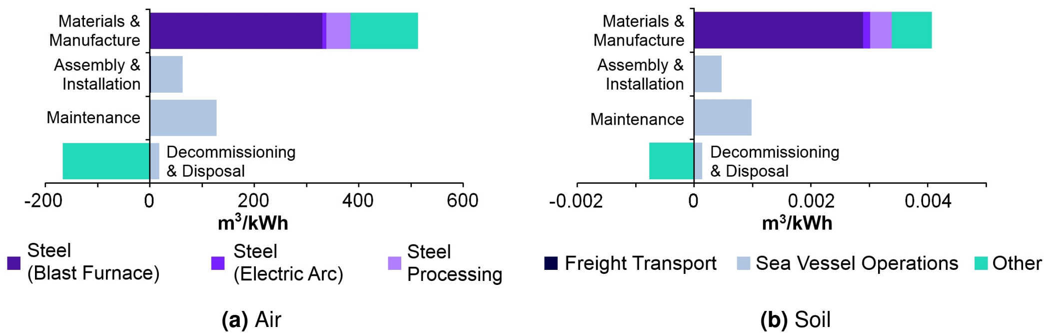


Figure 4.15: Human toxicity potentials

In contrast to air and soil, the human toxicity potential in water is dominated by material extraction and manufacturing processes, with sea vessel operations combined contributing less than 2 % to the total impacts (Figure 4.16).

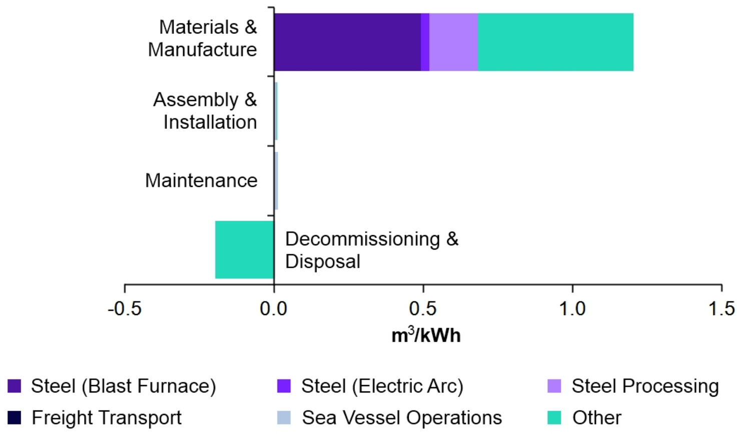


Figure 4.16: Human toxicity potential (water)

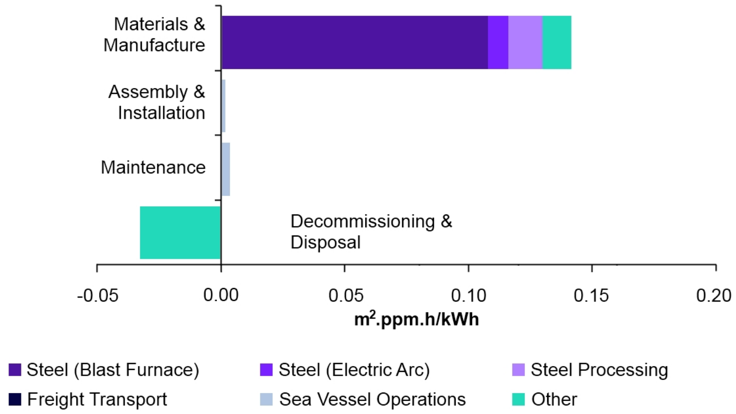
Ecotoxicity potential

Any pollutant emission that has a toxic effect on the organisms in an ecosystem will contribute to the ecotoxicity potential. However, similarly to substances considered toxic to humans, any substance can be ecotoxic if the dose is large enough. In determining the ecotoxicity of a pollutant, properties such as persistence and ability to bioaccumulate are also considered.

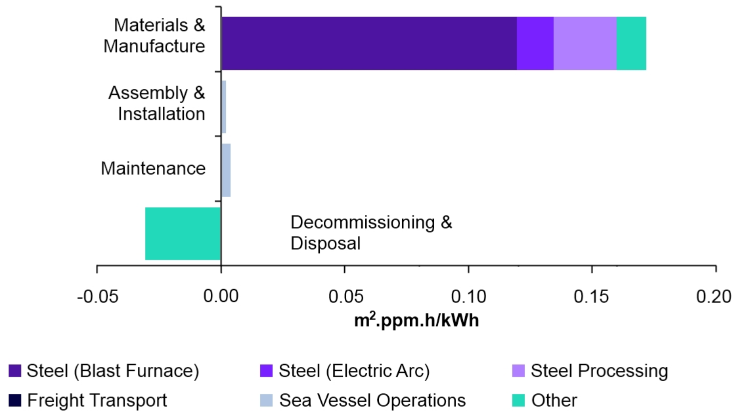
As with the human toxicity potential, the EDIP 2003 exposure factors for ecotoxicity are intended for use in combination with the EDIP 97 characterisation factors to identify site-specific impacts. These are considered in both aquatic ecosystems (acute and chronic) and terrestrial ecosystems (chronic exposure only) (Hauschild and Potting, 2005). The potential ecotoxic effects are expressed as the volume of the given medium required to absorb a pollutant emission without adverse effects on the ecosystem (Stranddorf *et al.*, 2005).

The ecotoxicity potentials of the Pelamis were found to be 9.2 m^3/kWh (chronic exposure) and 1.8 m^3/kWh (acute exposure) in aquatic ecosystems, and 0.0025 m^3/kWh for chronic exposure in terrestrial ecosystems. The relative contribution of different processes and life cycle stages to the aquatic ecotoxicity were found to be very similar for both acute and chronic exposure, as shown in Figure 4.17. In both cases steel extraction and processing are the dominant contributors, with the impact from other processes, including sea vessel operations, being relatively small.

The contribution of the different processes and life cycle stages to the potential terrestrial ecotoxicity is given in Figure 4.18. It can be seen that sea vessel operations contribute slightly more to this impact category, although steel extraction and processing is still dominant.



(a) Chronic exposure



(b) Acute exposure

Figure 4.17: Aquatic ecotoxicity potentials

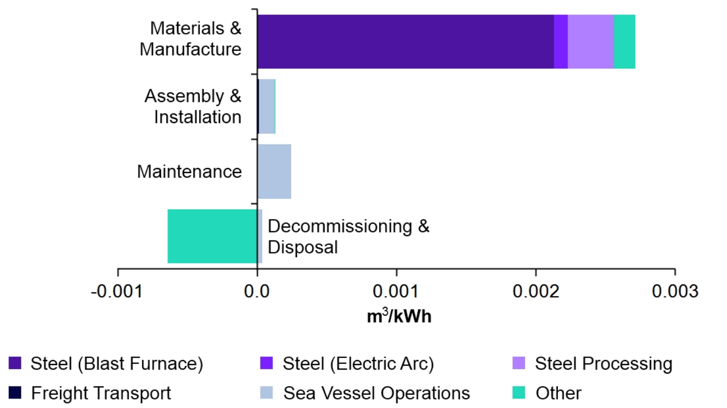
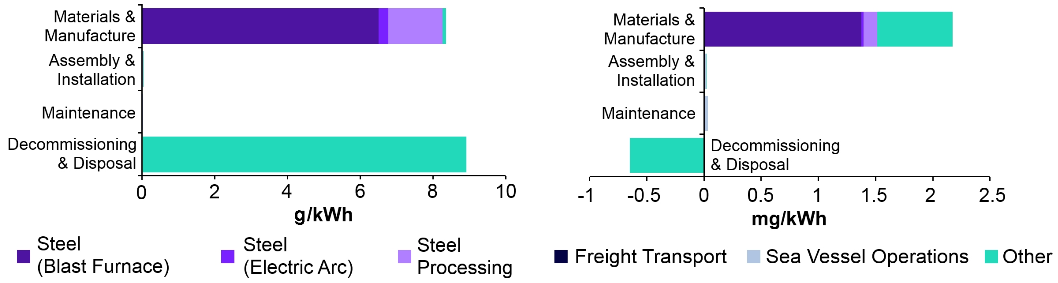


Figure 4.18: Terrestrial ecotoxicity potential (chronic exposure)

Waste

The impact assessment methodology implemented for classifying and characterising the waste impacts is taken from EDIP 97, and considers the mass of waste that goes to landfill (Hischier *et al.*, 2010). This is divided into four types: bulk waste, slag and ashes, hazardous waste and radioactive waste.

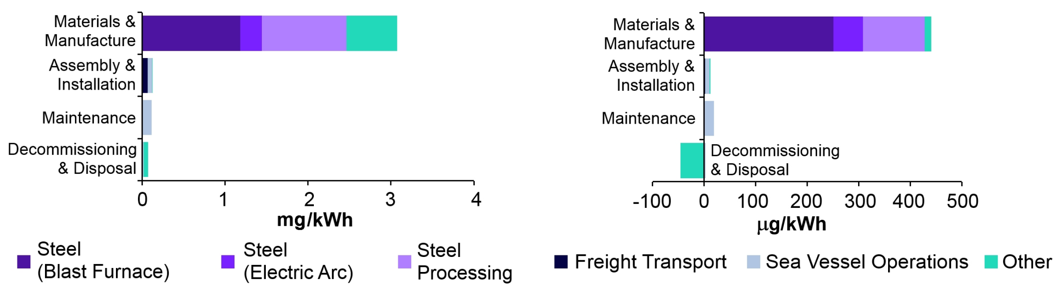
The life cycle of the Pelamis is expected to generate 17 g/kWh of bulk waste, of which 51 % is from disposal at the end-of-life, and 38 % is generated during the refining of steel in a blast furnace, as illustrated in Figure 4.19. In contrast, it can be seen that there is a (-)41 % credit to the hazardous waste category for recycling materials at the end-of-life. There is also a significant contribution from the processing and manufacture of materials other than steel, which contributes 42 % to the total hazardous waste of 1.6 mg/kWh.



(a) Bulk **(b)** Hazardous

Figure 4.19: Hazardous and bulk waste

During the Pelamis life cycle, 3.4 mg/kWh of slag and ashes will be produced, with 91 % of this in the materials and manufacturing stage, mostly from steel processing, as shown in Figure 4.20. The greatest contribution of freight transport to any impact category is in its 2 % contribution to the waste slag and ashes.



(a) Slag/ashes **(b)** Radioactive

Figure 4.20: Slag, ashes and radioactive waste

It was calculated that the life cycle of the Pelamis would produce 429 μg/kWh of radioactive waste, all due to the consumption of nuclear electricity. As can be seen in Figure 4.20, the majority of this was produced during the materials and manufacturing stage, where European average LCI data was applied. This data assumes that any electricity consumed is produced by the average European generation mix, of which nuclear energy is a significant contributor. A detailed investigation of the results found that 51 % of the total radioactive waste was produced by nuclear electricity generation in France. As nuclear electricity plays a much more modest role in the UK, it is possible that the actual radioactive waste impacts are lower for products and materials sourced locally.

Resources

It is useful to consider the consumption of finite resources in any product life cycle, but resources aren't considered in the EDIP 2003 methodology, so the method from EDIP 97 was applied. This calculates the total resource consumption on a mass basis of the pure resource, which was found to be 78 mg/kWh for the Pelamis life cycle, with 99 % attributable to the production of steel in a blast furnace (Figure 4.21).

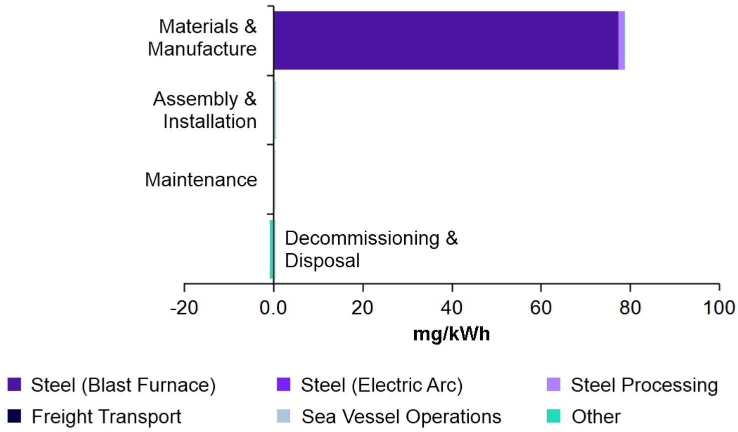


Figure 4.21: Resource consumption

4.3.3 Normalisation

Normalisation is an optional step within a life cycle impact assessment that allows the relative magnitude of each impact to be examined (ISO, 2006b). The normalisation references applied in the EDIP methodology are the annual background impacts per person in the area for which the impact is computed - either global or regional, depending upon the impact category (Stranddorf *et al.*, 2005). Due to a lack of data, the SimaPro version of the EDIP 2003 methodology does not include normalisation references for any ecotoxicity categories.

It can be seen in Figure 4.22 that, excluding ecotoxicity, the most significant impact categories for the Pelamis life cycle are aquatic eutrophication, human toxicity, bulk waste and radioactive waste. Global warming potential is considered to be only 3 % of the total normalised impact, suggesting that the focus on carbon footprint may be neglecting other impact categories. It is important to note, however, that the normalisation references for the EDIP methodology are based on background emissions from 1995, and do not attempt to assess the relative severity of changes in different impact categories. It could, therefore, be argued that a global warming potential of 2x10^{-6} person equivalents might actually be more damaging to the global ecosystem than an aquatic eutrophication potential (P) of 28x10^{-6} person equivalents. Such qualitative assessments are highly subjective, and may be assessed by the application of weighting factors.

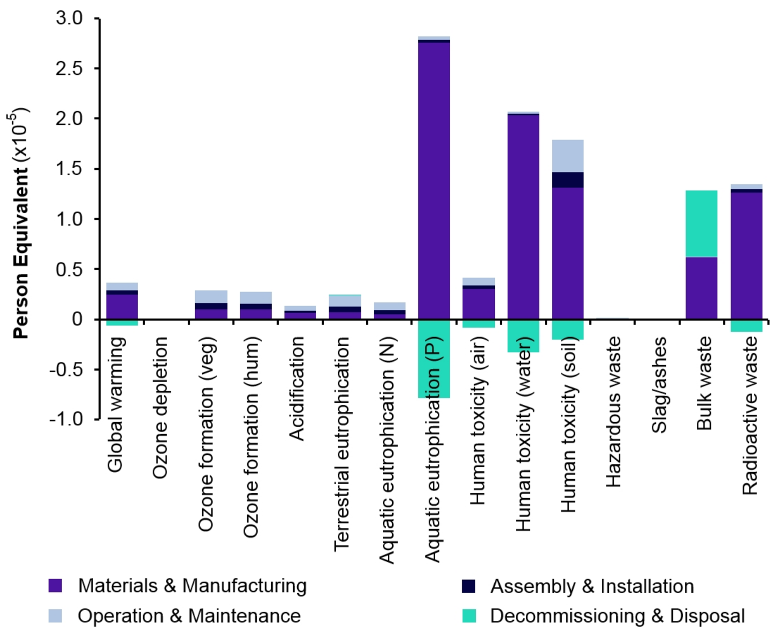


Figure 4.22: Normalised impact potentials for the Pelamis

4.3.4 Cumulative energy demand

It is of particular interest for power generation technologies, particularly renewable energy converters, to understand the energy return on investment (EROI). Figure 4.23 illustrates the principal energy flows through the Pelamis life cycle. Data for these energy flows is not explicitly available within the LCI, and energy is not included as an impact category in the EDIP 2003 methodology, so a Cumulative Energy Demand (CED) calculation was carried out. This is a form of Cumulative Energy Requirements Analysis that investigates both direct and indirect energy consumption throughout the life cycle of a device.

The CED methodology is based on that published by Ecoinvent, expanded for raw materials available in the SimaPro 7 database (Hischier *et al.*, 2010; Goedkoop *et al.*, 2008). This presents

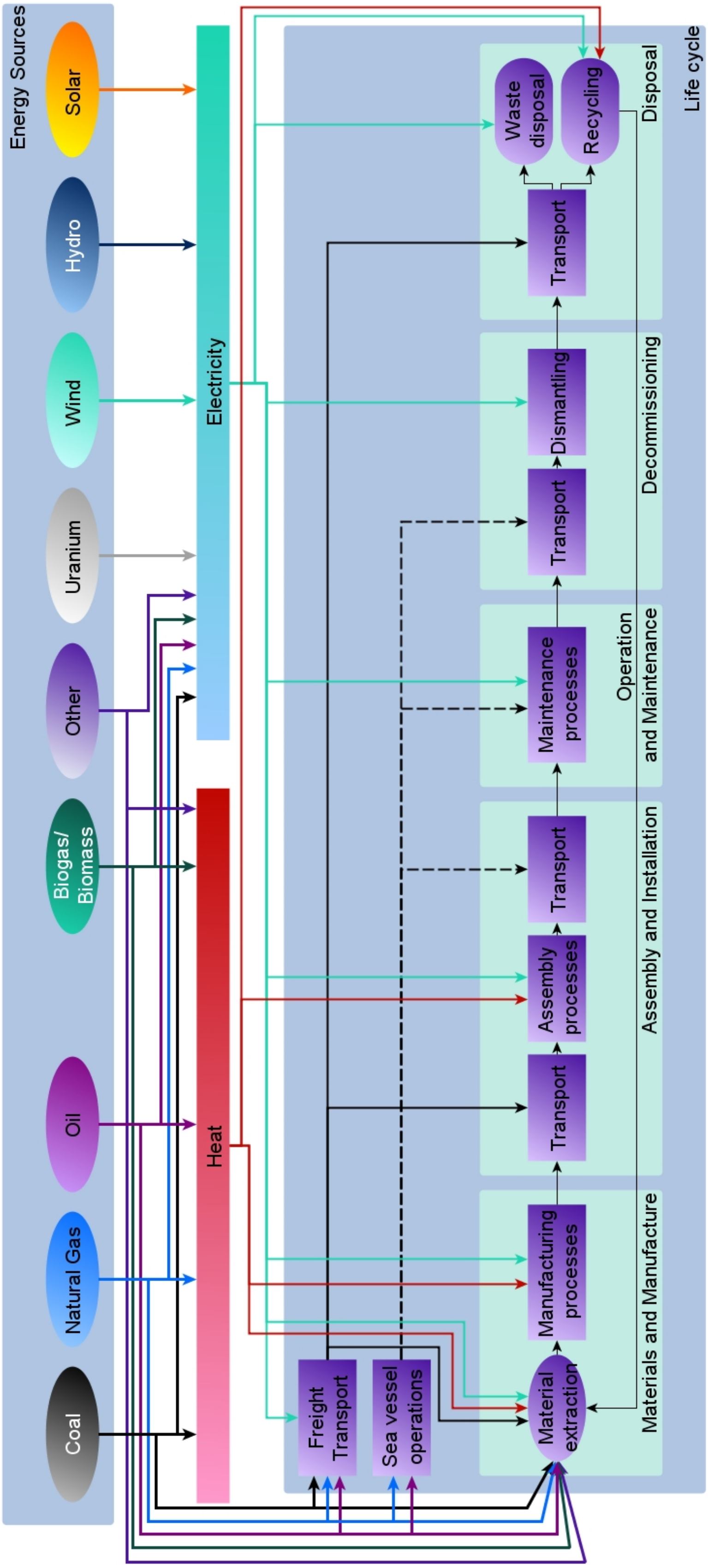


Figure 4.23: Main energy flows in the life cycle of the Pelamis

the results in eight categories according to the type of primary energy carrier; a division created by the different concepts existing for the characterisation of different primary energy carriers, suggesting that the resulting values may not be comparable across the subcategories. For the purposes of quantifying the energy invested in the Pelamis, however, the subcategories have been combined and the results are presented in Table 4.10. The total energy intensity of the Pelamis was found to be 411 kJ/kWh, corresponding to an EROI or energy ratio of 8.8, and an energy payback period of 27 months.

Category	Value (kJ/kWh)
Non renewable, fossil	364
Non-renewable, nuclear	31
Non-renewable, biomass	0.0016
Renewable, biomass	2.0
Renewable, wind, solar, geothermal	0.51
Renewable, water	14
TOTAL	411

Table 4.10: Cumulative energy demand

Figure 4.24 shows the contribution of different processes and life cycle stages to the total energy intensity. Again, the negative value for the disposal stage represents the avoided consumption due to recycling. The majority of energy consumption occurs during material extraction and manufacture, with almost 60 % in refining steel in blast furnaces. In contrast only 4 % is consumed in electric arc furnaces, despite the assumption that these produce 18.5 % of the steel used in the Pelamis (see Section 4.3.2). A further 25 % of the total energy consumption can be attributed to sea vessel operations for maintenance.

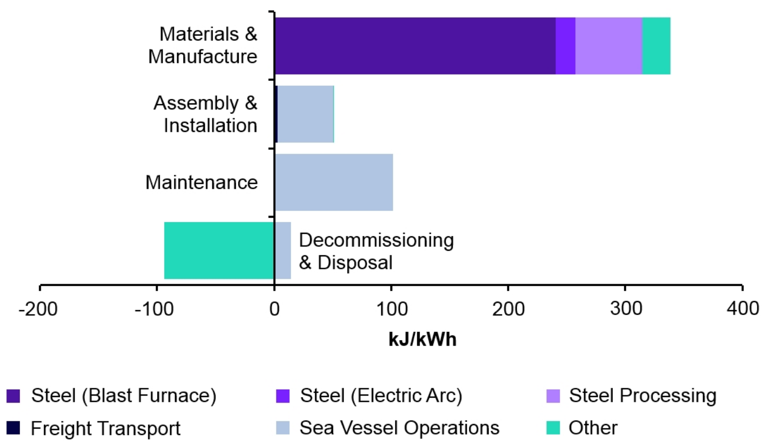


Figure 4.24: Contribution of significant processes to energy intensity

4.3.5 Consistency and completeness

This analysis aimed to investigate a broad range of environmental impacts of the whole life of the Pelamis WEC. The studied life cycle included all materials, manufacturing, transport, assembly, operations, maintenance, decommissioning and disposal processes. The particular assembly scenario under investigation assumed that the major steel components were processed at a steel fabrication yard, so no allowances were made for transport of much of the mass of the device. Inconsistencies may have been introduced by the quality of data gathered by Parker *et al.* (2007), particularly in the derivation of information from fabrication drawings; for example, it was observed that flame cutting only appeared in the data for the manufacture of the nose tube, suggesting that similar processes might have been omitted from the data for other parts. The sensitivity of the results to errors in input data was tested in the sensitivity analysis detailed in Section 4.4.2.

The manufacturing scenario was for a Pelamis P1 built in Scotland in 2006. Most of the base LCI data used in the calculation was average European data, although Swiss data was also applied when European or Scottish data was unavailable, generally for small items considered unlikely to have significant environmental impacts. In some cases the data was sourced from a single manufacturer, and the limitations of this inconsistency has been tested in the uncertainty analysis in section 4.4.1. Where possible, data was also selected to encompass information from 2006, as this is when the data was collected on the Pelamis manufacturing scenario, but otherwise more recent data was used.

Capital goods, such as lorries and machinery used in the manufacture of the Pelamis, were generally included in this analysis because they are included in the base Ecoinvent LCI data: the Ecoinvent data for road transport, for example, includes allowances for the operation of the vehicle, the production, maintenance and disposal of the vehicle, and the construction, maintenance and demolition of roads (Ecoinvent, 2010); however, data from other sources does not include the impacts of capital goods. It is unlikely that such an omission will have a significant impact on the results, as such data was primarily used in the calculation of the impacts of manufacturing processes that are very small in comparison to other life cycle stages.

Capital goods were also not included in the calculation of the impacts of sea vessel operations. This was because the data was approximated by scaling the Ecoinvent operational data for a barge according to the fuel consumption figures provided by PWP, and it was considered that the third order impacts of the specialist sea vessels required to install and maintain the Pelamis were unlikely to be similar to those of the average barge. The sensitivity of the results to this assumption was tested by re-running the analysis with third order impacts from the barge data included: as sea vessel operations are such a significant contributor across all impact categories, this did have a significant impact on the results, increasing the impacts by an average of 10 %, as shown in Table 4.11. The smallest change was in the resource consumption category and the largest in the production of radioactive waste, with the global warming potential increasing by

13 % and the energy intensity by 11 %. Further investigation is required to assess the true third order impacts of the particular specialist sea vessels used in the Pelamis, and these findings also raise questions about the convention of excluding capital goods from LCA, which is discussed in Section 2.5.

Impact potential			Increase
Global warming	30	g CO_2 eq/kWh	13 %
Ozone depletion	2.4	μg CFC-11 eq/kWh	7 %
Ozone formation (Vegetation)	0.41	m^2.ppm.h/kWh	5 %
Ozone formation (Human)	2.8 x10^{-5}	pers.ppm.h/kWh	5 %
Acidification	0.0028	m^2/kWh	7 %
Terrestrial Eutrophication	0.0052	m^2/kWh	4 %
Aquatic Eutrophication (N)	21	mg N/kWh	5 %
Aquatic Eutrophication (P)	9.7	mg P/kWh	16 %
Human toxicity (Air)	598	m^3/kWh	7 %
Human toxicity (Water)	1.1	m^3/kWh	11 %
Human toxicity (Soil)	0.0052	m^3/kWh	6 %
Ecotoxicity (Water, chronic)	10.4	m^3/kWh	13 %
Ecotoxicity (Water, acute)	2.0	m^3/kWh	13 %
Ecotoxicity (Soil, chronic)	0.0028	m^3/kWh	15 %
Hazardous waste	1.7	mg/kWh	6 %
Slag/ashes	3.9	mg/kWh	16 %
Bulk waste	18	g/kWh	2 %
Radioactive waste	589	μg/kWh	37 %
Resources (all)	79	mg/kWh	1 %
Energy	456	kJ/kWh	11 %

Table 4.11: Change in impact potentials due to inclusion of approximated capital goods data for sea vessel operations

4.4 Effect of Practitioner Decisions

As discussed in Section 2.5, it is a significant limitation of LCA methodology, and therefore also of carbon footprinting, that choices and assumptions made by the practitioner can affect the results. The international standards require a further analysis to be carried out to test the sensitivity of the results to data uncertainty, practitioner assumptions and calculation methods (ISO, 2006a,b). This section presents the results of a detailed sensitivity and uncertainty analysis, which allows the impact of the different practitioner decisions to be examined. The findings are also compared with those of the carbon and energy audit carried out by Parker *et al.* (2007) to identify the key decisions in estimating the carbon footprint and energy intensity of a steel wave energy converter.

4.4.1 Uncertainty analysis

The uncertainty analysis is an essential part of any LCA, testing the effect of the uncertainties in the input LCI data on the reliability of the results (ISO, 2006b). In this study it was carried out using the Monte Carlo method, which involves producing an uncertainty distribution by running the calculation many times with values randomly selected from a given probability density function for each variable. Probability distributions were therefore required for all input data taken from secondary sources, which included materials, recycled content, energy, manufacturing, assembly and freight transport processes.

The Ecoinvent database provides probability information for all data, generally assumed to have a lognormal distribution where the square of the geometric standard deviation covers the 95 % confidence interval. This standard deviation is estimated from the reliability of the input data by means of a pedigree matrix (Goedkoop *et al.*, 2008). In order to be consistent in this study, the uncertainty of data sourced from other references was also estimated using the same pedigree matrix.

The results of the uncertainty analysis are detailed in Table 4.12, and shown graphically in Figure 4.25. It can be seen that the uncertainty of the results varies significantly across the impact categories, with a typical 95 % confidence interval in the region of +60/-30 %. The greatest uncertainty is in the calculated human toxicity potential in water (+190/-50 %), and the lowest in the resource consumption (+12/-11 %).

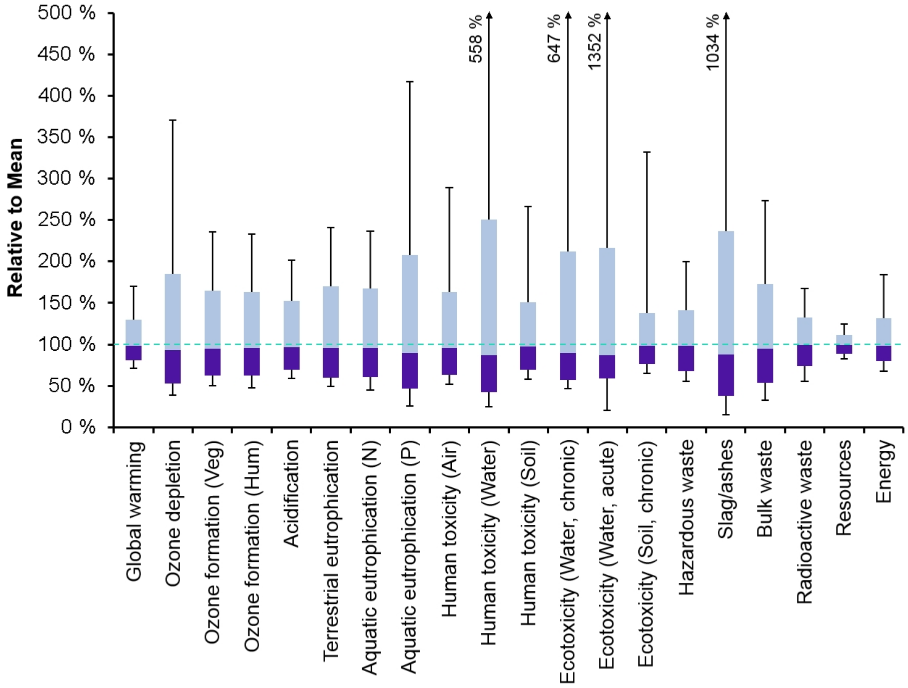


Figure 4.25: Uncertainty of impact potentials

Impact Potential	Median	SD	2.5%	97.5%	
Global warming	26	3.2	22	35	g CO_2 eq/kWh
Ozone depletion	2.1	0.79	1.2	4.2	μg CFC-11 eq/kWh
Ozone formation (Vegetation)	0.37	0.10	0.24	0.64	m^2.ppm.h/kWh
Ozone formation (Human)	2.5	0.67	1.7	4.3	x10^{-5} pers.ppm.h/kWh
Acidification	2.6	0.54	1.9	4.1	x10^{-3} m^2/kWh)
Terrestrial Eutrophication	4.8	1.4	3.0	8.7	x10^{-3} m^2/kWh
Aquatic Eutrophication (N)	19	5.3	12	33	mg N/kWh
Aquatic Eutrophication (P)	7.5	3.7	3.9	18	mg P/kWh
Human toxicity (Air)	532	147	352	907	m^3/kWh
Human toxicity (Water)	0.89	0.55	0.43	2.6	m^3/kWh
Human toxicity (Soil)	4.8	0.99	3.4	7.4	x10^{-3} m^3/kWh
Ecotoxicity (Water, chronic)	8.1	4.0	5.2	19	m^3/kWh
Ecotoxicity (Water, acute)	1.5	1.0	1.0	3.8	m^3/kWh
Ecotoxicity (Soil, chronic)	2.4	0.42	1.9	3.4	x10^{-3} m^3/kWh
Hazardous waste	1.6	0.30	1.1	2.3	mg/kWh
Slag/ashes	3.0	2.0	1.3	8.0	mg/kWh
Bulk waste	16	5.2	9.2	30	g/kWh
Radioactive waste	424	63	318	567	μg/kWh
Resources (all)	78	4.6	69	88	mg/kWh
Energy	404	54	328	540	kJ/kWh

Table 4.12: Results of uncertainty analysis

The Monte Carlo analysis found the global warming potential to be 26 g CO_2 eq/kWh +31/-18 %, and the energy intensity to be 404 kJ/kWh +34/-19 %; the resulting probability distribution for the global warming potential is shown in Figure 4.26. This magnitude of uncertainty is comparable to that in the analysis carried out by Parker *et al.* (2007), where the uncertainty in material embodied energy and carbon values taken from the Inventory of Carbon and Energy resulted in an error of ±36 % and ±38 % respectively for the calculated energy and carbon intensities.

4.4.2 Sensitivity to practitioner estimates

While the Monte Carlo analysis tests the sensitivity of the results to uncertainties in the secondary input data, significant variations may also be introduced by practitioner estimates and errors in the primary input data. The sensitivity of results to the accuracy of these values can be tested by varying them by a given range, in this case ±10 %, and presenting the results either as a percentage change or absolute deviation. The key values tested in the sensitivity analysis included the following:

- Annual energy output;
- Design life (also used as the temporal boundary);
- Primary input data, such as mass and distance information;

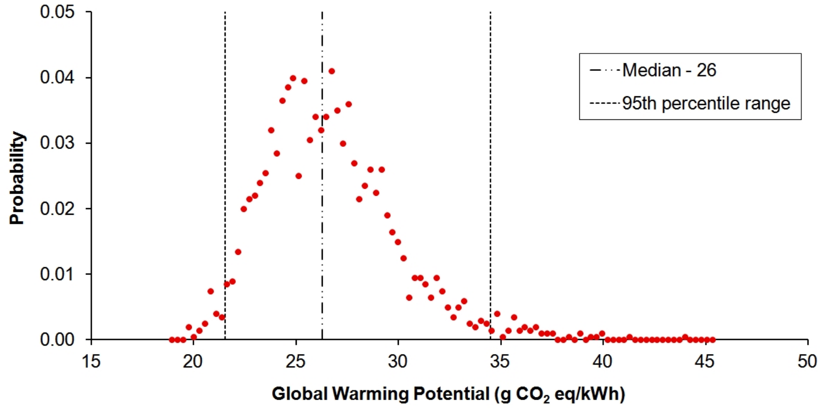


Figure 4.26: Probability distribution for GWP

- Cut-off criteria;
- Location of the steel fabrication yard and installation site;
- Recycling rate in the waste disposal profile.

Annual energy output

The results of all impacts are normalised per unit of output energy, which therefore requires an estimate of the lifetime energy production of the device. This is calculated from the estimated annual energy output and the design life. In this analysis the annual energy output was taken to be 2.97 GWh/yr, based on wave energy data for the given installation site combined with the manufacturer's performance data (Section 4.1.3). Figure 4.27 shows how a change of ±10 % in the estimated annual output affects all impacts equally with a +11/-9 % change across all categories.

In the particular scenario tested in this study the annual energy output of the Pelamis does not affect any aspects of the device life cycle, so is only used to normalise the results per unit of output energy and hence affects all impact categories equally. As the device continues to be developed, however, it is possible that maintenance could be scheduled according to the energy production, as this affects the rate of wear of components. In such a case the total life cycle environmental impacts of maintenance processes would be affected by the estimate of the annual energy output, and the accuracy of this value would, therefore, affect the accuracy of the impacts with the greatest contribution from maintenance processes, such as ozone formation and eutrophication.

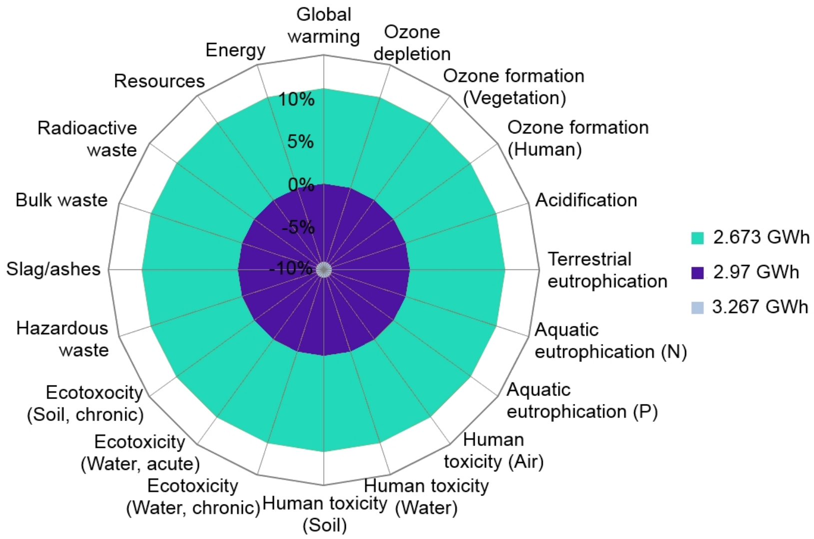


Figure 4.27: Sensitivity of impact potentials to annual energy production

Design life

The estimated design life of the Pelamis was also required for the calculation of the lifetime energy production, and to determine the maintenance requirements. Information from the manufacturer estimates this value at 20 years (Section 4.1.3) but, to date, no Pelamis wave energy converter has been proven to survive for this length of time in a real marine environment. The effect on the environmental impacts of changing this to values between 18 and 22 years was tested and the results are shown in Figure 4.28.

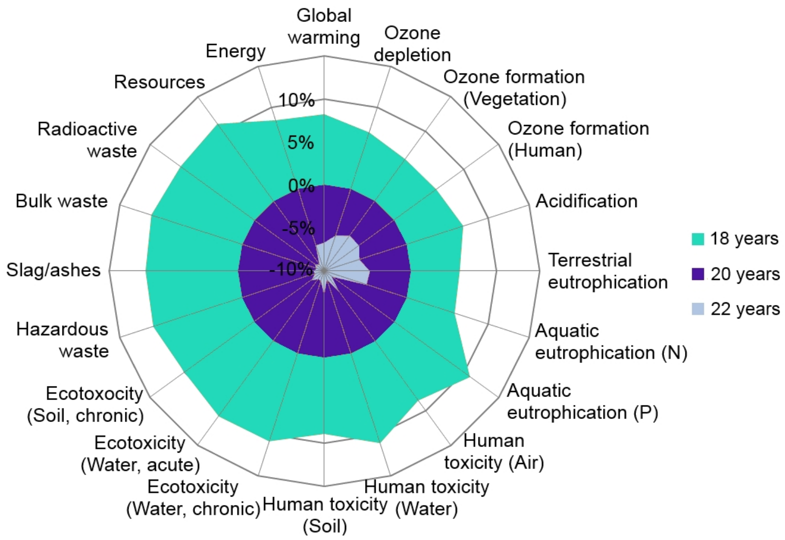


Figure 4.28: Sensitivity of impact potentials to design life

It was found that the environmental impacts were generally less sensitive to the estimated design life than the annual energy output, ranging from +6/-5 % for ozone formation and terrestrial eutrophication potentials to +11/-9 % for waste and toxicity categories. The effect of variations in design life on the estimated lifetime energy production should be the same as the effects of changes in annual energy output, with a decrease in design life leading to an increase in the normalised impacts. However, it can be seen that the corresponding decrease in maintenance requirements will mitigate this to some extent, most significantly in the impact categories most affected by the operation of sea vessels for maintenance purposes.

Primary input data

As discussed in Section 4.3.5, errors may have been introduced in the collection of primary data from the manufacturer. This data includes information about material mass and processing, freight transport distances, and installation, maintenance and decommissioning scenarios. In order to understand the range of uncertainty that could be introduced by errors in these input values, they were all varied by ± 10 % (Figure 4.29). While this method is not as sophisticated as the Monte Carlo simulation and may overestimate the uncertainty, it provides insight into the sensitivity of the results to the accuracy of the gathered data.

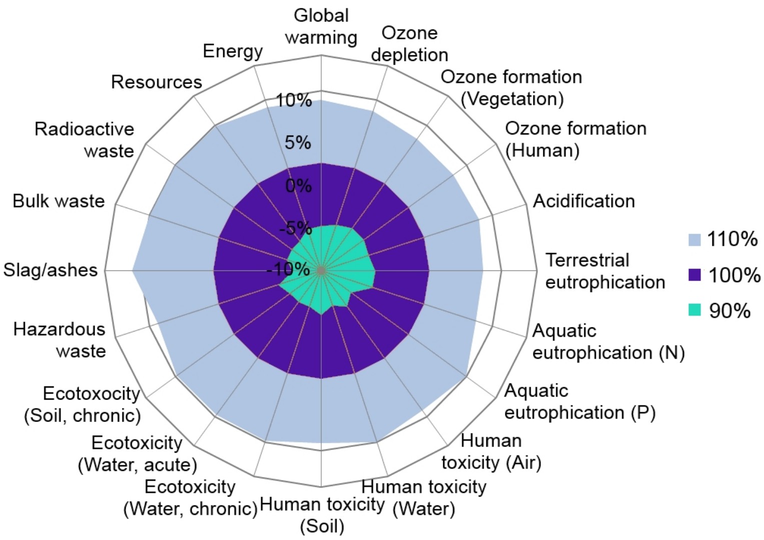


Figure 4.29: Sensitivity of impact potentials to accuracy of primary data

It can be seen that the variation in the results for many impacts is ± 10 %, suggesting that there is a direct correlation between the accuracy of the input data and the accuracy of the results. The exceptions to this are the slag and ashes, which has an error range of ± 11 %, and impact categories such as global warming potential and terrestrial eutrophication, which are less significantly affected.

Cut-off criteria

A cut-off criterion was applied to determine which pre-fabricated components to include in the analysis, excluding all those that contributed less than 10 % to the total cost, embodied carbon or energy in the preliminary carbon and energy audit (Section 4.2.1). It was found that varying this cut-off from 9 % to 11 % did not result in any changes to the components that were to be included or excluded.

Location

The location of the final installation site will affect the amount of wave energy available, the chosen location of the steel fabrication yard, and the distance of travel for sea vessels. The sensitivity of the results to changes in the assumed freight transport distances were previously tested in the examination of the effect of errors in the primary input data, and the change in available wave energy was considered when examining the effect of a change in annual energy output; however, transport of the power conversion modules to the steel fabrication yard and all sea vessel operations for installation, maintenance and decommissioning were not included.

In this analysis it was assumed that the wave farm would be located off the north-west coast of Scotland, with the steel fabrication yard on the nearest coast (Section 4.1.3). This section examines the sensitivity of the environmental impacts of the Pelamis to the distance of the steel fabrication yard from the Pelamis plant in Fife (originally taken to be 420 km), and from this yard to the final installation location (originally estimated at 322 km).

Figure 4.30 shows the results of the sensitivity analysis for distances to the steel fabrication yard of 378 to 462 km. It can be seen that the effect on the environmental impacts is very small, with a maximum range of ± 0.7 % for the slag and ashes category.

Varying the distance from the dockyard at the steel fabrication plant to the final installation site from 290 to 354 km, however, has a much more significant effect on the impacts, as can be seen in Figure 4.31. The categories that are most sensitive to these changes are those where sea vessel operations are important, such as the global warming potential, ozone categories, acidification, and some of the eutrophication categories. The greatest sensitivity is ± 5.5 % for terrestrial eutrophication.

In order to allow the results of this analysis to be adjusted for the widest possible number of different installation scenarios, the sensitivity to location was examined further and it was found that the relationship between each of the impact categories and the two distances was linear. Simple formulae have been developed to allow the normalised impact potentials to be estimated for any given installation location, as shown in Equation 4.1.

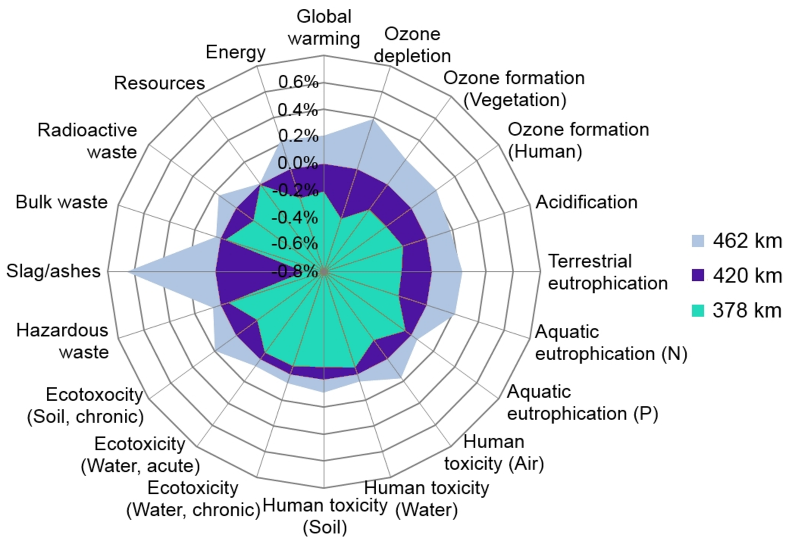


Figure 4.30: Sensitivity of impact potentials to location of steel yard

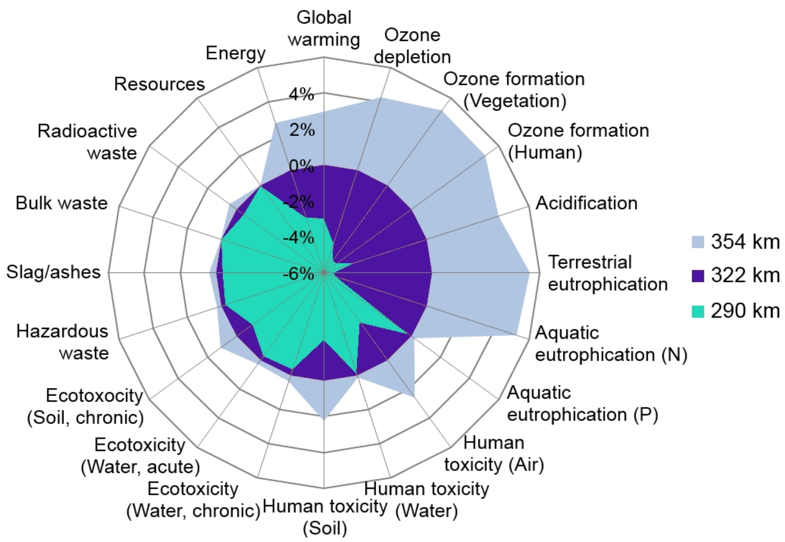


Figure 4.31: Sensitivity of impact potentials to the distance of the wave farm from the dockyard

$$IP = (a + bl_{steel} + cl_{offshore})/(20E) \quad (4.1)$$

where:

IP	=	Normalised impact potential (per kWh)
l_{steel}	=	Distance from Pelamis plant to steel fabrication yard (km)
$l_{offshore}$	=	Distance from dockyard to installation site (km)
E	=	Annual energy output (kWh)
a, b and c	=	Constants for each impact category (given in Table 4.13)

Note that this formula is a simplification of the results of this analysis, and cannot be used to determine the effect of a change in design life, assumed to be 20-years. Furthermore, this model has been developed for an installation scenario in the UK, and therefore installation in other countries may not have the same impacts.

Impact Potential	*a*	*b*	*c*	
Global warming	1.08 x10^{9}	7.74 x10^{4}	1.46 x10^{6}	g CO_2 eq
Ozone depletion	7.21 x10^{7}	1.24 x10^{4}	1.79 x10^{5}	μg CFC-11 eq
Ozone formation (Vegetation)	1.06 x10^{7}	1.27 x10^{3}	3.70 x10^{4}	m^2.ppm.h
Ozone formation (Human)	734	0.0862	2.48	x10^{-5} pers.ppm.h
Acidification	8.83 x10^{4}	6.99	206	x10^{-3} m^2
Terrestrial Eutrophication	1.30 x10^{5}	16.2	510	x10^{-3} m^2
Aquatic Eutrophication (N)	5.31 x10^{8}	6.20 x10^{4}	1.93 x10^{6}	mg N
Aquatic Eutrophication (P)	4.83 x10^{8}	6.49 x10^{3}	3.25 x10^{4}	mg P
Human toxicity (Air)	2.40 x10^{10}	1.37 x10^{6}	2.67 x10^{7}	m^3
Human toxicity (Water)	5.99 x10^{7}	765	2540	m^3
Human toxicity (Soil)	2.23 x10^{5}	6.45	205	x10^{-3} m^3
Ecotoxicity (Water, chronic)	5.23 x10^{8}	8390	6.15 x10^{4}	m^3
Ecotoxicity (Water, acute)	1.02 x10^{8}	1520	9440	m^3
Ecotoxicity (Soil, chronic)	1.28 x10^{5}	6.75	50.4	x10^{-3} m^3
Hazardous waste	9.13 x10^{7}	1320	7330	mg
Slag/ashes	1.80 x10^{8}	3.14 x10^{4}	2.35 x10^{4}	mg
Bulk waste	1.02 x10^{9}	8710	5050	g
Radioactive waste	2.38 x10^{10}	9.62 x10^{5}	3.90 x10^{6}	μg
Resources (all)	4.64 x10^{9}	4870	2.54 x10^{4}	mg
Energy	1.73 x10^{10}	1.31 x10^{6}	2.11 x10^{7}	kJ

Table 4.13: Constants for estimating the environmental impacts at alternative locations

Recycling rate

End-of-life recycling results in a credit at the disposal stage, so the recycling rate has a greater effect on impact categories that are sensitive to emissions and resource use from waste treatment, as shown in Figure 4.32. The assumed recycling rate for steel, copper and aluminium was taken to be 90 %, with all other materials going to landfill (Section 4.2.4). It was found that a decrease in recycling rate will result in an increase in all impact potentials, with a recycling rate of 99 % resulting in a decrease ranging from (-)0.1% for resource use to (-)5 % for bulk waste. Conversely, a recycling rate of 81 % raises the impact potentials. The toxicity and global warming potentials were also identified as being most sensitive to changes in assumed recycling rate.

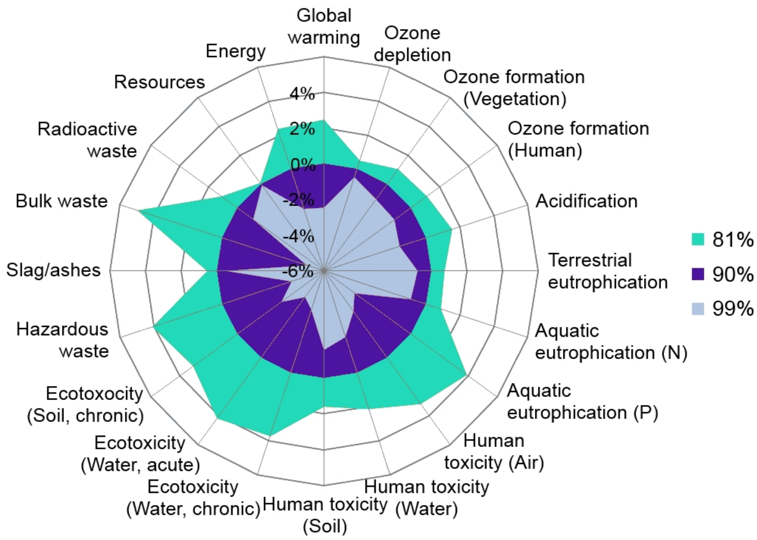


Figure 4.32: Sensitivity of impact potentials to end-of-life recycling rate

It is likely that the sensitivity of the impacts to the recycling rate is also a function of the assumed waste disposal scenario; in this analysis it was assumed that all waste that was not recycled was sent to landfill, which is typical in the UK, but will have different environmental impacts than the alternatives, such as incineration.

Summary

Figures 4.33 and 4.34 summarise the effects of variation in different practitioner estimates on the calculated global warming potential and energy intensity. It can be seen that the relationship between each factor and impact category is mostly linear, except for the estimated design life and annual energy output which are quadratic. The results are most sensitive to changes in estimated energy output and the accuracy of the primary input data, and least sensitive to the assumed distance from the Pelamis plant in Fife to the steel fabrication yard.

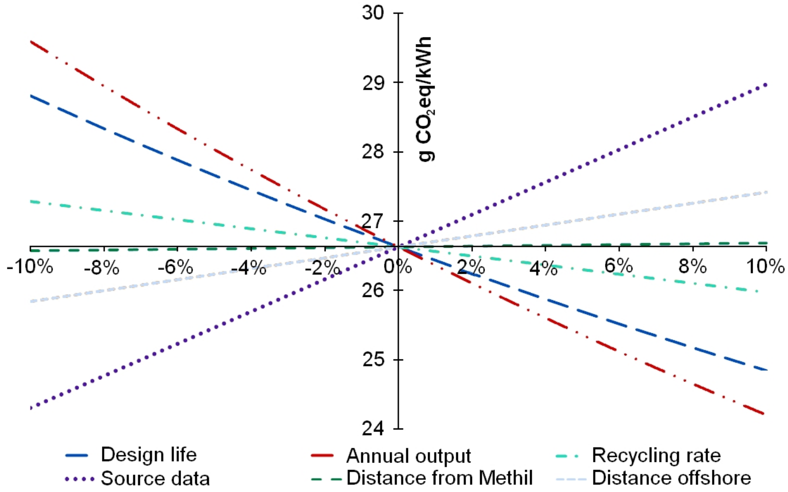


Figure 4.33: Sensitivity of GWP to different practitioner estimates

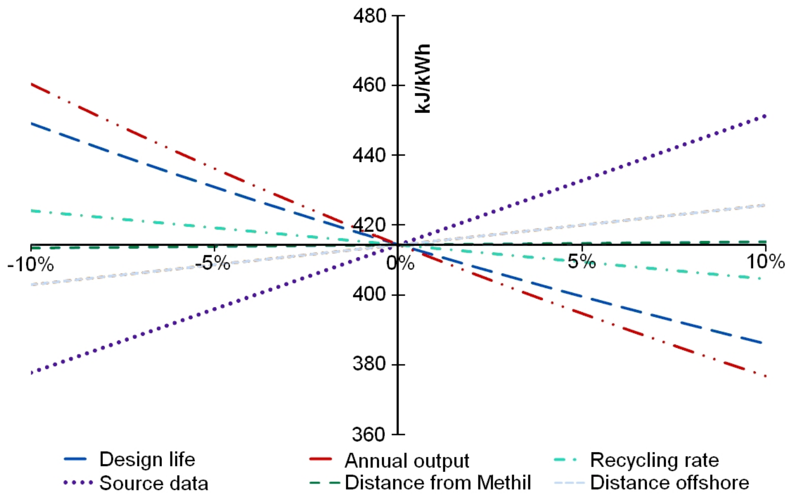


Figure 4.34: Sensitivity of energy intensity to different practitioner estimates

4.4.3 Life cycle impact assessment methods

Life cycle impact assessment (LCIA) involves the classification and characterisation of the results of the LCI into environmental impact potentials. The international standards allow the practitioner to define their own impact categories and characterisation factors at this stage, but normally one of several standard methods is applied (ISO, 2006b). Each of the standard methods, however, includes different impact categories and presents the results in different units: minimising opportunities for comparison between different studies.

In this analysis all impact potentials presented so far were calculated using the EDIP 2003 and CED LCIA methods. This section examines how this methodological choice affected the study, by presenting the resulting environmental impacts from several other popular LCIA methods. The impact potentials are then compared, where possible, across different LCIA methods, in order to assess the impact of this methodological choice on the findings of an LCA.

Selected LCI

In some situations it is useful to simply present a selection of LCI results, and therefore this method is provided with the Ecoinvent data (Hischier *et al.*, 2010). It involves the summation of selected substances on the sole basis of physical properties and does not take into account any environmental impacts or damages. The results of the selected LCI for this analysis are given in Table 4.14.

Inventory Results		
Non-methane volatile organic compounds (NMVOC)	17	mg/kWh
Carbon dioxide, fossil	25	g/kWh
Sulphur dioxide	49	mg/kWh
Nitrogen oxides	196	mg/kWh
Particulates, <2.5 μm	19	mg/kWh
Land occupation	576	mm^2/kWh
Organic water pollutants (BOD)	74	mg/kWh
Cadmium	0.0046	μg/kWh

Table 4.14: Selected LCI data

EDIP 97

EDIP 97 is the predecessor to EDIP 2003. Although it has now been superseded, it is included here for comparison with existing published studies that pre-date the release of the new method, and to examine the effect of the update on the calculated impact potentials. Generally the methodology is very similar to EDIP 2003, but some different characterisation factors and units of measurement have been applied; the method is described in detail in Stranddorf *et al.* (2005).

It can be seen from Table 4.15 that the results for waste and resource consumption are the same as those calculated with EDIP 2003 (given in Table 4.9, on page 87), which is due to the implementation of EDIP 2003 within SimaPro taking characterisation factors for these categories from the EDIP 97 methodology in accordance with the guidance provided by Hauschild and Potting (2005). Although EDIP 97 characterisation factors are also used in EDIP 2003 for the ecotoxicity and human toxicity categories, these have been modified by the EDIP 2003 exposure factors (see Section 4.3.2).

Environmental Impact Potentials		
Global warming (GWP 100)	27	g CO_2/kWh
Ozone depletion	2.2	μg CFC-11/kWh
Acidification	0.19	g SO_2/kWh
Eutrophication	0.58	g NO_3/kWh
Photochemical smog	12	mg ethene eq/kWh
Ecotoxicity (water, chronic)	51	m^3/kWh
Ecotoxicity (water, acute)	5.2	m^3/kWh
Ecotoxicity (soil, chronic)	0.24	m^3/kWh
Human toxicity (air)	9820	m^3/kWh
Human toxicity (water)	1.2	m^3/kWh
Human toxicity (soil)	0.0053	m^3/kWh
Bulk waste	17	g/kWh
Hazardous waste	1.6	mg/kWh
Radioactive waste	0.43	mg/kWh
Slag/ashes	3.4	mg/kWh
Resources (all)	78	mg/kWh

Table 4.15: Results from EDIP 97

Normalisation in EDIP 97 is based on the same methodology as that used in EDIP 2003, with the normalisation references being the annual background impact per person in the area of interest, but using 1990 as a reference year (Stranddorf *et al.*, 2005). The normalised results from this analysis are shown in Figure 4.35.

Eco-indicator 99

The Eco-indicator method is intended for use in comparative studies for optimising the design of a product or service, and is therefore based upon a normalisation and weighting system of three types of environmental damage: human health, ecosystem quality and resources (Goedkoop and Spriensma, 2001). As weighting factors are based on value-choices rather than objective measurements, this limits the application of the results; the ISO states that weighting “shall not be used in LCA studies intended to be used in comparative assertions intended to be disclosed to the public” (ISO, 2006b).

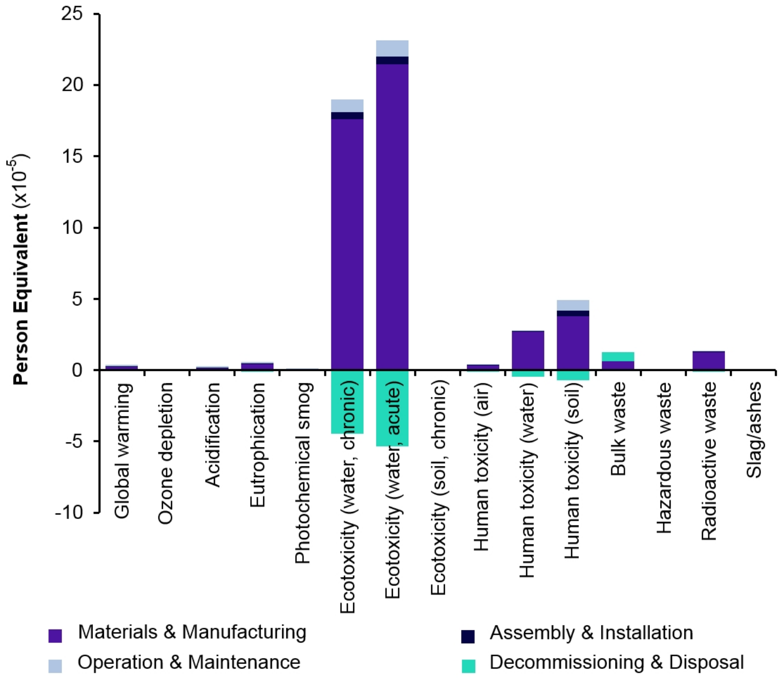


Figure 4.35: Normalised impact potentials from the EDIP 97 method

The units of measurement for the impact categories are divided into the types of environmental damage: damages to human health are expressed as Disability Adjusted Life Years (DALY); damages to ecosystem quality are expressed as the percentage of species that have disappeared in a certain area due to the environmental load (PDF); and resource extraction is related to a parameter indicating the quality of the remaining mineral and fossil resources (Goedkoop and Spriensma, 2001). The results for the Pelamis are shown in Table 4.16.

Damage category	**Impact**	
Carcinogens	$8.2\text{x}10^{-9}$	DALY/kWh
Resp. Organics	$2.1\text{x}10^{-11}$	DALY/kWh
Resp. Inorganics	$4.1\text{x}10^{-8}$	DALY/kWh
Climate Change	$5.6\text{x}10^{-9}$	DALY/kWh
Radiation	$6.3\text{x}10^{-11}$	DALY/kWh
Ozone Layer	$2.3\text{x}10^{-12}$	DALY/kWh
Ecotoxicity	0.051	PDF$\times$m^{2}yr/kWh
Acidification/Eutrophication	0.0012	PDF$\times$m^{2}yr/kWh
Land use	0.00031	PDF$\times$m^{2}yr/kWh
Minerals	16	kJ surplus/kWh
Fossil fuels	36	kJ surplus/kWh

Table 4.16: LCIA Results from Eco-indicator 99

A further normalisation and weighting step was applied to remove the dimensions from the results and enable comparison between the three damage categories, as illustrated in Figure 4.36. Goedkoop and Spriensma (2001) developed the normalisation and weighting factors by surveying a panel of members from a Swiss LCA interest group to identify the perceived severity of each type of damage to society.

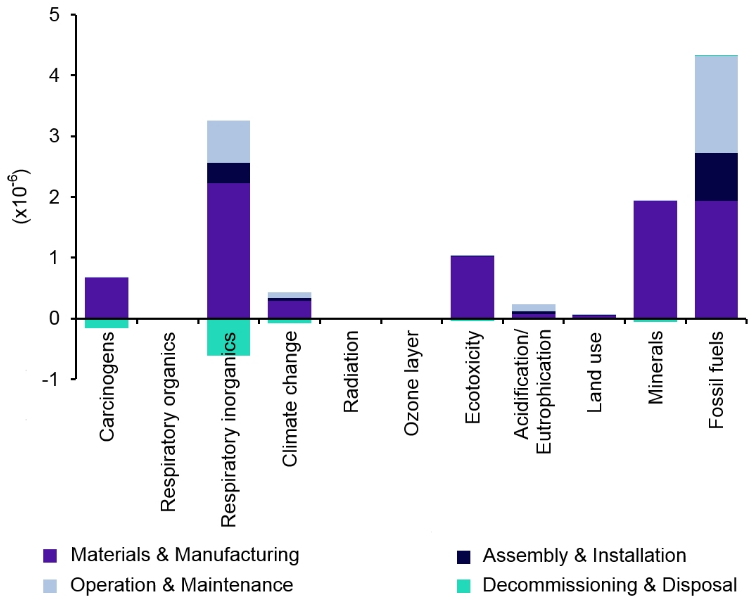


Figure 4.36: Normalised impact potentials from the Eco-indicator 99 method

Ecological scarcity

The method of ecological scarcity is a 'distance-to-target' method intended to deliver standardized, generic results that can be added and compared. It therefore only generates normalised and weighted results measured in eco-points (EP), with the weighting determined by a set of eco-factors. These are primarily based on environmental protection targets set by the Swiss government, and are intended to reflect the current environmental situation and the target situation aimed at by environmental policy. Alternative eco-factors have been developed for countries including Holland, Norway, Sweden, Belgium and Japan (Frischknecht *et al.*, 2009).

The results for the Pelamis from the Ecological Scarcity 2006 method, using Swiss eco-factors, are shown in Table 4.17.

Damage category	Impact	
Emission into air	30	EP/kWh
Emission into surface water	4.0	EP/kWh
Emission into ground water	0.0045	EP/kWh
Emission into top soil	0.040	EP/kWh
Energy resources	1.3	EP/kWh
Natural resources	0.50	EP/kWh
Deposited waste	2.8	EP/kWh

Table 4.17: LCIA Results from Ecological Scarcity 2006 method

Environmental Product Declaration (EPD)

The International EPD System was developed by the Swedish Environmental Management Council as a science-based, verified and comparable tool for communicating the environmental performance of products (The International EPD Cooperation, 2008). It introduces the concept of Product Category Rules (PCRs) to simplify calculations for a given market sector, and is widely used for power generation; most notably for mature technologies such as gas and nuclear power (AEA Energy and Environment, 2008a; Axpo, 2011; AEA Energy and Environment, 2009; Vattenfall, 2007).

The implementation within SimaPro reports the impact potentials for the specific categories required by a standard EPD; the results for the Pelamis are given in Table 4.18.

Impact Potential		
Global warming (GWP100)	27	g CO_2/kWh
Ozone layer depletion (ODP)	2.2	μg CFC-11/kWh
Photochemical oxidation	21	mg C_2H_4 eq/kWh
Acidification	150	mg SO_2 eq/kWh
Eutrophication	58	mg PO_4^{3-} eq/kWh
Non renewable energy consumption (fossil)	390	kJ/kWh

Table 4.18: LCIA results using the EPD method

CML 2 Baseline 2000

The CML 2 method was developed by the Institute of Environmental Sciences at Leiden University. It is a problem-oriented midpoint LCIA method, which presents the results as a set of impact potentials for given baseline categories. The characterisation factors are freely available on the Internet (Institute of Environmental Sciences, 2013).

The implementation within SimaPro is an extension of that described in the Ecoinvent report (Hischier *et al.*, 2010), and the results for the Pelamis are presented in Table 4.19.

Impact Potential		
Abiotic depletion	190	mg Sb eq/kWh
Acidification	160	mg SO_2 eq/kWh
Eutrophication	58	mg PO_4^{3-} eq/kWh
Global warming (GWP100)	27	g CO_2 eq/kWh
Ozone layer depletion (ODP)	2.2	μg CFC-11 eq/kWh
Human toxicity	86	g 1,4-DB eq/kWh
Fresh water aquatic toxicity	21	g 1,4-DB eq/kWh
Marine aquatic toxicity	25	kg 1,4-DB eq/kWh
Terrestrial ecotoxicity	230	mg 1,4-DB eq/kWh
Photochemical oxidation	7.5	mg C_2H_4/kWh

Table 4.19: Results from the CML 2 method

ReCiPe

ReCiPe is an LCIA methodology that builds upon both Eco-indicator 99 and CML 2, to include both midpoint and endpoint characterisation factors (Goedkoop *et al.*, 2012). This provides a consistent methodology for calculating both the environmental impact potentials (midpoint) and damage potentials (endpoint) associated with a device life-cycle. The developers of this method acknowledge that the calculation of midpoints is relatively scientifically robust, but produces abstract results that are difficult to interpret; while the calculation of endpoints involves some level of weighting that adds considerable uncertainty, but provides results that are easier to interpret. The results can be presented in one of three consistent sets of subjective choices, such as time horizon and assumed manageability, each identified by names: individualist (I), hierarchist (H) and egalitarian (E). Perspective I is based on the short-term interest, perspective H on the most common policy principles, and perspective E is the most precautionary. For this analysis the hierarchist perspective has been applied, and the results are presented in Tables 4.20 and 4.21. Further details of the methodology and the characterisation factors are freely available on the internet (ReCiPe, 2013).

It is of interest to note that the toxicity categories are reported in the same units as for CML 2, but the results are very different. As discussed in Section 2.4.2 and Baumann and Tillman (2004), there is considerable debate over the calculation methodologies for toxicity impact potentials, and this results in widely varying results between impact assessment methods.

Impact Potential		
Climate change	26	g CO_2 eq/kWh
Ozone depletion	2.3	μg CFC-11 eq/kWh
Human toxicity	11	g 1,4-DB eq/kWh
Photochemical oxidant formation	220	mg NMVOC/kWh
Particulate matter formation	91	mg PM10 eq/kWh
Ionising radiation	3.0	g U_{235} eq/kWh
Terrestrial acidification	160	mg SO_2 eq/kWh
Freshwater eutrophication	9.5	mg P eq/kWh
Marine eutrophication	79	mg N eq/kWh
Terrestrial ecotoxicity	2.7	mg 1,4-DB eq/kWh
Freshwater ecotoxicity	510	mg 1,4-DB eq/kWh
Marine ecotoxicity	530	mg 1,4-DB eq/kWh
Agricultural land occupation	350	mm^2a/kWh
Urban land occupation	200	m^2a/kWh
Natural land transformation	7.6	mm^2/kWh
Water depletion	170	cm^3/kWh
Metal depletion	32	g Fe eq/kWh
Fossil depletion	8.7	g oil eq/kWh

Table 4.20: Midpoint results from the ReCiPe(H) method

Embodied energy

The cumulative energy demand methodology has been observed to produce estimates of embodied energy that are up to 45 % higher than other LCIA methods (Davidsson *et al.*, 2012). In order to examine this further, a detailed analysis was carried out of the energy flows through the modelled Pelamis life cycle, using data from Ecoinvent to quantify the total energy consumption. Although no standard LCIA methodology has been applied, the use of a standard methodology is not prescribed within the international standards, provided that the methodology is reported in detail (ISO, 2006a).

The flow chart previously shown in Figure 4.23 on page 98 illustrates the flows of fuel and energy processes through the Pelamis Life Cycle. The model contains a number of 'energy conversion' processes, where the fuel source, usually measured by mass or volume, is converted to an energy value. While it would be possible to simply add these together, there are also processes within the model, such as transport, where the fuel consumption is never converted to an energy value, and therefore the results would be an underestimate.

Characterisation factors were estimated from a detailed analysis of the inventory of raw materials and processes. The first step was to examine all energy flows through the process network produced by the SimaPro model to identify the source fuels (such as coal) and processes (such as renewable electricity generation). The total available energy in these raw processes or fuels, or energy density, was then estimated from the Ecoinvent data. As each fuel or energy source

Damage category	Impact	
Climate change (human health)	3.7×10^{-8}	DALY/kWh
Ozone depletion	6.0×10^{-12}	DALY/kWh
Human toxicity	7.5×10^{-9}	DALY/kWh
Photochemical oxidant formation	8.7×10^{-12}	DALY/kWh
Particulate matter formation	2.4×10^{-8}	DALY/kWh
Ionising radiation	4.9×10^{-11}	DALY/kWh
Climate change (ecosystems)	2.1×10^{-10}	species.yr/kWh
Terrestrial acidification	9.3×10^{-13}	species.yr/kWh
Freshwater eutrophication	4.2×10^{-13}	species.yr/kWh
Terrestrial ecotoxicity	3.5×10^{-13}	species.yr/kWh
Freshwater ecotoxicity	1.3×10^{-13}	species.yr/kWh
Marine ecotoxicity	4.2×10^{-16}	species.yr/kWh
Agricultural land occupation	3.9×10^{-12}	species.yr/kWh
Urban land occupation	3.9×10^{-12}	species.yr/kWh
Natural land transformation	1.1×10^{-11}	species.yr/kWh
Metal depletion	2.3×10^{-3}	$/kWh
Fossil depletion	0.14	$/kWh

Table 4.21: Endpoint results from the ReCiPe(H/A) method

feeds multiple different energy conversion processes, often with very different energy densities, this involved taking a weighted average of the energy density using mass or volumetric data from the SimaPro process network. Correction factors, derived from the Ecoinvent data, were also applied to some fuels, particularly coal and uranium, to take into account the increase in energy density of the fuel as it is processed and refined; for example, 1 kg of hard coal burned in a German power plant produces 23.98 MJ of electricity, but 1.46 kg of raw 'coal, hard, unspecified, in ground' is required to produce the 1 kg of coal delivered to the power plant. It is, therefore, estimated that the energy density of the raw coal is 16.43 MJ/kg. This estimation process may underestimate the true energy density of the raw fuel, as it neglects any loss of fuel between extraction and energy conversion, but it is important to note that the consumption of energy at mines and processing plant will be accounted for in the mass/volume of fuel consumed. The characterisation factors applied in this study are shown in Table 4.22. Note that for biomass only the extracted energy has been included, as the majority of biomass consumption is wood as a material, rather than a fuel.

This detailed analysis found the energy intensity to be 360 kJ/kWh, corresponding to a pay-back period of 24 months (Table 4.23). 78 % of this embodied energy is associated with the manufacturing stage, while the negative values at the disposal stage are the credit associated with recycling of waste materials.

Fuel	Energy Density	
Coal, brown, in ground	8.42	MJ/kg
Coal, hard, unspecified, in ground	16.15	MJ/kg
Gas, natural, in ground	37.54	MJ/m^3
Oil, crude, in ground	44.57	MJ/kg
Peat, in ground	8.77	MJ/kg
Uranium, in ground	1.62×10^8	MJ/kg
Biogas, at storage	12.41	MJ/m^3
MOX fuel element for LWR, at nuclear fuel fabrication plant	1.14×10^8	MJ/kg
Energy Source	**Primary Energy Multiplier**	
Energy, raw	1	MJ/MJ
Blast furnace gas, burned in power plant	1	MJ/MJ
Electricity from waste, at waste incineration plant	1	MJ/MJ
Electricity, from biomass	1	MJ/MJ
Heat from waste, at waste incineration plant	1	MJ/MJ
Refinery gas, burned in flare	1	MJ/MJ
Wood chips, burned in furnace	1	MJ/MJ

Table 4.22: Characterisation factors for energy calculation

Life cycle stage	Energy Intensity (kJ/kWh)
Materials and manufacturing	280
Assembly and installation	49
Operation and maintenance	98
Decommissioning and disposal	-63
TOTAL	360

Table 4.23: Results of embodied energy analysis

Sensitivity of results to LCIA method

Although the variation in impact categories and measurement units across different LCIA methodologies makes it difficult to quantitatively assess the sensitivity of the LCA results to the chosen method, there are some instances where similar impact categories and units have been applied, allowing the results to be directly compared. Table 4.24 identifies comparable impact potentials across the given impact assessment methods.

Impact Potential	**EDIP 2003**	**EPD**	**ReCiPe Midpoint**	**EDIP 97**	**CML2**
Global warming	✓	✓	✓	✓	✓
Ozone depletion	✓	✓	✓	✓	✓
Acidification	-	✓	✓	-	✓
Photochemical oxidation	-	✓	-	✓	✓
Energy consumption	**CED**	**EPD**	**Embodied Energy**		
Energy intensity	✓	✓	✓	-	-
Damage category	**Eco-indicator 99**		**ReCiPe Endpoint**		
Climate change	✓	-	✓	-	-
Ozone depletion	✓	-	✓	-	-
Radiation	✓	-	✓	-	-

Table 4.24: Comparable results across different LCIA methods

The mean and percentage error were calculated for each of these impact potentials and the results are shown in Table 4.25. It can be seen that the global warming potential is the least sensitive to the impact assessment method, which is likely to be due to the comprehensive data published by the IPCC (IPCC, 2007). In contrast, there appears to be little consensus on the characterisation factors for photochemical oxidation potential, with results varying widely across the different methods. The results for the three damage categories that have been compared are weighted, and it can be seen that the added subjectivity of this step leads to a considerable discrepancy.

The variation in the calculated energy intensity is not as large as suggested by Davidsson *et al.* (2012), although the cumulative energy demand analysis did produce results that were 4 % and 13 % higher than those from the EPD and embodied energy methods respectively. The embodied energy analysis was re-run using the characterisation factors from the cumulative energy demand calculation, and produced results that were only 0.3 % higher. This confirms that the CED calculation is based on the fuel and energy flows through the process network, and demonstrates that the discrepancy is introduced by differences in the estimated energy density of the raw fuels.

Impact or Damage	Mean	Error	Unit
Global warming	26.6	±0.7 %	g CO_2 eq/kWh
Ozone depletion	2.24	±1.2 %	μg CFC-11 eq/kWh
Acidification	155	±4.8 %	mg SO_2 eq/kWh
Photochemical oxidation	13.5	±45 %	mg C_2H_4 eq/kWh
Energy intensity	390	±6.6 %	kJ/kWh
Climate change	2.1×10^{-8}	±74 %	DALY/kWh
Ozone depletion	4.2×10^{-12}	±44 %	DALY/kWh
Radiation	5.6×10^{-11}	±12 %	DALY/kWh

Table 4.25: Comparison of resulting impact potentials from different LCIA methods

Further examination found that the differences in estimated energy densities for coal and uranium introduced the greatest discrepancies between the results from the CED analysis and the embodied energy method: in the former the energy density of coal was 18 % higher than the values used in the embodied energy method, and the energy density of uranium was 250 % higher. It is likely that this is partially due to the assumptions made when determining the characterisation factors for the embodied energy method, particularly the assumption that no useful fuel is lost during extraction and processing; however, this does not account for all of the discrepancy, and Hischier *et al.* (2010) notes that characterisation of the energy intensity for uranium is currently disputed.

4.4.4 Recycling allocation method

As discussed in Section 4.2.4, the recycling of materials reduces the environmental impacts of a product, which appears as a recycling credit in a life cycle assessment. If the product is manufactured from recycled material that is, in turn, recycled at the end-of-life, the allocation of this credit is complicated by the potential for double-counting if it is assigned to both the waste material and the resulting product. Several recycling allocation methods have been developed to avoid this problem, and detailed descriptions of these, along with an examination of existing guidance on selecting an appropriate method, are provided in Section 2.4.2.

In order to consider the recycling credit for both the use of recycled material and end-of-life recycling, the 50:50 method was selected for the analysis presented in this chapter. This section examines the sensitivity of the results to this decision by applying two more recycling allocation methods to the Pelamis LCA: the recycled content method (RC) and the closed loop approximation method (CL). The three methods are described as follows:

- Recycled Content Method - The default SimaPro/Ecoinvent method, this uses data for the average mix of recycled and virgin input material and end-of-life recycling only results in avoiding some of the impacts of waste disposal.

- Closed Loop Approximation Method - It is assumed that all input material is virgin material, and recycling credit is applied at the waste disposal stage.
- 50:50 Method - Half of the recycled content credit is allocated to the input material, and half of the closed loop credit to the disposal of waste.

The recycled content method is the simplest to apply, as most LCI data already includes recycled content information. It is thought that this method is applied in most existing published studies, although the recycling allocation method is rarely clearly defined (see Chapter 3). The method is described by the following equation (Hammond and Jones, 2010):

$$E = (1-R)E_v + RE_R + (1-r)E_d \tag{4.2}$$

where:

E	=	Embodied impacts per unit of material
E_v	=	Embodied impact of virgin material
E_R	=	Embodied impact of recycled material
E_d	=	Embodied impact of waste disposal
R	=	Recycled content
r	=	Recycling rate at end-of-life

It can be seen that the first two terms $(1-R)E_v + RE_R$ are the embodied impacts of the mixture of virgin and recycled input material, as provided by Ecoinvent as average European data. The impact of waste disposal is described by $(1-r)E_d$, which ignores the credit or impact of recycling waste materials and only considers the impacts of disposing of the remaining waste, in this case to landfill.

The closed loop approximation method applies system expansion to include all of the recycling credit at the end-of-life, and is described in detail in Hammond and Jones (2010). In order to examine the implementation within SimaPro, the equation needs to be rearranged:

$$\begin{aligned} E &= (1-r)E_v + rE_R + (1-r)E_d \\ &= rE_R + (1-r)(E_v + E_d) \\ &= rE_R + E_v - rE_v + (1-r)E_d \\ &= E_v + r(E_R - E_v) + (1-r)E_d \end{aligned} \tag{4.3}$$

In this method no credit is given for the use of recycled content at the manufacturing stage, so the impacts of the input materials are simply E_v, and again the impacts of waste disposal to landfill are described by $(1-r)E_d$. The end-of-life recycling credit appears as the term $r(E_R - E_v)$. This will normally be negative because the embodied impact of recycling the material should be less than the embodied impact of the virgin material. (As LCA assigns positive values to pollutant emissions and environmental impacts, recycling credits will appear as negative values.) Ecoinvent (2010) provides recommendations for which processes to include to estimate $(E_R - E_v)$ for each metal.

The 50:50 method is designed to allow the benefits of both the use of recycled materials during manufacture, and the recycling of waste products at the end-of-life, to be examined. It is a combination of both of the above methods, and avoids double-counting by simply allocating 50 % of the credit for recycled content and 50 % of the credit for recycling of waste to the product. It can be described by the following equation, which is again rearranged to clarify the implementation within SimaPro (Hammond and Jones, 2010):

$$\begin{aligned} E &= \frac{1}{2}(1-R)E_v + \frac{1}{2}RE_R + \frac{1}{2}rE_R + \frac{1}{2}(1-r)E_v + (1-r)E_d \\ &= \frac{1}{2}E_v - \frac{1}{2}RE_v + \frac{1}{2}RE_R + \frac{1}{2}rE_R + \frac{1}{2}E_v - \frac{1}{2}rE_v + (1-r)E_d \\ &= (\frac{1}{2} + \frac{1}{2} - \frac{1}{2}R)E_v + \frac{1}{2}RE_R + \frac{1}{2}r(E_R - E_v) + (1-r)E_d \\ &= (1 - \frac{1}{2}R)E_v + \frac{1}{2}RE_R + \frac{1}{2}r(E_R - E_v) + (1-r)E_d \end{aligned} \tag{4.4}$$

In this case, the embodied impacts of the input material are given by $(1 - \frac{1}{2}R)E_v + \frac{1}{2}RE_R$, and the end-of-life recycling credit is half that found in Equation 4.3 and becomes $\frac{1}{2}r(E_R - E_v)$, which is implemented within SimaPro in the same way as for the closed loop approximation method. As with both other methods, the impact of waste disposal to landfill is given by $(1 - r)E_d$.

The results of the analysis with both the recycled content (RC) and closed loop (CL) methods are shown in Table 4.26, along with the 50:50 method for comparison. It can be seen that the closed loop approximation method generally gives the most optimistic results; this is because the estimated recycling rate applied in this analysis is significantly higher than the recycled content of the materials, and the CL method considers only end-of-life recycling. The exceptions to this are in resource consumption and waste categories: as the CL method assumes that all input material is virgin material, the resource consumption is increased; this increases the waste slag and ashes and decreases the radioactive waste, due to the steel being sourced from blast furnaces rather than electric arc furnaces; it also increases the bulk waste production. This suggests that omitting the impacts of waste that is recycled in the RC method may underestimate the total impacts, particularly in the waste categories.

This analysis demonstrates that the choice of recycling allocation method can have a significant effect on the results of an LCA, as illustrated in Figure 4.37. The calculated global warming potential using the CL method is 24 % lower than that calculated with the RC method, and

Impact Potential	RC	50:50	CL	
Global warming	30	27	23	g CO_2 eq/kWh
Ozone depletion	2.3	2.3	2.2	μg CFC-11 eq/kWh
Ozone formation (Vegetation)	0.41	0.39	0.37	m^2.ppm.h/kWh
Ozone formation (Human)	2.8 x10^{-5}	2.6 x10^{-5}	2.5 x10^{-5}	pers.ppm.h/kWh
Acidification	0.0029	0.0027	0.0024	m^2/kWh
Terrestrial Eutrophication	0.0053	0.0051	0.0049	m^2/kWh
Aquatic Eutrophication (N)	21	20	19	mg N/kWh
Aquatic Eutrophication (P)	9.9	8.3	6.8	mg P/kWh
Human toxicity (Air)	667	558	449	m^3/kWh
Human toxicity (Water)	1.6	1.0	0.5	m^3/kWh
Human toxicity (Soil)	0.0055	0.0049	0.0043	m^3/kWh
Ecotoxicity (Water, chronic)	10.5	9.2	7.9	m^3/kWh
Ecotoxicity (Water, acute)	2.0	1.8	1.5	m^3/kWh
Ecotoxicity (Soil, chronic)	0.0028	0.0025	0.0021	m^3/kWh
Hazardous waste	2.0	1.6	1.2	mg/kWh
Slag/ashes	3.3	3.4	3.4	mg/kWh
Bulk waste	16	17	18	g/kWh
Radioactive waste	474	429	384	μg/kWh
Resources (all)	62	78	95	mg/kWh
Energy	469	411	353	kJ/kWh

Table 4.26: Results of life cycle impact assessment with different recycling methods

the reduction in energy intensity is 25 %. The largest variation in the results is in the potential toxicity to humans via water, which is found to be 69 % lower if the CL method is applied instead of the RC method. The slag and ashes category and ozone depletion potential are the least affected by the choice of recycling allocation method.

4.4.5 Summary of findings

The results of the sensitivity analysis confirm that the necessary choices and assumptions made during this LCA did have a significant effect on the results. These practitioner decisions are summarised in Table 4.27 and compared with those for the earlier carbon and energy audit carried out by Parker *et al.* (2007). As the same primary data and estimates were used for both analyses, any variation in the results between the two studies will have been introduced by the secondary LCI data or methodological differences such as the scope and type of analysis, inclusion of capital goods, application of a cut-off criteria and recycling allocation method.

Figures 4.38 and 4.39 summarise the results of the sensitivity analysis for a selection of impact categories. It appears that, in all cases, the results are most sensitive to the uncertainty of the secondary LCI data. However, the results of the statistical analysis of the uncertainty information from Ecoinvent are not directly comparable to the simplified tests of the sensitivity to practitioner estimates, as these do not take any probability information into account but

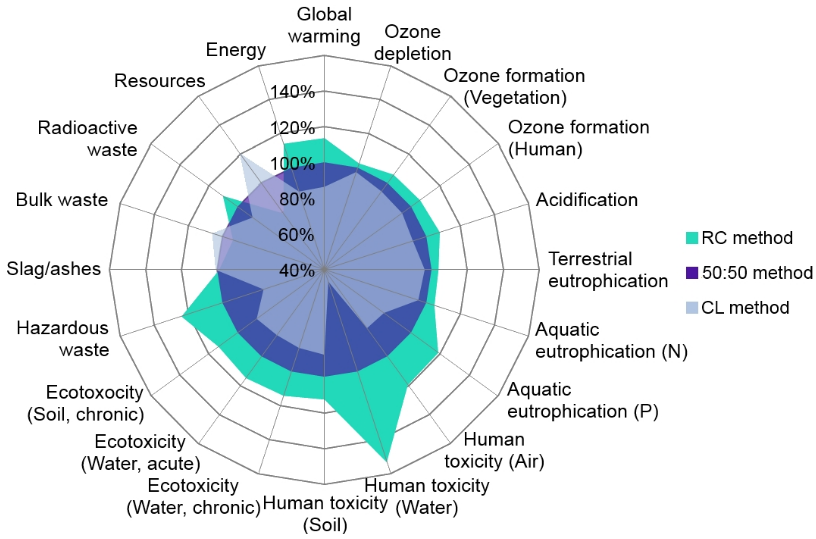


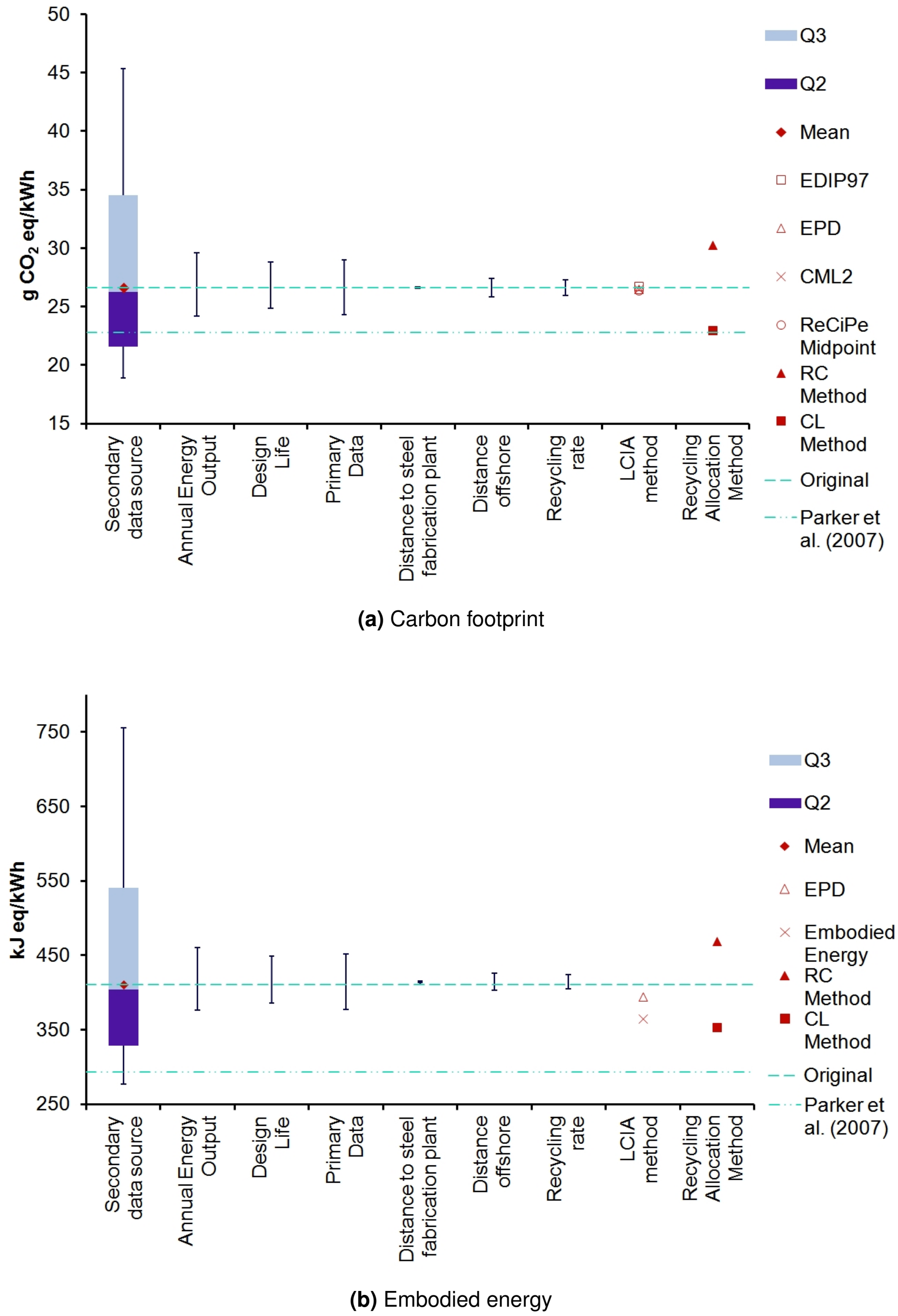
Figure 4.37: Effect of recycling method on results

simply investigate the effect of a ±10 % change; therefore, the true inaccuracies of these estimates could be much higher. Furthermore, the uncertainty of the input data is independent of the other practitioner choices tested in this sensitivity analysis, and therefore the results of a study using a different recycling allocation method, for example, would still have a similar uncertainty range.

Across all categories it was found that the sensitivity of the results to the estimated annual energy output, the accuracy of the primary input data gathered from the manufacturer and the choice of recycling allocation method were also significant; however, the relative effect of these different practitioner choices was not consistent across all impact categories. As can be seen from Figures 4.38 and 4.39, the ozone depletion and acidification potentials appear to be much less sensitive to the uncertainty of practitioner estimates and methodological choices than the GWP or energy intensity.

The sensitivity of the global warming potential and energy intensity to the uncertainty of the input data could be compared to that of the earlier Parker *et al.* (2007) study. It was found that the interquartile range from the new analysis is very similar to the sensitivity range introduced by the uncertainty of data from the Inventory of Carbon and Energy, which suggests that such uncertainty from secondary LCI data is common in LCA; as the quality of raw data improves, this uncertainty should decrease.

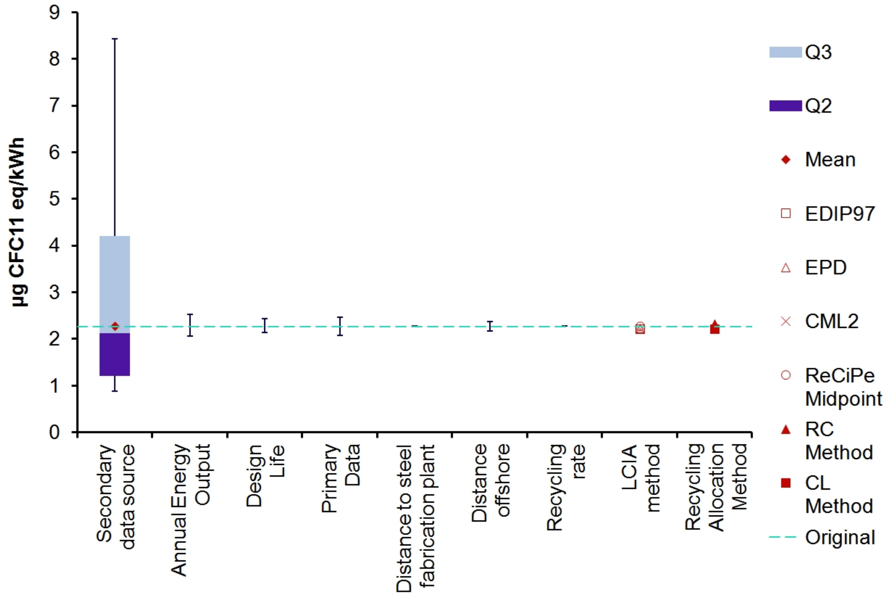
Significantly, the sensitivity analysis for the GWP indicates that the choice of recycling allocation method accounts for all of the discrepancy between the results of this analysis and Parker *et al.* (2007) (Figure 4.38a). Despite the difference in LCI data source, consideration of capital goods and cut-off criteria, and the inclusion of all greenhouse gas emissions rather than just carbon dioxide, the carbon footprint found by this analysis is the same as that found by



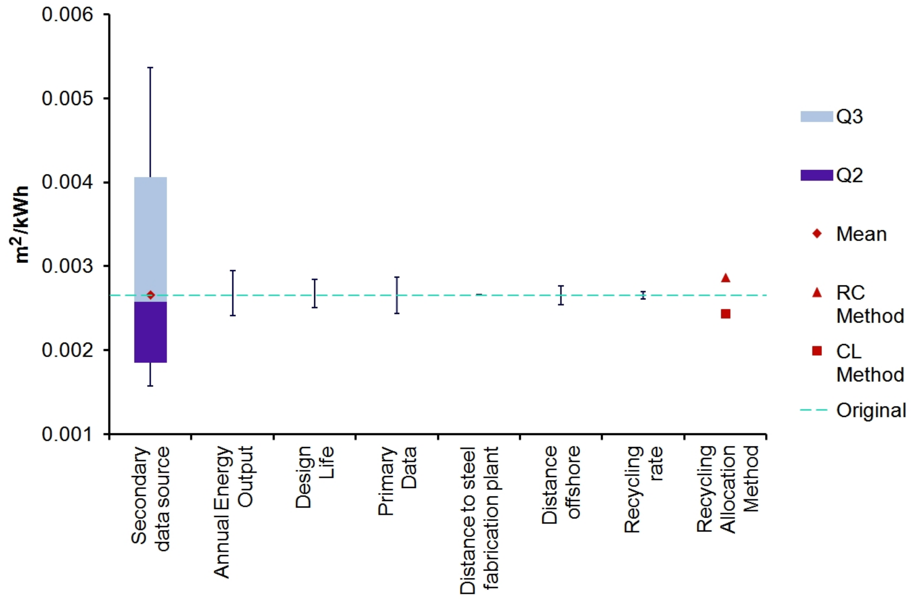
(a) Carbon footprint

(b) Embodied energy

Figure 4.38: Sensitivity of carbon and energy to practitioner decisions



(a) Ozone depletion potential



(b) Acidification potential

Figure 4.39: Sensitivity of ozone depletion and acidification to practitioner decisions

Methodology	Parker *et al.* (2007)	Current Analysis
Type of analysis	Partial process-based LCI	Process-based LCA
Life cycle stages	Cradle to grave	Cradle to grave
Waste disposal profile	90 % of steel recycled, remainder to landfill	90 % of metals recycled, remainder to landfill
Temporal boundary	20 years	20 years
Physical system boundary	Point of connection to grid	Point of connection to grid
Capital goods	Excluded	Included (Section 4.3.5)
Cut-off criteria	Unknown	10 % for pre-fabricated components
Functional unit	1 kWh output energy	1 kWh output energy
Annual energy output	2.97 GWh/year	2.97 GWh/year
Scope of analysis	CO_2 and energy only	EDIP 2003 impact categories
Primary data source	Manufacturer	Parker *et al.* (2007)
Secondary data source	Inventory of Carbon and Energy	Ecoinvent v2.2
Recycling allocation	Closed loop method	50:50 method

Table 4.27: Key practitioner choices and assumptions

Parker *et al.* (2007) when the same closed loop (CL) recycling allocation method is applied. This suggests that any increase due to the inclusion of all greenhouse gases was cancelled out by other methodological choices.

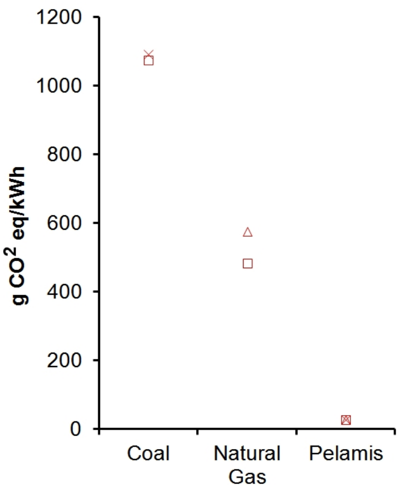
The sensitivity analysis for the energy intensity provides less conclusive results, as the application of the same closed loop recycling allocation method calculates the energy intensity to be 21 % higher than that found by Parker *et al.* (Figure 4.38b). It is likely that this remaining discrepancy is due to the consideration of upstream energy consumption and losses in the CED methodology, particularly the high estimates for the embodied energy of raw uranium (see Section 4.4.3).

4.5 Environmental Impacts of the Pelamis

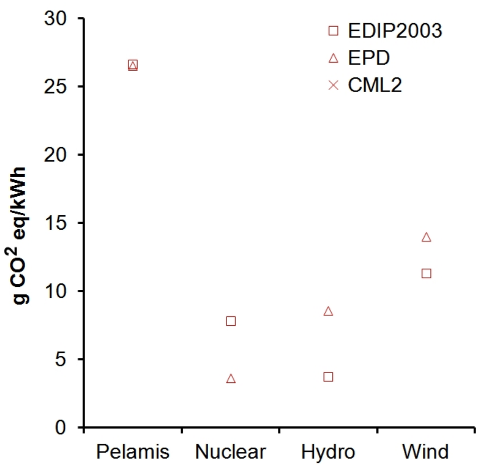
This detailed life cycle assessment provides comprehensive information about the environmental impacts of the Pelamis wave energy converter, but the value of this information lies in allowing the Pelamis to be compared with other power generation technologies, and in identifying the opportunities for reducing the environmental impacts of future models.

4.5.1 Comparison with other generators

The Pelamis is intended to be a low-carbon alternative to traditional power generation, and therefore the output energy should have a low carbon intensity while also providing a good energy return on investment. The carbon intensity of electricity from the Pelamis is compared to a range of other technologies in Figure 4.40. It can be seen that it is significantly lower than for fossil-fuelled generation but slightly higher than that for other mature low-carbon technologies, possibly due to the constraints of operating in a marine environment or the relative novelty of wave energy conversion. The potential for modifying the design of the Pelamis to reduce the environmental impacts of future models is discussed in Section 4.5.2.



(a) Fossil Fuels



(b) Low Carbon Technologies

Figure 4.40: Comparing GWP of the Pelamis with other technologies (Ecoinvent, 2010; Axpo, 2011; AEA Energy and Environment, 2008a; Vattenfall, 2011, 2013; Koornneef *et al.*, 2008)

Any viable energy generation technology must have a positive EROI. Furthermore, information about the EROI of a power generator can provide insight into its relative economic viability, without the influence of government policy and financial incentives. One of the challenges of comparing energy return on investment, is that most estimates for conventional generation include the embodied energy of the raw fuel. Figure 4.41 compares the EROI of several typical energy sources and the Pelamis. The values shown in the graph were taken from Murphy and Hall (2010), with uncertainty ranges highlighting the maximum and minimum values identified in their review. Note that the EROI for 'coal (mine-mouth)' and 'natural gas' have been adjusted for the efficiency of power generation; the assumed efficiencies were taken to be 41.5 % and 55.1 % for pulverised coal and CCGT respectively, calculated from published UK data and detailed in Section 7.2.1. These efficiencies may well be overestimates, providing optimistic EROI values for coal and gas-fired generation; however, the graph clearly shows that the EROI of the Pelamis compares well with generation from natural gas, nuclear, and wind plants.

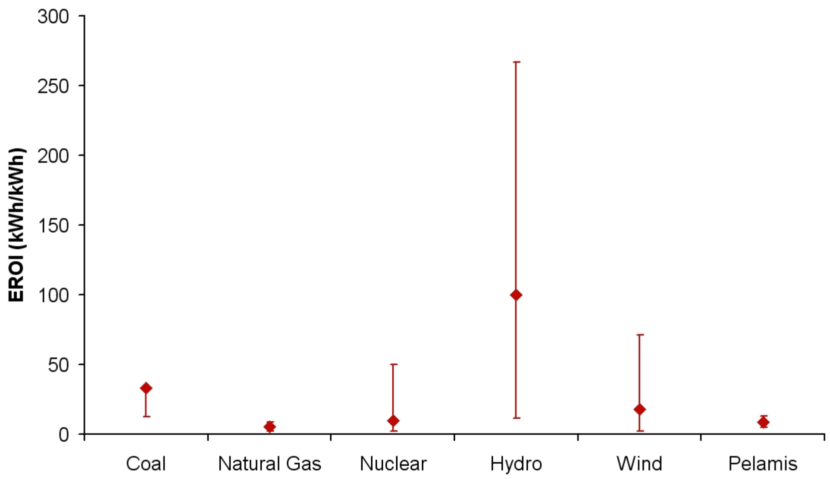
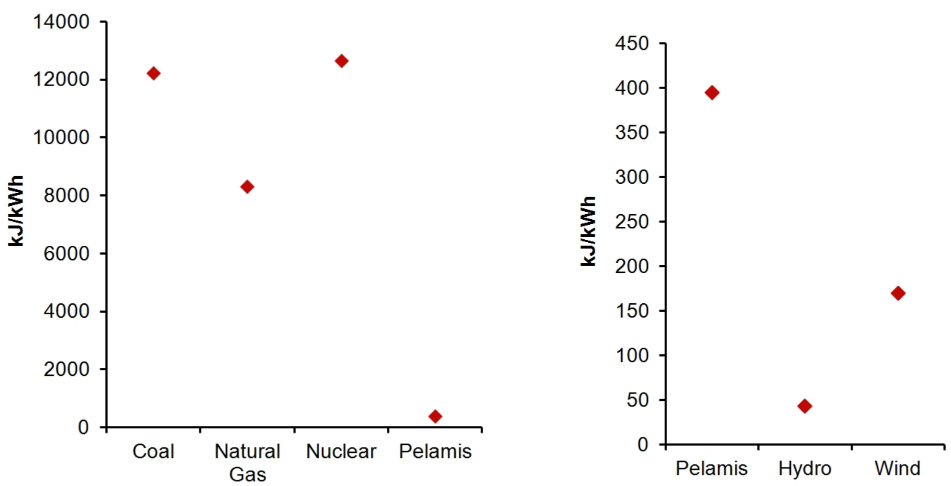


Figure 4.41: Comparison of the energy return on energy invested (Murphy and Hall, 2010)

Another benefit of a wave energy converter is that it extracts energy from a renewable resource. Figure 4.42 compares the non-renewable energy intensity of the different generation technologies (i.e. excluding all energy consumption from renewable sources). Similarly to the GWP, the energy intensity of electricity from the Pelamis was found to be significantly lower than that for conventional thermal power generation, and slightly higher than mature wind and hydro power.



(a) Conventional thermal generation

(b) Renewables

Figure 4.42: Non-renewable energy intensity for different types of generation (Ecoinvent, 2010; Axpo, 2011; AEA Energy and Environment, 2008a; Vattenfall, 2011, 2013)

In addition to having a low carbon footprint, there is an expectation that renewable energy generators will generally have much lower environmental impacts than conventional thermal power plants across all categories. Comparison of the results of this LCA with data from Ecoinvent confirms that the Pelamis has significantly lower impacts than coal and gas-fired generation in most categories, as illustrated in Figure 4.43 (Ecoinvent, 2010). Conventional nuclear generation, however, performs better than the Pelamis in some categories, such as ozone formation, eutrophication and bulk waste, although it has significantly higher impacts in others, most noticeably in radioactive waste production and ozone depletion potential.

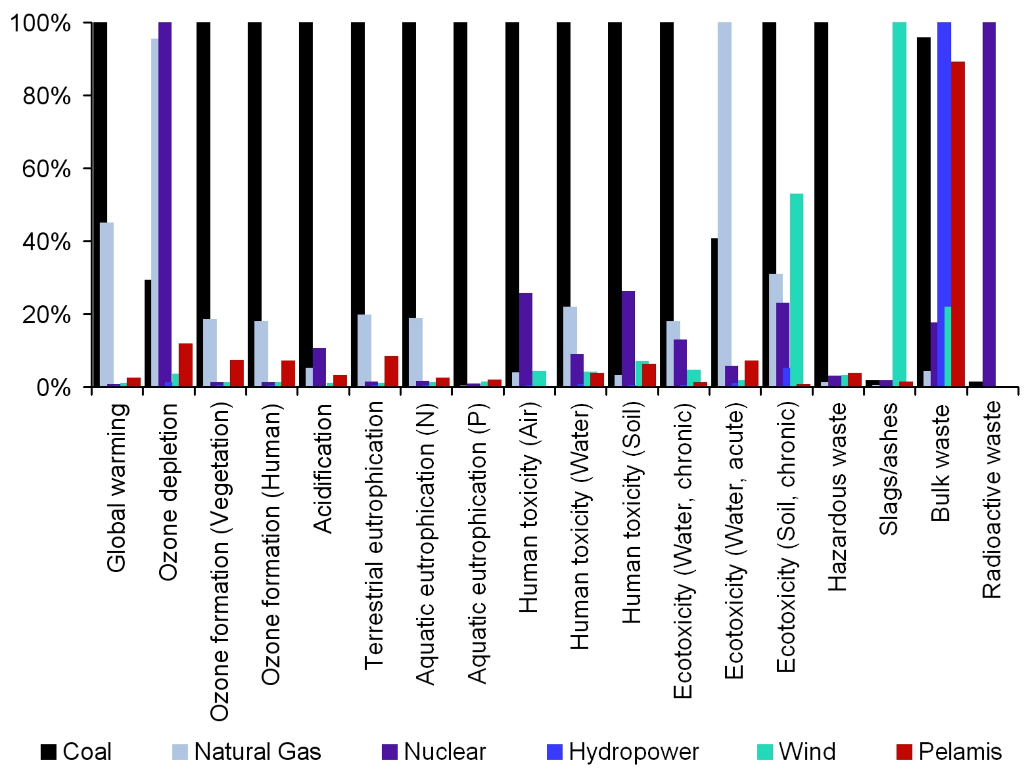


Figure 4.43: The relative impacts of different types of generation across all EDIP 2003 categories

4.5.2 Potential for improvement

The results of this comprehensive LCA highlight the life cycle stages with the most significant environmental impacts; this can be used to inform and guide future design developments. Section 4.3.2 shows that there are two life cycle stages that have high impacts across all categories: the consumption of raw materials, particularly steel; and the operation of specialist sea vessels, particularly for maintenance.

A large proportion of the total mass of the finished Pelamis P1 is the sand ballast, which has been shown to have negligible environmental impacts. Steel, however, accounts for over 99 % of the remaining mass and has significant impacts. In order to reduce the environmental impacts of the Pelamis, future design developments should consider reducing the quantity of steel within the device. Provided this does not result in an increase in consumption of other materials, it will have an environmental benefit; however, where there is an increased requirement for other materials, a further comparative analysis would be required to confirm that the benefits of reducing the use of steel are not outweighed by the increased impacts from the new material. Parker *et al.* (2007) did consider the effect on carbon footprint and energy intensity of replacing the steel in the main tube sections with concrete or glass reinforced plastic, but neither material has been selected for the latest Pelamis model, so they have not been considered in this analysis.

The environmental impacts could also be significantly reduced by simply increasing the recycled content of the steel: replacing virgin steel from blast furnaces with recycled steel from electric arc furnaces. This analysis assumed that the steel used in the Pelamis would be equivalent to the average European mix and contain 37 % recycled material. As described in Section 4.3.2, this is halved by the application of the 50:50 recycling allocation method to 18.5 %, but still accounts for only 5 % of the total global warming potential of steel production. This is because the global warming potential of steel from an electric arc furnace is only 21 % of that produced in a blast furnace, so replacing all of the steel with electric arc steel (given the constraints of the 50:50 method) would reduce the GWP to 22 g CO_2 eq/kWh, a reduction of 18 %. It can be seen in Section 4.3.2 that recycled steel has lower impacts across all impact categories, so such a change would reduce all environmental impacts, most significantly in the toxicity and acidification categories; although it may result in a slight increase to radioactive waste production if the electricity mix has a high proportion of nuclear energy.

The environmental impacts of operating specialist sea vessels are also significant within the life cycle of the Pelamis, and therefore are an area to target for improvement. Although the precise requirements for sea vessels is dependent upon the final location of the wave farm, any design developments that reduce the requirement for these, such as reducing the frequency of maintenance operations, will reduce the environmental impacts of the Pelamis.

4.6 Conclusions

Carbon footprint and embodied energy audits have their limitations: the focus on greenhouse gas emissions and energy consumption may neglect trade-offs with other environmental impacts, and the flexibility of the methodology allows practitioner choices to affect the results. The latter will have an impact on estimates of carbon and energy payback. This chapter presents a detailed life cycle assessment of the first-generation Pelamis wave energy converter, expanding an earlier carbon and energy audit carried out by Parker *et al.* (2007) to examine a broad range of environmental impacts. The chapter also includes a detailed analysis of the sensitivity of the results to key practitioner decisions, to identify those that have the most significant effect.

In accordance with the requirements of the International Standards, this analysis considered every stage of the device life cycle: from extraction of raw materials to disposal at the end of life (ISO, 2006a). All fundamental assumptions and base data were taken to be the same as those used by Parker *et al.* (2007), using data derived from Pelamis Wave Power's own records, facilitating comparison of the two studies and isolating the effects of practitioner choices. The analysis was carried out using LCI data from Ecoinvent, the 50:50 recycling allocation method, and the EDIP 2003 and CED impact assessment methods (Ecoinvent, 2010; Hammond and Jones, 2010; Hauschild and Potting, 2005; Hischier *et al.*, 2010).

A preliminary analysis suggested that the key practitioner decisions in calculating the carbon footprint and energy intensity were the LCI data source, the inclusion of primary energy multipliers and the recycling allocation method. The detailed assessment of the sensitivity of the results to practitioner decisions in the full LCA produced similar findings. The GWP was found to be 27 g CO_2 eq/kWh, 17 % higher than the carbon intensity calculated by Parker *et al.* (2007), and this discrepancy was found to be almost entirely due to the choice of recycling allocation method; significantly, the effect of differences in LCI data source, characterisation methodology and scope of carbon emissions appeared to have a minimal effect on the calculated carbon footprint. There were greater discrepancies in the calculated embodied energy intensity, which was found to be 411 kJ/kWh, 40 % higher than that calculated by Parker *et al.*, and accounted for by differences in source LCI data and energy characterisation factors, as well as the selected recycling allocation method. The uncertainty of the results due to the uncertainty of the source LCI data was found to be similar for both studies.

The detailed sensitivity analysis found that the results for all impacts were generally most sensitive to the estimated energy output of the Pelamis, the accuracy of the primary input data, and the choice of recycling allocation method. The choice of impact assessment method also affected the results, not least in that different methods often choose different units for certain impact categories, but the GWP (or carbon footprint) was found to be relatively consistent, probably due to the comprehensive data published by the IPCC (IPCC, 2007).

Comparison of the results of this LCA with those for other power generators found that the Pelamis generally has lower environmental impacts than coal and gas-fired generation, and performs similarly to or better than nuclear power generation in several categories. The carbon footprint was found to be slightly higher than that of the mature low-carbon technologies, which may be due to the relative novelty of wave energy conversion or the constraints of operating in a marine environment; however, it is significantly lower than for fossil-fuelled generation, and corresponds to a carbon payback period of 13 months (assuming it replaces the average mix of UK electricity). The energy return on energy invested was found to be similar to that of wind and coal-fired generation, with an energy payback period of 27 months.

The analysis also highlighted the areas where there could be significant potential to reduce the environmental impacts of future models. The impacts of the large quantity of steel used to form the main structure of the Pelamis are high, so any reduction in mass of steel, or an increase in recycled content, should decrease all environmental impacts. The impacts of sea vessel operations are also significant, so reductions in maintenance requirements or locating the wave farm nearer to a port would be beneficial.

This study confirms that practitioner choices can have a significant impact on the results of an LCA, including on the calculated carbon footprint; in the case of a wave energy converter largely constructed from highly-recyclable steel, the choice of recycling allocation method is very important, highlighting the need for consensus across the generation sector. The uncertainty of the secondary LCI data, sourced from databases such as Ecoinvent, also introduces considerable uncertainty to the calculated impacts, but this appears to be fairly consistent across different data sources (Ecoinvent, 2010). Uncertainties in characterisation factors, particularly for toxicity categories and primary energy consumption, also have a significant impact and can affect the comparability of different studies.

The variability of results introduced by practitioner choices and the LCA methodology will, therefore, have an impact on the estimated carbon and energy payback periods of all types of generating technologies. Their sensitivity to this variability is examined in greater detail in Chapter 8, where it is demonstrated that the carbon payback period of the Pelamis can range up to 19 months, and the energy payback period may be as much as 4 years if the worst case estimates for uncertainty of the input data are used. It is significant to note, however, that these payback periods remain well within the design life of the device. With regards to other types of variable output renewable generation, estimates of carbon and energy payback periods may have an associated uncertainty of as much as ± 30 %.

The results of this full LCA confirm that the Pelamis wave energy converter has low environmental impacts when compared to conventional power generation, particularly from fossil-fuels, while also providing a good energy return on investment.

Chapter 5

Carbon Displacement of Variable Renewable Energy - An introduction to the problem and current practices

5.1 Introduction

Any calculation of payback period has two sides: a cost and a saving; or - in the case of greenhouse gases (GHGs) - emissions and displacement. Previous chapters have concentrated on examining the reliability of carbon and energy footprints for renewable energy converters, and have demonstrated that these can be estimated with some confidence for the most established technologies. Once the lifetime energy consumption of a device is known, it is then relatively simple to calculate the energy payback period from the estimated annual energy production; however, this process is much less straightforward for carbon payback. The true GHG emissions savings arising from the use of variable renewable energy instead of other forms of power generation are poorly understood and the subject of fierce public policy debate.

Current practice in both scholarly research and policy implementation is to estimate the GHG emissions displacement as the average emissions of the whole network (Siler-Evans *et al.*, 2012). It is widely acknowledged, however, that the energy from variable renewable sources, such as wind, wave and tidal power, will actually replace the marginal generation. Ideally, the marginal emissions displacement would be found by identifying precisely which power plants respond to changes in the renewable energy production, but this marginal generation mix will vary according to changes in load at different times of day and throughout the year. Furthermore, the increased cycling of thermal generators in response to fluctuations in variable renewable energy supply may increase the carbon emissions of these generators.

This chapter examines the complexities of identifying the carbon displacement of variable renewable energy, with particular reference to the British grid: the limitations of the current practice are discussed in greater detail; the liberalised British electricity trading market is introduced; and an overview of existing research in this area is presented.

5.2 Current Practice

Estimates of the greenhouse gas emissions savings that arise from renewable power replacing other types of generation are already used to support planning applications and inform government policy. There is currently no reliable estimate for this displacement factor, so government guidance recommends using the annual average emissions of UK electricity (Defra, 2013; AEA Technology, 2005; White, 2004). This is published by the Department of Environment Food and Rural Affairs (Defra) and the Department of Energy and Climate Change (DECC), and was 460.02 g CO_2 eq/kWh for 2012 (Ricardo-AEA, 2012).

There has been considerable debate over the use of this figure. Historically, many estimates of carbon savings for renewable energy - normally published for pro-wind farm marketing - have assumed that wind power displaces coal-fired generation, resulting in several complaints to the Advertising Standards Authority (ASA, 2005, 2007a,b). In 2007 the ASA sought advice from the British electricity system operator, National Grid, who observed that the type of marginal plant being displaced by wind power would depend upon the relative prices of coal and gas, as dictated by the liberalised energy market. Significantly, they confirmed that an accurate displacement factor for wind power would lie somewhere between the emissions factors for coal- and gas-fired generation, taking into account seasonal variations throughout the year; this resulted in a recommendation from the ASA and the Committees of Advertising Practice (CAP) that the displacement estimate should be "based on up-to-date, generally accepted evidence that is representative of the current UK electricity-generating mix" (ASA, 2007b; CAP, 2013). The ASA concluded that estimating the displacement factor of wind power was highly complicated, a view supported by the GHG Protocol (WRI and WBCSD, 2007).

The current practice of assuming that the carbon displacement of renewable power generation is equal to the average grid emissions is, therefore, an approximation due to a lack of better information. The use of this figure assumes that electricity from renewable generators replaces that from all other forms of generation proportionally, which is not the case. As observed by National Grid: nuclear power stations, which can only change their output very slowly, do not react at all to changes in renewable power production; instead, renewable energy mostly replaces electricity from coal- and gas-fired power stations (ASA, 2007b). The high proportion of low-carbon nuclear energy in the UK mix significantly decreases the average carbon emissions of UK electricity, suggesting that actual emissions savings are higher than current estimates.

There are also significant efficiency penalties associated with operating coal- and gas-fired plant at reduced output - these increase the carbon emissions per unit of output energy from these generators, in turn decreasing the carbon displacement of renewable energy. Furthermore, additional reserve capacity may be required to cover the changes in renewable power output. The true carbon displacement will, therefore, depend upon decisions made within the electricity trading markets, the efficiency penalties of operating marginal power stations at lower output, and the additional emissions of any associated reserve capacity.

5.3 The British Electricity Transmission Trading Arrangements

In order to understand the complexities of determining the carbon displacement of variable renewable energy, it is necessary to understand how generators are dispatched within the constraints of the UK electricity markets. The UK power network is divided into two sections, geographically: the Northern Irish grid is connected to that of the Republic of Ireland, and electricity is traded on the Single Electricity Market for the island of Ireland; the island of Great Britain (England, Wales and Scotland) is on a separate grid, with the electricity market operating under the British Electricity Transmission Trading Arrangements (BETTA). The research presented in this thesis concentrates on the characteristics of only the British National Grid, shown in Figure 5.1.

BETTA came into effect on the 1st April 2005, uniting electricity trading for the whole of Great Britain into a single market. The resulting transmission network connects over 180 power stations to a grid consisting of over 25,000 km of overhead lines, and is operated by a single National Electricity Transmission System Operator: National Grid (Whiteford, 2011). The arrangements under BETTA allow generators to self-dispatch their plant, rather than being centrally controlled by the System Operator (SO). Bilateral trade between generators, suppliers, traders and consumers takes place across a series of markets operating on a rolling half-hourly basis. The wholesale market is divided into three sections, and is followed by a post-event settlement process, as illustrated in Figure 5.2. The forward and futures contract market and short-term bilateral market allow generators, suppliers and traders to buy and sell electricity as they wish - typically operating up to a year ahead of real time, with trading possible up to gate closure (1 hr before delivery). From gate closure until the end of the relevant half-hour settlement period the SO operates the balancing mechanism to ensure that supply and demand are continuously matched, resolve any transmission constraints and balance the system in real time. The final stage is the settlement of any cash flows arising from the balancing process and imbalances. BETTA is described in detail in the National Grid Seven Year Statement (National Grid plc, 2011).

5.3.1 Trading up to gate closure

The majority of electricity is traded in the forward and futures contract market, which typically operates far in advance of the settlement period, but can continue up until gate closure. The short-term bilateral markets (or power exchanges) operate over similar timescales, but tend to be concentrated into the last 24 hours; these are screen-based exchanges that allow fine-tuning of the trade contracts as forecasts become more accurate.

At gate closure all participants must notify the Settlement Administration Agent of their contract volumes, and provide minute-by-minute Final Physical Notifications (FPNs) to the SO. Any further trading will be administered under the balancing mechanism.

Figure 5.1: Map of electricity supply in Great Britain, from MacLeay *et al.* (2013)

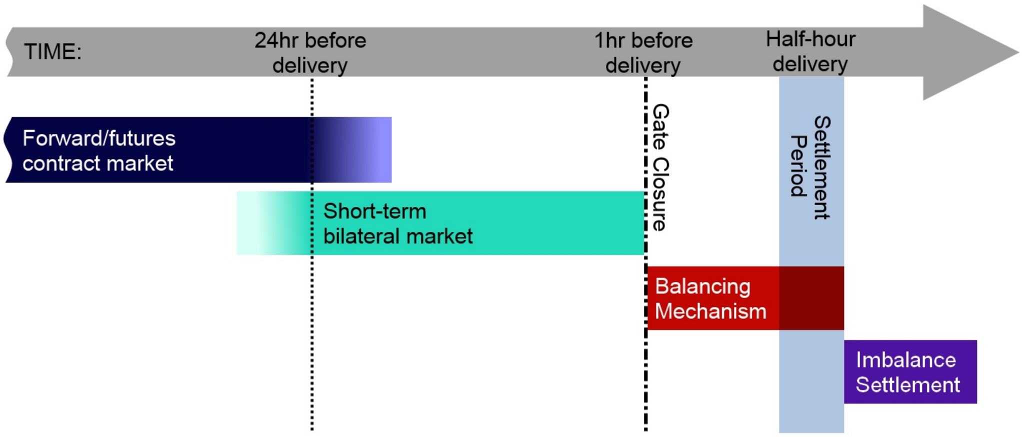


Figure 5.2: Overview of BETTA market structure, after National Grid plc (2011)

5.3.2 Balancing mechanism

The balancing mechanism (BM) is operated by the SO (National Grid) to balance any mismatches between supply and demand within the physical constraints of the system. It is guided by the Balancing and Settlement Codes (BSC), administered by a non-profit-making entity called Elexon, and operates through a system of bids and offers: values that indicate the willingness of a unit to deviate from its FPN, which are submitted to the system operator at gate closure. A bid is a price to reduce the power on the network (either by decreasing generation or increasing demand), while an offer is the price to increase power on the network (either by increasing generation or reducing demand). Up to five 'bid-offer pairs' may be submitted for levels above the FPN, and five below. When the SO requires a unit to deviate from its FPN it will issue a 'bid-offer acceptance', with a corresponding Bid-Offer Acceptance Level (BOAL).

Although only about 5 % of total system demand is traded through the balancing mechanism, it is this market that reflects the effects of short-term fluctuations in supply from variable renewable energy sources (Whiteford, 2011). The BSC requires the publication of BM data on the Balancing Mechanism Reporting Service to allow market participants access to information facilitating trade and self-dispatch, and much of this is publicly available (Elexon, 2013c).

5.3.3 Imbalances and settlements

All units must take part in the settlement process, which is run by the Settlement Administration Agent, Elexon. This takes place after the end of the settlement period and resolves any outstanding payments. Power flows are metered in real time, and any imbalances between the actual flows and the contract values set at gate closure (adjusted for any accepted bids and offers) are identified. Imbalance volumes are settled at the imbalance prices (the system sell price and system buy price), which acts as an incentive to market participants to effectively manage contractual positions before gate closure.

5.3.4 Participants

The basic unit of participation in BETTA is the 'BM unit'. This can be a single generator, an entire power station, a major consumer, or a grid supply point. All transmission, generation, supply and distribution license holders, any power stations directly connected to the transmission network, and all those capable of exporting at least 100 MW must sign up to the BSC, and will have at least one BM unit ID. Participation in the forward and futures contract market, short-term bilateral market and balancing mechanism is optional, but all BM units must provide a physical notification to the SO at gate closure and participate in the settlement process (National Grid plc, 2011). Low-capacity embedded generation will need to have an arrangement with their distribution network operator to allow appropriate notification at the relevant grid supply point.

Variable renewable energy sources, such as wind, are not usually controlled, so they will export energy whenever it is available, except when constrained by the system operator. These generators do not, therefore, take part in the balancing mechanism, instead having to accept the imbalance prices when they deviate from their forecast FPN. It is considered one of the limitations of BETTA that it penalises variable-output renewable generation in this way (Whiteford, 2011). In contrast, the increase in wind and other variable renewable energy on the network increases the short-term fluctuations of the supply - in turn increasing trading on the short-term power exchanges and the balancing mechanism, and favouring plant that can offer operational flexibility at the best price (Whiteford, 2011). Operation of the system under BETTA, therefore, makes it very difficult to predict precisely which plant will respond to changes in renewable energy production and the corresponding carbon displacement.

5.3.5 Reserve capacity and efficiency

Further ancillary services necessary for balanced grid operation are managed by National Grid outside of the framework of the balancing mechanism. This includes the contracting of any reserve capacity - such operational reserves will need to increase with increasing penetration of variable-output renewable generation on the grid. 'Spinning reserve', where the generator is already running so it can respond quickly to unplanned fluctuations in electricity supply and demand, is normally provided by operating power stations at reduced load. Such part-loading of conventional thermal plant has an efficiency penalty: at lower loads the efficiency of the generator decreases, which increases the fuel consumption per unit of energy, and thus the carbon intensity of the power generated. This will affect the carbon displacement of renewable power generation. Efficiency penalties are discussed in more detail in Chapter 7.

5.4 Existing Estimates of Marginal Emissions

Although it is common practice to estimate the emissions savings of renewable energy generation to be the average emissions of grid electricity, this provides only a rough approximation of the emissions displacement of variable-output renewable power generation (WRI and WBCSD, 2007). A number of attempts have therefore been made to quantify the *marginal emissions displacement* - the reduction in GHG emissions due to the displacement of conventional generation - on electricity networks around the world.

Marginal changes can be considered over different time frames and, for this application, three are of particular interest (Hawkes, 2010):

- Seconds to 1.5 hours ahead - short-term 'balancing' impact
- 1 hour to 1 year ahead - systematic energy trading impact
- 5 to 15 years ahead - long-term infrastructure impact

The short-term impact is the effect of unpredictable fluctuations in renewable power generation (or demand) on generation dispatch and, therefore, carbon emissions (in Great Britain managed through the BM). Forecast changes in generation from variable sources (or demand), however, will affect the planned output of generators at gate closure resulting in a systematic impact on grid carbon emissions. Longer-term changes, such as the increase in installed capacity of renewable generation, will affect grid emissions by affecting the network infrastructure, including the grid topology and the commissioning/decommissioning of other power stations.

In order to estimate the carbon payback period of a new renewable energy installation, it is necessary to quantify the short-term and systematic emissions savings per unit of energy generated, as it is normally a target for payback to be achieved within a few years. Longer-term effects of the increased penetration of such renewables on the network, along with any planned network developments, should be taken into account when considering emissions savings over the whole lifetime of the installation. Several different methods exist for estimating both the build margin (long-term infrastructure) and operating margin (short-term and systematic) emissions in a grid, with guidance and recommendations for these provided by the Greenhouse Gas Protocol (WRI and WBCSD, 2007).

A number of studies have been published that examine the marginal impacts of both demand- and supply-side changes on power network emissions. These can be divided into three categories: those that concentrate on identifying the *marginal displacement factor* (MDF) of variable-output renewable power generation; those that investigate the *marginal emissions factor* (MEF) of changes in demand (the MDF is often considered to be equivalent to this); and those that look at the long-term *carbon abatement potential* of increased penetration of variable-output renewables. Where these studies consider renewable power generation, it is generally wind power, as this is the most widely installed form of variable-output generation, with a relatively high penetration on many existing large power networks.

The MDF of variable-output renewable generation has not been extensively studied, but the work that has been carried out in this area has consistently found that the MDF is significantly different from the average emissions factor (AEF) of the corresponding network. Gil and Joos (2007) and Farhat and Ugursal (2010) applied two different methodologies to examine the marginal effects of wind generation on networks in Canada, and both concluded that the carbon displacement per unit of wind energy output would be higher than the average carbon emissions. However, in their analysis of networks in the USA, Kaffine *et al.* (2011) identified that the effect of efficiency penalties would significantly reduce the estimated MDF of wind power generation, although their work does not provide corresponding AEFs for the networks being investigated.

One study, carried out by Udo (2011), suggests that the MDF of wind energy in Ireland is significantly lower than current estimates - in the region of only a few g CO_2/kWh. This work, however, does not appear to have been peer-reviewed, and no explicit estimate of the marginal displacement of wind energy is presented. The analysis was also based on instantaneous measurements of wind generation as a proportion of total demand, and would therefore be subject to the influence of the merit-order (as observed by Hawkes (2010) - see Sections 5.4.1 and 5.4.2): as the Irish power system does not have any nuclear generators, it is likely that carbon-intensive coal-fired generation provides much of the baseload, so the carbon intensity of electricity is higher at times of low demand; the proportional contribution of wind generation is also higher at times of low demand.

The marginal emissions factor of demand-side fluctuations has been much more broadly studied for a number of networks around the world (Bettle *et al.*, 2006; Marnay *et al.*, 2002; Hawkes, 2010; Siler-Evans *et al.*, 2012). The marginal displacement of variable renewable energy may be the same as this, as it is likely that the dispatchable generators on any network will respond to fluctuations in supply as though they are negative changes in demand (Farhat and Ugursal, 2010). This assumption of equivalency, however, neglects the impacts of forecasting errors, which may be different for supply- and demand-side fluctuations, and doesn't consider the effect of increasing penetration of renewable energy on the efficiency of conventional generators. The MEF may still be a good approximation, as it is possible that these effects are negligible - one study examined the impact of forecasting errors on the MDF of wind power on the Belgian network and found that it wasn't significant (Delarue *et al.*, 2009).

The majority of published studies have found that marginal emissions factors are significantly higher than average emissions factors for the same power networks, suggesting that current estimates of emissions savings are significantly underestimated (Bettle *et al.*, 2006; Hawkes, 2010). However, two analyses found that the MEFs of demand-side changes were not consistently higher than the AEFs (Marnay *et al.*, 2002; Siler-Evans *et al.*, 2012): in a network with the majority of the baseload supplied by coal-fired generation, for example, Siler-Evans *et al.*

found the the AEF to be 35 % higher than the MEF; however, in another network with low-carbon nuclear or hydro power providing a significant proportion of baseload, the AEF was lower than the MEF by 25 %. As a reasonable proportion of baseload in Britain is provided by low-carbon nuclear power, the findings of these analyses support those of both Bettle *et al.* and Hawkes. Furthermore, both Hawkes and Voorspools and D'Haeseleer (2000a) observe that the MEF is likely to reduce over time as the most polluting power stations are decommissioned and replaced with lower carbon alternatives (see Section 5.4.4).

Many published studies are based on theoretical dispatch models that assume there is a clearly defined merit-order for dispatch of generators across the network, but this may not reflect the actual situation following market liberalisation (Bettle *et al.*, 2006; Hawkes, 2010). The treatment of power station commissioning and decommissioning also varies in the analysed marginal emissions scenarios, and few studies consider the impact on emissions of the efficiency penalties of part-loading conventional plant or providing reserve capacity. An empirical approach may provide better opportunities to consider market fluctuations and the effect of efficiency penalties.

5.4.1 Theoretical dispatch approach

Theoretical dispatch models are valuable tools to simulate the operation of a network in a range of possible current and future scenarios. These models can be used to identify the marginal generator and thus the marginal emissions. The simplest theoretical approach is to assume a merit order for the given system. Merit-order dispatch models prioritise the dispatch of generators for least cost, with the underlying assumption that the generators with the lowest operational costs are always dispatched first, followed by the more expensive generators, until the demand for the given time period is met. To illustrate the concept, a load duration curve for a day within a system operating with merit-order dispatch is shown in Figure 5.3. This stacked graph shows the proportion of a day that each type of generation will be operating - with the cheapest generation at the bottom, and the most expensive at the top. The marginal emissions curve is derived from this by identifying the emissions intensity of the next generator to be brought on-line or taken off-line. Usually such analyses assume that the emissions intensity of each generator is constant, therefore ignoring efficiency penalties and producing a discontinuous step function: in this example the marginal generator will be nuclear at times of very low demand and the marginal emissions factor will be near zero, but this will only occur for about 2 hours a day; at times of very high demand, for about 12 hours per day, the marginal generator will be oil-fired, with a marginal emissions factor of around 1 kg CO_2eq/kWh. The average marginal emissions factor over a given time period is determined from the total system loads at the times that supply- or demand-side changes took place.

Simple merit-order based analyses of marginal emissions have been published by Gil and Joos (2007); Bettle *et al.* (2006) and Marnay *et al.* (2002). The merit orders were derived

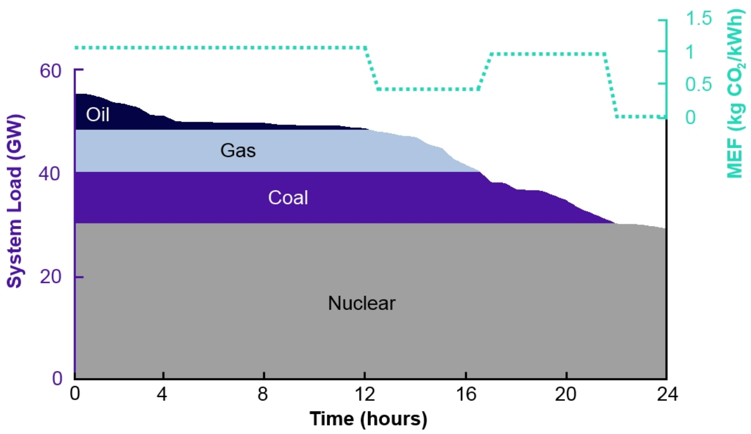


Figure 5.3: Marginal emissions and load duration curve derived from merit-order dispatch (after Hawkes (2010))

from real generator dispatch information by extracting the observed utilisation factors of the different generation technologies for the relevant networks. Unusually, Bettle *et al.* derived 8 different merit orders for the network in England and Wales, in order to take daily and seasonal variations into account. This analysis investigated the marginal emissions savings of a range of emissions reduction measures and also considered the long-term impacts of several future generation scenarios. It found that the true emissions reductions of demand-side interventions on the England and Wales grid could be as much as 50 % greater than those estimated from the AEF of the system.

The analysis carried out by Gil and Joos used a similar approach to Bettle *et al.*, combining empirical data with the merit-order approach to examine the marginal displacement of wind on the network in Ontario. Once the marginal emissions curve was identified, it was combined with a real load duration curve to derive the time-marginal emissions. Wind energy output was simulated from real wind-speed data from Kansas, USA, to calculate the weekly-average marginal displacement. This study also examined the relationship between the weekly-average displaced emissions of wind power, and the coefficient describing the correlation between the time-marginal emissions and the wind energy production, and found it to be linear. This linear relationship provides the opportunity for the MDF of wind power in Ontario to be quickly estimated for a range of different production scenarios, and also simplifies the estimation of the yearly average MDF.

Marnay *et al.*, however, used a slightly different approach to form the merit order in their analysis of the MEF of the Californian network in 1999: the empirical data was examined and baseload generation (such as nuclear power), along with ‘must take’ resources (such as wind power) were placed low down in the merit order, in order of probable dispatch; controllable thermal generation was then placed higher in the merit order and arranged according to capacity factors derived from the empirical data. The empirical data was then used to build up the load duration curve and thus identify the load-following marginal generators, calculating the marginal emissions factor from their emissions intensities.

The use of more sophisticated dispatch models avoids one of the limitations of a simple merit-order approach: that it implies that the emissions rate is a discontinuous step-function of system load. In reality, it can be observed that the logistics of plant operation, transmission constraints, plant availability and liberalised markets cause a combination of generators to respond to changes in system load, introducing much greater complexity to the emissions rate. More advanced models can simulate the output of each generator given their specific constraints, such as start-up costs and minimum up- and downtimes, thus providing more realistic estimates of the marginal generation mix.

Marnay *et al.* carried out a second analysis to identify the MEF of the Californian network in 1990, using the 'Elfin' forecasting model to simulate the system, and verifying the results against empirical data from that year. The marginal emissions factor was calculated by finding the AEF for 1990, and then re-running the model with a load decrement of 3 % to identify the corresponding change in total emissions. The 3 % load decrement was chosen following a preliminary analysis that found that the MEF was constant only for marginal changes up to 5 % of the total system load - possibly due to greater load decrements resulting in significantly different network configurations within the model, which may not truly reflect the effect of short-term marginal fluctuations. The results from the two methods were not significantly different, despite the evolution of the network over nine years, although Marnay *et al.* did observe that the MEF was highly dependent upon the assumed order of probable dispatch.

Voorspools and D'Haeseleer (2000a) also identified the MEF of particular demand-side changes by simulating the dispatch of generators for both a reference and alternative scenario, and examining the resulting change in emissions. This used the PROMIX-B tool, detailed in Voorspools and D'Haeseleer (2000b), which simulates the dispatch of generators according to cost information for the Belgian network prior to market liberalisation. Specific demand technologies with the potential to affect emissions were selected, and the effects of both promoting and prohibiting these technologies were examined by adding their demand characteristics to the simulation model. The effects of the evolution of the power system were also considered. It was found that a change in demand for any given application would not result in a change in generation from all plant equally, with the actual change in emissions affected by both supply- and demand-side fluctuations, which are unlikely to be independent.

A more recent study examined the marginal displacement of wind power on the Belgian network (Delarue *et al.*, 2009). This analysis applied an advanced unit-commitment model combined with economic dispatch, and looked at the emissions savings for a range of different load and wind profiles. The annual-average marginal displacement was estimated by comparing the simulated load and wind profiles to empirical data and taking a weighted average of the carbon savings for the closest approximations. Although this analysis did not compare the marginal displacement values to the average emissions, it examined the impact of forecast errors and found that they had little effect on the resulting MDF. It also found the marginal emissions displacement of wind power to be about 10 % lower than the possible savings of demand-side reductions as calculated by Voorspools and D'Haeseleer, although this difference may have been introduced by different assumptions in the network simulations.

In contrast to the other analyses, that published by Lund *et al.* (2010) used consequential LCA to calculate the marginal environmental impacts of demand-side changes with the test case of the Danish energy network. Firstly, hour-by-hour simulations were run in the EnergyPLAN model for several possible network configurations in 2030, in order to identify the annual average marginal contribution of each type of generator technology. Consequential LCA was then used to calculate the corresponding emissions and other environmental impacts for the given marginal technology mix.

While several different analyses have been identified that are based on theoretical dispatch models, these all share a shortcoming that they are ultimately based on a merit-order or least-cost optimisation. It is difficult to develop accurate models of liberalised energy markets, where decisions about generator dispatch may be made years in advance of the settlement period, and fluctuations in energy prices can cause unpredictable short-term changes in dispatch priorities. Despite this, however, an empirical analysis carried out by Hawkes (2010) did find a clear relationship between the MEF and system load on the British grid following market liberalisation, reflecting that the system still operates under the influence of a merit-order, although the MEF does not follow the step function that would be expected if the merit-order were the only dispatch constraint.

5.4.2 Empirical approach

Given the limitations of analysing the marginal displacement or emissions of supply- and demand-side changes using theoretical models, particularly in liberalised energy markets, more recent studies have been based on detailed empirical generation dispatch data. One of the first of these was published by Hawkes (2010), which analysed data for the British grid from 2002 to 2009. This estimated the MEF of demand-side changes from a linear regression of the relationship between changes in system CO_2 emissions and corresponding changes in demand. These values were derived from empirical half-hourly data for every generator on the British transmission network, using fixed emissions factors for each generator type to estimate the corresponding CO_2 emissions. Disaggregation of the dataset also allowed the underlying trends and characteristics to be examined with variations in total system load, time-of-day, season and date, and it was found that over very short time scales the fluctuations in MEF could be attributed to the regular scheduled dispatch of certain types of generator, such as hydro.

The methodology developed by Hawkes was also applied to a systematic analysis of MEFs for American networks (Siler-Evans *et al.*, 2012). This used detailed data from the Environmental Protection Agency (EPA) for CO_2, NOx and SO_2 emissions and gross power output for every fossil-fuelled generator greater than 25 MW. In examining eight separate regional networks it was found that the marginal CO_2 emissions factor was not always higher than the corresponding average emissions factor, particularly in regions where coal provided the base load. This analysis, however, may have been limited by the lack of data for non-fossil-fuelled generation, as the effects of any low-carbon technologies operating on the margin would not be accounted for.

Kaffine *et al.* (2011) avoided these limitations of the EPA dataset by sourcing additional empirical wind generation and temperature data for an analysis of the marginal displacement factor of wind power on three of the American networks. A reduced-form model was developed to describe the CO_2, NO_x or SO_2 emissions in terms of wind generation, average hourly temperature and other control variables. The MDF of wind power was extracted for the given empirical data by regression analysis (temperature is a key determinate of electricity demand, so was used in place of empirical demand data). The results of this study confirmed that the emissions displacement of wind power is dependent upon the displaced fuel source, which is a function of the existing network infrastructure and the installed capacity of wind on the network.

The highly detailed operational data analysed in the studies discussed so far is not available for all networks, so, in their examination of ten provincial power networks in Canada, Farhat and Ugursal (2010) instead sourced empirical information about the mix of fuel sources used for marginal generation. In order to estimate the annual average MEFs, it was assumed that the marginal generating technologies would contribute to marginal changes in the same proportions as their contribution to the average generating mix. This assumption limits the validity of the results, as merit-order theory (which has been found to hold true to some extent in a liberalised electricity market) would suggest that the ratio of baseload to marginal generation will not be the same for all different fuel types. The analysis was further extended to examine monthly/seasonal average MEFs for three of the provincial networks, where more detailed empirical data was available for the mix of fuel sources and their contribution to marginal generation. The additional detail in the measured data avoids the limitation of the methodology for calculating annual average marginal emissions rates.

While the majority of analyses based on empirical data draw the same broad conclusions on the difference between marginal and average emissions factors of power networks, they produce very different results, suggesting that analyses based on theoretical dispatch models may not truly calculate the marginal emissions of a given power network. Furthermore, the findings show that the actual marginal emissions or displacement of demand- and supply-side changes to a network are highly dependent upon the installed generating mix and relative fuel prices.

5.4.3 Efficiency penalties

As discussed in Section 5.3, conventional thermal power stations operate at a lower efficiency when generating at part load, resulting in a higher emissions intensity. The increased cycling of these power stations due to fluctuations in variable renewable energy supply will have an impact on the marginal emissions displacement. Very few existing studies consider the effect of these efficiency penalties when estimating MDF or MEF values. The empirical analysis published by Hawkes (2010), for example, estimates the CO_2 emissions of generation on the British network from average emissions intensities for each type of generator. Although these are calculated from empirical fuel consumption and emissions data, they do not capture the detail of any increase in emissions intensity attributable to marginal changes in supply or demand.

Only four studies have been identified that do consider this effect. The first of these was the analysis of the MEF of Belgian electricity published by Voorspools and D'Haeseleer (2000a). This applied a network simulation model to determine the time-varying power output of each generator and calculate the corresponding CO_2 emissions. The model considered generator efficiency at the actual instantaneous working regime in the emissions calculations, as it was identified that the specific emissions would not necessarily be constant for each generator (Voorspools and D'Haeseleer, 2000b). This study found that the marginal emissions of demand-side changes were similar in magnitude to the average emissions of the existing network (before the consideration of any long-term infrastructure impacts), demonstrating that the impact of efficiency penalties on marginal emissions are significant.

In the USA, the detailed measured emissions and power output data published by the EPA for every large fossil-fuelled generator will be affected by the actual operating efficiency of the plant, and therefore all studies based on this information implicitly consider efficiency penalties (Siler-Evans *et al.*, 2012; Kaffine *et al.*, 2011). In their analysis of the marginal displacement of wind power, Kaffine *et al.* found the MDF for one regional network to be 39 % lower than that found in an earlier study when only average plant emissions intensities were considered, again demonstrating that the consideration of efficiency penalties is important.

The study published online by Udo (2011) also considered the effect of efficiency penalties. This was based upon Eirgrid data that includes carbon emissions calculated from individual heat-rate curves, and therefore efficiency fluctuations, for each power station. As has been previously discussed, it is difficult to draw definitive conclusions from this study, but there is clear scope for further analysis to be carried out on this data. Udo does, however, highlight a limitation of applying heat-rate or efficiency curves to estimate the instantaneous emissions intensity of each power station: such analyses may not consider any degradation in the heat-rate caused by frequent ramping in response to the fluctuations of variable-output renewable power generation. This should also be considered in further research.

5.4.4 Infrastructure changes

The short-term and systematic marginal displacement of variable renewable energy can be used to improve the accuracy of carbon payback calculations and provide information on the historical impact of such technologies. When considering the impact of longer-term large-scale investment or trying to identify the lifetime emissions savings, however, the effect on emissions of changes to the infrastructure must also be considered. There are two different approaches for examining the marginal infrastructure impact: those that identify the impact of infrastructure changes on the marginal emissions of changes in supply or demand, and those that quantify the carbon abatement potential per unit increase in the penetration of variable-output renewable generation.

Some studies that examine the marginal emissions of demand- and supply-side fluctuations do consider the impact of long-term changes in infrastructure. These range in scope from simply examining historical trends (Siler-Evans *et al.*, 2012; Marnay *et al.*, 2002), to carrying out a detailed scenario analysis of future options (Farhat and Ugursal, 2010; Voorspools and D'Haeseleer, 2000a; Lund *et al.*, 2010). Siler-Evans *et al.*, for example, specifically investigated the trend in marginal CO_2 emissions over time and found little change, concluding that estimates based on recent historical data would be valid for several years. In contrast to this conclusion, however, Bettle *et al.* (2006) considered the long-term impacts of future changes to the English and Welsh generating mix and found that they could be significant. This work involved a merit-order based analysis of several scenarios for long-term infrastructure development, based on both projections published by the British Department of Trade and Industry and some potential extreme scenarios. Hawkes (2010) then built upon this work by estimating the MEF for a scenario encompassing future planned changes to the network, as published in the National Grid Seven Year Statement. As the original calculation was an analysis of detailed historical data for the British network, the future scenario appears to have been simulated by recalculating the CO_2 emissions with upgraded emissions intensities. This study also found that planned changes would significantly reduce the MEF.

The analysis published by Farhat and Ugursal (2010) was based on identifying the marginal generating mix, and therefore a range of different future scenarios could be considered by applying the same methodology. Specifically the effect of alternative demand projections, planned generator developments, and the option of replacing all existing coal capacity with hydro and gas power were examined. Again this found that the impact of infrastructure changes on marginal emissions could be significant.

The advantage of network simulation models is that they are normally developed to forecast a range of different future scenarios. Voorspools and D'Haeseleer (2000a) and Lund *et al.* (2010) both identified the impact of future changes to the network on MEFs by comparing the planned network development (used as a base case) to several alternative scenarios. These found that plans to de-carbonise networks, decommission highly polluting coal-fired generation, and increase the proportion of natural gas-fired turbines on networks would decrease marginal emissions factors. However, Lund *et al.* also found the forecast MEFs to be higher than estimates based on the assumption that the future marginal technology would be solely gas-fired generation, as the marginal demand was met by a mix of technologies.

All of the above are examinations of the effect of infrastructure changes on the MEF of demand-side changes. However, despite qualitative assessments of existing data (such as Kaffine *et al.* (2011)), and a consideration of the effect of increased wind capacity on the MEF of the Danish network (Lund *et al.*, 2010), no studies have been identified that specifically examine the effect of future network developments on the marginal displacement factor of variable-output renewables. Instead a number of analyses have been published that examine the marginal change in emissions due to an incremental increase in renewable penetration on existing networks, commonly referred to as the carbon abatement potential. These model the infrastructure impacts of an increase in renewable generation capacity and identify the marginal changes in average emissions attributable to this increase.

Analyses of the carbon abatement potential of renewable energy have been carried out for a number of networks around the world. Theoretical dispatch models are applied to simulate network operation for a range of different renewable energy penetrations and identify the marginal emissions from the corresponding change in average emissions factor. While most studies consider the impact of increased wind power penetration, other forms of renewable energy generation have also been considered, such as geothermal, concentrated solar power and photovoltaic installations (Hart and Jacobson, 2012).

The simplest studies examine only the effect of an increase in renewable energy capacity on the network, assuming that there are no other significant changes to infrastructure or demand. In one such analysis Valentino *et al.* (2012) modelled the network in Illinois, USA, for 2006, examining the effect of increasing wind generation capacity with particular reference to the efficiency penalties of both cycling and start-up/shut-down of thermal generators. This found that, despite these efficiency penalties, an increase in wind capacity would have decreased the annual carbon emissions for the year. A similar analysis was also included in the paper published by Delarue *et al.* (2009). This examined the effect of increasing installed capacity of wind power on the marginal emissions displacement, and found a linear relationship between emissions reductions and wind penetration.

In a real system, however, there will be changes to the network infrastructure over time, partially in response to the increased renewable generation capacity. Although Denny and O'Malley (2006) do not specifically examine the impact of such network developments on the carbon abatement potential of wind power on the Irish grid, their study incorporates planned changes to network infrastructure. This analysis examines the impact of accurate forecasting on the carbon abatement potential, and finds that dispatch decisions based on forecast output of wind generators results in greater emissions reductions as wind capacity is increased.

Other studies consider a range of different scenarios so that the impact of infrastructure developments can be considered. The detailed analysis published by Hart and Jacobson (2012), for example, considered a wide range of different scenarios when examining the marginal impact of an increase in penetration of renewables on the Californian network. This study compared the carbon abatement potential of several different renewable resources and control schemes, considered different portfolio combinations, and tested the sensitivity of the results to changes in demand. The findings demonstrate that a combination of different approaches will have the greatest impact on carbon emissions.

It can be seen that there is no clear consensus on the methodology for investigating the long-term impact of infrastructure changes on marginal emissions. However, existing studies consistently find that current trends towards the decarbonisation of networks, coupled with improvements in technology, will reduce both the marginal and average displacement of variable renewable energy over time. While this will increase the carbon payback period of future renewable energy installations, it shows that total greenhouse gas emissions from electricity generation are expected to decrease.

5.5 Marginal Emissions Displacement in Great Britain

As has been previously discussed, it is current practice to assume that the carbon displacement of variable-output renewable generation on the British grid is equal to the average emissions of UK electricity, despite an acknowledgement that such generation will actually only replace the output from generators operating on the margin. A number of studies of networks around the world have shown that the marginal emissions of supply- and demand-side fluctuations can differ significantly from the average emissions. In networks where the baseload generation is mostly low-carbon, which is the case in Great Britain, the marginal emissions factor has been shown to be generally higher than grid-average emissions (Bettle *et al.*, 2006; Hawkes, 2010; Siler-Evans *et al.*, 2012). However, the actual marginal emissions displacement of renewable energy will depend upon the topology of the network in which is operating, and therefore accurate values for the British situation can only be inferred from an analysis of the British grid (Kaffine *et al.*, 2011).

Two studies have attempted to identify the marginal emissions of the British grid, both examining the change in emissions due to marginal changes in demand (Bettle *et al.*, 2006; Hawkes, 2010). Although it is thought that the marginal displacement of fluctuating renewable energy supply will be the same as the marginal emissions of changes in demand, this assumption has not been tested. Furthermore, the analysis by Bettle *et al.* is based upon a theoretical order of generator dispatch, and may not, therefore, truly reflect the operation of the British network following market liberalisation. There is scope for the empirical analysis published by Hawkes to be further expanded to examine the marginal emissions displacement of variable renewable energy and identify whether it is similar to the marginal emissions of demand-side changes.

Analyses of the marginal emissions on other networks around the world have also found that the efficiency penalties associated with operating thermal generators at part load can have a significant impact on emissions reductions (Siler-Evans *et al.*, 2012). No studies have been identified that examine this effect on the British grid, however, although concerns have been raised in the media and elsewhere that the increased cycling of thermal plant will mitigate any emissions reductions attributable to renewable energy generation; therefore, there is not currently a generally accepted value for the true marginal displacement of variable renewable energy in Great Britain.

Chapter 6

Historical Analysis of the Marginal Emissions Displacement of Wind Power in Great Britain

6.1 Introduction

As has been discussed in the previous chapter, there is some debate over what generation is actually replaced by variable output from renewables, and the corresponding reduction in greenhouse gas (GHG) emissions. Large base load generators, such as nuclear power stations, do not respond to marginal fluctuations in supply or demand (Figure 6.1), and therefore the marginal changes in emissions are unlikely to be the same as the average emissions factors (AEFs). Indeed, studies on a number of networks around the world have found that the changes in GHG emissions associated with marginal changes in demand (the marginal emissions factors - MEFs) are often significantly different from the AEFs (Hawkes, 2010; Bettle *et al.*, 2006; Gil and Joos, 2007; Farhat and Ugursal, 2010). This confirms that the AEF may not be a good approximation for the carbon displacement of variable-output renewable generation, and raises questions over the validity of current carbon payback estimates.

The analysis presented in this chapter identifies the greenhouse gas displacement of variable-output renewable generation by examining the real marginal effects of wind power on the British grid (wind power is currently the only type of variable renewable energy large enough to be operationally metered and explicitly influence generator dispatch). The methodology is based upon and extends that developed by Hawkes (2010), who analysed historical data to identify the marginal emissions rate of demand, and thus evaluate the impact on CO_2 emissions of demand-side interventions. In this study, similar historical generation data is examined to identify the effect on the network GHG emissions of changes in wind power output, isolating these from any marginal effects of fluctuating demand. The resulting marginal displacement factor (MDF) of wind power is compared to the MEF of changes in total system output to identify whether the response of generator dispatch to variable renewable energy production is the same as the response to the more conventional fluctuations in demand.

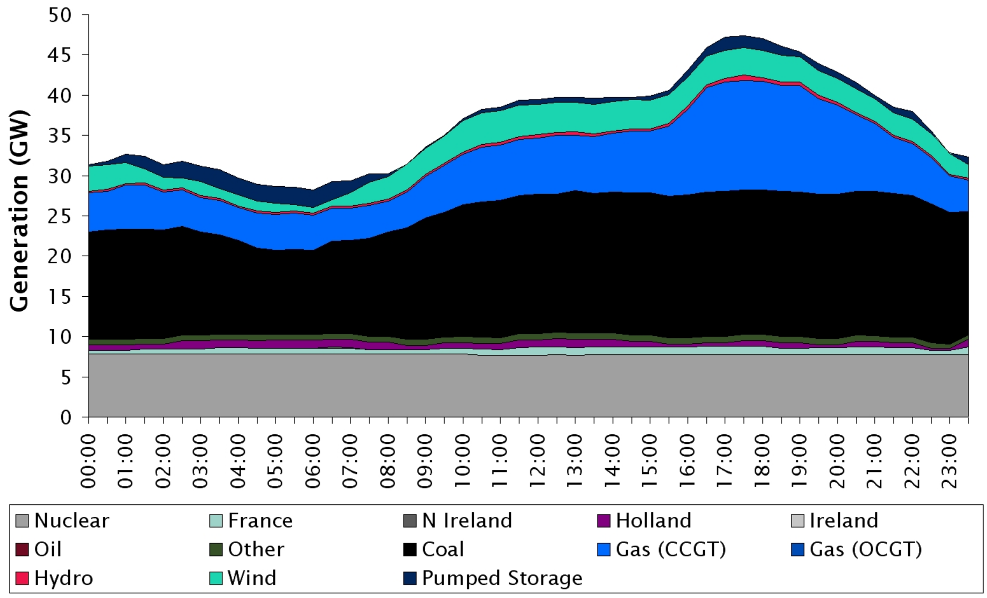


Figure 6.1: Typical winter's day on the British grid (9th December 2012) Elexon (2013a)

The specific aim of this work is to develop a real picture of the GHG emissions that are offset by variable-output renewable energy generation, and understand how these compare to the AEF of the network. The analysis focusses on the short-term and systematic impacts of such generation on carbon emissions, and does not explicitly examine the impacts of any long-term infrastructure changes, as it is based on an analysis of real historical data, rather than future projected scenarios. An examination of changes in emissions displacement over time is, however, presented, to examine whether any relationship between long-term infrastructure changes and marginal emissions displacement can be identified from this work. As this analysis is based upon data aggregated by fuel type, the effect of variable output of renewable generation on the efficiency of conventional plant is not considered; however, Chapter 7 further develops the analysis to examine detailed market data for individual generators and thus consider the effect of these efficiency penalties. This work is the first step in a process to develop more accurate estimates of the true impact on carbon emissions of variable-output renewable generation.

6.2 Analysis

6.2.1 Data sources

Operational data from the British National Grid is publicly available online (Elexon, 2013c). This is real historical information that includes data archives from the balancing mechanism (BM) dating back to the 1st January 2002 as well as historic data of generation aggregated by fuel type (Elexon, 2013a). The latter, which has been published since November 2008, is the main source of the data used in the analysis presented here. It contains operationally metered data for all large generators on the National Grid, thus avoiding the limitation of the BM data which only includes contracted production values. Although the majority of embedded generation, including all bio-energy, solar, wave and tidal power, is not operationally metered and therefore not included in the dataset, data is provided for all units reporting to the balancing mechanism. Currently the only form of variable renewable energy included in this is wind power, and approximately two-thirds of total energy from wind generation in Great Britain is supplied from operationally metered sites (Hemingway, 2012).

The analysis carried out by Hawkes (2010) mostly pre-dates the publication of this operationally metered data, so it was based upon Final Physical Notification (FPN) power levels for each BM unit. These are the contracted values at gate closure, 1 hour before the settlement period. These values may not reflect actual power outputs (see Section 5.3) and, therefore, the resulting marginal emissions factor will capture only the systematic impacts and ignore short-term marginal effects. In contrast to Hawkes' focus on demand-side fluctuations, this study examines the marginal displacement of variable-output renewable generation, which are highly subject to forecast inaccuracies, and therefore the use of FPN data could introduce a significant error (illustrated in Figure 6.2). Furthermore, wind farms tend to report FPN values as half-hourly step functions, as shown in Figure 6.3, but actual outputs fluctuate much more quickly. The response of conventional generators to these short-term fluctuations will be managed through the balancing mechanism following gate closure, and such contractual adjustments are not included in the FPN values. As the historic data of generation is based upon metered operational values, the analysis of this data will avoid these problems. A significant limitation of this operationally metered data, however, is that it is aggregated by fuel type, rather than presented per BM unit. This precludes an examination of the detailed operational impacts of fluctuations in variable renewable power output on individual generators, which may affect efficiency and the self-use of fuel. These are examined in the development of this analysis that is presented in Chapter 7, using detailed generator data extracted from the Balancing Mechanism Reporting Service.

The operational data is classified into twelve different supply types according to fuel, generator technology or international interconnector, and network operational carbon emissions are calculated from this power data by applying emissions intensities for each type of supply. As has been previously discussed, the greenhouse gas emissions of power generation are a matter of considerable debate, so average emissions intensities for each supply type were derived from historical data published by the UK government or Ecoinvent (Ecoinvent, 2010), summarised in Table 6.1 and Figure 6.4. Further details of the calculation process are given later. It can be seen that the calculated estimates vary from year to year, so overall emissions factors were approximated from the weighted average of the annual factors, based on the number of months included for each year (the analysed data runs from November 2008 to June 2013). There is

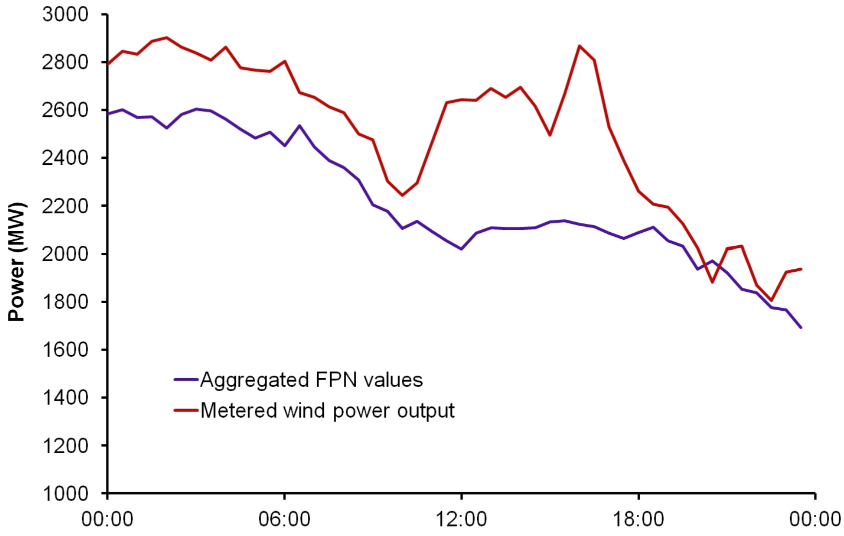


Figure 6.2: Comparing reported FPN and actual metered data for 15th June 2012

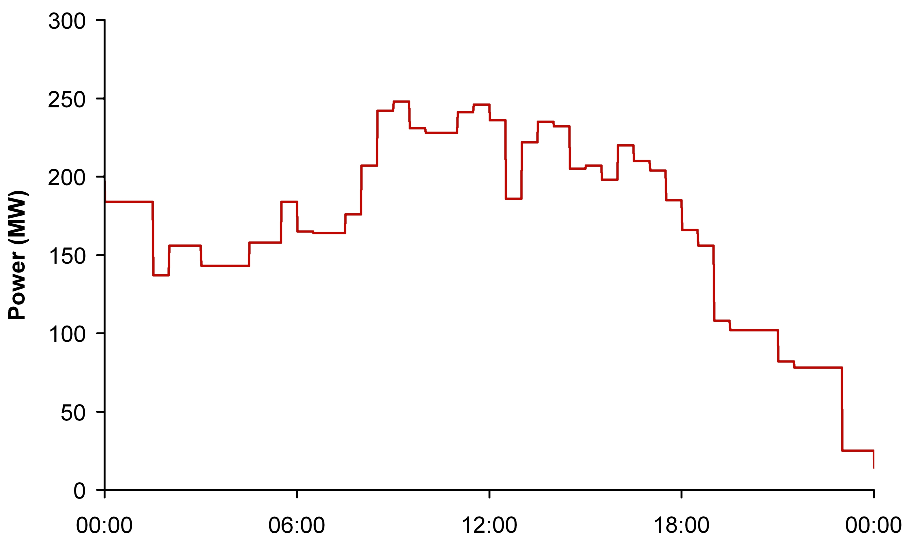


Figure 6.3: FPN data for Whitelee wind farm on 15th June 2012 (Elexon, 2013c)

still a significant disparity between the derived data and that from Ecoinvent (2010), the LCA Harmonization Project (NREL, 2013d) and Hawkes (2010), which will add uncertainty to the resulting carbon payback estimates. The published government data is, however, specific to the UK.

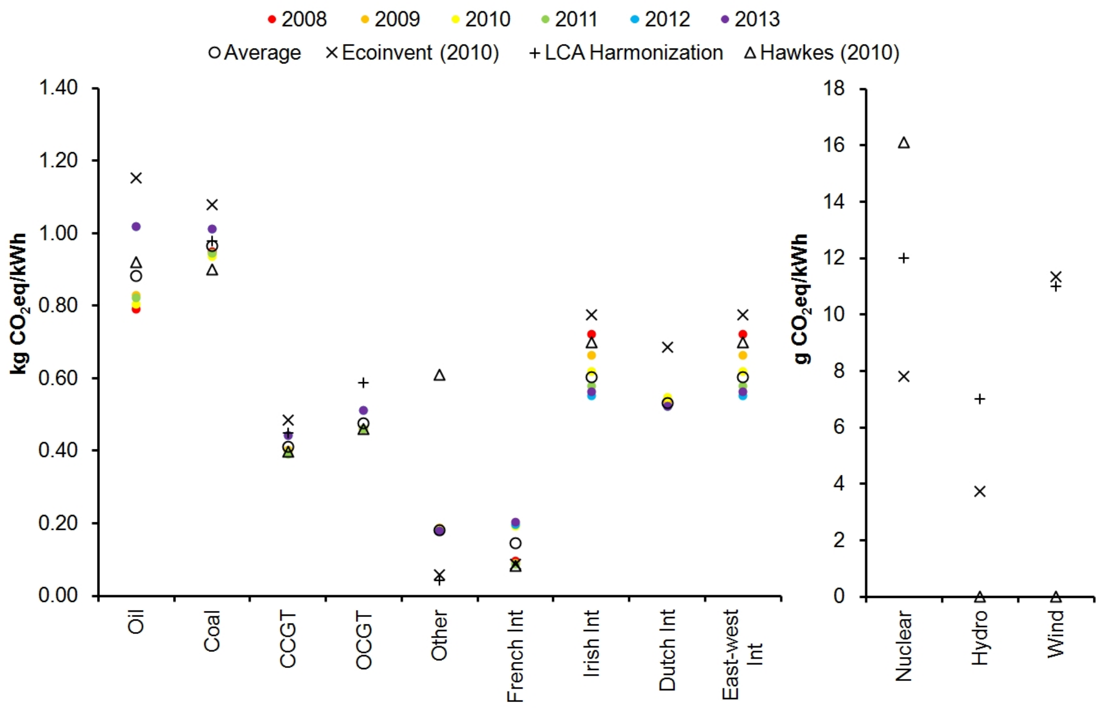


Figure 6.4: Comparison of emissions intensities (Ecoinvent, 2010; NREL, 2013d; Hawkes, 2010)

Table 6.2 shows an example of how the emissions intensities were calculated from published annual data: the majority of the information was sourced from the Digest of UK Energy Statistics (DUKES, MacLeay *et al.* (2013)) with additional information from Defra/DECC Greenhouse Gas Conversion Factors (Ricardo-AEA, 2013). Note that, unlike the analysis published by Hawkes, transmission and distribution losses are not considered in this analysis, except for imports from the international interconnectors, as this study is focussed on the emissions displacement of supply-side interventions. The analysis of the emissions intensity of energy in pumped storage plant, and more detailed information on the treatment of power flow through interconnectors, is provided in Section 6.2.3.

Supply Type	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**[1]	**Weighted Average**	**Ecoinvent**[2]	**Harmonization Project**[3]	**Hawkes (2010)**
	(Calculated with the method presented in Table 6.2)									
Nuclear	-	-	-	-	-	-	-	**0.0078**	0.0120	0.0161
Oil	0.7906	0.8297	0.8051	0.8210	1.0188	1.0188	**0.8820**	1.1514	-	0.9194
Coal	0.9492	0.9431	0.9348	0.9439	1.0124	1.0124	**0.9640**	1.0792	0.9790	0.9008
CCGT[a]	0.3991	0.4029	0.3968	0.3909	0.4429	0.4429	**0.4117**	0.4839	0.4490	0.3978
OCGT	0.4599	0.4599	0.4599	0.4599	0.5109	0.5109	**0.4763**	-	0.5880	0.4600
Other	0.1855	0.1872	0.1821	0.1797	0.1787	0.1787	**0.1817**	0.0588	0.0430	0.6100
French interconnector[4]	0.0965	0.0884	0.1923	0.0883	0.1963	0.2026	**0.1463**	0.0873	-	0.0830
Irish interconnector[4]	0.7211	0.6633	0.6195	0.5791	0.5519	0.5625	**0.6033**	0.7761	-	0.6990
Dutch interconnector[4]	0.5384	0.5374	0.5476	0.5227	0.5227	0.5227	**0.5318**	0.6859	-	-
East-west interconnector[4]	0.7211	0.6633	0.6195	0.5791	0.5519	0.5625	**0.6033**	0.7761	-	-
Hydro	-	-	-	-	-	-	-	**0.0037**	0.0070	0.0000
Wind	-	-	-	-	-	-	-	**0.0113**	0.0110	0.0000

Values used in this analysis are shown in bold

1. 2012 data applied, except for interconnector exports, which are sourced from Ricardo-AEA (2013)
2. Source: Ecoinvent database v2.2 with IPCC 2007 LCIA method (Ecoinvent, 2010)
3. Source: LCA Harmonization Project (NREL, 2013d)
4. Source: Defra/DECC GHG conversion factors 2012 (or relevant year) (Ricardo-AEA, 2011). Interconnector imports include transmission and distribution losses, and all relevant well-to-tank conversion factors. Interconnector imports are examined in more detail in Section 6.2.3.

Table 6.1: Emissions intensities of generation (kg CO_2 eq/kWh)

a. CCGT - Combined Cycle Gas Turbine. OCGT - Open Cycle Gas Turbine.

Supply Type	Emissions intensity of fuel[1] (kg CO_2 eq/kWh)	Fuel/energy use in power stations[2] (GWh)	Gross electricity supplied[2] (GWh)	Average efficiency of gross electricity supplied[3]	Average GHG content of gross electricity supplied (kg CO_2 eq/kWh)
Oil	0.30704[4]	9076	2735	30.1 %	1.0188
Coal	0.34457	399253	135888	34.0 %	1.0124
CCGT[5]	0.20435	214146	94080	46.1 %	0.4429
OCGT[5]	0.20435	-	4091	40.0 %	0.5109
Other	0.03895[6]	61471	13400	21.8 %	0.1787

1. Source: Defra/DECC GHG conversion factors 2012, Annex 1 (AEA, 2012). For all other years these have been sourced from Defra/DECC GHG conversion factors 2009, Annex 1 (Hill, 2009)
2. Source: Digest of UK Energy Statistics 2013, Table 5.6 (MacLeay *et al.*, 2013)
3. Average efficiency is defined as the GWh of electricity supplied divided by the GWh of fuel consumed.
4. Based on the assumption that all oil-fired generators consume diesel.
5. CCGT/OCGT breakdown is calculated based on an assumption of 40 % efficiency of OCGT.
6. "Other" generators are currently coal-fired stations converted to run on biomass. Data on the emissions intensity that for "wood pellets" from Defra/DECC GHG conversion factors 2012, Annex 9 (AEA, 2012), with all other data from DUKES based on "thermal renewables" (MacLeay *et al.*, 2013).

Table 6.2: GHG emissions factor calculations for 2012

6.2.2 Isolating marginal effects of wind power

The method applied in this analysis was based upon that developed by Hawkes (2010) to examine the marginal emissions of demand-side fluctuations on the British grid (the same method was later applied by Siler-Evans *et al.* (2012) for a similar analysis of networks in the USA). As the marginal emissions factor is not the carbon intensity of the demand at a fixed point in time, but instead the change in CO_2 emissions caused by a given change in demand, the method involves examining the relationship between changes in demand and corresponding changes in CO_2 emissions. In the original analysis Hawkes extracted these from the detailed empirical data, and plotted the resulting vector of change in CO_2 emissions against the vector of change in system demand, finding that the results were broadly in a straight line. Linear regression was then applied to determine the gradient of this trend, and thus the average change in emissions resulting from a unit change in demand - the marginal emissions factor. This is illustrated in Figure 6.5.

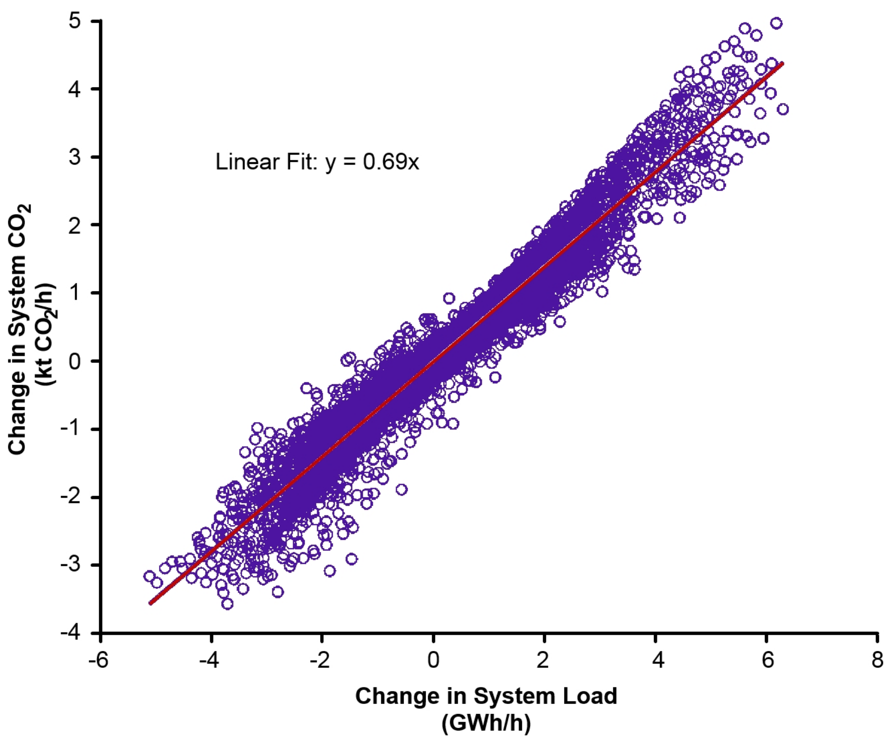
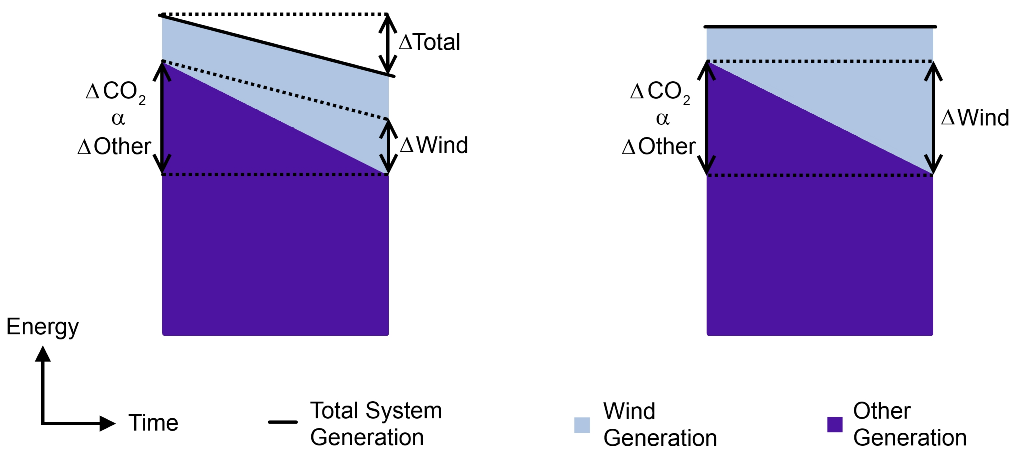


Figure 6.5: Linear relationship between changes in demand and CO_2 emissions (after Hawkes (2010))

It can be seen that there is a clear relationship between changes in demand and CO_2 emissions, but there are also many residuals, which represent the impact of other network effects, such as network constraints, weather effects, planned and unplanned outages and fluctuations in demand and wind power output. For the work presented here, the marginal effects of changes in wind generation must be isolated from the strong influence of changes in total system generation, by examining the relationship between a change in wind power output and change in emissions where the change in total generation is zero, as illustrated in Figure 6.6.



(a) Both wind and system generation changing

(b) Only wind changing

Figure 6.6: Isolating the marginal emissions displacement of wind power from the marginal effects of change in system generation

While it may be possible to identify specific data points with a negligible change in total system generation and run the marginal analysis on these, it would result in the majority of the available information being discarded. The application of multiple linear regression, however, allows the effects of both the changes in total generation and in wind power output to be considered, as described by Equation 6.1, and illustrated by plotting the values in three dimensions and fitting a planar surface (Figure 6.7). It can be seen that the relationship between the change in emissions and the change in wind output is given by the two-dimensional line where the change in total system generation is zero (Figure 6.8): the gradient of this line is the marginal displacement factor of wind power. A further benefit of this analysis method is that it also calculates the marginal emissions factor of total generation, so that the MEF and MDF can be compared.

$$\Delta C = a\Delta P_s + b\Delta P_w + c \tag{6.1}$$

where:

ΔC	=	Change in GHG emissions (t CO_2 eq/h)
ΔP_s	=	Change in total system generation (MWh/h)
ΔP_w	=	Change in wind power output (MWh/h)
a	=	Marginal emissions factor (MEF) (kg CO_2 eq/kWh)
b	=	Marginal displacement factor (MDF) (kg CO_2 eq/kWh)
c	=	Constant representing other network effects (t CO_2 eq/h)

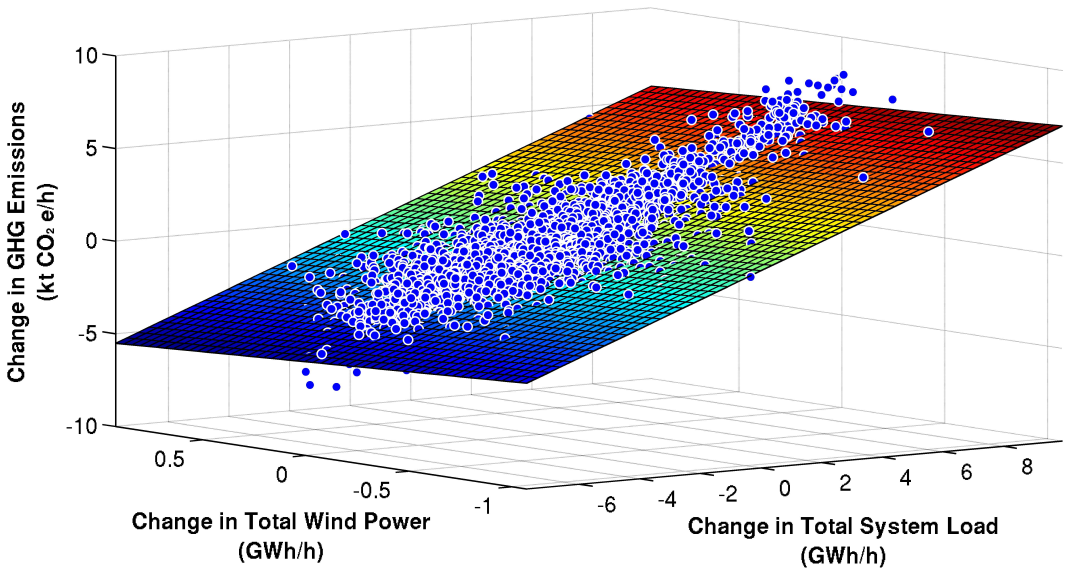


Figure 6.7: Multiple linear regression to isolate the impact of changes in wind power output from changes in total generation

6.2.3 Detailed method

The analysis presented here was carried out with Matlab (MathWorks, 2011), and the first step is to extract half-hourly data from the National Grid operational dataset. This is reported in two different forms: instantaneous power levels at 5-minute intervals, and half-hourly levels that are the average instantaneous measurements over half-hour time periods. It is published quarterly, and includes data from November 2008 to the present. In order to truly examine the short-term marginal emissions, this analysis has been based upon the instantaneous measurements; however, to reduce processing time and provide maximum comparability with data from the balancing mechanism, power levels were extracted from these measurements at half-hourly intervals. A preliminary examination of the resulting MDF has found that this lowers the

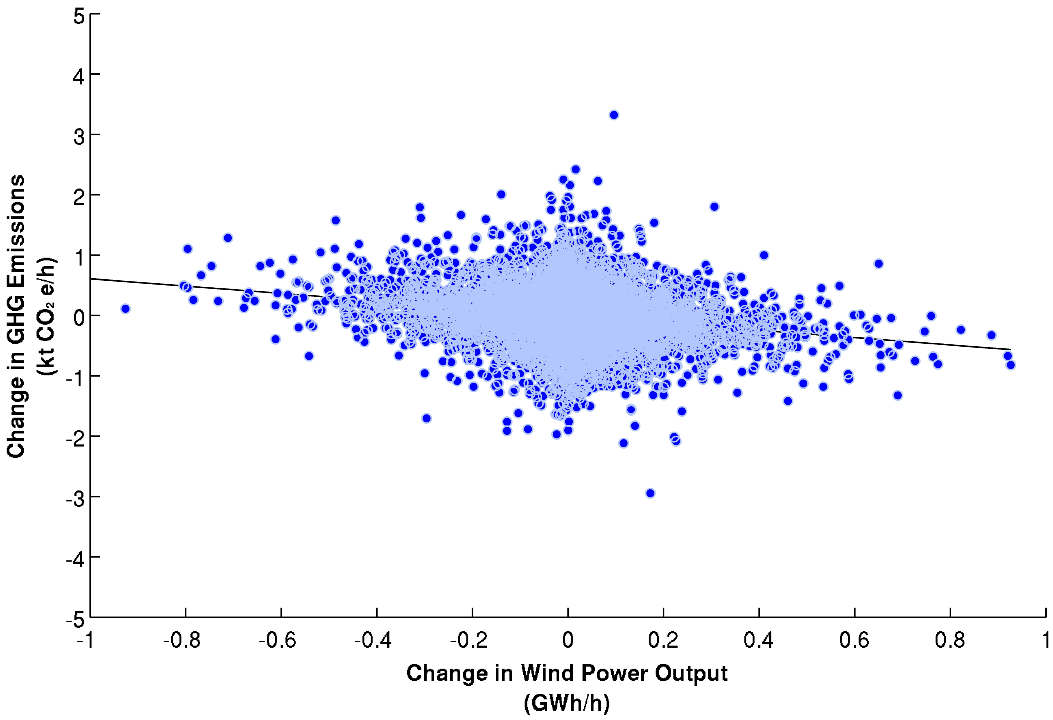


Figure 6.8: Linear relationship between change in GHG emissions and change in wind power output

estimate by 0.7 %, and introduces a slightly greater uncertainty in the results of the linear regression, which is likely to be due to the decrease in number of data points.

As discussed in the previous section, this analysis is based upon changes in the total wind power output, the total system generation and the total system greenhouse gas emissions. The raw dataset provides information on the first of these, but the total system generation and corresponding GHG emissions must be calculated for each time stamp.

Total system generation

The total system generation is not simply the sum of all power from each different type of supply reported in the historical data, as the raw data includes both international exports and energy absorption by pumped storage plant as negative values. The focus of this analysis is to examine the changes in GHG emissions due to marginal changes in power supply (both system-wide and focussed on wind power), and therefore only international imports should be considered: all exports across the interconnectors are removed from the calculation at this stage by correcting all negative values to zero. Negative values representing the consumption of power by pumped-storage hydro plants are, however, included in the calculation as a decrease in total system production because these plants simply introduce a time-delay to the consumption of the energy, rather than removing it from the system completely.

System GHG emissions

The instantaneous system GHG emissions are the sum of the GHG emissions from each type of supply, which are calculated by applying emissions intensity values from Table 6.1 to the output power levels, as described in Equation 6.2.

$$C = \sum_{i=1}^{13} e_i P_i \tag{6.2}$$

where:
C = GHG emissions (t CO_2 eq/h)
i = Integer representing the type of supply
e = Emissions intensity of given type of supply (kg CO_2 eq/kWh)
P = Power output of given type of supply (MW)

Table 6.1 does not include an estimated emissions intensity for pumped storage, as this is affected by the generation mix at the time that the energy is stored. Figure 6.9 describes the calculation process for estimating the emissions intensity for power consumption and production by pumped storage hydro at each time step: when these plants are storing energy the carbon intensity is calculated from the generation mix; and when they generate power the carbon intensity is calculated from the weighted average of all the stored emissions.

Generally, transmission and distribution losses have not been considered in the estimation of the carbon intensity of each type of supply, as this analysis is focussed on the effect of supply-side changes; however, these losses have been included in the carbon intensity of the power supplied by international interconnectors, as they will occur upstream of the connection to the British grid (Ricardo-AEA, 2011).

Half-hourly changes

In order to identify the marginal emissions factors, the changes in emissions and generation must be extracted. This is achieved by simply calculating the difference between each value and that from the previous half hour, and correcting these to report the results per hour (as shown in Equation 6.3): if the original dataset included N observations, there will be N-1 changes for each of the three variables of interest.

$$\delta C_i = 2(C_{i+1} C_i) \qquad \text{for} \quad i = 1 : N-1 \tag{6.3}$$

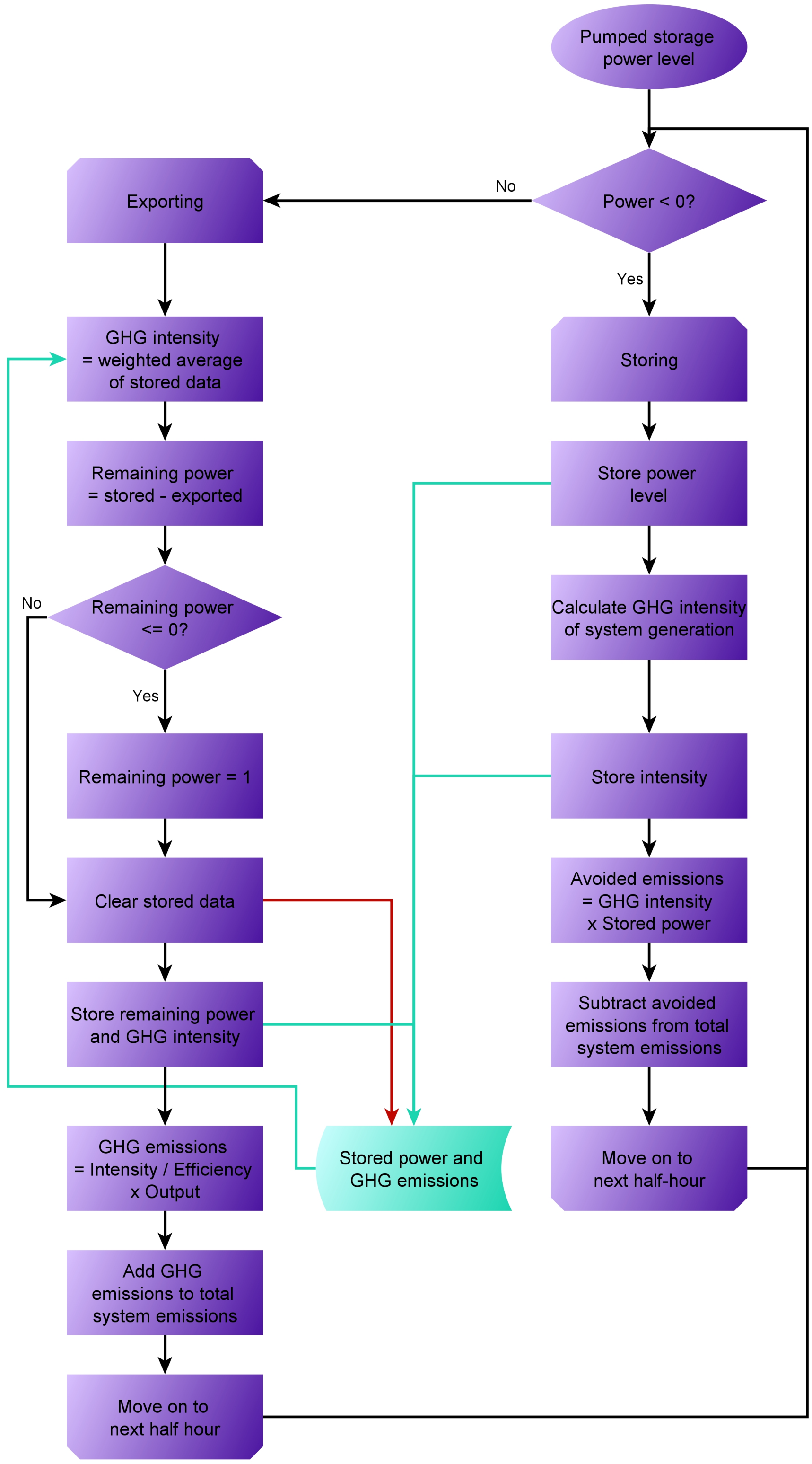


Figure 6.9: Calculation process for finding the carbon intensity of power consumed or generated at pumped storage stations

Calculation of marginal emissions factors

The final step before carrying out the multiple linear regression is to discard any outliers. As the raw dataset is not verified by National Grid or Elexon before publication, it is subject to reporting errors, particularly where no data is received for a given supply type so there is an erroneous reported power output of zero. This is most noticeable in the three largest types of generation - nuclear, coal and CCGT (combined cycle gas turbine) - where the output might be reported to drop from 5 GW to 0 GW and back to 5 GW over the course of an hour. It is highly unlikely for this to ever happen in reality, and the resulting change in GHG emissions is a significant outlier, so measurements were identified as outliers and discarded if any of the following were true:

- Nuclear, coal or CCGT power level equals zero
- Change in total system generation is greater than 12 GW
- Change in total GHG emissions or wind output is infinite

The process of discarding outliers resulted in only 0.15 % of the data being removed from the analysis, with the remainder comprising over 80,000 data points. A three-dimensional scatter graph was created from these values, and the multiple linear regression was achieved by fitting a planar surface using the linear-least squares algorithm, as illustrated in Figure 6.7. In order to reduce the influence of any remaining outliers, the robust bisquare weights method was applied, which achieved a high goodness of fit, and the 95 % confidence bounds of the results are also reported.

Sub-sets of data

Analysing the entire dataset provides a useful generalised result for the MDF of wind power and the corresponding MEF of system wide generation on the British network, but does not provide details of the underlying trends and characteristics. In order to examine these, the analysis was further refined by disaggregating the data according to several categories of interest (a process often referred to as 'binning') and re-running the linear regression on the resulting sub-set. The specific categories that were examined included the time-of-day, season, year, instantaneous wind power output and the contribution of wind power to total generation. As the changes were calculated by finding the difference between a given reported value and that from the previous half-hour, the reported time-of-day could be more accurately described as 15 minutes earlier, but any error introduced by this is minimal. In order to examine the trend between MDF and instantaneous wind power output or wind contribution, the average power level or contribution was found for the two time stamps included in the change calculation, and the bins were defined evenly across the minimum to maximum values.

6.3 Results

Figure 6.10 shows the planar fit for the analysis of the entire dataset, covering the period from November 2008 to the end of June 2013[1]. This is a close fit, with a coefficient of determination (R^2 value) of 0.96. This analysis found the marginal displacement factor of wind power to be 0.628 kg CO_2 eq/kWh $\pm$ 0.017, with the marginal emissions factor of total system generation being 3 % higher at 0.648 kg CO_2 eq/kWh $\pm$ 0.001. This small difference suggests that the system does respond to fluctuations in wind power output similarly to fluctuations in demand, so the MEF of the system may be a good approximation for the marginal displacement of variable-output renewables.

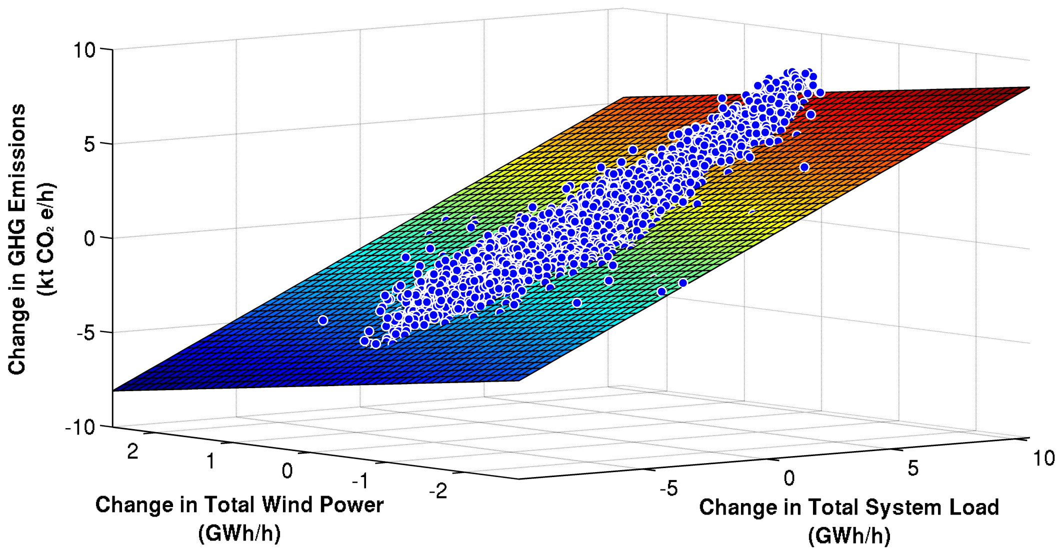


Figure 6.10: Relationship between changes in GHG emissions, system generation and wind power output (data from November 2008 to June 2013)

The system-average emissions rate for this same data, calculated from the total emissions and total system generation over the entire time period, was 0.510 kg CO_2 eq/kWh. This confirms that the marginal changes in wind power output do displace the more carbon-intensive generation, such as coal and CCGT: from the fixed carbon intensities of these types of generation (and assuming that wind power only displaces these), the result suggests that the displaced generation mix was approximately 60 % CCGT and 40 % coal-fired generation for this time period.

1. Note that the wind output and total system generation are measured in GWh/h. This is due to the historical data from National Grid being reported as an instantaneous power output, rather than a measure of energy. The calculated instantaneous GHG emissions are therefore measured in kt CO_2 eq/h, and the unit of measurement for the gradients will be kt CO_2 eq/GWh, or kg CO_2 eq/kWh.

The relationship between the changes in wind power output and GHG emissions is shown in Figure 6.11. It can be seen that the linear trend is not as clearly defined as that between GHG emissions and total system generation, which is likely to be due to the small proportion of wind power on the network resulting in smaller half-hourly fluctuations. The negative gradient, however, confirms that an increase in wind power output will result in a decrease in GHG emissions. There is also a significant increase in the magnitude of the residuals as the change in wind power output approaches zero, which represent changes in GHG emissions caused by other network effects, such as scheduled ramping of generators, unplanned outages and network constraints.

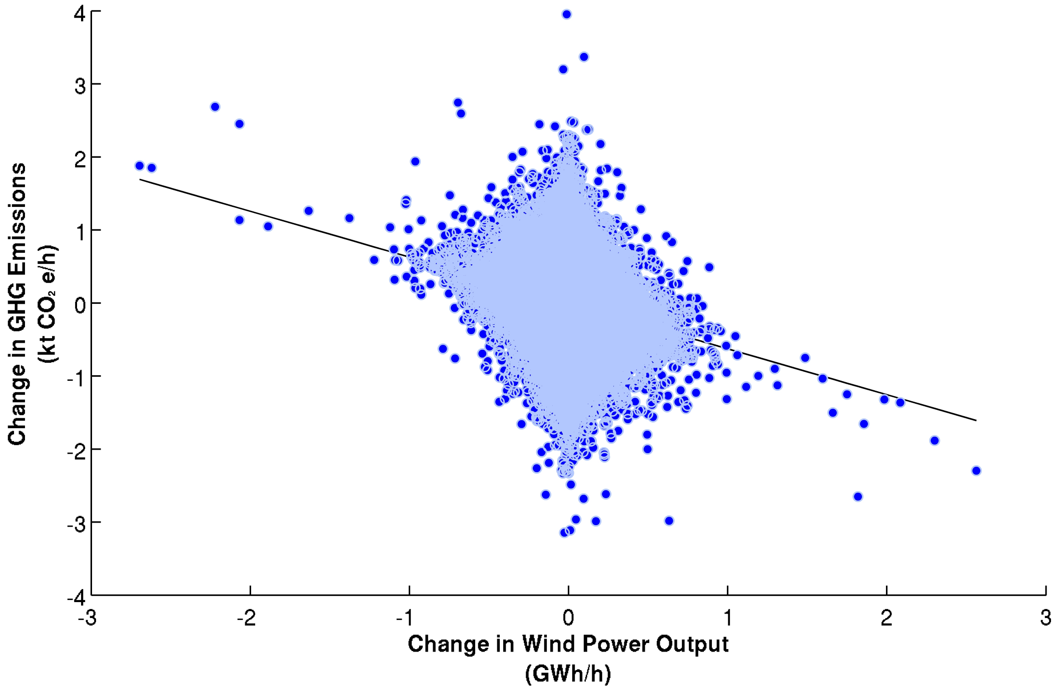


Figure 6.11: Relationship between change in wind power and change in GHG emissions

The method applied in this analysis also calculates the marginal emissions factor of changes in total system generation, which were found to be very similar to the marginal emissions of demand-side changes in the analysis by Hawkes (2010), despite being based on measured output data from a later period. The system marginal emissions factor was found to be only 6 % lower than Hawkes' estimate, which may be attributable to changes in the generation mix caused by price fluctuations favouring higher-carbon fuels for base load generation and thus increasing the proportion of lower-carbon fuels at the operating margin. As with the study carried out by Hawkes, this work found both the marginal emissions factor of generation and the marginal displacement factor of wind power to be significantly higher than the value of 0.46 kg CO_2 eq/kWh recommended by the UK government for calculation of carbon emissions savings (Ricardo-AEA, 2012). This suggests that current calculated carbon payback times for variable renewables may be significant overestimates; however, it is important to note that this analysis does not take into account the impact of efficiency penalties on the emissions of conventional plant and thus marginal displacement factors, which is addressed in greater detail in Chapter 7.

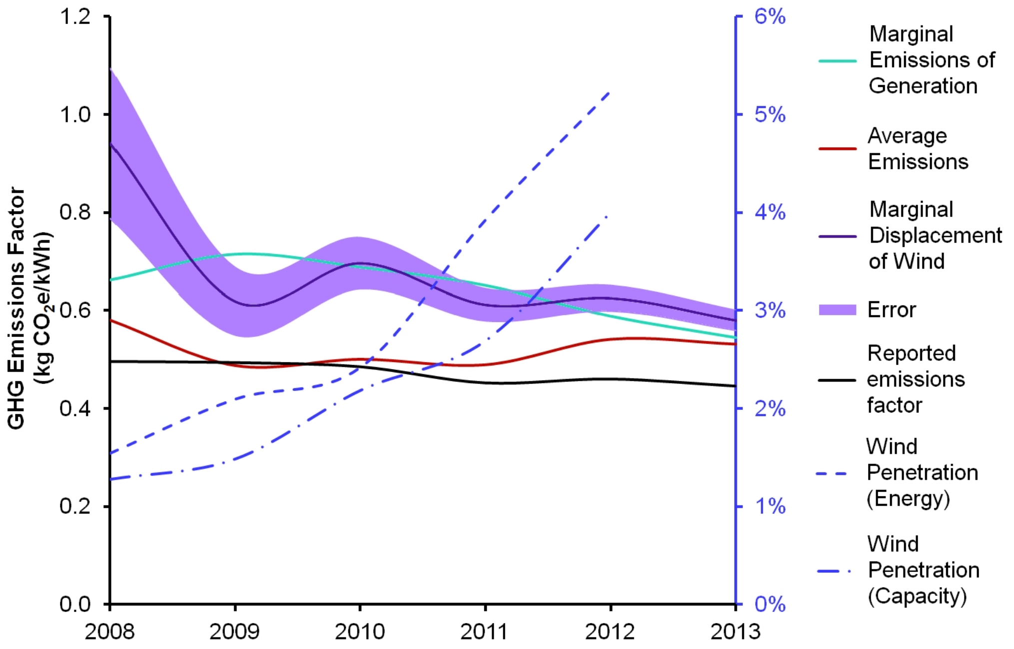
6.3.1 Trends over time

While the generalised result presented in the previous section is of interest, it does not provide insight into the change in marginal emissions displacement over time. Furthermore, it is current practice to estimate carbon payback period of renewable generation using the average emissions of UK electricity published annually by Defra/DECC (Ricardo-AEA (2013), see Chapter 5). The analysis was, therefore, re-run with the dataset split by year, as shown in Figure 6.12. It can be seen that the general trend is a decrease in both the marginal displacement factor of wind and the marginal emissions factor of generation. In contrast, both the calculated system-average emissions rate, and the reported emissions factor of UK electricity have decreased at a much lower rate.

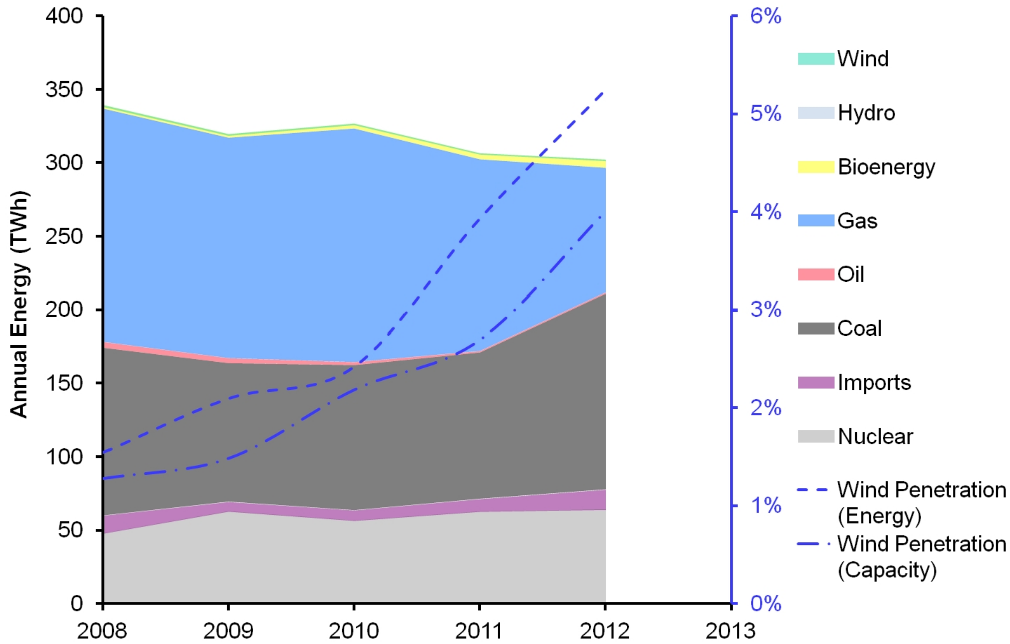
The general decrease in both marginal and average emissions may be due to the increasing proportion of wind generation on the network, which is also shown on the graph. It can be seen that there has been a significant increase in wind penetration since 2008 (the wind penetration on the network has been calculated from both the gross energy supplied by major UK power producers, as reported in Table 5.6 of MacLeay *et al.* (2013), and from the installed capacity in Great Britain, published in Table 5.8, as these values are slightly different due to reported capacity figures including some de-rating using estimated capacity factors).

In order to investigate whether the trend has been influenced by the underlying generation mix, the gross energy supplied from all major power producers is also shown in Figure 6.12. Significantly, it can be seen that there has been an increase in high-carbon coal-fired generation in recent years, and a corresponding decrease in gas-fired generation (which has a lower carbon intensity). This suggests that the price of coal has fallen relative to natural gas, which will increase the proportion of coal-fired generation on the network and result in a slight increase in the system-average emissions. Merit-order theory suggests that such a price change would, however, be expected to result in a decrease in marginal emissions, as the marginal generator will tend to be the more expensive. This is reflected in both of the calculated marginal emissions factors, supporting the findings of Hawkes (2010) that, even in an opaque liberalised energy market, the merit order still has some influence.

Figure 6.12 also shows significant fluctuations in the marginal displacement factor of wind from year to year, particularly at lower penetrations. In order to investigate this more fully, the raw dataset was disaggregated further and a marginal analysis was carried out on monthly data, with the results shown in Figure 6.13. It can be seen that the magnitude of the fluctuations and the corresponding uncertainty of the results have decreased as the installed capacity of



(a) Marginal emissions factors



(b) Generation mix

Figure 6.12: Examining the relationship between annual trends in marginal emissions and generation mix

wind has increased. This may be due to the calculation methodology: it is based upon changes in wind output and corresponding changes in GHG emissions - as the installed capacity increases, the changes in wind power generation will be larger, allowing a clearer relationship between changes in wind output and GHG emissions to be observed. It is also likely that, as the installed capacity of wind farms increases, the accuracy of forecasts are improved and the fluctuations in wind power output are more distributed across the transmission network, allowing for greater consistency in the scheduled response from conventional generation. While the MEF of changes in system generation does not follow the same fluctuations as the MDF of wind power, it shows the same general trends suggesting that it may be a good approximation for the marginal displacement of variable-output renewable generation.

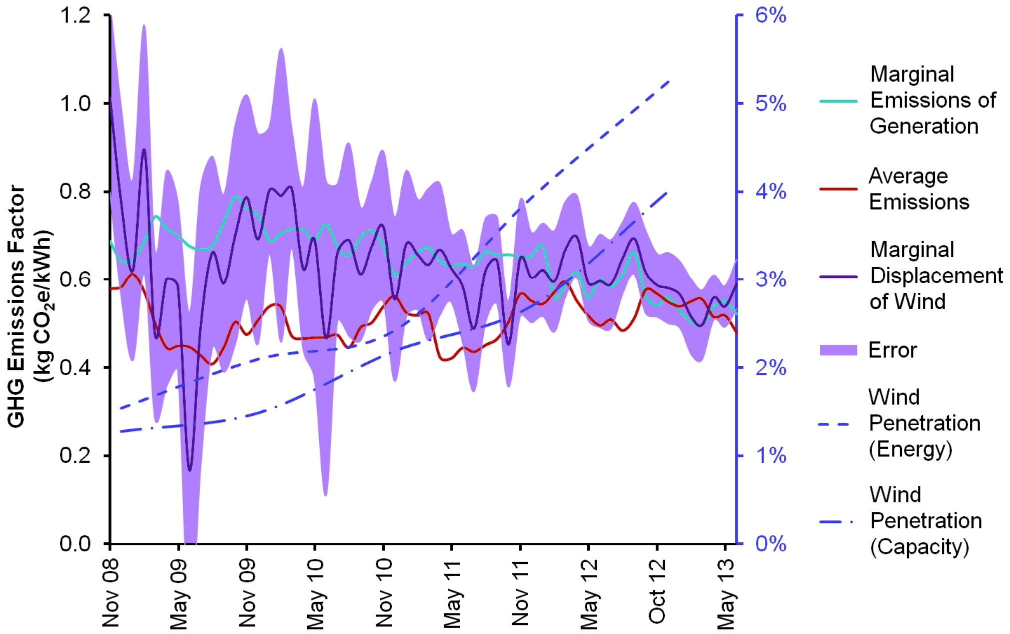


Figure 6.13: Monthly fluctuations in calculated marginal emissions

This annual analysis also provides the opportunity to compare the results of this study with those of Hawkes (2010) for 2009, where the two analyses overlap. Hawkes found the marginal emissions factor of demand to be approximately 0.7 kg CO_2 eq/kWh in 2009, while this study calculated the marginal emissions factor of total system generation to be 0.716 kg CO_2 eq/kWh for the same year. The similarity of these results suggests that the relationship between fluctuations in total supply or demand and greenhouse gas emissions is strong enough to hold despite different raw datasets and carbon intensity assumptions.

6.3.2 Seasonal trends

In order to examine whether the fluctuations observed in the monthly analysis might be attributed to seasonal fluctuations, the influence of time of year was examined by binning the data points according to the month for all years, and calculating the emissions factors for each bin, with the results shown in Figure 6.14. The maximum system generation for each bin is also shown as an indicator of seasonal fluctuations in demand.

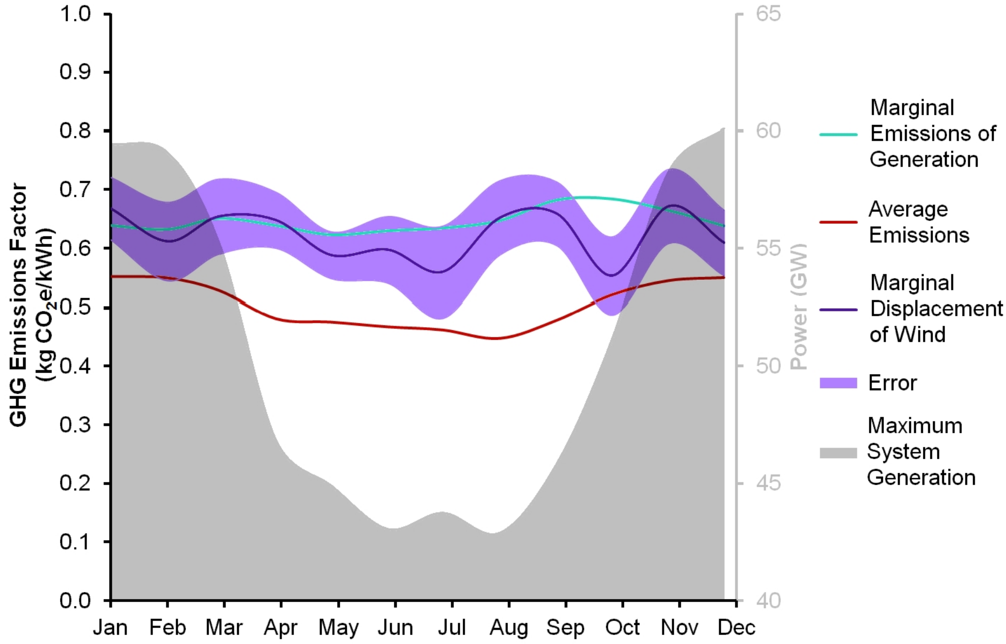


Figure 6.14: Seasonal fluctuations in calculated marginal emissions - mean values shown for each month from all 4 years 8 months of data

This analysis found no clear seasonal trend in the marginal displacement factor of wind power; however, a further examination of the monthly data found that there was some correlation with the maximum wind power output, as illustrated in Figure 6.15. This is examined further in Section 6.3.4. Figure 6.14 also shows that there was no seasonal trend in the marginal emissions factor of system generation, in contrast to the findings of Hawkes (2010) which indicated that it would be higher in the summer months. This discrepancy may be due to the analyses being based on different years, with recent changes in fuel prices significantly affecting the dispatch of conventional generation. Also, the MEF of system generation follows the same general trend as the MDF of wind power but with fewer fluctuations, suggesting that it might provide a better approximation for the emissions displacement of variable renewable energy than the system-average emissions, which are consistently lower.

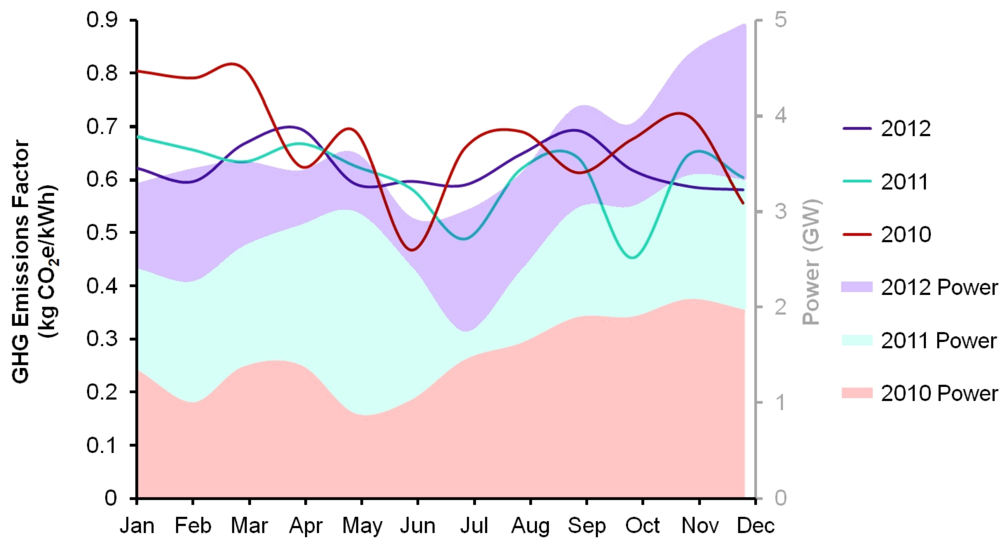


Figure 6.15: Relationship between fluctuations in marginal displacement factor and maximum wind power output

6.3.3 Time of day

A further disaggregation of the results involved investigating how the marginal emissions fluctuate with time of day, examining the findings for each half-hourly settlement period averaged over the entire dataset (Figure 6.16). It can be seen that, even when averaged over many years, both the marginal displacement factor of wind power and the marginal emissions factor of system generation are highly variable between settlement periods. The maximum system generation for each settlement period is also shown in Figure 6.16, and it appears that the highest MEFs coincide with the times of day when system generation is changing most rapidly. Furthermore, both the MDF and MEF appear to decrease at times of higher system output in direct contrast to the findings of Hawkes (2010), and again suggesting that recent changes in the relative prices of coal and gas have had a significant effect on marginal emissions factors.

Again the marginal emissions factor of system generation and the marginal displacement factor of wind power are more closely correlated than the system-average emissions, with the former fluctuating less significantly from month-to-month.

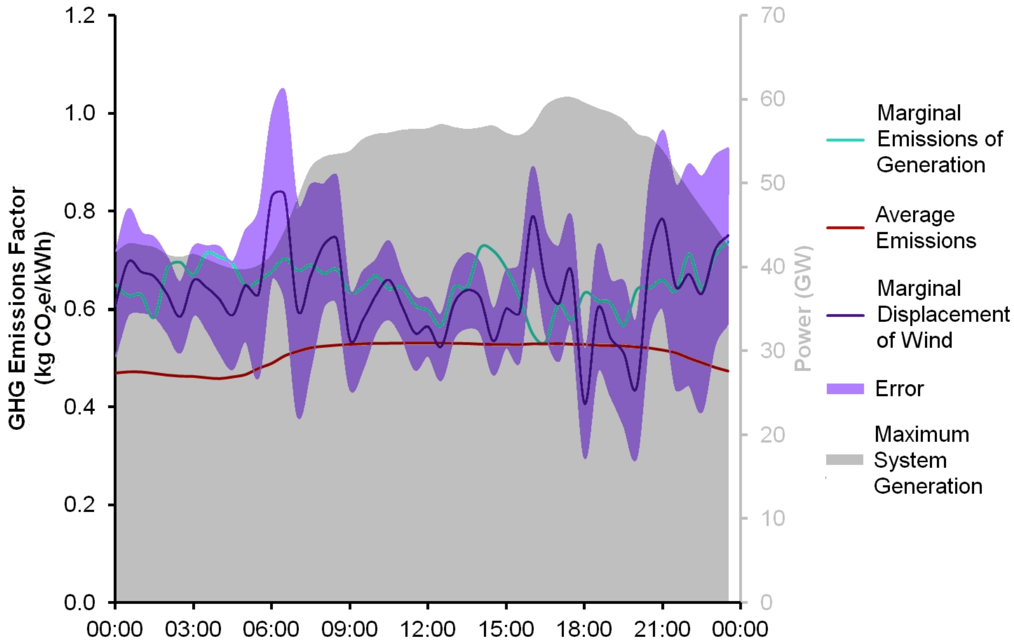


Figure 6.16: Marginal and average emissions as a function of time of day

6.3.4 Wind output level

In the seasonal analysis it was observed that there was some correlation between the calculated marginal displacement factor of wind power and the instantaneous power output. In order to investigate this further, the data was binned according to wind power output and marginal analyses were carried out with the results summarised in Figure 6.17. It can be seen that the uncertainty in the calculated MDF decreases with increased wind production, which is likely to be due to the greater influence of fluctuations in wind power on generator dispatch at higher outputs. Furthermore, both the MDF and MEF decrease when wind power production is higher; again suggesting that coal is before CCGT in the merit order. These results also suggest that the marginal emissions factor of total generation may be a more accurate estimate for the marginal displacement factor of wind power than the calculated results, as the greater number of data points significantly reduces the fluctuations, and both sets of results follow very similar trends.

One question that is of particular interest to policy makers is the effect on emissions of increased penetration of wind on the network. This has been examined in terms of annual trends in Section 6.3.1, but showed that both the average and marginal emissions are more strongly influenced by the fuel mix of conventional generation dictated by fuel prices. In order to identify what impact the increase in wind power production has had on the system, a further analysis was carried out, binning the data according to the instantaneous wind power output as a proportion of total system generation. The results of this analysis are shown in Figure 6.18. As can be expected, there is considerable uncertainty in the marginal displacement factor of

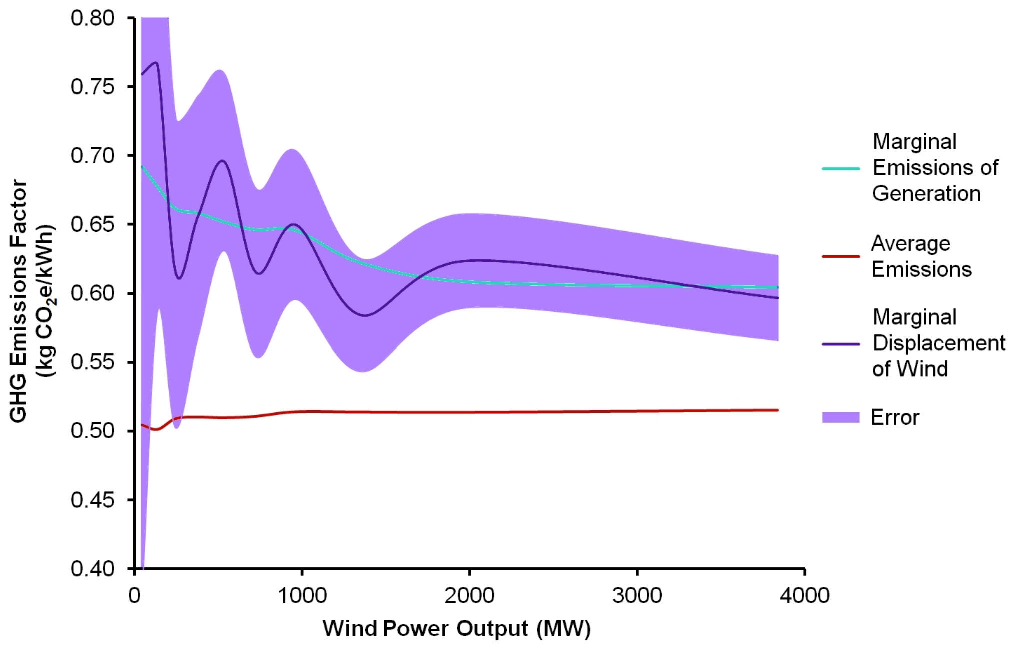


Figure 6.17: Relationship between marginal/average emissions and instantaneous wind power output

wind power when the contribution is low, but as it increases there is a definite trend for both the MDF and MEF to increase. Significantly, the calculated system-average emissions decrease when wind provides a higher proportion of total system generation, suggesting that increased wind power production does decrease system emissions. This analysis is, however, limited by the same problems as that published by Udo (2011): both of these results could be explained by the fact that higher contributions from wind are likely to occur at times of lower demand, when nuclear power provides a higher proportion of total generation and high-carbon coal-fired generation is likely to be the marginal generator. The results are different to those found by Udo due to the presence or absence of nuclear generators on the British and Irish networks.

6.4 Conclusions

In order to accurately estimate the carbon payback period of variable-output renewable generation, the true GHG emissions displacement of such technologies must be identified. Current payback calculations in Great Britain assume that this is equal to the average annual emissions of the entire system; however, the actual displacement is likely to be higher than this, as low-carbon nuclear generation does not respond to marginal changes in renewable energy supply. The analysis presented in this chapter has examined real historical operational data from the British electricity grid to estimate the marginal displacement factor of wind power

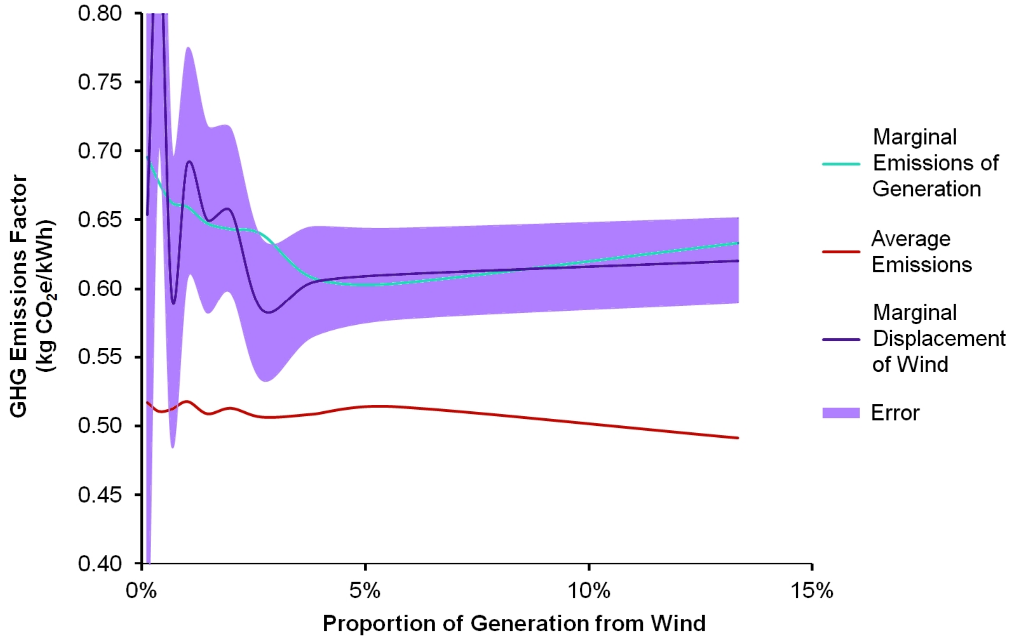


Figure 6.18: Relationship between marginal and average emissions and wind generation as a proportion of total system output

and understand how it compares to the system-average emissions, as well as the marginal emissions factor of changes in total system generation. Wind power is the only form of variable-output renewable generation currently operationally metered, and it is thought that the effects on generator dispatch, and thus marginal emissions, will be similar for other variable-output renewable technologies.

The average marginal displacement factor of wind power for the period from November 2008 to June 2013 was found to be 0.63 kg CO_2 eq/kWh, slightly lower than the marginal emissions factor of total generation at 0.65 kg CO_2 eq/kWh, but higher than the system-average emissions rate of 0.51 kg CO_2 eq/kWh. This shows that wind power is offsetting a mixture of generation, likely to be primarily coal and CCGT.

The data was disaggregated in several ways to investigate any annual, seasonal or hourly trends, and any relationship with wind power output. All of these analyses found that both the MDF and MEF were consistently higher than the calculated system-average emissions. Furthermore, the annual analysis demonstrated that the marginal emissions factors were also higher than the corresponding annual average emissions published by the UK government (Ricardo-AEA, 2013). This suggests that current estimates of carbon payback periods for variable-output renewable generators may be significant overestimates.

The trends showed that both the marginal displacement factor of wind power and the marginal emissions factor of total generation are highly influenced by the generation mix. Recent decreases in these marginal rates correspond to an increase in coal-fired power generation and a decrease in energy from gas - this is likely to be due to relative changes in coal and gas prices leading to an increase in lower-carbon CCGT operating on the margin. This supports the findings of Hawkes (2010) that the merit-order concept still holds true to some extent in the liberalised energy market.

The analysis also shows that the uncertainty of the calculated MDF is decreasing as the installed capacity of wind on the grid increases. This might be partly attributable to the calculation methodology, but is also likely to reflect the impact of improved forecasting accuracy and the technical benefits of variable-output generation being distributed throughout the electricity grid. Significantly, no clear trend was found in the seasonal or daily analysis of the marginal displacement factor, with results fluctuating considerably even when averaged across the entire dataset; instead a much clearer relationship was found between marginal displacement and instantaneous wind power output, with both the MDF and MEF decreasing at higher outputs.

The marginal emissions factor of total system generation was generally found to be a good approximation for the marginal displacement factor of wind power, with differences of up to $\pm$ 10 % for recent years. This is of particular relevance at times of low wind power output when the MDF is subject to significant calculation errors. Furthermore, despite differences in the base data and carbon intensity estimates, the calculated annual average MEF for 2009 was very similar to that found by Hawkes (2010) for the same year.

6.4.1 Further work

The analysis presented in this chapter is the first step in a process to identify a robust and reliable figure for the marginal displacement of variable-output renewable generation on the British grid. One significant limitation of this work, however, is that it does not consider the effect of efficiency penalties on the emissions intensity of conventional generators: power stations operating at a reduced output, either in response to an increase in renewable power generation or to provide reserve capacity, will be operating at a lower efficiency, which increases the fuel consumption per unit of energy and thus the carbon emissions. This question is addressed in the analysis presented in Chapter 7.

There are also significant limitations in the historical data that is analysed in this Chapter: firstly, it only includes operationally metered data, and therefore most generation embedded in the distribution network, including all existing bio-energy, solar, wave and tidal power and a lot of wind power, is not considered; secondly, the data only includes information for power stations actually exporting electricity, and therefore may ignore the emissions associated with warming power stations prior to grid connection. Improved data collection from generators and network operators would be necessary to expand the historical analysis to consider any impacts these might have on marginal emissions.

This work is a historical analysis of the marginal emissions displacement of renewable power generation since November 2008. In order to inform network development, planned decommissioning of conventional plant, and the design of new renewable energy installations, a forecasting model needs to be developed. The findings of this analysis suggest that the marginal emissions of the grid are highly dependent upon generator dispatch as influenced by fuel prices, so such a model could also examine the impact on GHG emissions of alternative dispatch priorities.

Chapter 7

The Effect of Efficiency Penalties on the Marginal Displacement of Wind Power

7.1 Introduction

As has been discussed in previous chapters, the carbon payback period of renewable generators is currently estimated by assuming that they offset the average emissions of UK electricity, but this is an approximation due to a lack of better information (see Chapter 5). Recent analyses, including that presented in Chapter 6, have attempted to provide more accurate estimates by examining historical data to identify the marginal change in greenhouse gas emissions attributable to marginal changes in demand or wind power output. However, such analyses of the British grid are significantly limited by the lack of detailed historical emissions data for generation, with the GHG emissions instead estimated from power output data. This requires assumptions to be made about the carbon intensity of different types of generation, which are a matter of considerable debate (see Chapter 3). Furthermore, such values are unlikely to be constant for every generator: part-loading of fossil-fuelled power stations has an efficiency penalty that will increase the fuel consumption and thus carbon emissions per unit of energy generated. Both the analysis presented in Chapter 6 and that carried out by Hawkes (2010) have assumed that the carbon intensity of power generation can be represented by a set of constants, finding that the marginal emissions factor is considerably higher than the average emissions. However, similar marginal analyses of networks in the USA have used empirical emissions data, and found that the marginal emissions factors were often similar to, and sometimes lower than, the corresponding average emissions (Siler-Evans *et al.*, 2012; Kaffine *et al.*, 2011). This suggests that the effect of efficiency penalties can be significant.

This chapter investigates the effect of efficiency penalties of the two most significant fossil-fuelled generating technologies, coal and CCGT, on the marginal emissions displacement of wind power on the British grid. It builds upon the study presented in the previous chapter (Chapter 6) by deriving carbon intensity curves from efficiency data for typical coal and CCGT plant, and applying these to detailed historical power output profiles for each generator derived from published balancing mechanism data. Multiple linear regression analysis is again applied to determine the marginal emissions factors for both wind power fluctuations and changes in demand.

7.2 Analysis

Where possible, the data and methodology used in this analysis is the same as that presented in Chapter 6 so this chapter only includes information where the data sources or method deviate from the earlier study. Reference must be made to Chapter 6 for full details of the analysis presented here.

7.2.1 Efficiency penalties

The GHG emissions intensity of power generation is related to generator efficiency by Equation 7.1. The Department for Energy and Climate Change (DECC) estimates the emissions intensities of coal and natural gas in the British market to be 0.39988 kg CO_2 eq/kWh and 0.22674 kg CO_2 eq/kWh respectively (AEA, 2012). Applying Equation 7.1 and the average emissions intensities detailed in Table 6.1, which were also calculated from data published by the UK government (MacLeay *et al.*, 2013; Ricardo-AEA, 2013), this suggests that the average efficiency of coal-fired power stations is 41.5 %, and CCGT is 55.1 %. However, the efficiency of such power stations is not constant and instead depends upon the relative power output, as illustrated in Figure 7.1. In order to determine the effect of such part-load efficiency penalties on the GHG emissions intensity of coal and CCGT power generation, typical efficiency curves were derived from published data.

$$\mathrm{GWP_{elec}} = \frac{\mathrm{GWP_{fuel}}}{\eta} \tag{7.1}$$

where:

$\mathrm{GWP_{elec}}$	=	Emissions intensity of output electricity (kg CO_2 eq/kWh)
$\mathrm{GWP_{fuel}}$	=	Emissions intensity of input fuel (kg CO_2 eq/kWh)
η	=	Power station efficiency

The typical efficiency curve shown in Figure 7.1 for a CCGT plant was developed from generic information published by Kehlhofer *et al.* (1999). Firstly, a typical plant capacity was estimated from the average generating capacity of all CCGT units reporting to Elexon in 2012 (Elexon, 2013b; Enappsys, 2013), and found to be 492 MW. The maximum efficiency of a power station of this size was then extracted from Figure 2-4 on p18 of Kehlhofer *et al.* (1999). By combining this maximum efficiency value with data from a curve of relative efficiency against relative

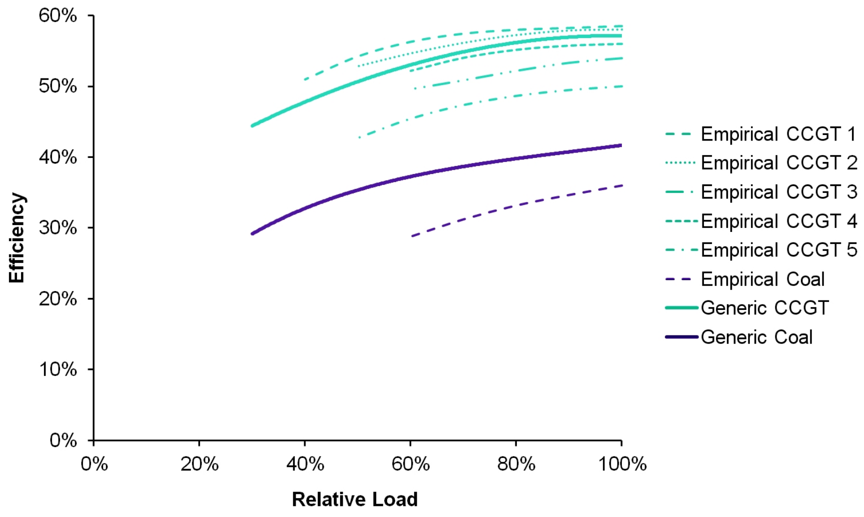


Figure 7.1: Typical efficiency curves for coal and CCGT power stations

power output (Figure 8-3, p211), the efficiency curve shown in Figure 7.1 was created. It can be seen that the efficiency of 55 % (the calculated average for British CCGT plant) corresponds to a power output of 75 %. Further validation of this generic curve was carried out by comparison with empirical data provided by a major generating company, which shows that it is a good match for several different types of common CCGT generator; the most significant discrepancy being the assumption that stable operation can be achieved down to 30 % of full load. As discussed later in this section, generator start-up and shut-down has been approximated by extrapolating the part-load curves, so this assumption does not have an impact on the results of this analysis. There is scope for further work to develop the modelling of generator start-up and shut-down.

No published efficiency curves were identified for the subcritical pulverised-coal power stations currently in operation on the British grid, so the efficiency curve shown in Figure 7.1 was derived from generic data for boiler and turbine efficiencies, based on the following equation (Sgourinakis, 2009):

$$\eta_{\text{full load}} = \eta_{\text{boiler}} \times \eta_{\text{turbine}} \times (1 - \textit{parasitic load}\%) \tag{7.2}$$

At part load, the parasitic load, which is usually 1.5 to 3 % of gross output power, can be assumed to be very small, so the relative efficiency (actual efficiency divided by full-load efficiency) can be estimated from the relative boiler and turbine efficiencies (Equation 7.3). The turbine efficiencies were estimated from a typical Willians Line, taken from Sgourinakis (2009) and described by Equations 7.4 to 7.6, and the relative boiler efficiency curve was taken from Sorour (2008). The typical generating capacity was estimated to be 488 MW (the mean generating capacity of all coal-fired BM units), and the maximum efficiency was again taken from Figure 2-4 of Kehlhofer *et al.* (1999) to create the curve shown in Figure 7.1. Validation of this curve by comparison with data from a major generating company suggests that this efficiency estimate is optimistic, although the empirical data is from only a single generator, and is therefore not representative of the British coal-fired generating mix. The initial average efficiency estimate from reported emissions data for coal-fired power generation was 42 %, corresponding to the full-load efficiency of the modelled curve, and suggesting that the efficiency of the generator used to create the empirical data might be below average. Significantly, however, the shape of the modelled curve is similar to that of the empirical data, and an examination of the impact of using the empirical curve in the analysis found that it decreased the resulting MDF estimate by less than 1 %.

$$\eta_{\text{rel part load}} = \eta_{\text{rel boiler at part load}} \times \eta_{\text{rel turbine at part load}} \tag{7.3}$$

$$P_{\text{in turbine at part load}} = 0.88P_{\text{out}} + 0.12 \tag{7.4}$$

$$\eta_{\text{turbine}} = \frac{P_{\text{out}}}{P_{\text{in}}} \tag{7.5}$$

$$\eta_{\text{rel turbine at part load}} = \frac{P_{\text{rel out}}}{(0.88P_{\text{rel out}} + 0.12)} \tag{7.6}$$

Once typical efficiency curves had been identified for coal and CCGT plant, similar curves could be created for the GHG emissions intensity based on Equation 7.1, and shown in Figure 7.2. The equations describing these curves, given in Equations 7.7 and 7.8, were used to calculate the part-load GHG emissions intensity for each coal and CCGT BM unit on the British network. The GHG emissions intensity for all other types of supply was assumed to be constant, and average emissions intensities for each of these were derived from historical data published by the UK government, as detailed in Section 6.2.1.

$$GWP_{\text{coal}} = 6.4P_{\text{rel}}^6 - 29.0P_{\text{rel}}^5 + 54.7P_{\text{rel}}^4 - 56.1P_{\text{rel}}^3 + 33.9P_{\text{rel}}^2 - 12.0P_{\text{rel}} + 3.1 \tag{7.7}$$

$$GWP_{\text{CCGT}} = 0.14P_{\text{rel}}^6 - 0.68P_{\text{rel}}^5 + 1.49P_{\text{rel}}^4 - 1.91P_{\text{rel}}^3 + 1.69P_{\text{rel}}^2 - 1.05P_{\text{rel}} + 0.71 \tag{7.8}$$

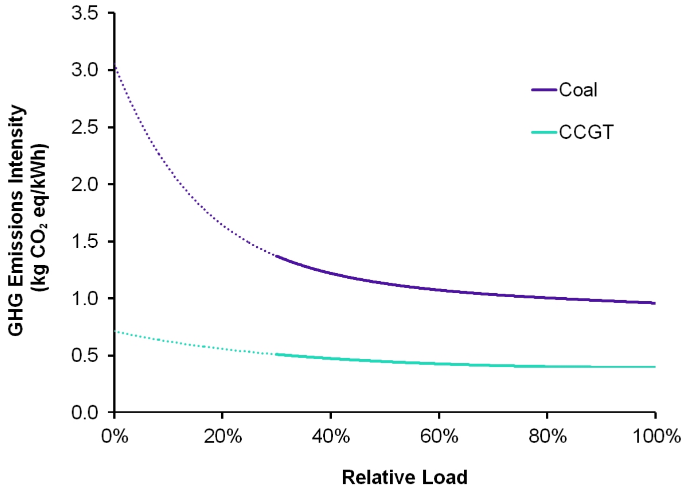


Figure 7.2: GHG emissions intensity curves for coal and CCGT power stations

Start-up and shut-down

Generator start-up and shut-down are complicated processes that have very different fuel consumption characteristics from part-load operation. Typical start-up of a CCGT plant, for example, begins by running the generator as an electric motor to bring the gas turbine up to low speed; the fuel is then ignited, the gas turbine is synchronised with the grid, and the load is increased to around 25 %. Operation is maintained at this level while the pressure in the heat recovery steam generator rises to the minimum operating pressure and is then raised to around 50 %; once the steam meets the required operating parameters of the steam turbine, it can also be run up and synchronised. For a cold-start it will be several hours before the steam turbine can accept all available steam (Environment Agency, 2011).

The emissions intensity curves shown in Figure 7.2 do not accurately describe the GHG emissions of generator start-up and shut-down. It is expected that these emissions will not contribute significantly to the marginal displacement of wind power, but further work is required to confirm this. For the purposes of the analysis presented here, it has been assumed that the start-up and shut-down emissions can be approximated by extrapolating the curves representing the effect of part-load efficiencies, as shown by the dotted lines in Figure 7.2.

7.2.2 Data sources

In order to take efficiency penalties into account when calculating the instantaneous GHG emissions of a generator, the GHG emissions intensity must be calculated from the instantaneous relative power output. The analysis presented in Chapter 6 was based on empirical data that was aggregated by fuel type (Elexon, 2013a), so the part-loading effects of each individual generator could not be determined. In order to consider efficiency penalties, detailed power output profiles are required for each generator, but operationally metered data is not publicly available for individual generators on the British grid. Approximate curves were therefore derived from data contained in daily file archives of BMRA messages published by Elexon (Elexon, 2013c).

Deriving power output profiles

As detailed in Chapter 5, the Balancing Mechanism Reporting Agent (BMRA), Elexon, publishes detailed information on the contracted output of all generators (or BM units) signed up to the Balancing and Settlement Code (Elexon, 2013c). This includes details of the planned power output level as contracted at gate closure; balancing bids and offers; prices; and any export or import limits set by the grid operator due to physical constraints. This data is reported for each half-hourly settlement period throughout the day. Although not all BM units take part in the balancing mechanism, they must all provide a final physical notification (FPN) of their contracted power production at gate closure, 1 hour before the settlement period.

Power output curves can be extracted for each BM unit from the published FPN data, as with those examined by Hawkes (2010); however, these FPN levels are the contracted values at gate closure, and therefore may not reflect the actual power outputs following planned and unplanned changes up to, and during, the settlement period. The response of conventional coal and CCGT power stations to short-term fluctuations in wind power output are likely to occur following gate closure. In order to better approximate the actual output of each BM unit, power output curves for this analysis were developed by also considering bid-offer acceptances and any changes to export and import limits set by the system operator (SO).

Elexon publishes daily summary files, in comma separated value format, containing all messages that pass through the balancing mechanism reporting system in 24 hours, such as planned operating levels, export limits, prices, and system operating information (Elexon, 2013c). These contain data on the contracted output of every generator for every minute of the day. The following data was extracted from these daily summary files to develop the power output profiles:

- **Final Physical Notification (FPN)** - This is published once for each settlement period for every BM unit, and is submitted at gate closure. It provides details of the level of generation (or demand) that the BM unit expects to export (or import) during the settlement period. It is reported as point power levels at the beginning and end of each settlement period, with additional intermediate levels reported as necessary, along with their corresponding times.

- **Bid-offer Acceptance Level (BOAL)** - This represents any purchase (or sale) of power through acceptance of offers (or bids) during the operation of the balancing mechanism. BOALs will not be published for every BM unit, as they do not all take part in the balancing mechanism. Power levels are reported for the beginning and end of any acceptance, as well as any necessary intermediate points. Acceptances may cross settlement period boundaries and may also modify or replace earlier BOAL values.
- **Maximum Export Limit (MEL)** - This is set by the system operator for each settlement period, and describes the maximum power output levels of a BM unit as a set of point values and associated times. It can be adjusted by the SO at any time so may be re-declared at any point up to and during the settlement period. Normally this will be equal to the maximum generating capacity of the unit, but it may be decreased in response to planned or unplanned system outages, or frequency fluctuations.
- **Maximum Import Limit (MIL)** - Distribution grid supply points, pumped storage power stations, interconnectors and some generators are BM units with the capacity to import power. Like the MEL, the MIL describes the maximum power levels that a BM unit can import as a series of point values and associated times for each settlement period, and may be re-declared at any time.

The first step in building the power output profiles for each BM unit was to extract FPN data from the published daily data, as shown in Figure 7.3a. This is relatively straightforward, as only one FPN message is issued per BM unit per settlement period; the times and corresponding power levels were extracted from each message and ordered sequentially.

The next step was to identify any acceptances of bids and offers during the balancing mechanism that would modify the FPN power levels. This is slightly more complicated, as BOAL data is issued each time a bid or offer is accepted, so may replace a previous acceptance. For each BOAL message, the times and corresponding power levels were extracted, and overlaid or inserted into the FPN output profile for the corresponding BM unit. Each BOAL was processed sequentially so that acceptances issued later would always replace the values in earlier acceptances if they corresponded to the same time. The resulting power output profile is illustrated in Figure 7.3b.

The final step was to check if the export or import limits had been exceeded. Generally only one set of limits is published for each settlement period, but these may be modified at any time, so it was assumed that the values reported later would always replace those reported earlier. For each MEL or MIL message, the times and corresponding power levels were extracted, and either replace or be added to any previously extracted values, as shown in Figure 7.3c. Once the limit profiles had been developed, they were compared to the contracted power levels (FPN data as modified by any BOAL values), and the output or input power levels were reduced as necessary (Figure 7.3d). It can be seen that the resulting power output curve in Figure 7.3d differs considerably from the initial contracted power output levels reported in the FPN at gate closure.

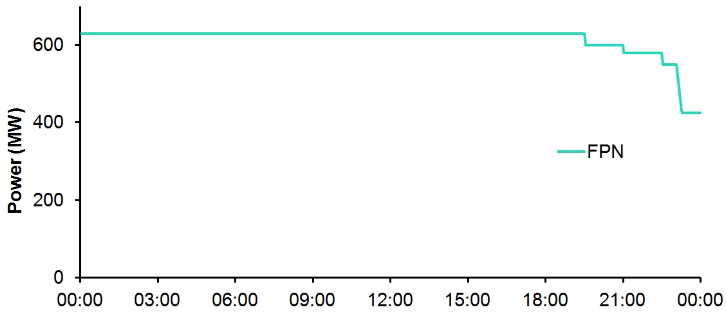
Initially this method was applied to produce estimated power output profiles for all 410 BM units reporting to Elexon from November 2008 to June 2013. (Although the historical BMRA data archive contains information for operation of the BM units dating back to January 2002, this start date was chosen to match that of the analysis presented in Chapter 6.) This involved the extraction of data from over 1700 BMRA message files for over 80 000 settlement periods, which was carried out using Matlab (MathWorks, 2011).

Verifying the power output profiles

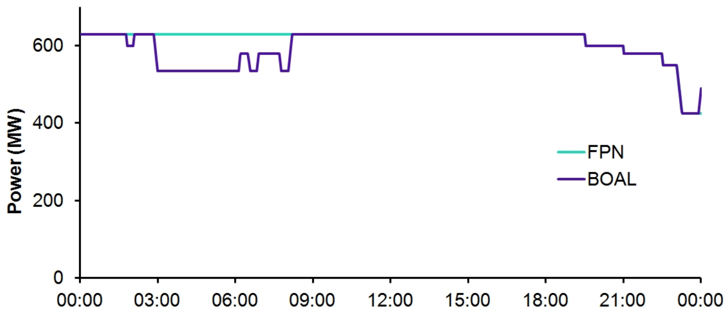
Although the power output profiles, such as that shown in Figure 7.3, are derived from the best available information, they may not reflect the actual output of each BM unit, as information about imbalances between the actual and contracted generation is not made publicly available. The accuracy of these curves was therefore verified by aggregating the power levels by fuel type at half-hourly intervals and comparing these with the historical metered data published by National Grid, as shown in Figure 7.4 (Elexon, 2013a). All operationally metered generators are BM units that must report their FPN to the system operator at gate closure, so both datasets should reflect the output from the same set of generators. The degree of correlation between these was identified graphically, by plotting the estimated power level against the empirical power level for each time stamp, and finding the gradient of the linear fit. A gradient, or correlation factor, of 1 represents a perfect correlation between the results.

The calculated correlation factors are summarised in Table 7.1. This confirms that the estimated power output profiles derived from the BMRA messages for coal and CCGT plant correlate well with the measured historical data, as might be expected of controllable plant that take part in the balancing mechanism. However, considerable discrepancies were found with some of the other data, particularly hydropower, wind and the East-West Interconnector. These discrepancies represent imbalances between the contracted and actual power output levels, likely to be due to forecasting errors or, in the case of the East-West Interconnector, reporting errors during the first few months of testing and operation (it opened in September 2012).

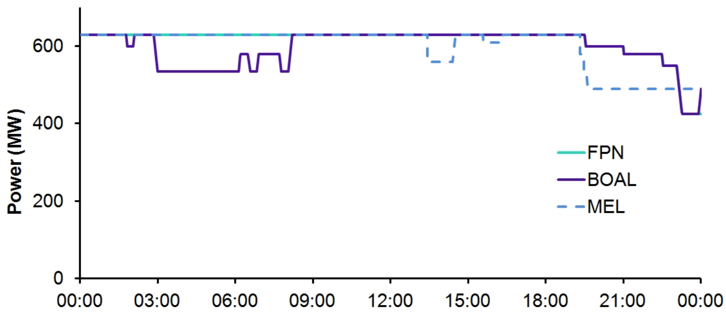
The half-hourly changes in output are of particular significance for this marginal analysis. The complete lack of correlation between the estimated and measured half-hourly variations in wind power output demonstrates the difference in the shape of the power output profiles from one time stamp to the next. It can, therefore, be seen that the power output profiles derived from the BM messages for wind power do not provide a good approximation of the actual output. In order to examine the effect of efficiency penalties for coal and CCGT power stations, however, detailed power output profiles are only required for these particular BM units, and there is a



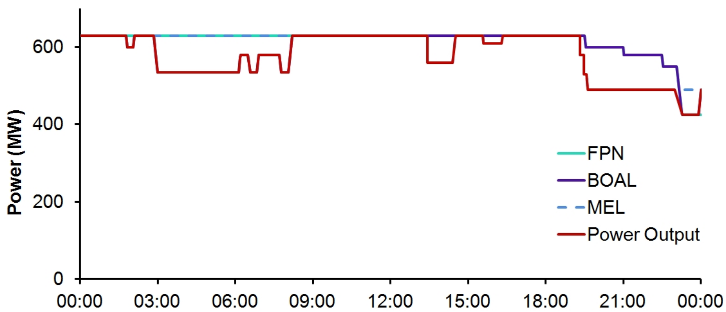
(a) FPN



(b) After BOAL



(c) MEL



(d) Estimated power output profile

Figure 7.3: Development of estimated power output profiles from BMRA messages (Drax 3 Generator, 18th February 2009)

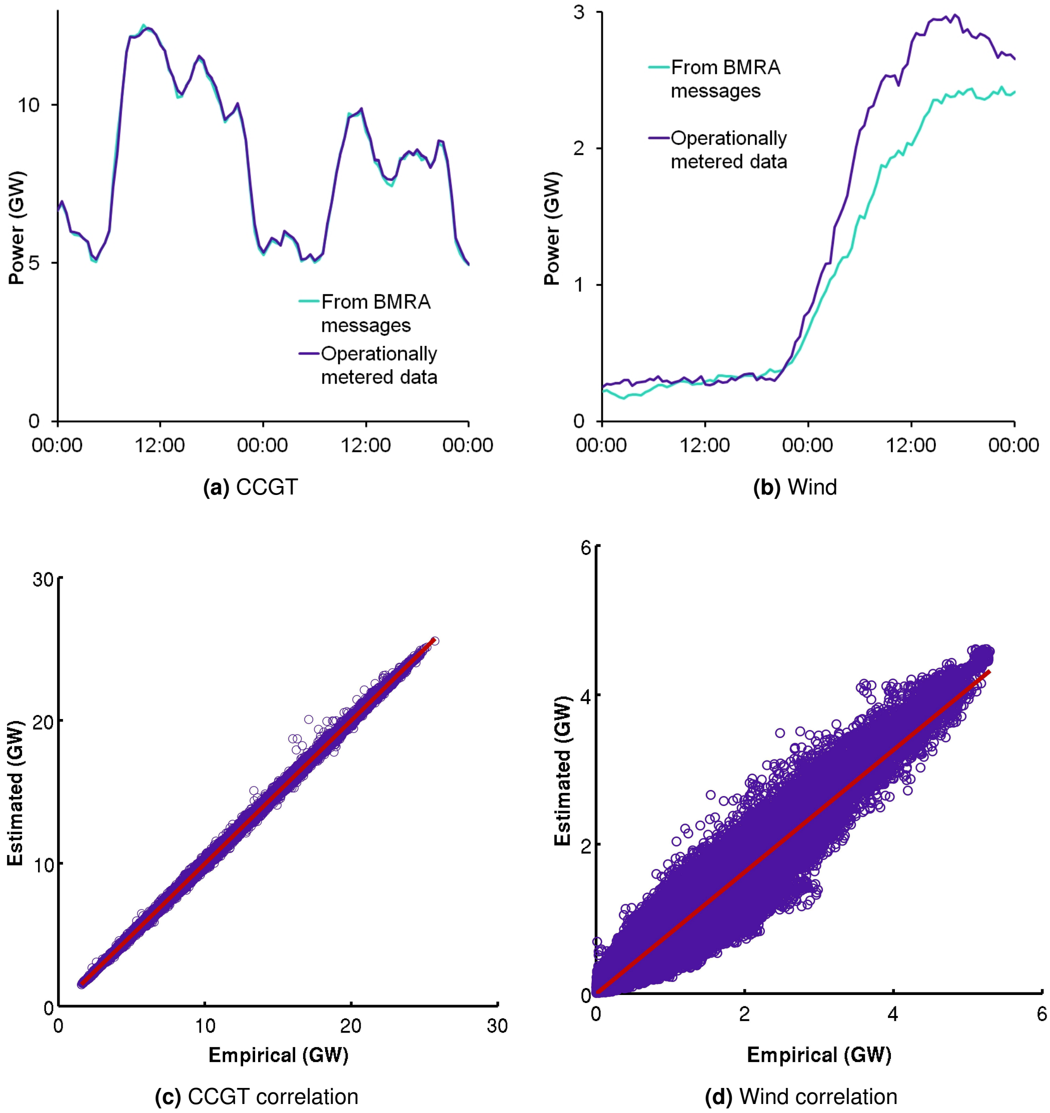


Figure 7.4: Determining the correlation between estimated and empirical power output (power curves shown for 21st July 2012)

Supply Type	Correlation Factor
Nuclear	1.00
Oil	1.01
Coal	0.97
CCGT	1.00
OCGT	1.04
Other	0.57
French interconnector	0.95
Irish interconnector	1.03
Dutch interconnector	1.00
East-west interconnector	0.13
Hydro	0.75
Pumped Storage	0.99
Wind	0.82
Changes in total system load	0.97
Changes in total wind power	0.17

Table 7.1: Correlation factors between metered data and data derived from BMRA messages

good correlation between the estimated and empirical data for these plants. Empirical data from the National Grid, as described in Section 6.2.1, was used for all other types of supply to avoid unnecessary errors being introduced by any imbalances (Elexon, 2013a).

Generating capacity

The GHG emissions intensity curves described in Section 7.2.1 allow the instantaneous GHG emissions intensity of the output from a coal or CCGT power station to be calculated based upon its relative power output. The instantaneous power output can be taken from the power output profiles derived from the BMRA messages, but in order to convert this into a relative power output, the maximum generating capacity of the plant must be provided. This was taken from published data of registered BM units (Elexon, 2013b; Enappsys, 2013).

7.2.3 Detailed method

The methodology for this analysis is the same as that described in detail in Chapter 6, based on a method developed by Hawkes (2010). It uses multiple linear regression to extract the marginal displacement factor of wind power and the marginal emissions factor of demand from detailed half-hourly operational data. In the analysis presented here the only deviation from the methodology presented in Chapter 6 is that required to incorporate the varying GHG emissions intensities of coal and CCGT power stations.

Power data

The first step in this analysis is to extract and aggregate half-hourly power data for each type of supply. This was mostly taken directly from the dataset developed for the analysis in Chapter 6, but for coal and CCGT it was extracted from the data derived from the BMRA messages. This involved sampling and aggregating the power output profiles for each BM unit at half-hourly intervals. The total system generation and half-hourly changes in both wind and total generation could then be calculated as described in Section 6.2.3.

GHG emissions data

The additional complexity in this analysis was in incorporating the effect of efficiency penalties on the GHG emissions of coal and CCGT plant. Firstly, GHG emissions profiles were developed for each coal or CCGT BM unit from its corresponding power output profile, generating capacity and Equation 7.7 or 7.8. This data was then sampled at half-hourly intervals and aggregated by fuel type. The half-hourly GHG emissions from all other types of supply were calculated from the aggregated half-hourly power data and the fixed emissions intensities given in Table 6.1. The GHG emissions of pumped storage were calculated as described in Section 6.2.3, based on the total system GHG emissions at the time that power was imported, and taking the efficiency of the plant into account. Total system GHG emissions were calculated as the sum of all emissions at a given time stamp, and half-hourly changes were calculated as described by Equation 6.3.

Calculation of marginal emissions factors

As with the analysis presented in Chapter 6, outlying data points were discarded before the final analysis step. The same criteria for discarding outliers were selected, resulting in only 0.15 % of data points being removed from the calculation.

7.3 Results

The planar fit for the whole dataset from November 2008 to June 2013 is shown in Figure 7.5. Again the fit is very good, with a coefficient of determination (R^2 value) of 0.96. When the efficiency penalties of coal and CCGT generation are taken into account, the marginal displacement factor of wind power is found to be to be 0.562 kg CO_2 eq/kWh $\pm$ 0.014. This is 11 % lower than the MDF estimated in Chapter 6 from fixed values of carbon intensity (0.628 kg CO_2 eq/kWh), demonstrating that the increased level of part loading of generators due to fluctuations of wind power does increase the carbon emissions of conventional generation. This displacement factor is, however, 9 % higher than the calculated average emissions

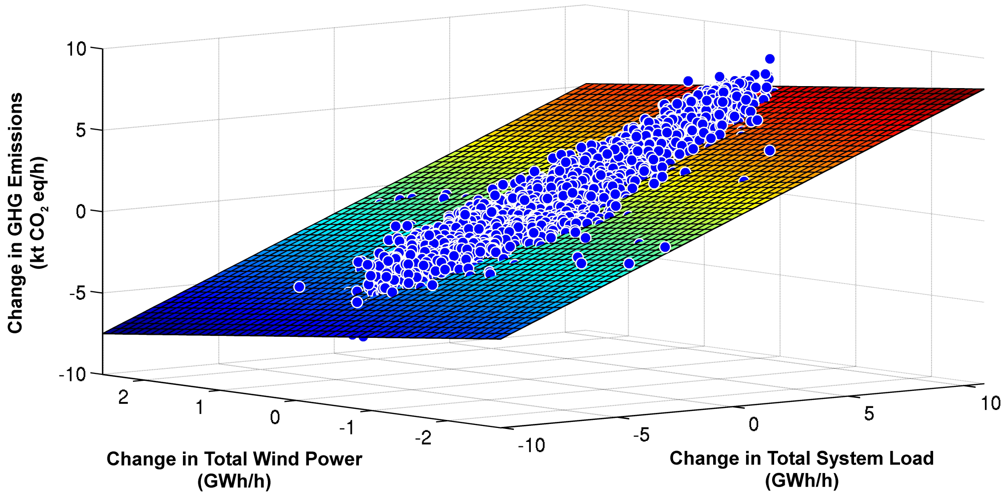


Figure 7.5: Relationship between changes in GHG emissions, system generation and wind power output (data from November 2008 to June 2013)

factor over the corresponding time period, so the actual emissions displacement will be higher than estimates based on average emissions.

The marginal emissions factor of total system generation was also calculated, and found to be 0.604 kg CO_2 eq/kWh $\pm$ 0.001, 7 % lower than that estimated in Chapter 6, but 7 % higher than the estimated MDF of wind power. In the earlier analysis with fixed carbon intensities, the MEF was only 3 % higher than the MDF, so the marginal emissions of demand-side fluctuations are less significantly affected by efficiency penalties than the emissions displacement of wind power; contrary to the findings of Chapter 6, the system does not respond similarly to fluctuations in demand and supply, and neither the marginal emissions of demand or the system-average emissions provide a good approximation for the emissions displacement of wind power.

As can be seen in Figure 7.6, the linear trend between changes in wind output and GHG emissions is not as clearly defined as that between emissions and total system generation. The general distribution is very similar to that found in the previous analysis, albeit with a different gradient. The increase in fluctuations of GHG emissions where fluctuations in wind power output are small are likely to represent changes caused by other network effects, such as scheduled ramping of generators, unplanned outages and network constraints. Further work could include an investigation of these effects.

In line with the earlier study, and that published by Hawkes (2010), this analysis found that both the marginal emissions factor of generation and the marginal displacement factor of wind power

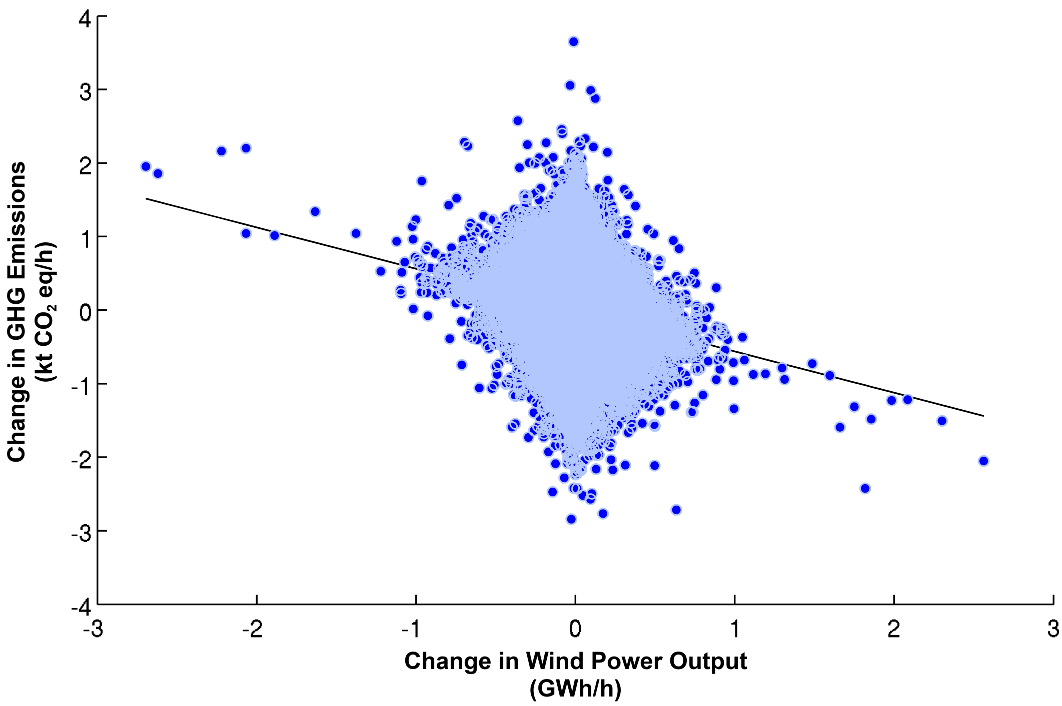


Figure 7.6: Relationship between change in wind power and change in GHG emissions

were higher than the emissions displacement estimate of 0.46 kg CO_2 eq/kWh recommended by the UK government (Ricardo-AEA, 2012), even when efficiency penalties have been taken into account. The impact of this variation on payback estimates is examined in Chapter 8.

7.3.1 Temporal trends

Again, the data has been divided into sections and examined with regards to any trends or changes in marginal emissions displacement over time, seasonally and throughout the day. As can be seen from Figures 7.7, 7.8, 7.9 and 7.10 the general shape of the curves is the same as that found in Chapter 6 when efficiency penalties were not considered (the MDF without efficiency penalties is shown as a dotted line); however, the effect of the inclusion of efficiency penalties in the analysis has decreased the estimated MDF.

The data shows that there is no clear relationship between the MDF and time of day or season, although the uncertainty of the marginal displacement seems to be greatest in the autumn, and also during the morning pick-up and evening drop-off in demand. The MEF appears to decrease slightly during daily times of high demand, but no clear seasonal trend emerges.

There is also a clear trend for the MDF to be converging with the average emissions factor over time, towards a value of 0.5 kg CO_2 eq/kWh and the estimates are becoming more consistent. This may be attributable to changes in coal and gas prices, but is also likely to be an effect

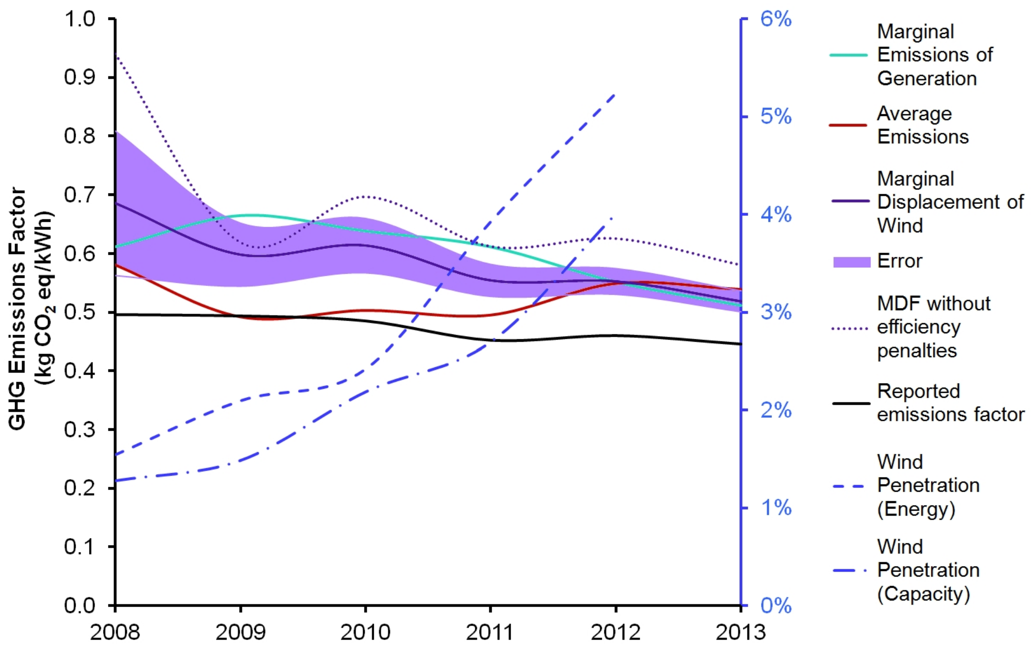


Figure 7.7: Annual trends in calculated marginal emissions

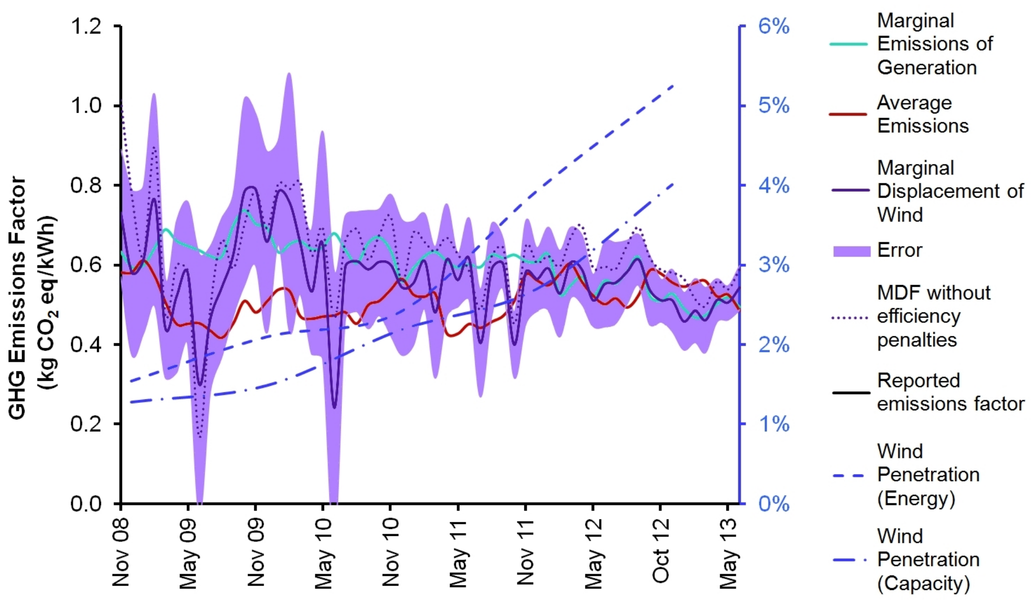


Figure 7.8: Detailed monthly fluctuations in calculated marginal emissions

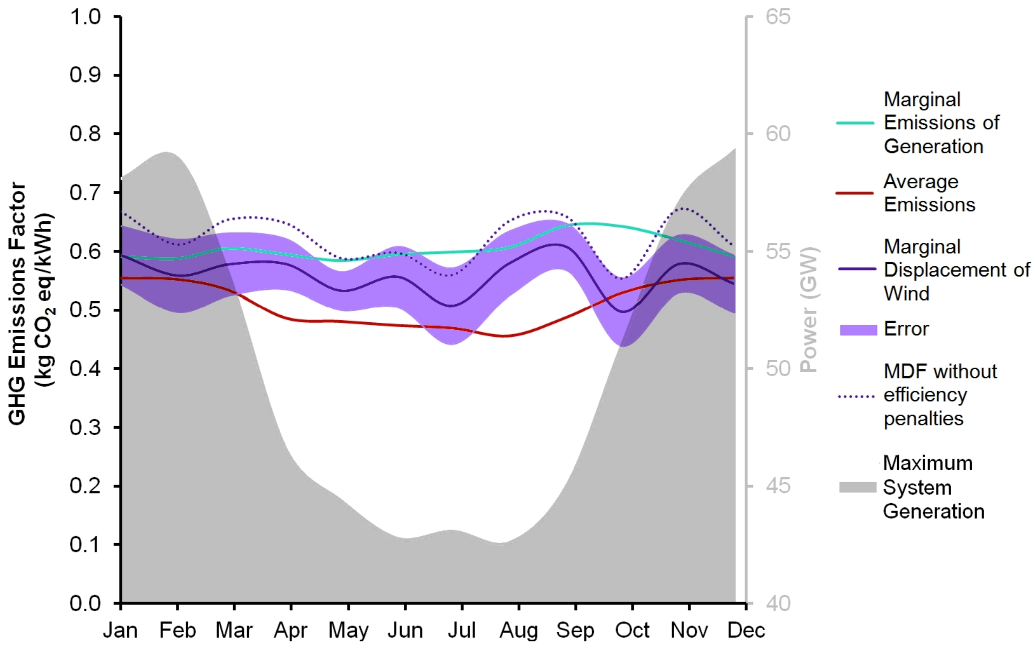


Figure 7.9: Seasonal fluctuations in calculated marginal emissions

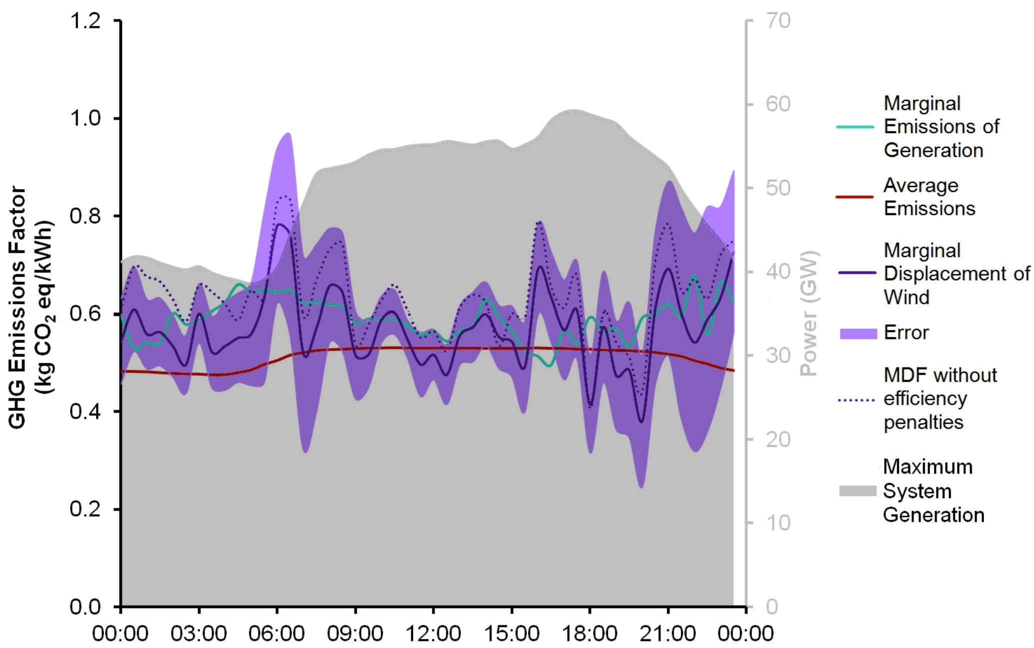


Figure 7.10: Marginal and average emissions as a function of time of day

of better forecasting coupled with an increase in installed capacity of wind on the UK grid. The calculated average MDF for 2012, however, is 0.55 kg CO_2 eq/kWh, 20 % higher than the UK-average emissions for that year reported by the government.

7.3.2 Wind output level

The effect on the marginal displacement factor of increased penetration of wind on the British grid was investigated by binning the data according to wind power output and running several marginal analyses. The results of this are summarised in Figure 7.11. It can be seen that both the MDF and MEF decrease at higher wind power output levels, suggesting that they tend towards replacing CCGT plant rather than coal fired generation. Significantly the MDF stabilises to a linear trend that is converging with the average emissions factor, suggesting that an increase in output from variable renewables will continue to decrease the marginal emissions displacement. However, as the increased output corresponds to an increase in installed capacity, there are fewer data points in the higher bins, and these are likely to be restricted to more recent years, so this may reflect a trend over time. Additional data is required to confirm this. Significantly, the MEF of total system generation is found to stabilise to a value of around 0.57 kg CO_2 eq/kWh.

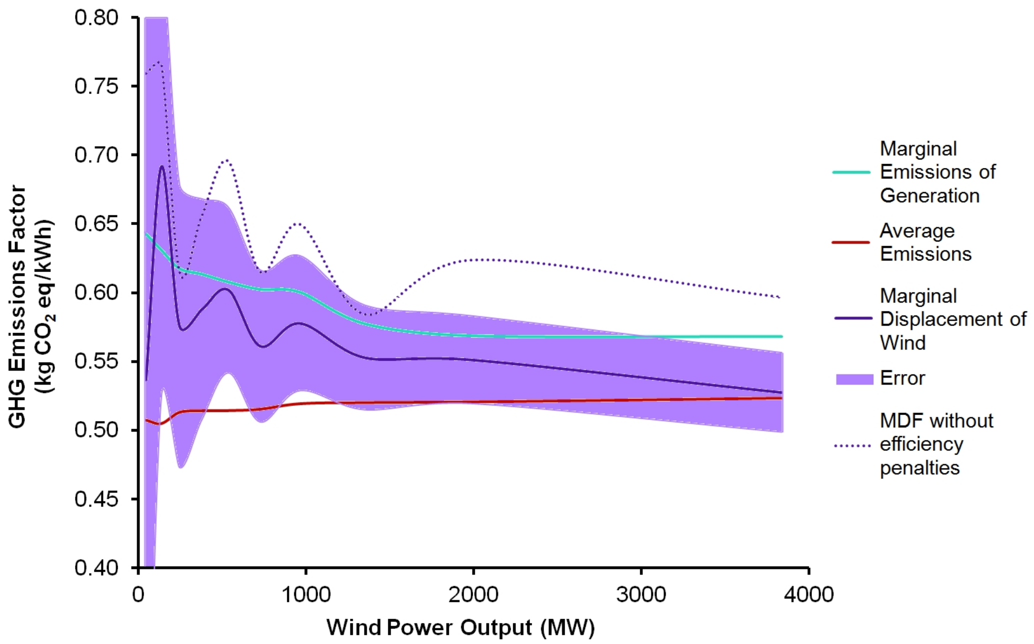


Figure 7.11: Relationship between marginal/average emissions and instantaneous wind power output

The effect on emissions of increased penetration of wind on the network is of particular interest. The examination of annual trends in Figure 7.7 found no clear relationship, due to the strong influence of the changing fuel mix of conventional generation dictated by fuel prices. By binning the data according to the instantaneous wind power output as a proportion of total system generation, the impact of an increase in wind penetration has been examined. As can be seen from Figure 7.12, the trends were similar to those found in Chapter 6, but with much lower values. At a contribution greater than 5 % of total output, the marginal displacement factor seems to stabilise at around 0.54 kg CO_2 eq/kWh. However, as with the analysis presented in Figure 7.11, there are many fewer data points at these higher values, and they are likely to come solely from more recent years, so further analysis of more data, as it becomes available, will increase the certainty of these results.

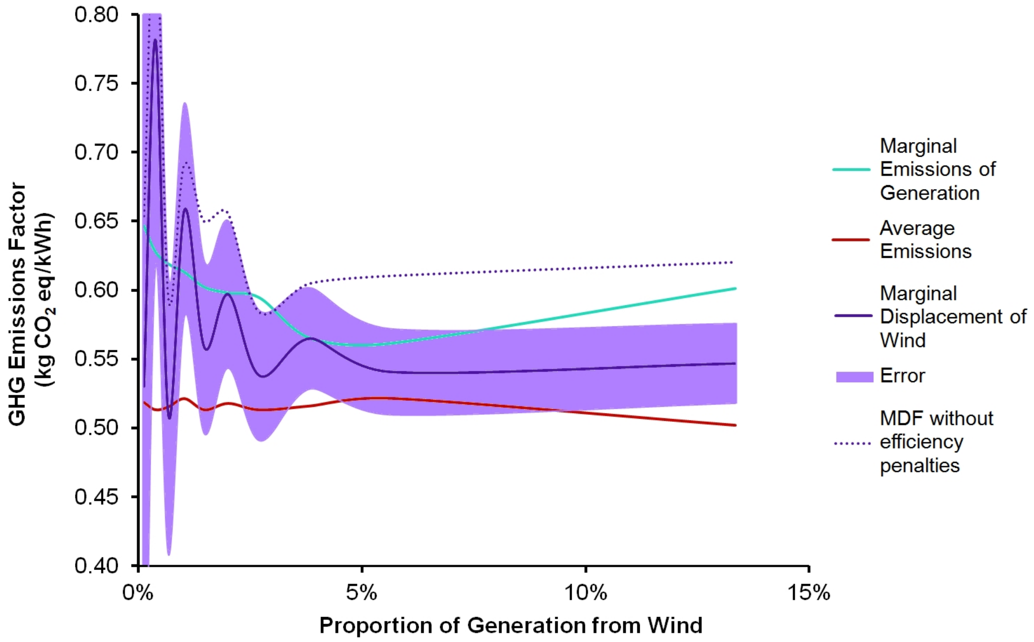


Figure 7.12: Relationship between marginal and average emissions and wind generation as a proportion of total system output

Note also that this review is limited by the fact that higher contributions from wind are likely to occur at times of lower demand, when the contribution of nuclear power to the generation mix is higher and high-carbon coal-fired generation is operating on the margin.

7.4 Conclusions

Reliable figures for the actual emissions savings of wind power are required to accurately estimate the carbon payback period of variable-output renewable generation and inform policy decisions. Estimates based on average emissions factors will be biased by the inclusion of low-carbon nuclear generation in the British mix, which does not respond to short-term fluctuations in supply or demand. The study presented in Chapter 6 examined the marginal displacement of wind power in Great Britain, and found that wind generation mostly replaced the output from carbon-intensive coal and gas-fired power stations; however, the findings of this analysis were limited by the assumption of constant GHG emissions factors for generation.

The analysis presented in this chapter builds upon the earlier study, taking into account the decrease in efficiency, and thus increase in GHG emissions intensity, of coal and CCGT generation at part load. Generic efficiency curves were derived from published information to determine the relationship between emissions intensity and power output. Published data from the balancing mechanism was then used to create detailed power output profiles for every coal and CCGT generator reporting to the system operator, from which the corresponding GHG emissions were estimated. This data was combined with real historical operational data for all other supply types to estimate the marginal emissions displacement of wind power.

This study found that the consideration of efficiency penalties decreased the calculated marginal displacement factor of wind power by 11 % to 0.56 kg CO_2 eq/kWh, averaged over the period from November 2008 to June 2013. This was 7 % lower than the estimated marginal emissions factor of total generation at 0.60 kg CO_2 eq/kWh, but still 9 % higher than the calculated system-average emissions rate of 0.52 kg CO_2 eq/kWh. The estimated MEF was also 7 % lower than that calculated when efficiency penalties were ignored, confirming that the impact of efficiency penalties on both the marginal displacement of variable-output renewable generation and the marginal emissions of total generation is significant.

This study also investigated temporal trends and the relationship between marginal emissions rates and wind power output, and found these to be very similar to those presented in Chapter 6, although the absolute values were consistently lower. Both the estimated MDF and MEF were observed to be converging with the calculated system-average emissions factor over time, towards a value of 0.5 kg CO_2 eq/kWh. This is likely to be an effect of better forecasting, increased installed capacity of wind power, and changes in the relative prices of coal and gas.

The relationship between wind power output and marginal displacement factor was also investigated, and it was found that the latter decreases with increasing generation. Examining this with wind output as a proportion of total generation found that the MDF tended towards a value of 0.54 kg CO_2 eq/kWh. This analysis is still very approximate, however, as the number of available data points decreases at higher levels of wind generation, and these data points are also likely to be spread over a smaller time frame.

The consideration of efficiency penalties when estimating marginal emissions of the British grid was, therefore, found to have a significant effect, with the marginal displacement factor of wind power much lower than that calculated by assuming constant GHG emissions intensities for coal and CCGT plant. The effect on the marginal emissions factor of total system generation was not so significant; however, both the MDF and MEF were found to be very similar to the calculated system-average emissions rate, with all three values falling within $\pm$ 5 % of each other in recent years. This suggests that the system-average emissions rate may be a reasonable approximation for the marginal displacement of variable-output renewables, but it is worth noting that the calculated marginal displacement factor for 2012 remains 20 % higher than the published UK-average for that year (Ricardo-AEA, 2012).

7.4.1 Further work

The study presented here provides a significant insight into the real GHG emissions displacement of variable-output renewable generation in Great Britain, attempting to answer questions about the actual response of conventional generation and the effect of such a response on its operating efficiency. One limitation of this work, however, is that the GHG emissions characteristics of power station start-up and shut-down are not considered, and were instead approximated by an extrapolation of the part-load efficiency curves. Models of the GHG emissions during start-up and shut-down need to be developed to further refine the results and determine whether start-up and shut-down emissions are significant.

Similarly to the work presented in Chapter 6, this analysis is limited by the availability of generator data. It was again based on data from operationally metered generators reporting to the system operator, and therefore excludes most embedded generation and a significant proportion of existing installed capacity of renewable energy. Furthermore, information about off-grid fuel consumption due to events such as pre-warming the power stations was not available, and therefore the associated emissions have not been included. In order to develop a more accurate picture of the emissions displacement of renewable energy, more detailed data is required from network and generator operators.

In order to conclusively determine the effect of increasing penetrations of variable-output renewable generation on the British grid, further data is required for a larger installed capacity. Such information should become available over the next few years, as there is a large increase expected in installed capacity of wind, wave and tidal generation. In the meantime, further work is required to create a detailed forecasting model that can better inform future network developments, power station commissioning and decommissioning and, possibly, generator dispatch.

Chapter 8

Carbon and Energy Payback Periods of Variable Renewable Generation in Great Britain

8.1 Introduction

The work presented throughout this thesis has focussed on the reliability of the values that are used to estimate the carbon and energy payback periods of renewable power generation. The carbon payback period is the time taken for the displaced emissions to match the life cycle carbon footprint of the generator installation, while the energy payback period is the time for the total production to equal the lifetime energy consumption. Uncertainties in the estimated carbon footprint, embodied energy, displaced emissions or energy output will introduce uncertainties to the estimated carbon or energy payback period. The analysis presented here builds upon that detailed in previous chapters to develop more robust and reliable estimates of the carbon and energy payback of variable-output transmission-connected renewable generation in Great Britain - specifically wind, wave and tidal power.

8.2 Carbon Payback Period

In order for a renewable energy generator to achieve a net reduction in GHG emissions, the carbon payback period should be significantly shorter than the intended lifetime of the installation. The carbon payback period is calculated from Equation 8.1. If the carbon footprint (CF) is reported per unit of output energy, then the payback can also be calculated from the marginal emissions displacement factor (MDF) and the design life of the installation (Equation 8.4), assuming that the estimated annual energy output (E_{out}/yr) in the calculation of the carbon footprint was correct.

$$\text{Payback period} = \frac{\text{Lifetime emissions}}{\text{Annual emissions displacement}} \tag{8.1}$$

$$\text{Lifetime emissions} = CF \times E_{out}/yr \times \text{life} \tag{8.2}$$

$$\text{Annual emissions displacement} = MDF \times E_{out}/yr \tag{8.3}$$

$$\text{Payback period} = \frac{CF}{MDF} \times \text{life} \tag{8.4}$$

Ideally, carbon payback should be achieved within a short period of time, and carbon payback estimates may not be valid if they are longer than a couple of years. This is because the GHG emissions intensity of a network fluctuates, and is likely to decrease in the future, affecting both the emissions from any electricity consumption and the displaced emissions. As it has been found that the greatest emissions occur during the extraction of materials and initial manufacturing or construction stage for most types of renewable generation (see Chapter 3), carbon payback estimates calculated with current displaced emissions are likely to be underestimates if the payback period is too long. In this analysis, an illustrative estimate of 2 years has been selected as an ideal maximum payback period, as it is 10 % of a typical renewable generator design life. Where electricity consumption during the generator life cycle is significant, such as for flood pumping in tidal barrage, changes in the emissions intensity of electricity over the device life cycle should have been taken into account in calculating the carbon footprint.

Figure 8.1 shows estimated carbon payback periods for wind, wave and tidal devices installed in the UK, assuming a marginal emissions displacement factor of 562 g CO_2 eq/kWh, from the analysis presented in Chapter 7. Carbon footprint estimates are taken from those summarised in Chapter 3, with ranges for wind power taken from Dolan and Heath (2012), and estimates for wave power from Parker *et al.* (2007); Walker and Howell (2011); Soerensen and Naef (2008) and Chapter 4; tidal stream converters from Douglas *et al.* (2008); Rule *et al.* (2009); and tidal barrage from Woollcombe-Adams *et al.* (2009); Kelly *et al.* (2012). As can be seen, most current estimates place the payback period within 2 years for wind, wave and tidal stream devices. Tidal barrage devices, which have a much longer design life of around 120 years, also have a longer expected carbon payback time, with the analysis by Kelly *et al.* (2012) producing an estimated payback period of 12 years when the impacts of flood pumping are included.

The range of values shown in Figure 8.1 reflects the variations introduced to the estimated payback period by variations in the carbon footprint. These variations in published values of carbon footprint for renewable energy converters are caused by differences in the technology, calculation methodology or assumptions in the initial study. In the LCA Harmonization Project, NREL attempted to remove some of these variations by harmonising specific assumptions and methodological choices (Dolan and Heath, 2012), and the effect on the estimated payback period for wind power is shown in the harmonised values in Figure 8.1.

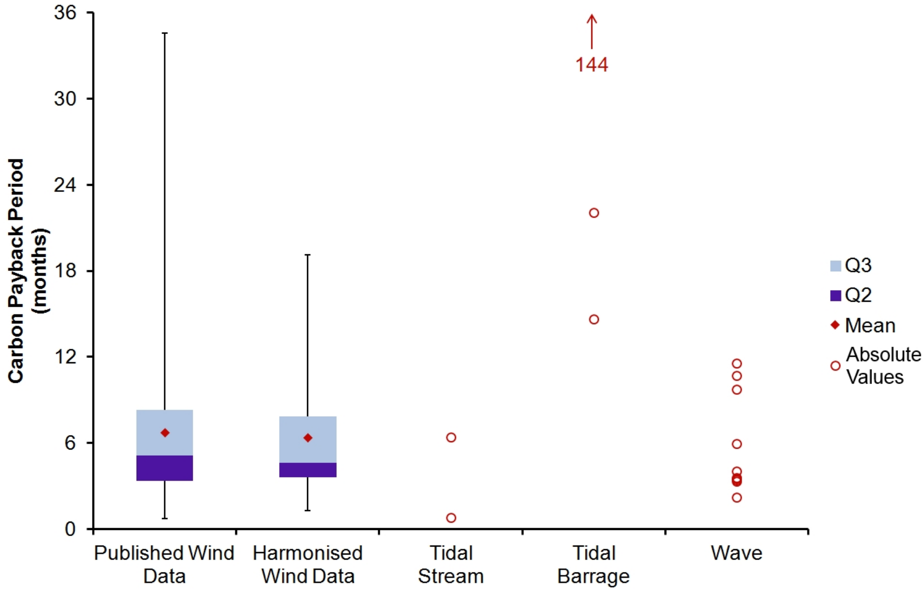


Figure 8.1: Range of carbon payback periods estimated from current published carbon footprints (Dolan and Heath, 2012; Parker *et al.*, 2007; Walker and Howell, 2011; Soerensen and Naef, 2008; Douglas *et al.*, 2008; Rule *et al.*, 2009; Woollcombe-Adams *et al.*, 2009; Kelly *et al.*, 2012)

The statistical ranges presented by Dolan and Heath (2012) do not, however, reflect the uncertainty of the results, but rather the range in published estimates. Each of these published values will have a corresponding uncertainty due to errors in the raw LCI data and assumptions inherent in the calculation. The impact of these on the uncertainty of a carbon footprint calculation was examined in Chapter 4, with a comprehensive sensitivity analysis on the results of a full LCA of the Pelamis wave energy converter. Figure 8.2 illustrates the findings of this sensitivity analysis with regards to the estimated carbon payback period (assuming a marginal displacement factor of 562 g CO_2 eq/kWh). Despite considerable uncertainty being introduced by the choice of recycling allocation method and uncertainties in the raw LCI data, it can be seen that payback is still expected well within the 20-year design life.

8.2.1 Emissions displacement

Chapter 5 highlights the challenges with estimating the emissions displacement of variable-output renewable generation. Current estimates of payback period are based on average emissions of power generation, estimated by the UK government to be 460 g CO_2 eq/kWh (Ricardo-AEA, 2012); however, an examination of the marginal displacement factor (MDF) of wind power estimated it to be 562 g CO_2 eq/kWh (considering efficiency penalties). It is thought that this latter value is the most accurate estimate of the true emissions displacement of variable-output renewable generation on the British grid. Figure 8.3 shows the estimated carbon payback

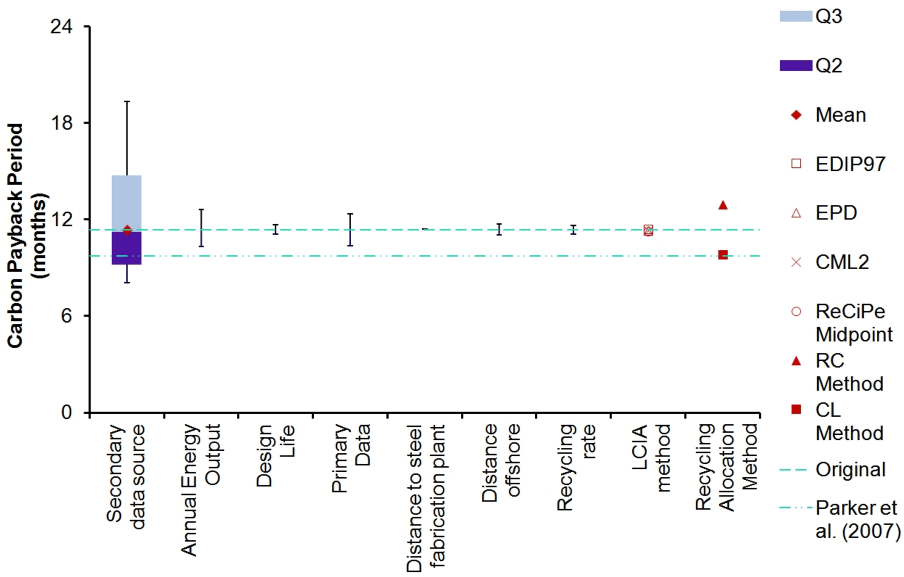


Figure 8.2: Sensitivity analysis of the carbon payback period for the Pelamis wave energy converter

period of wind power based on the harmonised results from Dolan and Heath (2012), and applying a range of different emissions displacement estimates. Calculated average emissions factors (AEFs) and MDFs are taken from Chapters 6 and 7, and the uncertainties are considered in presenting the ranges shown here. It can be seen that the variation in emissions displacement estimates doesn't have a significant effect on the payback period of wind farms, with all estimates falling within two years.

The estimated emissions displacement can, however, have a significant effect on the carbon payback period of renewable generators with a much higher carbon footprint. Chapter 3 found considerable discrepancies in the estimated carbon footprint of tidal barrages, with the inclusion of flood pumping having significant impacts. Figure 8.4 shows how the estimated payback period can range from 12 to 15 years if different values of emissions displacement are applied. The error bars reflect the uncertainty in the MDF values calculated in Chapters 6 and 7.

As previously discussed, the emissions displacement of variable-output renewable generation is likely to decrease over time. Equation 8.4 shows that carbon payback will be achieved as long as the emissions displacement is greater than the carbon footprint. Ideally the payback period should be considerably shorter than the design life. Figure 8.5 shows how the payback ratio (payback period as a proportion of design life) is affected by the emissions displacement for the range of carbon footprints considered in Figure 8.1. (The range shown for wind power is derived from the harmonised interquartile range of the summary published by Dolan and Heath (2012).) It can be seen that the payback period will be less than 10 % of the design life (2 years, if the design life is estimated at 20 years) for wind, wave and tidal stream devices

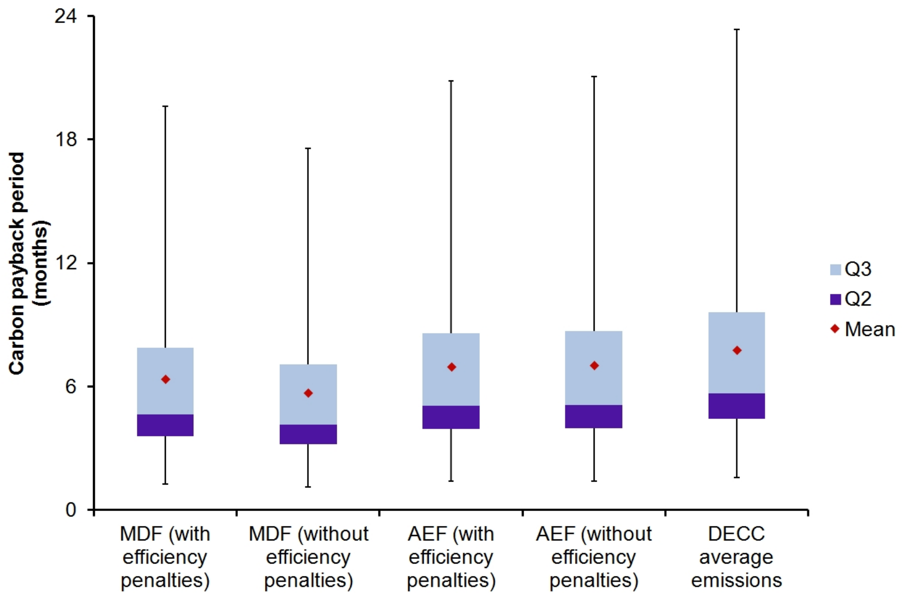


Figure 8.3: Comparison of effect of emissions displacement on carbon payback period for wind farms (based on harmonised carbon footprints from Dolan and Heath (2012))

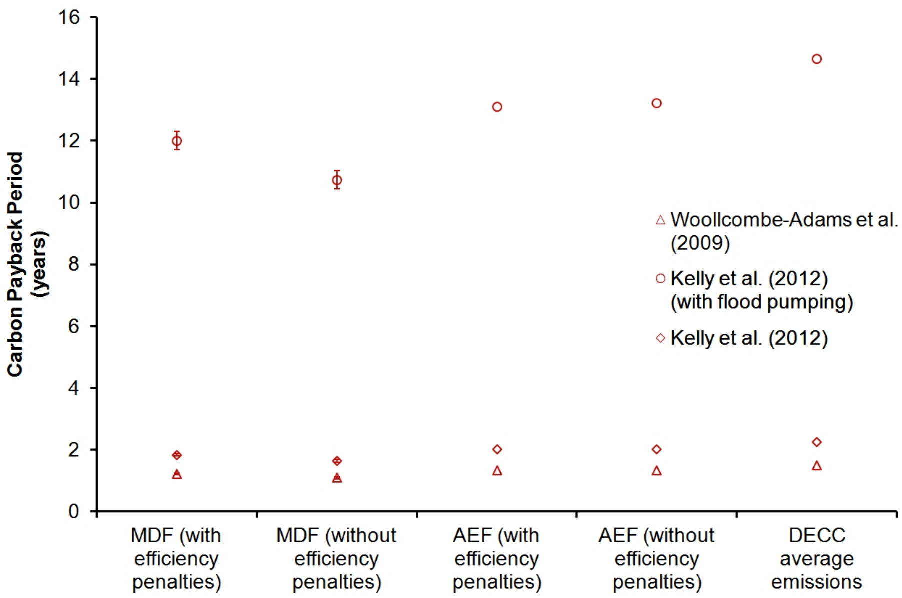


Figure 8.4: Comparison of effect of emissions displacement on carbon payback period for tidal barrages

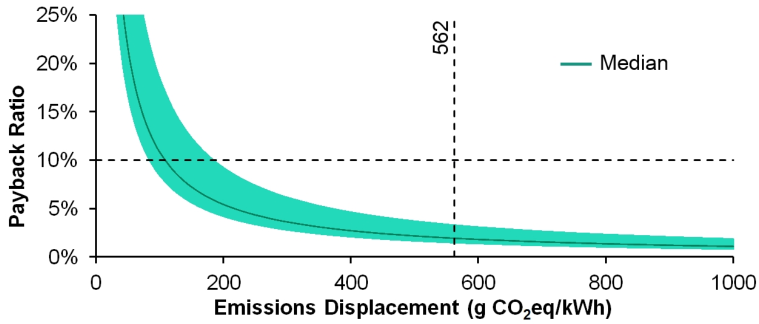
provided the emissions displacement is greater than 250 g CO_2 eq/kWh. For tidal barrage with flood pumping (Kelly *et al.*, 2012), carbon payback should be achieved within a quarter of the design life (30 years) if the emissions displacement does not fall below 225 g CO_2 eq/kWh, with 90 years of operation remaining.

8.2.2 Annual energy output

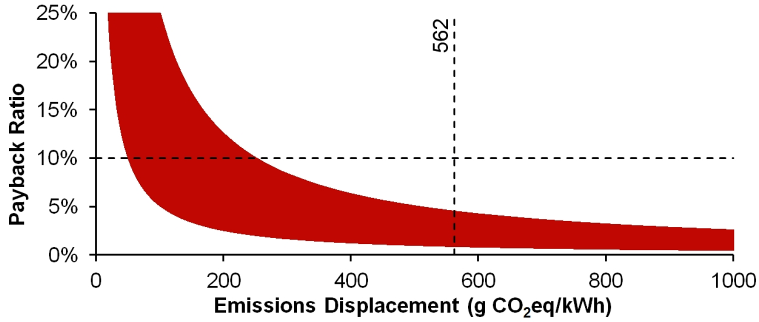
The assumed annual energy production can also affect the carbon payback period. In Section 3.4.1 a number of studies examining the carbon footprint of wind power were reviewed, with their results presented in Table 3.1. These studies assumed that the annual energy production per turbine would range from 3.3 GWh (Guezuraga *et al.*, 2012) to 19.5 GWh (Wagner *et al.*, 2011). Some of this variation will be due to different turbine ratings and greater wind availability offshore, as onshore estimates only ranged from 3.3 to 12.1 GWh (Ardente *et al.*, 2008); however, both of the studies that based their annual output estimates on empirical data found them to be below 4 GWh (Guezuraga *et al.*, 2012; Vattenfall, 2013). The effect of annual production estimates on payback ratio was examined for the studies detailed in Chapter 3, and are illustrated in Figure 8.6.

8.3 Net Reduction in Carbon Emissions

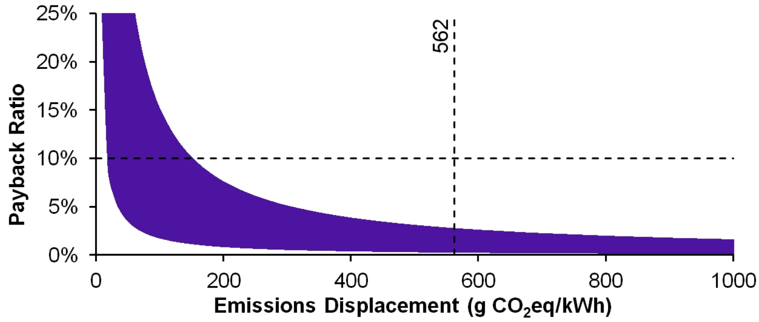
Carbon payback period is a measure of how long it will take for a renewable energy converter to offset the carbon emissions of its life cycle; however, in order to understand the impact of such devices on the carbon emissions of the electricity network the net emissions reductions should be calculated (Equation 8.5). If it is assumed that the emissions displacement, as characterised by the MDF, remains constant over the entire lifetime of the device, then the lifetime emissions displacement can also be calculated from the estimated carbon footprint (CF) and MDF, along with an estimate of the annual energy output (E_{out}/yr), as shown in Equation 8.8. For the purposes of these calculations it is assumed that the lifetime of the wind, wave and tidal stream devices is 20 years, and 120 years for the tidal barrage, unless stated otherwise; it is unlikely that the emissions displacement will remain constant over this period, so the impact of a reduced average lifetime MDF is explored in Figures 8.7, 8.8 and 8.9.



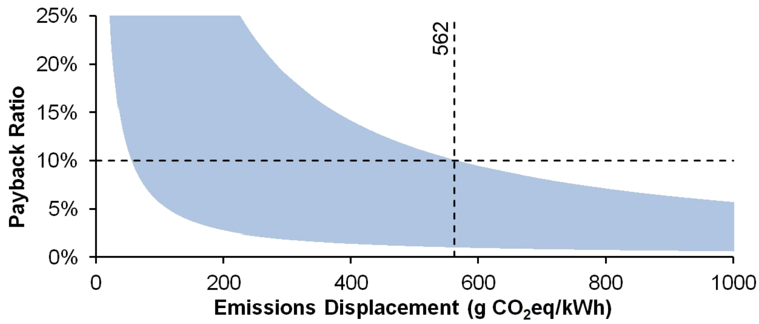
(a) Wind



(b) Wave

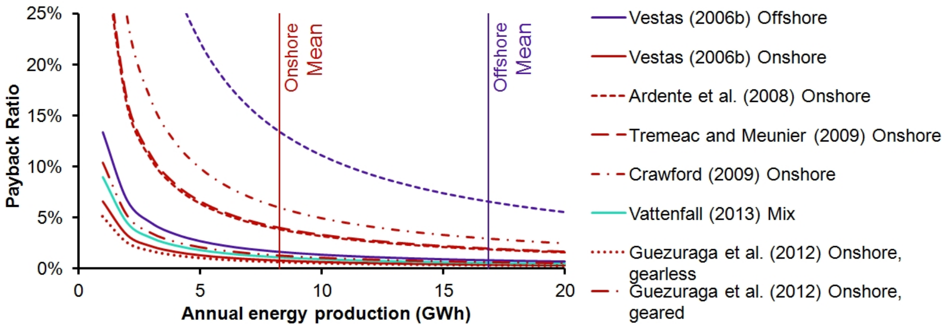


(c) Tidal Stream

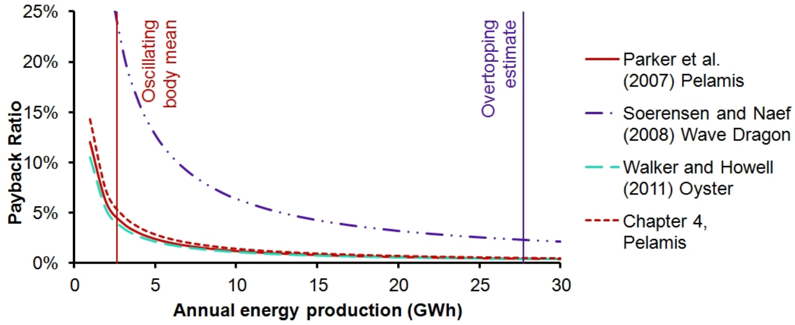


(d) Tidal Barrage

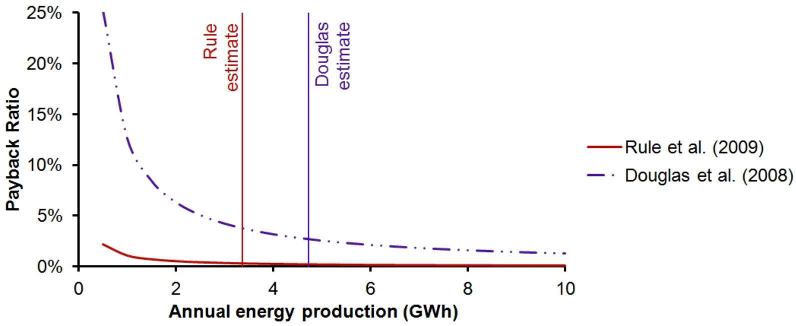
Figure 8.5: Carbon payback ratio as a function of emissions displacement



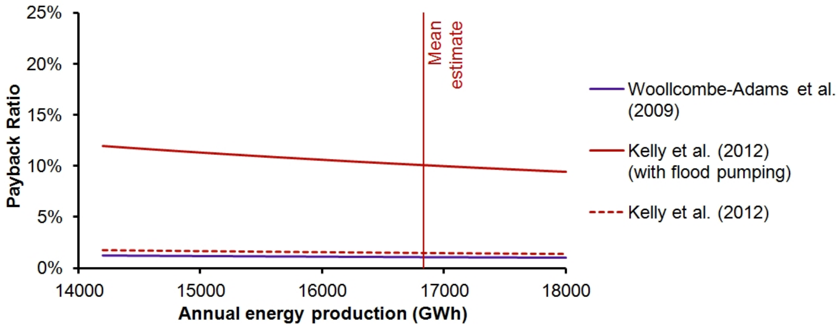
(a) Wind



(b) Wave



(c) Tidal Stream



(d) Tidal Barrage

Figure 8.6: Carbon payback ratio as a function of annual energy output

$$\text{Lifetime displacement} = \text{Annual displacement} \times \text{life} - \text{Lifetime emissions} \tag{8.5}$$
$$\text{Annual displacement} = MDF \times E_{out}/yr \tag{8.6}$$
$$\text{Lifetime emissions} = CF \times E_{out}/yr \times \text{life} \tag{8.7}$$

$$\begin{aligned}\text{Lifetime displacement} &= MDF \times E_{out}/yr \times \text{life} - CF \times E_{out}/yr \times \text{life} \\ &= (MDF - CF) \times E_{out}/yr \times \text{life} \end{aligned} \tag{8.8}$$

It is clear that the net emissions reductions will be greater the bigger the annual energy output (up to the levels where there are significant infrastructure impacts) and the difference between the emissions displacement and the carbon footprint. As can be seen in Figures 8.7 and 8.8 all three of these factors can have a significant effect on estimated net emissions reductions. As long as the emissions displacement is greater than the carbon footprint, the net emissions reduction will be positive, as demonstrated with the carbon payback period. Figure 8.9 examines the impact of a lower emissions displacement, with ranges reflecting the additional variation from energy output and carbon footprint estimates within the published literature (summarised in Chapter 3). Note that the energy output is estimated for a single device, and therefore the emissions savings of the tidal barrage installation are several orders of magnitude greater than for the other technologies.

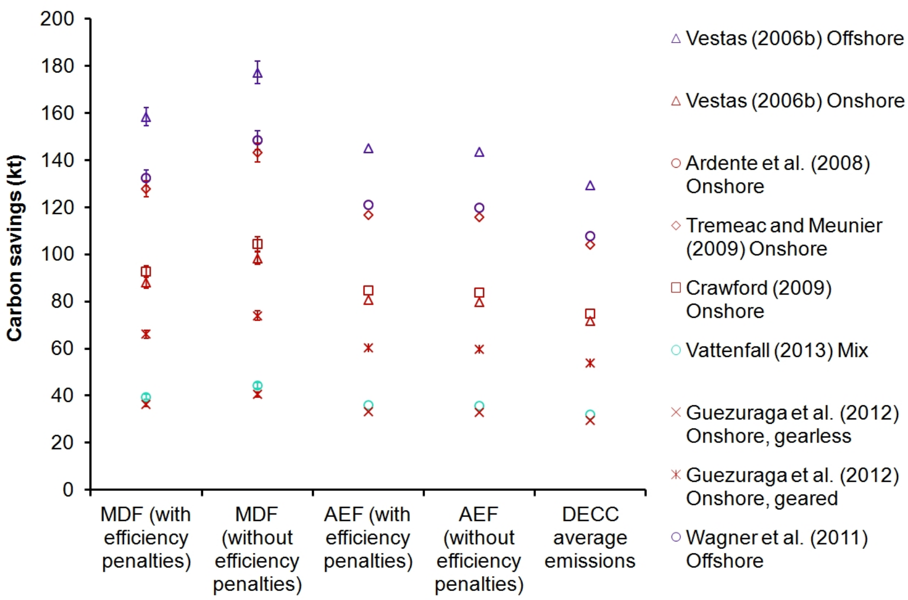


Figure 8.7: Net lifetime emissions reductions for wind power with a range of carbon footprint, annual output and emissions displacement estimates

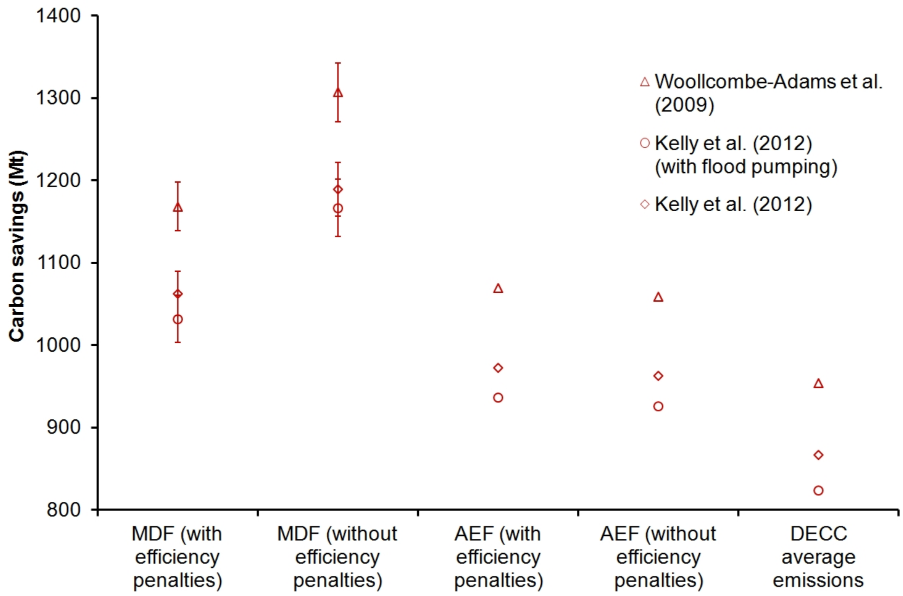


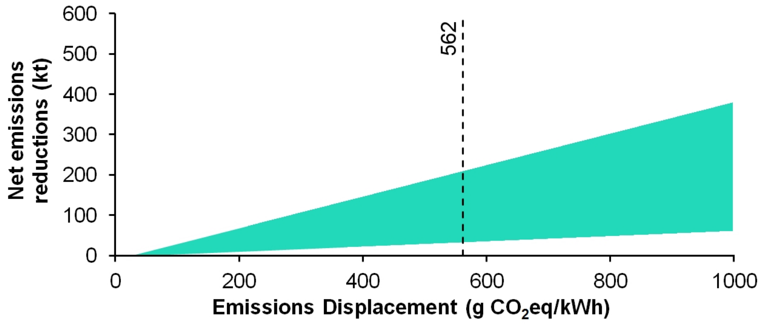
Figure 8.8: Net lifetime emissions reductions for tidal barrages with a range of carbon footprint, annual output and emissions displacement estimates

A further test of the sensitivity of the net emissions reductions to uncertainties in the carbon footprint data is illustrated in Figure 8.10 where the sensitivity analysis of the Pelamis study from Chapter 4 is expanded to examine the sensitivity of the net emissions reductions to uncertainty in data and practitioner assumptions. As has been observed in the previous figures, the assumed annual energy output and design life introduce the greatest variation to the results, with a 10 % change in either resulting in a 10 % change in emissions reductions. In contrast the uncertainty of input data only introduces an error of -3/+1 %, and the choice of recycling allocation method only affects the results by ± 1 %.

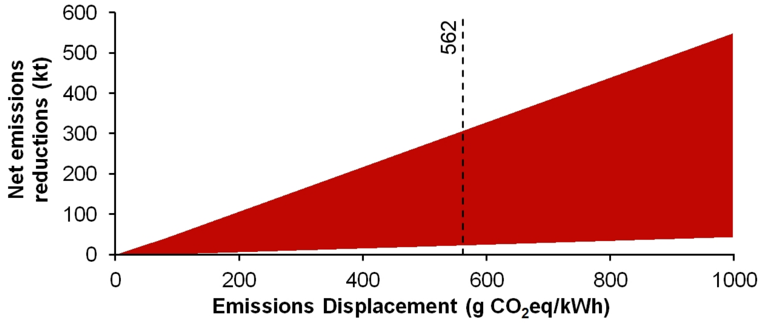
8.3.1 Net carbon reduction of existing wind generation capacity

There is considerable installed capacity of wind power in Great Britain. A review of published metered data (Elexon, 2013a) has found that the average annual energy production from November 2008 to June 2013 was 8328 GWh for the entire fleet, and is likely to have been responsible for some reduction in GHG emissions.

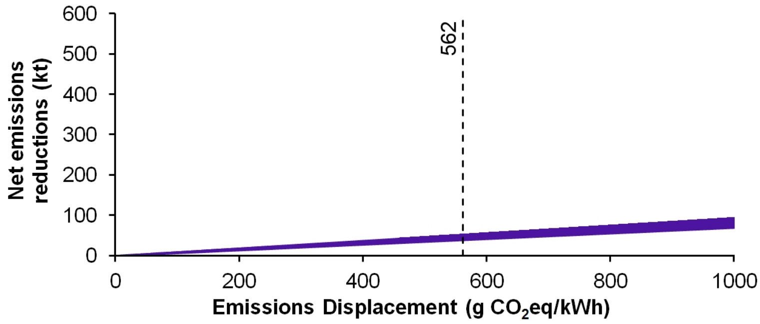
The first step in estimating this net emissions reduction was to identify reasonable carbon footprints for wind farms in Great Britain. The harmonisation process carried out by Dolan and Heath (2012) removed much of the uncertainty caused by differences in practitioner assumptions and methodology, but the carbon footprint estimates are still based on assumed energy outputs that may not reflect the actual energy production of wind farms in Great Britain. In order to represent the British situation more accurately, the harmonised carbon footprints



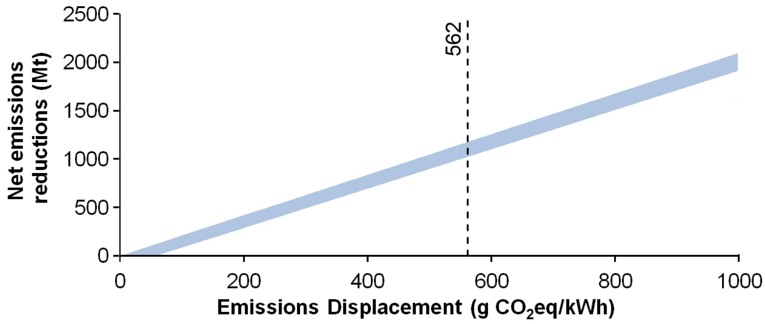
(a) Wind



(b) Wave



(c) Tidal Stream



(d) Tidal Barrage

Figure 8.9: Net emissions reduction as a function of emissions displacement

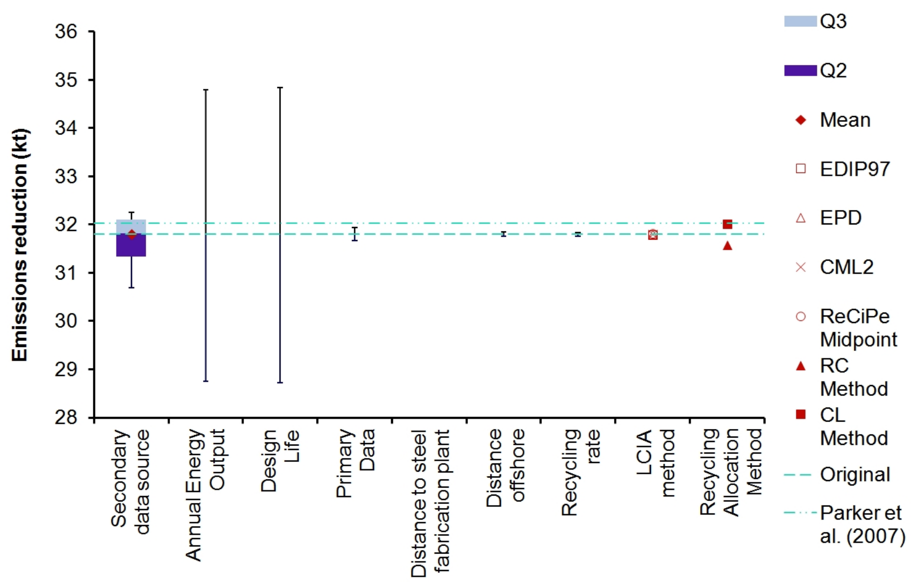


Figure 8.10: Sensitivity analysis of the net emissions reductions for the Pelamis wave energy converter

published by Dolan and Heath (2012) were adjusted to represent the historical average capacity factor in the UK.

The capacity factor of power generation is the annual output divided by the maximum potential annual output (generation at nameplate capacity for the whole year). Dolan and Heath (2012) assumed standard capacity factors of 30 % for onshore wind and 40 % for offshore wind. Average capacity factors for the UK were calculated from historical data (MacLeay *et al.*, 2013), and are shown in Table 8.1. It can be seen that these are significantly lower than those used in the LCA Harmonization Project. In order to adjust the carbon footprint (CF) estimates, the ratio of harmonized capacity factor to actual capacity factor was calculated, so that it could be applied according to Equation 8.9. The average ratios were taken for offshore and onshore farms, and where the study considered a mix of onshore and offshore, the capacity ratio was calculated as a weighted average of the two based on the installed capacity in the UK, as shown in Table 8.1.

$$CF_{\text{new}} = CF_{\text{original}} \times \frac{\text{Capacity Factor}_{\text{original}}}{\text{Capacity Factor}_{\text{new}}} \tag{8.9}$$

The net emissions reduction over the period from November 2008 to June 2013 was then estimated using Equation 8.10, derived from Equation 8.8, and is illustrated in Figure 8.11. The ranges include the uncertainty ranges for emissions displacement estimates detailed in Chapters 6 and 7, annual energy production (E_{out}/yr) was taken to be 8328 GWh, and the 'time' was 4.66 years.

Onshore	**2008**	**2009**	**2010**	**2011**	**2012**
Installed Capacity (MW)	2824	3477	4045	4638	5893
Generation (GWh)	5788	7553	7140	10384	12121
Capacity factor	23 %	25 %	20 %	26 %	23 %
Ratio	1.28	1.21	1.49	1.17	1.28
Average ratio	1.29				
Offshore					
Installed Capacity (MW)	586	951	1341	1838	2995
Generation (GWh)	1305	1754	3044	5126	7463
Capacity factor	25 %	21 %	26 %	32 %	28 %
Ratio	1.57	1.90	1.54	1.26	1.41
Average ratio	1.54				
Mix					
Average ratio	1.35				

Table 8.1: Calculating wind capacity factor from data published by MacLeay *et al.* (2013)

$$\text{Displacement} = (MDF \times \text{time} - CF \times \text{life}) \times E_{out}/yr \qquad (8.10)$$

This graph shows that the existing installed capacity has displaced enough greenhouse gas emissions in the last four and a half years to offset its own carbon footprint and continue to reduce the emissions associated with power generation. The best estimate from the work presented in this thesis is that the net emissions reduction of wind power was 18 - 20 Mt CO_2 eq from November 2008 to June 2013. The actual figure is likely to be slightly higher than this, as it was assumed that there was no generation before November 2008, and therefore the total lifecycle carbon footprints were subtracted from the emissions savings over this period. This reduction represents around 3 % of the total GHG emissions from power generation over this period, which were reported to be approximately 150 Mt CO_2 eq/yr (DECC and National Statistics, 2014).

8.4 Energy Payback Period

While the main driver for the development of new renewable generation installations is currently the reduction of GHG emissions, they must also produce a viable energy return on energy investment and have an energy payback period significantly shorter than their design life. This is of particular interest when considering renewable energy as an alternative to conventional fossil fuels, with supplies expected to become increasingly scarce and require more energy to extract and refine.

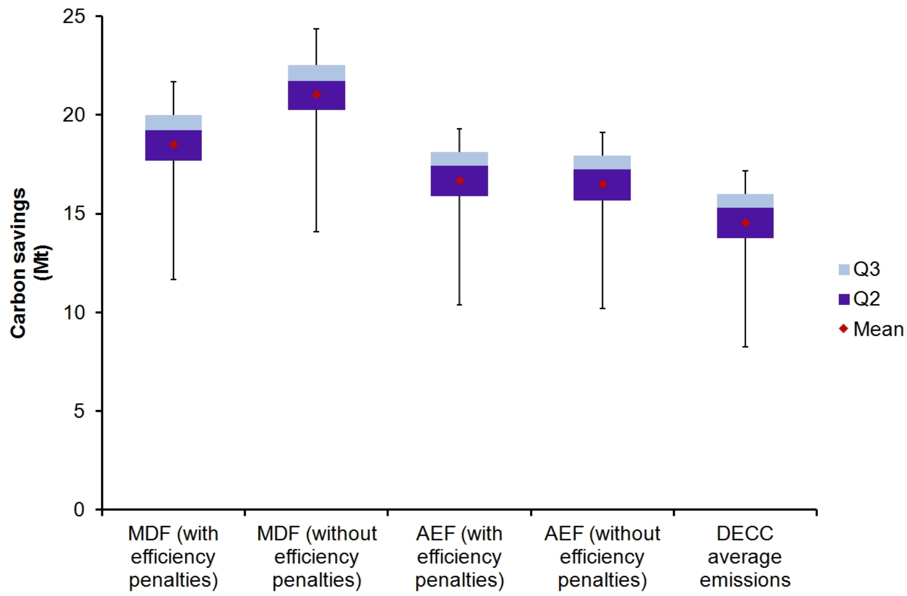


Figure 8.11: Estimates of net emissions reductions of the installed wind capacity in Great Britain from November 2008 to June 2013

The energy payback period is defined as the length of time required to recover all of the energy invested during the life cycle of the renewable energy generator. It is conventionally calculated as the ratio between the primary embodied energy and the annual energy output - comparing primary energy consumption with electricity generation (Equation 8.11). There is a suggestion, however, that this comparison is unfair, and that a more consistent index would be to consider the ‘primary energy payback period’, where the output electrical energy is converted to the equivalent primary energy required to generate this electricity (Tremeac and Meunier, 2009). The limitation of the latter method is that the conversion of output electricity into a primary energy equivalent introduces additional uncertainties, as assumptions need to be made about the type of generation displaced by renewable energy converters - similarly to a calculation of the GHG emissions displacement. In order to avoid this additional uncertainty, this chapter does not consider the primary energy payback period, but there is scope for further work to examine this and develop a truer picture of the energy payback of variable-output renewable generation.

$$\text{Payback period} = \frac{\text{Lifetime energy Consumption}}{\text{Annual energy production}} \tag{8.11}$$

$$\text{Lifetime energy consumption} = EE \times E_{out}/yr \times \text{life} \tag{8.12}$$

$$\text{Annual energy production} = E_{out}/yr \tag{8.13}$$

$$\text{Payback period} = EE \times \text{life} \tag{8.14}$$

The energy payback period can be directly calculated from the cumulative energy demand per unit of output power, or embodied energy (EE), calculated from a life cycle assessment (Equation 8.14). As embodied energy is conventionally reported in kilojoules per kilowatt-hour, however, this needs to be divided by 3600 to convert it into a dimensionless ratio, so Equation 8.14 becomes:

$$\text{Payback period} = \frac{EE}{3600} \times \text{life} \tag{8.15}$$

In comparison with the calculation of carbon payback there are fewer opportunities for uncertainty to be introduced to an energy payback calculation if the displaced primary energy consumption is not considered. Variations in estimates of energy payback period arise from uncertainty in the embodied energy estimates (including the assumed annual electricity production), and the assumed generator lifetime. Figure 8.12 shows the range of energy payback periods that have been calculated for typical wind, wave and tidal power converters, with embodied energy data taken from the studies summarised in Chapter 3. As most of the embodied energy estimates considered in this analysis were based on similar estimated design lives for the generators, these were assumed to be 20 years for wind, wave and tidal stream devices, and 120 years for tidal barrage. It can be seen that there is significant variation in the estimated payback periods, although they are all significantly shorter than the design lives, and most are less than two years. The exception is the tidal barrage installation, where embodied energy estimates from Kelly *et al.* (2012) produce energy payback periods of 5 and 32 years for a Severn Barrage excluding or including flood pumping respectively. As the design life used in this study is 120 years, the longer payback period is not as much of a problem as it would be with other types of energy converter.

It is likely that the range of estimates produced for each type of renewable energy converter is attributable to variations in the technology, embodied energy analysis methodology, and uncertainties in the calculation. In order to examine the impact of methodological variations and data uncertainties on the embodied energy, the sensitivity analysis for the Pelamis LCA (presented in Chapter 4) was modified to examine the sensitivity of embodied energy estimates. It can be seen in Figure 8.13 that very little variation is introduced by changes in the design life, as these cancel each other out in the calculation, and instead the uncertainty of secondary data introduces the most significant errors with the interquartile range being +30/-20 % on the mean estimate. The uncertainty of secondary LCI data is an ongoing problem with LCA, as it affects all calculation results. Significant variation is also introduced by the different recycling allocation methods.

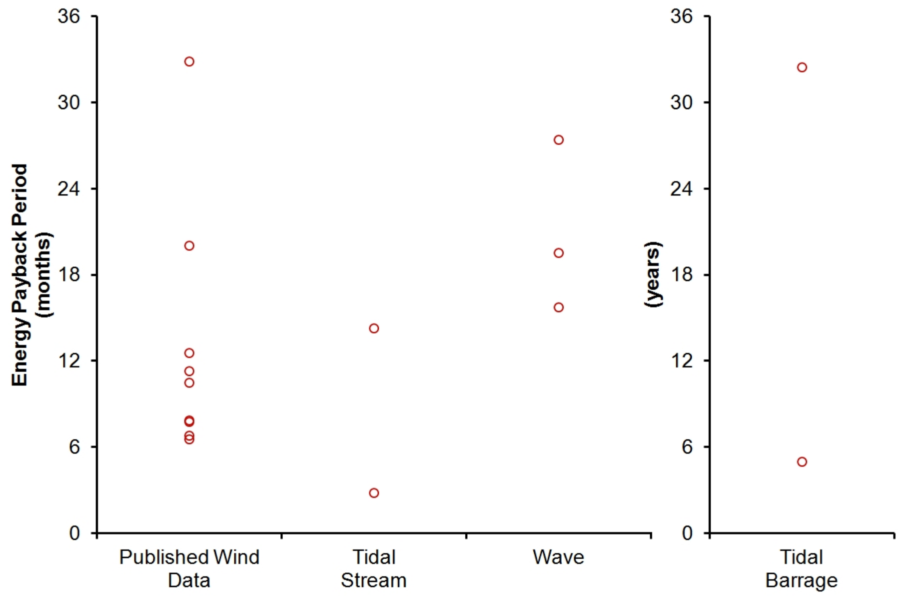


Figure 8.12: Energy payback periods for a range of variable-output renewable energy converters

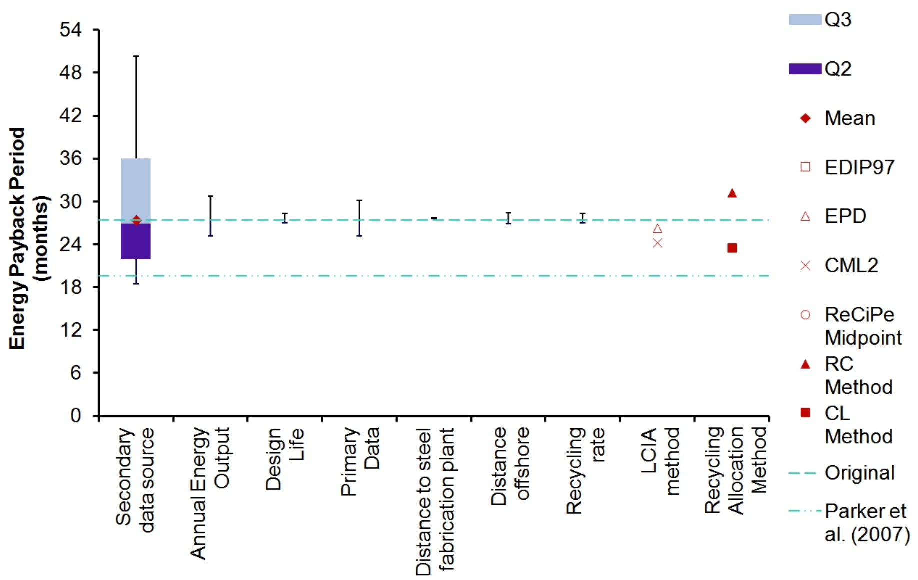


Figure 8.13: Sensitivity analysis for energy payback for the Pelamis WEC

8.4.1 Annual energy output

As with the carbon payback period and net reduction in emissions, the annual energy output can significantly affect the energy payback period of a renewable energy generator. Figure 8.14 illustrates this by examining the energy payback ratio (energy payback period as a proportion of design life) for a range of different technologies and embodied energy estimates. Devices with a lifetime of 20 years should be able to payback within two years - a payback ratio of 10 %. The mean annual energy output per device has been calculated from the values used in published studies, and it can be seen that onshore wind turbines and wave energy converters are close to exceeding a payback ratio of 10 % at this value. This highlights that renewable generators must be correctly sited to produce a good energy output and achieve energy payback.

8.5 Energy Return on Investment

Another metric commonly used to demonstrate the viability of electricity generation, particularly for renewable energy converters, is the energy return on investment (EROI). This is the ratio of the usable output energy to the energy consumption of the product life cycle (Equation 8.16) and it can be calculated from the embodied energy estimate by Equation 8.19.

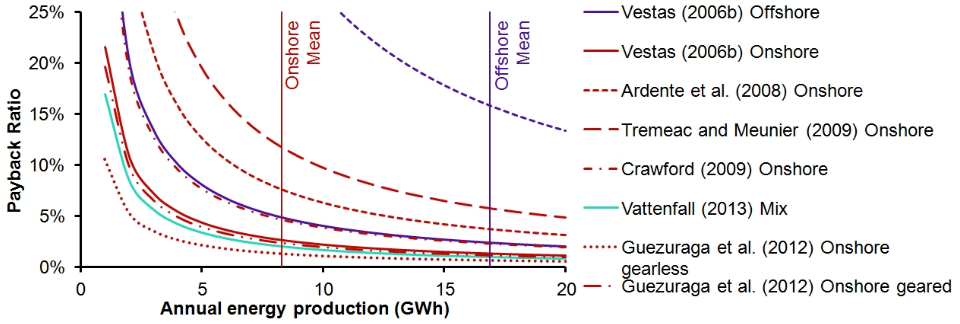
$$EROEI = \frac{\text{Lifetime energy output}}{\text{Lifetime energy consumption}} \tag{8.16}$$

$$\text{Lifetime energy output} = E_{out}/yr \times \text{life} \tag{8.17}$$

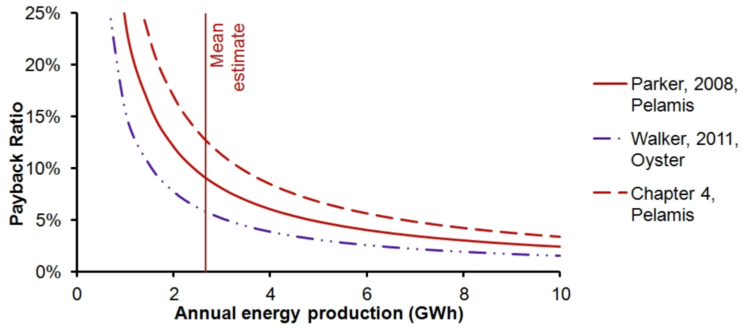
$$\text{Lifetime energy consumption} = EE \times E_{out}/yr \times \text{life} \tag{8.18}$$

$$\begin{aligned} EROEI &= \frac{E_{out}/yr \times \text{life}}{EE \times E_{out}/yr \times \text{life}} \\ &= \frac{1}{EE} \\ \text{Correct to make EE dimensionless} & \\ &= \frac{3600}{EE} \end{aligned} \tag{8.19}$$

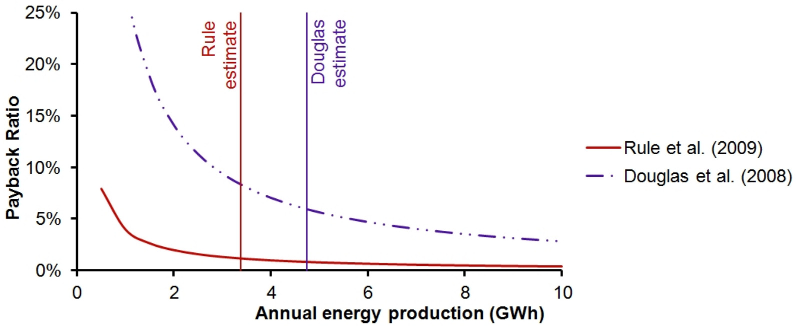
The range of estimated EROI values that can be calculated from published estimates of embodied energy is shown in Figure 8.15. It can be seen that these vary considerably, with the largest discrepancies between the two studies for tidal stream technologies, which are based on very different assumptions of design life. In order to set these values in context, they are further compared to the EROI for some typical fuels used for power generation in the USA in Figure 8.16 (Murphy and Hall, 2010). It can be seen that one of the estimates for the EROI of a tidal stream turbine (from the embodied energy estimate published by Rule *et al.* (2009))



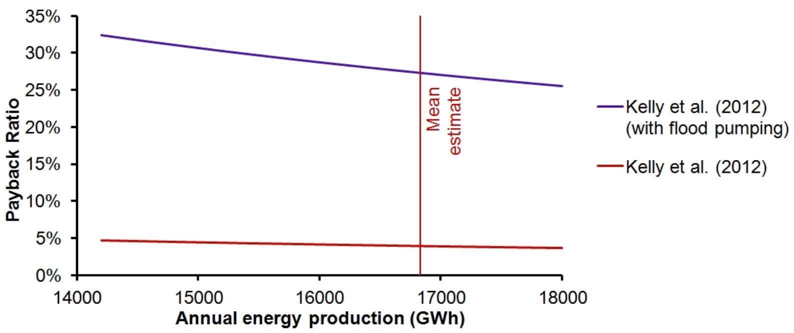
(a) Wind



(b) Wave



(c) Tidal Stream



(d) Tidal Barrage

Figure 8.14: Energy payback ratio as a function of annual energy output

appears to be a significant overestimate, but in general the energy return on investment for these technologies is comparable to natural gas and nuclear energy.

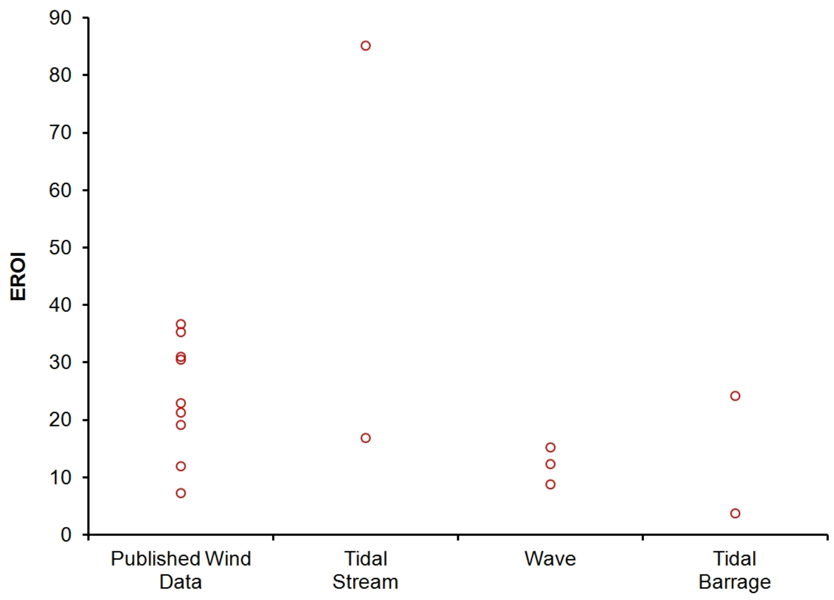


Figure 8.15: Range of EROI values calculated from published estimates of embodied energy

These estimates of EROI are based solely on the calculated embodied energy. Uncertainties in this value, either from the range of methodological choices and assumptions made during the calculation, or from the estimated annual energy production, will introduce uncertainties into the calculated EROI. The magnitude of these uncertainties has been examined by both carrying out a sensitivity analysis of the EROI of the Pelamis WEC (based on the sensitivity analysis presented in Chapter 4) and examining the effect of changing the assumed annual energy production per device for the embodied energy estimates examined for Figure 8.15.

Figure 8.17 shows the effect of uncertainties in the calculated embodied energy of the Pelamis on the EROI. It can be seen that, again, the uncertainty of the source LCI data, and different choices in recycling allocation method have the greatest impact on the results, with the energy return on investment comparing favourably with conventional fossil fuels.

The actual delivered energy from renewable generators will not only be dependent on their efficiency, but also on the weather, environment, local electricity grid topology and network power flows. Figure 8.18 examines how changes in the annual energy production can affect the EROI. It can be seen that the actual energy output will have a significant effect on the EROI. Furthermore, although it appears that onshore wind might have a higher EROI than offshore wind, onshore wind is unlikely to achieve such a high energy output, and therefore these results do not show definitively which will produce the better return.

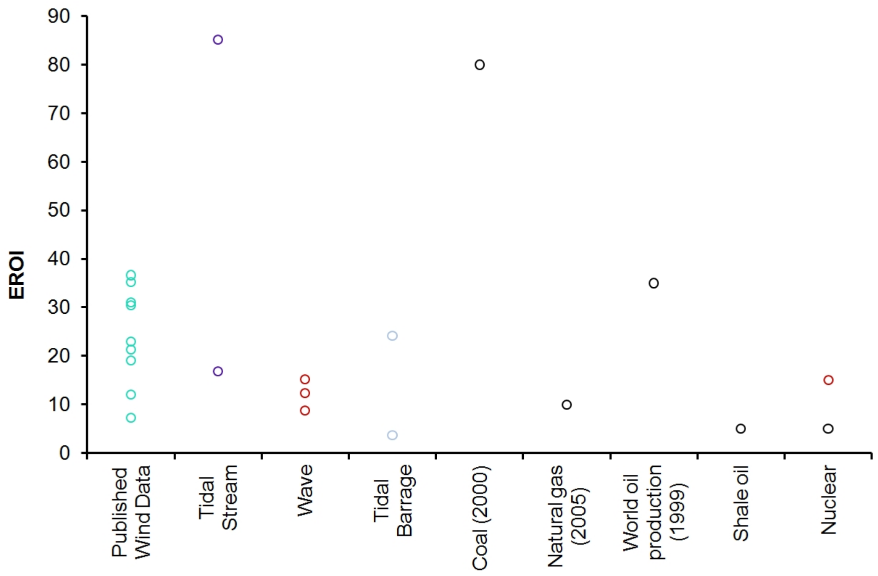


Figure 8.16: Comparison of calculated EROI with values for fuels in the USA (Murphy and Hall, 2010)

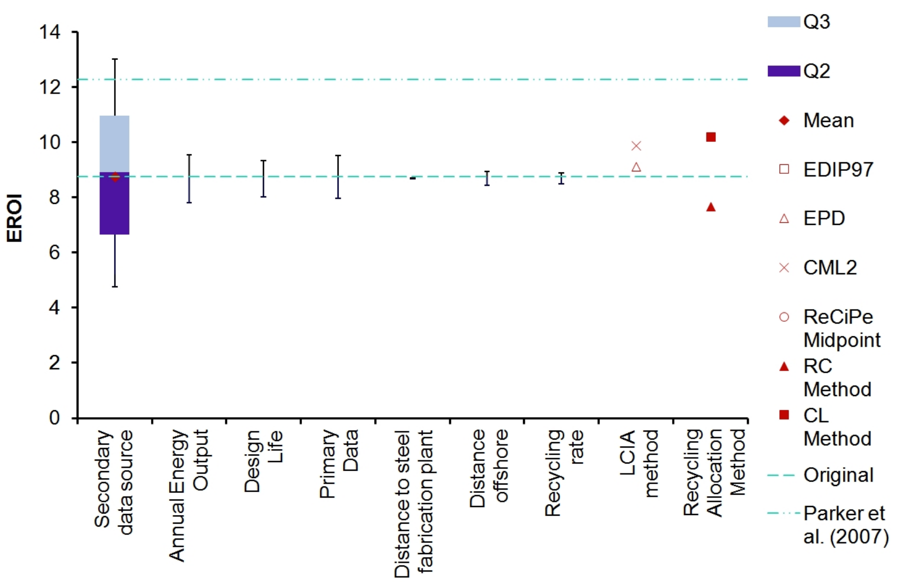
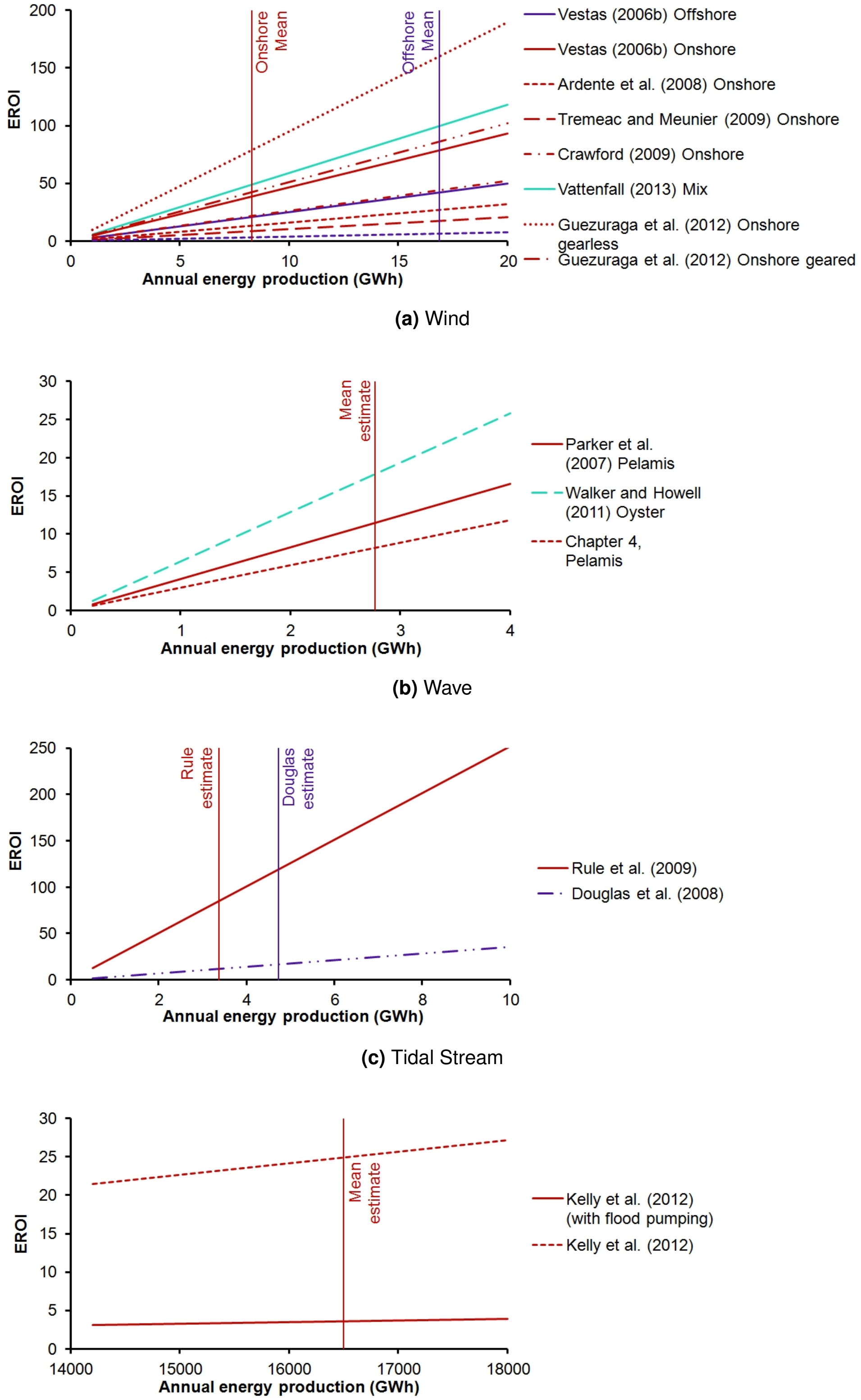


Figure 8.17: Sensitivity analysis of the EROI of the Pelamis WEC to data uncertainties, methodological choices and practitioner assumptions



(a) Wind

(b) Wave

(c) Tidal Stream

(d) Tidal Barrage

Figure 8.18: EROI as a function of annual energy output

8.5.1 Net energy output of existing wind generation capacity

In order to develop a realistic picture of the net energy return of the existing installed wind in Great Britain, the same values as those applied to calculate the net reduction in emissions of British wind from November 2008 to June 2013 were used to calculate the net energy output (see Section 8.3.1). The calculation was based upon Equations 8.21 and 8.22, with original embodied energy, annual output and capacity factor values taken from the original selection of references detailed in Section 3.4.1.

$$\text{Net energy return} = \text{Energy output} - \text{Lifetime energy consumption} \quad (8.20)$$

$$= E_{out}/yr \times \text{years} - EE_{\text{new}} \times E_{out_{\text{original}}} \times \text{life}$$

Correct to make EE dimensionless:

$$= E_{out}/yr \times \text{years} - \frac{EE_{\text{new}}}{3600} \times E_{out_{\text{original}}} \times \text{life} \quad (8.21)$$

$$EE_{\text{new}} = EE_{\text{original}} \times \frac{\text{Capacity Factor}_{\text{original}}}{\text{Capacity Factor}_{\text{new}}} \quad (8.22)$$

The results are shown in Figure 8.19. It can be seen that the existing wind fleet in Great Britain not only achieved energy payback, but also generated a further 38 TWh of energy, around 8.3 TWh per year.

8.6 Conclusions

This chapter draws together the work presented in the rest of the thesis to examine the reliability of carbon and energy payback estimates for variable-output transmission-connected renewable generation in Great Britain, taking into account uncertainties in the underlying carbon footprint, embodied energy, GHG emissions displacement and energy output estimates. It found that these values were most sensitive to uncertainties in input LCI data, choices of recycling allocation method and estimates of annual energy output. The uncertainties introduced by the first two issues should decrease as more LCAs are carried out, but the latter is dependent upon the precise installation location of the energy converter. As current estimates are mostly based on forecast values, it is important to ensure that new devices are sited correctly to achieve their maximum potential energy output.

The review presented here shows that carbon payback should be achieved within two years for wind, wave and tidal stream devices under current conditions. The payback period for tidal barrage is longer, in the region of 10 % of its 120-year design life, but there will still

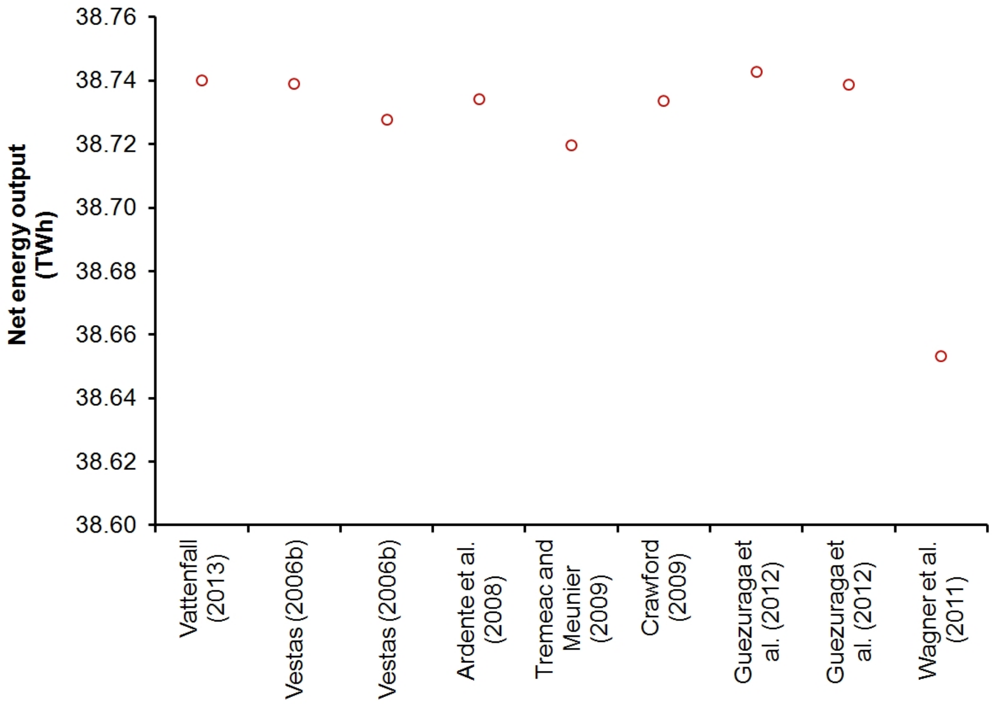


Figure 8.19: Net energy output of wind power in Great Britain from November 2008 to June 2013, based on published carbon footprint estimates

be a significant net emissions reduction. While it is likely that expected moves away from coal-fired generation towards lower-carbon energy sources will decrease the emissions displacement of variable-output renewables, the results presented in this chapter show renewable energy converters will still have a reasonable carbon payback period as long as the marginal emissions displacement does not fall below 250 g CO_2 eq/kWh. It seems unlikely that the emissions displacement will reach this value in the near future, as it is approximately half the current emissions of gas-fired generation: to achieve this, fossil fuels would need to supply less than half of British electricity demand, or be fitted with effective carbon capture and storage - both of these solutions require significant investment and development which is likely to take time. Once this threshold is achieved, it is likely that there will already be a significant installed capacity of renewable energy converters of all types, including wind, wave and tidal, and new cutting-edge technologies will need to demonstrate even better low-carbon credentials than those that currently exist.

The net GHG emissions reductions are also of interest when comparing different generation technologies, but are subject to considerable uncertainties. This analysis found that estimates of carbon payback periods provided clearer evidence of a net return on investment, as they were significantly less sensitive to estimates of design life or annual energy output; once the payback period has been exceeded, then the device will be reducing emissions on the network, irrespective of fluctuations in marginal displacement factor and any small changes in design life and energy production. Such limitations do not apply to a historical analysis of emissions reductions over a given time period, so the net emissions reduction of the current installed wind capacity in Great Britain was estimated for the period from November 2008 to June 2013. It was found to be approximately 19 Mt CO_2 eq, a reduction of 3 % on the total emissions of power generation over that time. During this period, wind was responsible for an average annual energy production of 3 % of UK electricity demand (calculated from MacLeay *et al.* (2013)).

Although GHG emissions reduction is the principal driver for the development of renewable generation, such devices must also achieve energy payback within a fraction of their design life in order to be viable. This energy payback, and the corresponding return on investment, was found to be most sensitive to changes in annual energy production, and uncertainties in the input data used to calculate the lifetime embodied energy of the device. Despite this, estimates suggest that most renewable energy converters should achieve energy payback within two years (except for tidal barrage, where the payback period is significantly longer).

Another common metric for measuring the energy performance of a generator is the Energy Return on Investment (EROI). Again, this was found to be sensitive to estimates of annual energy production, uncertainty of LCI data, and embodied energy calculation methods, but the range of estimated EROI for renewable power generation technologies was found to be comparable to published estimates for natural gas and nuclear power. The net energy output of wind power on the British grid from November 2008 to June 2013 was found to be around 38 TWh.

It is thought that the primary energy payback period is a more accurate metric of energy payback time than current estimates comparing electricity output with primary energy consumption. In order to understand the displaced primary energy consumption of electricity generation from variable renewable energy converters, however, a marginal analysis is required. The analysis method applied in Chapters 6 and 7 could be adapted in future work to consider the displacement of primary energy consumption, in order to allow a more accurate picture to be developed of the true energy payback period and return on investment of renewable energy converters.

The analysis presented in this chapter, and throughout this thesis, provides strong evidence that the existing installed wind capacity in Great Britain has already achieved carbon and energy payback, and will continue to contribute to reducing the emissions from power generation.

Chapter 9

Conclusions

This chapter presents an overview of the findings of this research and draws overall conclusions. The implications of these and the contribution to knowledge are also reviewed, and the central hypothesis is answered. Opportunities for further work are also discussed.

9.1 Thesis Summary

This research was a detailed examination of the reliability of carbon and energy payback period estimates for variable-output renewable generators, using Great Britain as a case study. The first section (Chapters 2, 3 and 4) concentrated on the calculation of carbon footprint and embodied energy estimates that are used to represent the 'investment' aspect of any carbon or energy payback calculation. The second section (Chapters 5, 6 and 7) then examined the complexities of estimating the marginal greenhouse gas (GHG) displacement of variable-output renewable generation, representing the 'returns' in the carbon payback. Finally, Chapter 8 brought these two together to examine the implications of these underlying uncertainties on the estimates of carbon and energy payback periods.

9.1.1 Variability of carbon footprint and embodied energy estimates for power generation

Many studies have been published that examine the environmental impacts, and in particular the carbon footprint and embodied energy, of power generating technologies; however, the findings of these vary widely, making it difficult to draw definitive conclusions. A review of existing studies was carried out in Chapter 3, and it was found that these variations are introduced through differences in the studied technology, underlying performance assumptions and applied calculation methodologies, as well as uncertainties in raw data. Furthermore, the quality of the publications also varies, with a lack of transparency limiting the opportunities to identify where discrepancies in results might arise. Considerable work has been done by the National Renewable Energy Laboratory (NREL) in the USA to reduce some of the variations in published carbon footprints by harmonising methodologies and assumptions, but over 85 % of the studies considered in the initial review were discarded because they did not meet the basic criteria for quality, relevance and transparency (NREL, 2013b). Harmonisation was found to reduce the variation in results, with a particularly narrow distribution of calculated carbon footprints for wind power. The broader range of harmonised results for other energy sources were attributed to variations in technology vintage, efficiency and fuel quality, which suggests that process-based LCAs of similar generation technologies and fuels may not be too severely affected by practitioner assumptions (Dolan and Heath, 2012).

A further extended review of existing published studies of carbon footprints and embodied energy is presented in Chapter 3. Variations in published estimates were found to be largely due to differences in boundary definition, age of data sources, rigour of the analyses and assumed power outputs or capacity factors.

Significant gaps were also identified in the existing literature, with very few published studies for marine energy converters, run-of-river hydro plants, and gas-cooled nuclear reactors, in particular. The latter is an old technology that is prevalent in Great Britain, but will eventually be replaced with pressurised water reactors; however the carbon footprint and embodied energy are of interest because these generators supply a high proportion of British electricity, and are likely to continue to do so for some time. Marine energy is an emerging sector with many different types of technology, so there is a call for more studies to be published to corroborate the findings of existing studies and inform future developments.

9.1.2 Environmental impacts of a wave energy converter

Technologies to extract energy from the waves and tides continue to develop, with marine energy having the potential to supply a significant proportion of UK demand; however, the environmental impacts of these devices are relatively poorly understood, with few published studies examining the carbon footprint and embodied energy, and none considering the broader environmental impacts. In order to address this problem, a full life cycle assessment of the Pelamis wave energy converter was carried out. Every stage of the device life cycle was considered, including materials; manufacturing processes; transport; sea vessel operations for installation, maintenance and decommissioning; and disposal.

One of the benefits of such a comprehensive analysis is that it can help to identify opportunities for further reducing the environmental impacts. The analysis found that the greatest impacts across all categories were due to steel conversion processes (mostly in a blast furnace) and sea vessel operations. Reducing the mass of steel in the device or increasing the recycled content would have significant environmental benefits across all categories except radioactive waste.

Another opportunity for reducing the overall environmental impacts lies in reducing the requirements for sea vessel operations: alternative installation locations could have significantly lower environmental impacts, and improvements in the reliability of the device could reduce the requirement for sea vessel operations for maintenance.

The carbon footprint of the Pelamis was found to be 27 g CO_2 eq/kWh and the embodied energy 411 kJ/kWh. All environmental impacts were found to be low when compared to conventional power generation, particularly from fossil-fuels, and the energy return on investment was also found to be competitive.

9.1.3 Reliability of carbon footprint and embodied energy estimates

The ranges of carbon footprint and embodied energy estimates considered in Chapter 3 do not necessarily reflect their accuracy. Even with the statistical results provided by the LCA Harmonization Project, equal weight is given to multiple reported values from the same study, which could introduce a bias towards particular assumptions or methodological choices. In order to assess the reliability of these values, this research included a comprehensive review of areas where uncertainty may be introduced to carbon footprint and embodied energy estimates, and examined the relative impacts of these.

Several key methodological limitations and practitioner assumptions were identified that can introduce uncertainty into carbon footprint and embodied energy estimates of power generation. These were examined in a case study life cycle assessment of the Pelamis wave energy converter, and, in order of decreasing impact, the most significant were found to be: the uncertainty introduced by LCI datasets; the choice of recycling allocation method; the choice of impact assessment method or characterisation factors; the annual energy output and design life; and the assumption of manufacturing and installation locations. It is expected that these sources of uncertainty will be similar for other types of renewable generation but, despite this, it appears that estimates of carbon footprint for wind power are converging over time, which suggests that the most significant variation for power generation might be introduced by differences in technology, particularly for coal, nuclear and marine, and site-specific impacts, such as deforestation.

9.1.4 Carbon footprint and embodied energy in context of wider environmental impacts

One of the concerns surrounding the focus on carbon footprinting, in particular, is that potential trade-offs with other environmental impacts are being neglected. The analysis of the Pelamis wave energy converter, discussed in Section 9.1.2, allowed the carbon footprint and embodied energy to be examined within the broader scope of impacts that can be calculated in a life cycle assessment. This found that all categories of environmental impact were dominated by steel processing, particularly in a blast furnace, and the operation of sea vessels. Furthermore, a comparison with published impacts for other types of generation found that the Pelamis performed better than fossil-fuelled generation in most categories. It is, therefore, possible to conclude that measures to reduce the carbon footprint and embodied energy of this device are also likely to reduce other environmental impacts, or, at the very least, maintain the emissions of other harmful substances well below the levels set by fossil-fuelled generation. It is also likely that such a conclusion will apply to similar renewable energy converters formed predominantly from steel, including wind turbines and some tidal energy converters.

9.1.5 Emissions displacement of variable renewable generation in Great Britain

There is currently no reliable estimate for the marginal emissions displacement of renewable energy such as wind, wave and tidal power - it is often approximated as the system-average emissions or the marginal emissions of demand-side fluctuations (see Chapter 5). Such data is necessary to calculate the carbon payback period of variable-output renewable generators, and is a function of the types of generator that respond to fluctuations in the supply; therefore a function of the topology of the network on which the renewable generator is operating. The system-average emissions are likely to be an underestimate in Great Britain, as they include a significant contribution from low-carbon nuclear generation, which is known not to respond to short-term fluctuations in supply or demand. A number of studies of other networks around the world have shown that the marginal emissions of both supply- and demand-side fluctuations can differ significantly from the average emissions of the corresponding network.

The analysis presented in Chapters 6 and 7 addressed the key faults in existing work in this area, analysing historical data of wind power on the grid in Great Britain - wind power is currently the only type of variable-output renewable generation reporting to the system operator. The initial calculation, presented in Chapter 6, was based upon aggregated output data, and found the marginal displacement factor (MDF) to be 0.63 kg CO_2 eq/kWh, averaged over November 2008 to June 2013, slightly lower than the calculated marginal emissions factor of total generation at 0.65 kg CO_2 eq/kWh, but higher than the system-average emissions rate of 0.51 kg CO_2 eq/kWh for the same period. This shows that wind power is offsetting a mixture of generation, likely to be primarily coal and CCGT, and that current payback period estimates based on the UK-average emissions for electricity may well be an underestimate. Disaggregation of the data to investigate temporal and other trends found that both the MEF and MDF were consistently higher than the system-average emissions. They were also found to be highly influenced by the generation mix - an effect of fluctuations in the relative prices of coal and gas. It is expected that the electricity system in Great Britain will respond in a similar way to fluctuating output from marine energy converters, so that their carbon emissions displacement will also be higher than the system-average. This is significant in this emerging sector where the carbon footprint of the devices is often higher than for a typical wind turbine, and, therefore, the carbon payback period is more critical. This analysis was, however, limited by not considering the impact of fluctuating power output on the efficiencies of conventional plant.

9.1.6 The impact of efficiency penalties in conventional plant on the emissions displacement of variable renewable generation

The effect of efficiency penalties of coal and CCGT power stations on both the marginal displacement of variable renewable generation and the marginal emissions of total system generation in Great Britain were examined in Chapter 7. This built upon the methodology developed in Chapter 6, again based on reported data for wind power to represent the variable renewable generation, but used detailed individual power output profiles for each coal and CCGT power station, derived from published BMRA messages. The GHG emissions were then calculated from part-load emissions intensity curves.

This analysis found that efficiency penalties do have a significant impact on both the marginal displacement of wind power and the marginal emissions of total generation, decreasing them by 11 % and 7 % respectively for the period from November 2008 to June 2013. However, at 0.56 and 0.60 kg CO_2 eq/kWh for the MDF and MEF respectively, both estimates remain higher than the calculated system-average emissions of 0.52 kg CO_2 eq/kWh. Again, it is expected that such values will be similar for marine energy converters, so the effect of efficiency penalties must be taken into consideration when estimating the carbon displacement.

An investigation of the trends over the studied time period found that both the estimated MDF and MEF were converging with the calculated system-average emissions factor over time, towards a value of 0.5 kg CO_2 eq/kWh. All three values were found to be within $\pm$ 5 % of each other for recent years suggesting that the system-average emissions rate may be a reasonable approximation for the marginal displacement of variable-output renewables, but it is worth noting that the calculated marginal displacement factor for 2012 remains 20 % higher than the UK-average for that year (Ricardo-AEA, 2012).

9.1.7 Carbon and energy payback periods

This research was ultimately directed towards understanding the reliability of carbon and energy payback periods for variable-output renewable generation, using Great Britain as a case study. Uncertainties can be introduced from the underlying carbon footprint, embodied energy, GHG emissions displacement and energy output estimates.

One of the key findings of this analysis was that the carbon payback period of a variable-output transmission-connected wind, wave or tidal stream generator will be less than 10 % of the design life (2 years, if the design life is estimated at 20 years) provided the emissions displacement is greater than 250 g CO_2 eq/kWh. Even with the uncertainty introduced by the carbon footprinting methodology, there will be carbon reductions from such renewable generators as long as decarbonisation of the electricity supply is not too rapid, as demonstrated by the findings of the detailed sensitivity analysis for the Pelamis presented in Figure 8.2.

The range of energy payback periods was found to be slightly longer, with estimates for wind, wave and tidal stream devices all falling within 3 years.

Other metrics that are often considered alongside the payback periods are the net emissions reduction and the energy return on investment (EROI). These were found to be significantly more sensitive to uncertainties in lifetime energy production than the estimates of payback periods, due to being calculated over the whole design life, but the EROI of renewable energy converters was also found to compare well with conventional technologies.

As an example of existing variable renewable generators, it is possible to confidently estimate that wind farms in Great Britain both offset their own lifetime GHG emissions and were responsible for a further emissions reduction of 18 - 20 Mt CO_2 eq over the period from November 2008 to June 2013. Furthermore, the net energy output over this time period was around 38 TWh - even given the uncertainties introduced by the LCA methodology, the evidence provided in this thesis demonstrates that the existing installed wind capacity in Great Britain has already achieved carbon and energy payback, and will continue to contribute to reducing the emissions from power generation.

9.2 Implications

This section examines the implications of this research for carbon and energy audits, renewable power developments and government policy.

9.2.1 Carbon footprints and embodied energy estimates

With a considerable number of published studies estimating the carbon footprint and embodied energy of power generation technologies, there is confusion over their accuracy, reliability and the reasons for variations in their results. This research includes a review of existing published studies, particularly wind and marine energy converters; identifies the key issues to consider when assessing the reliability of a study; and highlights the key methodological choices to make when carrying out such an analysis. This information may be of use to anyone with an interest in the carbon footprint and embodied energy of power generation technologies - particularly policy makers, renewable energy converter manufacturers, and power station operators.

Reliable estimates of the carbon footprint and embodied energy of power generation vary due to differences in technology, location, and calculation methodology. A summary of the range of existing published carbon footprints and embodied energy estimates is given in Chapter 3. While these values should not be used to estimate the impacts of a particular installation, they do provide reliable information on the relative impacts of different technologies.

Several key methodological choices were identified that can have a significant impact on the carbon footprint or embodied energy estimate, particularly:

- **Type of analysis** - Process-based LCA may underestimate impacts, while hybrid LCA is likely to overestimate them;
- **Functional unit** - The functional unit for all types of power generation is a unit of output energy, normally 1 kWh;
- **Physical boundary** - The study should consider the entire system up until the point of connection with the grid;
- **Life cycle stages** - Best practice recommends that all life cycle stages of both the generator and the fuel should be included in the analysis;
- **Data sources** - Data sourced from well recognised and documented databases may introduce an uncertainty of as much as ± 30 % to the estimated carbon footprint or embodied energy;
- **Recycling allocation method** - Care must be taken to avoid double-counting, and the credit for end-of-life recycling is usually higher than the credit for recycled content of raw materials (due to typically higher assumed recyclability than recycled content);
- **Design life** - This has a significant impact on results and should be realistic;
- **Annual energy output** - A sensitivity analysis should demonstrate the sensitivity of the results to the estimated output;
- **Characterisation factors** - Consensus has yet to be reached on the primary energy intensity of many fuel carriers, so results may vary considerably;
- **Scope of GHGs** - Ideally should include all greenhouse gases, but only CO_2, CH_4 and N_2O will also provide a good approximation, and in many cases the emissions of CO_2 alone will be sufficient;
- **Land-use change** - The impacts of land-use change are location specific, but there may be considerable impacts with construction on peat lands, deforestation, or inundation of land.

9.2.2 Methodology for estimating marginal emissions displacement of variable-output renewables

The marginal emissions displacement of renewable energy such as wind, wave and tidal power is usually approximated as the system-average emissions or the marginal emissions of demand-side fluctuations. The methodology developed in this thesis can be applied to historical data for any network over any time frame to examine both the marginal displacement of variable-output renewables, and the marginal emissions of fluctuations in system generation. In order to calculate the former, there must be some existing capacity, but the method allows marginal emissions displacements for wind, wave or tidal power to be calculated either independently or as a group. This provides information on the marginal operation of the network, which is of particular interest in an opaque liberalised market, and calculates more accurate estimates of the actual marginal displacement for carbon payback calculations, which are required by renewable energy developers, planners and policy makers.

9.2.3 Effect of efficiency penalties on marginal emissions displacement

It is expected that the marginal emissions displacement of variable-output renewable generators on the British grid will be higher than the system-average emissions, as a large proportion of base load is provided by low-carbon nuclear power; however, the effect of the efficiency penalties of part-loading conventional plant was previously unknown. The analysis presented in Chapter 7 found that this effect was significant, and that the calculated system-average emissions would provide a good approximation for the marginal displacement of wind, wave or tidal power in Great Britain. However, UK-wide average emissions would remain an underestimate, with the calculated marginal displacement factor of wind power being 20 % higher than the reported average emissions for 2012, even taking efficiency penalties into consideration.

The best estimate of the marginal displacement of wind power for November 2008 to June 2013 was 0.56 kg CO_2 eq/kWh. The marginal emissions of total generation were 0.60 kg CO_2 eq/kWh and the calculated system-average emissions 0.52 kg CO_2 eq/kWh for the same time period. These robust and reliable estimates confirm that marginal emissions are higher than average emissions, and may be valuable in estimating carbon payback and emissions reductions of all types of variable-output generator, or identifying potential savings from demand-side management.

9.2.4 Carbon and energy payback

Estimates of carbon and energy payback periods of variable-output renewable generators are sensitive to uncertainties in carbon footprint and embodied energy, assumed lifetime energy production and GHG emissions displacement. An examination of the values summarised in Chapter 3, found that the carbon payback period of variable-output transmission-connected wind, wave or tidal stream generators is expected to be less than 2 years as long as the emissions displacement is greater than 250 g CO_2 eq/kWh. Similarly, energy payback periods are all estimated to be below 3 years. This conclusion is of significance for renewable energy advocates and policy makers, as it demonstrates that renewable generators are viable options for energy supply, and provides a threshold for a reasonable carbon payback.

A detailed examination of the sensitivity of carbon and energy payback periods to practitioner assumptions in the initial carbon footprint and embodied energy calculation was carried out for the Pelamis (Chapter 8). This also demonstrated that, with an emissions displacement of 560 g CO_2 eq/kWh (Chapter 7), payback should be achieved within 2 and 3 years for carbon and energy respectively.

This research has also concluded that, over the period from November 2008 to June 2013, the entire fleet of wind farms in Great Britain was responsible for a net emissions reduction of approximately 19 Mt CO_2 eq, and a net energy production of 38 TWh, assuming that all carbon and energy payback took place during this time frame - itself a very conservative estimate.

This demonstrates that wind power has been reducing GHG emissions of UK electricity, while also providing an energy return on investment. It is expected that emerging marine energy technologies will also achieve a similar result, if new installations are also planned with enough care to ensure a good power output.

9.3 Recommendations for Further Work

9.3.1 Further analyses of the carbon footprint and embodied energy of generation technologies

Despite the considerable number of published studies examining the carbon footprint and embodied energy of generation technologies, many gaps remain. Further work could include detailed analyses of emerging technologies, particularly marine energy converters, but also of established technologies, such as wind farms. Although many studies have been carried out on the latter, they are mostly based on theoretical installations; in order to confirm the relevance of these analyses, it would be of benefit to carry out detailed carbon and energy audits of real wind farms in Great Britain. One particular question to answer is whether existing studies represent the true impacts of land-use change, as many wind farms in Scotland are built on peat lands or replace forests. There is a similar issue with current estimates of the carbon footprints and embodied energy of solar photovoltaics, where the installation location may have a significant impact; current studies are mostly based on installation at lower latitudes, where the insolation is much higher. In the marine energy sector there is considerable research being carried out into the available resource at different possible installation locations, but the carbon impacts of any sea bed disturbance appear to have been largely ignored.

9.3.2 Development of a carbon footprint and embodied energy calculation tool

One of the limitations of existing carbon and energy audits is that they are location dependent. Further research is required to investigate the possibility of adapting the output of existing studies to be implemented for specific installation locations; for example, extracting the impacts associated with transport distances and change in land use from the original calculation, and enabling these to be modified for alternative locations. An attempt has been made to achieve this for the Pelamis wave energy converter. Such an analysis would be even more straightforward for the second-generation device which is manufactured in a factory in Leith, irrespective of final installation location. There is scope to develop a tool containing manufacturer-specific life cycle information for a range of renewable energy converters, to enable the impacts of prospective developments to be quickly assessed.

9.3.3 Refinement of marginal displacement analysis

There is also scope for further work to refine the analysis of the marginal emissions displacement of variable renewable generation: the analysis presented in this thesis does not accurately model the GHG emissions of power station start-up and shut-down, pre-warming or running reserve capacity. Furthermore, frequent ramping of fossil-fuelled generators could degrade the heat rate and affect the efficiency, and the effect of this on GHG emissions should also be considered. Another limitation with the current work is that it only considers transmission-connected generation and thus neglects the considerable capacity of embedded generation on the British network.

Further investigation should also be carried out into the significant change in GHG emissions (Figure 6.8) when there is no change in system generation or variable renewable output. This could be achieved by expanding the multiple linear regression analysis to consider other network effects, such as scheduled ramping of generators, unplanned outages and network constraints.

9.3.4 Further marginal analyses

The methodology for analysing the marginal displacement of variable-output renewables could be applied to other networks where detailed historical power output data is available. One such network of interest would be that in Ireland, which has no nuclear generation and a large installed capacity of wind power.

The methodology could also be adapted to consider other marginal effects; for example, the primary energy displacement of variable-output renewable generation. There is a suggestion that the primary energy payback period is a more accurate metric of energy payback time than the current estimates that compare electricity output with primary energy consumption, but this requires an accurate estimate of the displaced primary energy consumption of electricity generation. Such an analysis would simply require the replacement of GHG emissions intensity values for each type of supply with equivalent primary energy multipliers.

9.3.5 Development of forecasting model

A significant limitation of the marginal displacement analysis presented here is that it is historical. In order to be able to truly assess the impact on GHG emissions of a new renewable energy installation, a forecasting model is required. By basing it on existing models of network operation, it could provide both temporal and spatial detail of emissions displacement factors. It would be of value in informing decision making by examining the net emissions reductions of proposed changes to the network; for example, the commissioning and decommissioning of power stations, proposed routes of new transmission lines and potential locations for any new renewable energy installations.

9.4 Thesis Conclusion

Despite renewable energy sources being ‘carbon-free’, resources are consumed and pollutants are emitted during the life cycle of the renewable energy generators. With the incentive for developing renewable generation being the continued drive to decarbonise electricity supplies in an attempt to mitigate climate change, these energy converters must achieve a net reduction in greenhouse gas emissions over their lifetimes. Furthermore, concerns over the decreasing availability of fossil fuels mean that such generators must also provide a good energy return on investment to remain viable.

Carbon and energy payback periods are often calculated for renewable generation technologies to demonstrate that these targets will be achieved. Developed from the concept of economic payback, these are calculated from estimates of life cycle carbon footprint, embodied energy, GHG emissions displacement and energy output. However, existing estimates for carbon footprint and embodied energy vary widely, and there is some debate over the actual emissions displacement of variable-output renewable generation, with current calculations based on system-average emissions rather than the marginal emissions displacement. This raises doubts over the accuracy of existing carbon and energy payback estimates.

A key conclusion of this research was that carbon payback should be achieved within 2 years for wind, wave and tidal stream generators, and within a quarter of the design life for tidal barrages, provided the marginal emissions displacement remains above 250 g CO_2 eq/kWh. With the most reliable estimate for the marginal displacement of variable-output renewable generation in Great Britain being 560 g CO_2 eq/kWh for November 2008 to June 2013, considerable decarbonisation is required to reach this threshold. Furthermore, analysis of the reported output from such generation over the same time frame demonstrates that the entire fleet of wind turbines in Great Britain not only paid back their carbon and energy between November 2008 and June 2013, but also achieved a further emissions reduction of approximately 19 Mt CO_2 eq and an output of around 38 TWh. Even when the most extreme uncertainties are taken into account, the net emissions reduction is greater than 10 Mt CO_2 eq and the net energy output remains greater than 38 TWh. It is expected that similar reductions should be possible with emerging marine energy technologies, although further analysis of the impact of any new installations is required to confirm this.

It is, therefore, possible to confirm the hypothesis that, despite the considerable scope for uncertainties to be introduced to carbon and energy payback estimates, variable-output renewable energy generators in Great Britain do deliver a net reduction in greenhouse gas emissions over their lifetimes, and also produce a viable energy return on energy investment.

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Appendix A

Data selection for LCA of Pelamis

A.1 Data Selection

In the analysis presented in Chapter 4, life cycle inventory data was mostly sourced from the Ecoinvent database (Ecoinvent, 2010), with additional information from PlasticsEurope (2005), ELCD (European Commission, 2013) and manufacturer's datasheets or environmental product declarations. The names of each material and process selected for this analysis is shown in Table A.1.

A.2 Uncertainty

Data from the Ecoinvent database is provided with an uncertainty value, assumed to be a log-normal distribution, with the square of the geometric standard deviation covering the covering the 95 % confidence interval. This geometric standard deviation is estimated from a pedigree matrix, reproduced in Goedkoop *et al.* (2008), with each data point assessed with regards to six criteria plus a 'basic' uncertainty factor. The 95 % confidence interval is then calculated as shown in Equation A.1 (Goedkoop *et al.*, 2008), with the factors U_1 through to U_6 representing scores in the pedigree matrix, and U_b being the basic uncertainty factor derived from a table published in Frischknecht *et al.* (2007):

$$SD_{g95} = \sigma_g^2 = e^{\sqrt{[ln(U_1)]^2+[ln(U_2)]^2+[ln(U_3)]^2+[ln(U_4)]^2+[ln(U_5)]^2+[ln(U_6)]^2+[ln(U_b)]^2}} \tag{A.1}$$

In order to estimate uncertainty ranges from data derived from manufacturer's information, the same pedigree matrix was used in this analysis. Table A.2 details the uncertainty ranges applied to each of these values, with the scores used to estimate each of these also provided.

Stock Material	Source	Process Name
Steel (virgin)	Ecoinvent	Steel, converter, low-alloyed, at plant/RER
Steel (recycled)	Ecoinvent	Steel, electric, low-alloyed, at plant/RER
Stainless steel	Ecoinvent	Chromium steel 18/8, at plant/RER
Aluminium (virgin)	Ecoinvent	Aluminium, primary, at plant/RER
Aluminium (recycled from new scrap)	Ecoinvent	Aluminium, secondary, from new scrap, at plant/RER
Aluminium (recycled from old scrap)	Ecoinvent	Aluminium, secondary, from old scrap, at plant/RER
Copper (virgin)	Ecoinvent	Copper, primary, at regional storage/RER
Copper (recycled)	Ecoinvent	Copper, secondary, at regional storage/RER
Copper tungsten 20	Ecoinvent	As for copper, above
Brass	Ecoinvent	Brass, at plant/CH
Bronze	Ecoinvent	Bronze, at plant/CH
Zinc	Ecoinvent	Zinc, primary, at regional storage/RER
Cast iron	Ecoinvent	Cast iron, at plant/RER
Aluminium oxide	Ecoinvent	Aluminium oxide, at plant/RER
Nylon 6	Ecoinvent	Nylon 6, at plant/RER
Polyamide 11 (in MV circuit breaker)	Ecoinvent	Nylon 6, at plant/RER
Polyamide 66 (in MV circuit breaker)	Ecoinvent	Nylon 66, at plant/RER
Polyurethane	Ecoinvent	Polyurethane, rigid foam, at plant/RER
PVC pipe	Plastics Europe	PVC Pipe E, Industry data 2.0
PVC	Ecoinvent	Polyvinylchloride, at regional storage/RER
Polycarbonate	Ecoinvent	Polycarbonate, at plant/RER
Polycarbonate+FB30	Ecoinvent	Polycarbonate, at plant/RER
EPDM	Ecoinvent	Synthetic rubber, at plant/RER
Polypropylene	Plastics Europe	Polypropylene injection moulding, Industry data 2.0
Polyester	Ecoinvent	Polyester resin, unsaturated, at plant/RER
PTFE	Ecoinvent	Tetrafluoroethylene, at plant/RER
Transformer insulation material	Ecoinvent	Polyvinylchloride, at regional storage/RER

Table A.1a: Processes selected for use in LCA of Pelamis

Stock Material	Source	Process Name
Impregnation resin	Ecoinvent	Epoxy resin, liquid, at plant/RER
Paint	Ecoinvent	Alkyd paint, white, 60% in solvent, at plant/RER
Epoxy resin	Ecoinvent	Epoxy resin, liquid, at plant/RER
Epoxy resin-Fe10	Ecoinvent	Epoxy resin, liquid, at plant/RER
Glass-flake paint	Hempel (2010a)	Materials only
Epoxy primer/topcoat		Taken to be glass-flake paint without glass
Epoxy primer	Ecoinvent	Epoxy resin, liquid, at plant/RER
Curing agent	Hempel (2010b)	Materials only
Xylene	Ecoinvent	Xylene, at plant/RER
n-butanol	Ecoinvent	2-butanol, at plant/RER
p-tert-butylphenol	Ecoinvent	Phenol, at plant/RER
m-xylylene-diamine	Ecoinvent	Ethylenediamine, at plant/RER
Ethanol	Ecoinvent	Ethanol from ethylene, at plant/RER
Ethylbenzene	Ecoinvent	Ethyl benzene, at plant/RER
2,2,4- and 2,4,4-trimethylhexamethylene diamine	Ecoinvent	Trimethylamine, at plant/RER
2,4,6-tris(dimethylaminomethyl)phenol	Ecoinvent	Phenol, at plant/RER
3-(2-aminoethylamino)propyltrimethoxysilane	Ecoinvent	Trimethylamine, at plant/RER
Sand	Ecoinvent	Sand, at mine/CH
Glass reinforced plastic (GRP)	Ecoinvent	Glass-fibre reinforced plastic, polyamide, injection moulding at plant/RER
Glass	Ecoinvent	Flat glass, uncoated, at plant/RER
Glass flakes	Ecoinvent	Flat glass, uncoated, at plant/RER
Sulphur hexafluoride	Ecoinvent	Sulphur hexafluoride, liquid, at plant/RER
Transformer oil	Ecoinvent	Lubricating oil, at plant/RER
Porcelain	Ecoinvent	Ceramic tiles, at regional storage/CH
Wood	Ecoinvent	Sawn timber, softwood, planed, air dried, at plant/RER
Water	Ecoinvent	Water, process and cooling, unspecified natural origin/RER

Table A.1b: Processes selected for use in LCA of Pelamis

Energy Sources	**Source**	**Process Name**
Natural gas	Ecoinvent	Natural gas, high pressure, at consumer/GB
Fuel oil	Ecoinvent	Heavy fuel oil, burned in industrial furnace 1MW, non-modulating/RER
Electricity	Ecoinvent	Electricity, medium voltage, at grid/GB
Diesel	Ecoinvent	Diesel, burned in chopper/RER
Metal Processing		
Steel rolling	Ecoinvent	Hot rolling, steel/RER
Drawing of pipes	Ecoinvent	Drawing of pipes, steel/RER
Drawing of wire	Ecoinvent	Wire drawing, steel/RER
Sand casting	Donohoe (2001)	
Machining	Ecoinvent	Milling, steel, average/RER
Flame cutting	Ecoinvent	50m per m^2 of Welding, gas, steel/RER
Welding	Ecoinvent	Welding, arc, steel/RER
Abrasive for jet blasting	Ecoinvent	Sand, at mine/CH
Compressed air supply for jet blasting	Ecoinvent	Compressed air, average installation, <30kW, 8 bar gauge, at supply network/RER
Compressed air supply for painting	Ecoinvent	Compressed air, average installation, >30kW, 7 bar gauge, at supply network/RER
Electronics		
Transformer	ABB (2007)	Materials only
Main generator	ABB (2011)	Materials only
MV switch-disconnector cubicle	ABB (2010)	Materials only
MV SF6 circuit breaker	ABB (2001)	Materials only

Table A.1c: Processes selected for use in LCA of Pelamis

Assembly Equipment	Source	Process Name
60T crane	SWF (2011)	18 kWh of electricity per hour
Fork lift truck	Caterpillar (2011)	91.3 MJ of diesel per hour
Transport		
Rigid lorry	Ecoinvent	Transport, lorry 3.5-16t, fleet average/RER
Articulated lorry	Ecoinvent	Transport, lorry >16t, fleet average/RER
Sea freight	Ecoinvent	Transport, transoceanic freight ship/OCE
Barge	PWP	290 l of fuel per day
Multicat	PWP	1710 l of fuel per day
Tug	PWP	1490 l of fuel per day
Inspection vessel	PWP	500 l of fuel per day
Sea vessel fuel consumption	Ecoinvent	Operation, barge/RER x Density of diesel / 0.00939
Waste landfill		
Steel	Ecoinvent	Disposal, steel, 0% water, to inert material landfill/CH
Aluminium	Ecoinvent	Disposal, aluminium, 0% water, to sanitary landfill/CH
Tin plate	Ecoinvent	Disposal, tin sheet, 0% water, to sanitary landfill/CH
Tin sheet	Ecoinvent	Disposal, tin sheet, 0% water, to sanitary landfill/CH
Other metals	ELCD	Landfill of ferro metals EU-27
Slag	Ecoinvent	Disposal, slag, unalloyed electr. steel, 0% water, to residual material landfill/CH
Steel dust	Ecoinvent	Disposal, dust, unalloyed EAF steel, 15.4% water, to residual material landfill/CH
PVC	Ecoinvent	Disposal, polyvinylchloride, 0.2% water, to sanitary landfill/CH
PE	Ecoinvent	Disposal, polyethylene, 0.4% water, to sanitary landfill/CH
PET	Ecoinvent	Disposal, polyethylene terephtalate, 0.2% water, to sanitary landfill/CH
PP	Ecoinvent	Disposal, polypropylene, 15.9% water, to sanitary landfill/CH

Table A.1d: Processes selected for use in LCA of Pelamis

Waste landfill	Source	Process Name
PS	Ecoinvent	Disposal, polystyrene, 0.2% water, to sanitary landfill/CH
PUR	Ecoinvent	Disposal, polyurethane, 0.2% water, to sanitary landfill/CH
PVDC	Ecoinvent	Disposal, polyvinylchloride, 0.2% water, to sanitary landfill/CH
Rubber	Ecoinvent	Disposal, plastics, mixture, 15.3% water, to sanitary landfill/CH
Other plastics	Ecoinvent	Disposal, plastics, mixture, 15.3% water, to sanitary landfill/CH
Wood	Ecoinvent	Disposal, wood untreated, 20% water, to sanitary landfill/CH
Paint	Ecoinvent	Disposal, paint, 0% water, to inert material landfill/CH
Glass	Ecoinvent	Disposal, glass, 0% water, to inert material landfill/CH
Cement	Ecoinvent	Disposal, building, cement (in concrete) and mortar, to final disposal/CH
Textile	ELCD	Landfill of textiles EU-27
Biodegradable waste	ELCD	Landfill of biodegradable waste EU-27
Dry inert material	Ecoinvent	Disposal, inert material, 0% water, to sanitary landfill/CH
Damp inert material	Ecoinvent	Disposal, inert waste, 5% water, to inert material landfill/CH
Recycling		
Steel - input	Ecoinvent	Iron scrap, at plant/RER
Steel - output	Ecoinvent	Pig iron, at plant/GLO
Aluminium - input	Ecoinvent	Aluminium scrap, old, at plant/RER
Aluminium - output	Ecoinvent	Aluminium, primary, at plant/RER
Copper - input	Ecoinvent	Copper, secondary, from electronic and electric scrap recycling, at refinery/SE
Copper - output	Ecoinvent	Copper, primary, at refinery/GLO

Table A.1e: Processes selected for use in LCA of Pelamis

Material	**Uncertainty** (σ_g^2)	**Pedigree Matrix Score** $\mathbf{U_1, U_2, U_3, U_4, U_5, U_6, U_b}$
Recycled content of steel	1.1094	From Classen *et al.* (2009)
Recycled content of aluminium	1.2	From Classen *et al.* (2009)
Recycled content of copper	1.07	From Classen *et al.* (2009)
Paint - kg/m^2	1.4	2,4,4,1,3,5,3
Glass-flake paint	1.331	2,4,2,3,3,5,4
Epoxy primer/topcoat	1.331	2,4,2,3,3,5,4
Curing agent	1.331	2,4,2,3,3,5,4
Metal Processing		
Flame cutting	1.316	4,4,0,0,3,0,0
Compressed air supply for painting	1.401	2,4,4,1,3,5,2
Sand casting		
Water	1.043	1,2,2,2,1,2,0
Additional steel	1.066	1,2,2,2,1,2,9
Sand	1.066	1,2,2,2,1,2,4
Natural gas	1.066	1,2,2,2,1,2,8
Oil	1.066	1,2,2,2,1,2,8
Sand reuse	1.066	1,2,2,2,1,2,4
Electricity	1.066	1,2,2,2,1,2,2
Carbon dioxide emissions	1.066	1,2,2,2,1,2,21
Particulate emissions	1.503	1,2,2,2,1,2,38
Water emissions to air	1.043	1,2,2,2,1,2,0
Waste water	1.043	1,2,2,2,1,2,0
Waste disposal	1.043	1,2,2,2,1,2,0
Abrasive jet blasting		
Abrasive	1.59	2,4,2,3,4,5,4
Compressed air supply	1.46	4,4,4,1,3,5,2
Particulate emissions	1.59	2,4,2,2,3,3,38
Waste disposal	1.84	2,4,2,3,4,5,10

Table A.2a: Uncertainty estimates for LCA of Pelamis

Materials for Electrical Equipment	**Uncertainty (σ_g^2)**	**Pedigree Matrix Score $\mathbf{U}_1$, $\mathbf{U}_2$, $\mathbf{U}_3$, $\mathbf{U}_4$, $\mathbf{U}_5$, $\mathbf{U}_6$, $\mathbf{U}_b$**
Transformer	1.324	1,4,2,1,3,5,4
Main generator	1.324	1,4,2,1,3,5,4
MV switch-disconnector cubicle	1.324	1,4,2,1,3,5,4
MV SF6 circuit breaker	1.324	1,4,2,1,3,5,4
Assembly Processes		
60T crane	1.46	4,4,4,1,3,5,2
Fork lift truck	1.46	4,4,4,1,3,5,2
Sea Vessel Operations		
Barge fuel consumption	2.06	1,4,1,1,1,5,5
Multicat fuel consumption	2.06	1,4,1,1,1,5,5
Tug fuel consumption	2.06	1,4,1,1,1,5,5
Inspection vessel fuel consumption	2.06	1,4,1,1,1,5,5

Table A.2b: Uncertainty estimates for LCA of Pelamis

Appendix B

Publications

The work described in this thesis has been reported in the following publications:

1. Thomson, R. C., Harrison, G. P. and Chick, J. P., How eco-friendly is wave power? A full life cycle assessment of a wave energy converter, Poster presented at *UKERC Annual Assembly* (First prize), 2011
2. Thomson, R. C., Harrison, G. P. and Chick, J. P., Full life cycle assessment of a wave energy converter. In *IET Conference on Renewable Power Generation (RPG 2011), Edinburgh, UK*, 6-8 Sept, 2011. doi: 10.1049/cp.2011.0124
3. Thomson, C., Harrison, G. and Chick, J., Life cycle assessment in the marine renewable energy sector, In *The LCA XI International Conference, Chicago, IL, United States*, 4-6 October, 2011.
4. Thomson, R. C., Harrison, G. P. and Chick, J. P., Marginal greenhouse gas offset for renewable energy in the UK, In *The LCA XII International Conference, Tacoma, WA, United States*, 25-27 September, 2012.
5. Thomson, C., Greenhouse gas emissions savings from wind power, Poster presented at *SET for BRITAIN 2013, Houses of Parliament, London, UK*, 18 March, 2013.
6. Thomson, R. C., Harrison, G. P. and Chick, J. P., Greenhouse gas emissions savings from wind power, Poster presented at *Global Energy Systems conference, Edinburgh, UK*, 26-28 June, 2013.

HOW ECO-FRIENDLY IS WAVE POWER?

A full life cycle assessment of a wave energy converter

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Pelamis Wave Energy Converter

Motivation

Questions are arising over the environmental impacts associated with the process of converting low-carbon energy sources into electrical power. Impacts of renewable generation arise indirectly from the manufacture, operation and decommissioning of generators and network infrastructure. A detailed understanding of such impacts will help inform future developments of the energy system.

While studies have been carried out for a range of energy technologies, to date few have been carried out in the marine renewables sector, and these mostly concentrate on carbon emissions and embodied energy.

Results

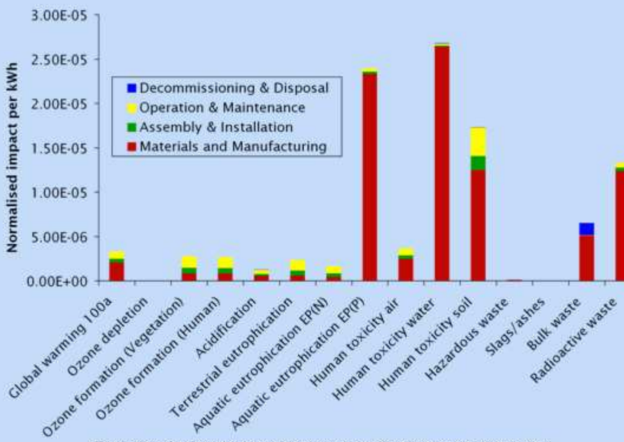

Fig 2: Graph of environmental impacts normalised per person equivalent

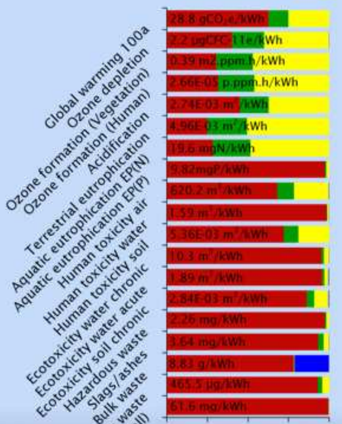

Fig 4: Graph of characterisation results

- Global warming potential: 28.8 gCO_2e/kWh
  Payback: 1 yr 2 months
- Energy intensity: 378 kJ/kWh
  Payback: 2 yrs 1 month
- Generally, the greatest impact arises during the materials & manufacturing stage, mostly from steel production.
- Shipping operations are also a significant contributor.
- Significant radioactive waste impacts arise due to the contribution of nuclear power to British grid electricity.

The Pelamis

The Pelamis is emerging as a leading wave energy converter, with a new commercial-scale site currently under development off the coast of Shetland in the UK. This semi-submerged snake-like offshore device has four cylindrical sections linked by power conversion modules at the hinged joints. The motion induced by the passing wave front pumps hydraulic fluid through generators to produce power.

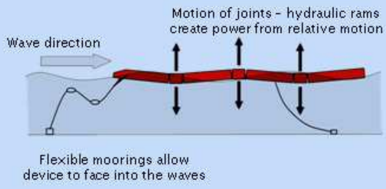

Fig 1: Pelamis operation

Life Cycle Assessment (LCA)

LCA is a methodology for systematically analysing resource use and pollutant emissions at each stage of a device life cycle. The full process is described in ISO 14040[1] and involves characterising the inventory results as a set of identifiable consequences.

This analysis used SimaPro software and the EDIP2003 impact assessment method. Raw data was mostly provided by the manufacturer or taken from the Ecoinvent database.

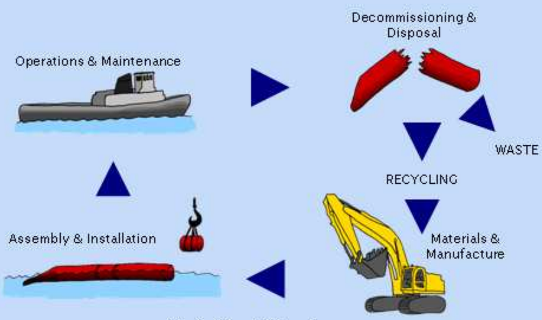

Fig 3: Pelamis life cycle

Discussion

Full life cycle results compare well with published information for wind power, with the Pelamis performing significantly better in the human toxicity categories.[2]

Carbon and energy intensity figures are ~30% greater than Parker *et al.*[3] This is probably due to variations in practitioner assumptions and raw data.

The greatest environmental impact across all categories arises from the processing and raw materials in steel.

Images of the Pelamis Wave Energy Converter are taken from http://www.pelamiswave.com

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FULL LIFE CYCLE ASSESSMENT OF A WAVE ENERGY CONVERTER

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Keywords: Life cycle analysis, wave energy

Abstract

The Pelamis wave energy converter is emerging as one of the most promising devices to harness the available power in the waves. This study examines the environmental impacts of the device, presenting the results as a set of impact potentials, and demonstrating that it performs well in comparison to other renewable energy converters and fossil-fuelled generators.

1 Introduction

The continued drive to mitigate climate change by reducing Greenhouse Gas (GHG) emissions has led to an increase in demand for low-carbon energy sources. This has resulted in the development of new technologies to harness renewable energy. However, while the energy sources are themselves 'carbon-free', there are wider environmental impacts associated with the process of converting the energy into electrical power. In order to make informed decisions for future developments of the energy system, it is therefore necessary to develop a detailed understanding of the life cycle environmental impacts that arise indirectly from power generation due to the manufacture, operation and decommissioning of generators and network infrastructure.

In the United Kingdom (UK) the Government has introduced ambitious targets to decarbonise the electricity supply, with the latest carbon budget aiming to reduce average emissions from generation from current levels of around 500 gCO_2/kWh to around 50 gCO_2/kWh by 2030 [3]. It is expected that marine energy will be an important contributor, with resources believed to have the potential to supply around 20 per cent of electricity demand [5].

The Pelamis Wave Energy Converter (WEC) is emerging as one of the most promising devices to harness this available power. Developed by Pelamis Wave Power Ltd, the P1 version of this semi-submerged offshore device was successfully installed at the world's first commercial wave farm at Aguaçadoura, off the coast of Portugal, in 2008. The experience gained has been fed directly into the development of the second-generation P2 device, currently on test at the European Marine Energy Centre. Several projects are currently in the development stages, with lease agreements having been agreed for two farms comprising around 70 devices off the coast of Scotland [13]. It is therefore important to understand the life cycle impacts of these devices. To date very few life cycle assessments have been carried out in this sector, and many of these concentrate only on carbon emissions and embodied energy.

In 2007 an in-depth life cycle carbon and energy audit was published by Parker *et al.* [12] on the Pelamis P1 device, based on detailed data from the manufacturer. This study found that the energy and carbon intensities were 293 kJ/kWh and 23 gCO_2/kWh respectively. The current paper builds upon the work carried out by Parker *et al.* by expanding the analysis to cover a broad range of environmental impacts. In particular this includes an expansion of the carbon analysis to include all GHG emissions. This will involve creating an inventory of all environmentally significant resource use and pollutant emissions at each stage of the device life cycle, from 'cradle-to-grave', and then characterising these according to their 'impact potential'. This detailed study will allow better comparison with existing and future generating technologies.

2 Life Cycle Assessment

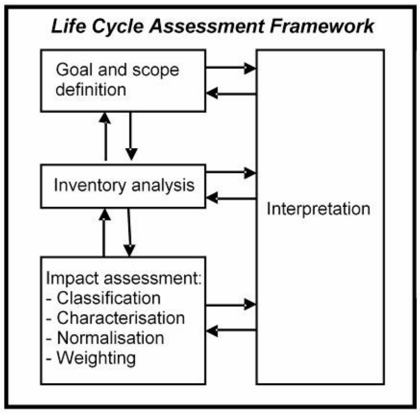

Figure 1: Life cycle assessment framework [4]

Life Cycle Assessment (LCA) is an established technique for identifying and evaluating the inputs, outputs and potential environmental impacts of products or services. The process is illustrated in Figure 1. It involves systematically analysing resource use and pollutant emissions at each stage of the product life cycle; from extraction of raw materials, through manufacture and operation to decommissioning and disposal. The detailed results are then described as a set of identifiable consequences or 'impact potentials'. This mature methodology is governed by the ISO 14040 series of international standards [1], and has already been applied to a range of energy technologies and networks.

The results of this comprehensive analysis will highlight the components, materials or stages of the life cycle with the largest environmental impacts. This information can be used in design development and marketing product environmental credentials, and will also be valuable in planning the development of an environmentally-sustainable energy system. More information on LCA can be found in reference [4].

3 The Pelamis Wave Energy Converter

Figure 2: Pelamis wave energy converter [13]

The Pelamis is a semi-submerged snake-like offshore wave energy converter. The P1 version is 120 m long, 3.5 m in diameter and rated at 750 kW (Figure 2). It has four cylindrical sections linked by three power conversion modules at the hinged joints. The compliant moorings allow the Pelamis to face into the oncoming waves, and the joints flex vertically and horizontally (heave and sway) as the wave front passes. This motion is resisted by hydraulic rams housed within the power conversion modules. These rams pump high-pressure oil into banks of accumulators, which are drained at a constant rate through hydraulic motors, in turn driving induction generators. The resistance of the rams can be tuned to provide a resonant response in small sea states to maximise power capture, and can also assist in protecting the device from potentially damaging storm waves.

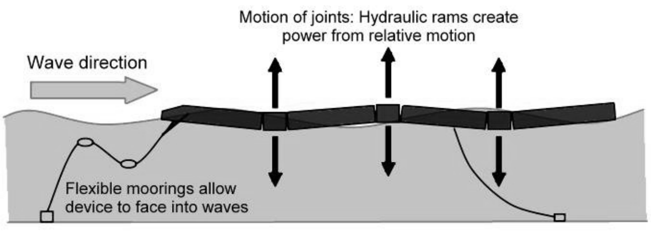

Figure 3: Side view of the Pelamis [12]

In order to enable comparison with the analysis published by Parker *et al.* in 2007 [12], many of the fundamental assumptions and base data have been kept the same in the current study. Therefore, in line with these earlier assumptions, it is estimated that the power output of a single device will average 2.97 GWh/year over the design life, if installed in a typical site off the northwest coast of Scotland. The successful installation at Aquaçadoura found that the Pelamis did perform as expected, so this assumption is still considered to be valid [13].

4 Analysis

The current study was carried out with one of the leading LCA software tools, SimaPro (version 7.2 PhD). Life cycle inventory data is mostly sourced from the Ecoinvent database, published by the Swiss Centre for Life Cycle Inventories, as this dataset is recognised as one of the most comprehensive sources of cradle-to-gate resource use and emissions data for materials, transport and other processes in Europe [2].

4.1 Goal and Scope Definition

The clear definition of a goal and scope is an integral part of any LCA [1]. The current study is intended to expand earlier work to provide an assessment of the broader environmental impacts of the Pelamis WEC, contributing to the wider body of research on the environmental impacts of power generation, and informing future design developments.

The system boundary of the current study will include the entire life cycle from "cradle-to-grave" (Figure 4). Physically the analysis includes the device, its moorings and sub-sea connecting cable, but excludes all downstream electrical components. The functional unit will be one kilowatt-hour of output power (1 kWh), with a calculation reference flow of 1 Pelamis device, producing an average of 2.97 GWh/year over its 20-year life (see section 3).

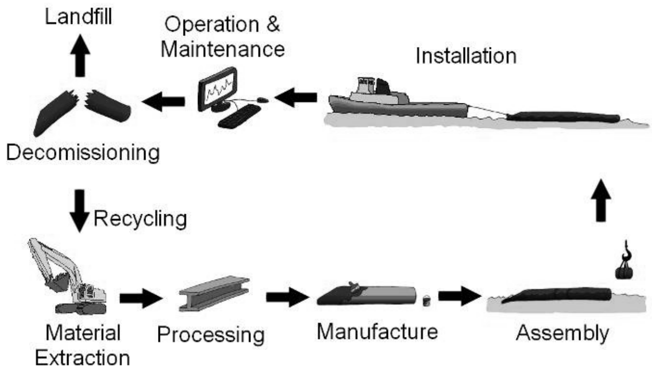

Figure 4: Pelamis Life Cycle

In line with the assumptions made by Parker *et al.* [12], the current study presents a generic case for the production of a single device, based on materials data for the first production machines. The same fixed scenario of manufacture, assembly and deployment has been defined. Later versions of the device and different installation scenarios will have different impacts to those presented here.

The current study assumes that all major components and sub-components are manufactured in the UK and subject to UK energy statistics and transport distances. It is assumed that the typical wave farm in which the device will be deployed is within 200 miles of a commercial port (implying a travel time of 24 h at 6 knots). For the purposes of calculating the carbon payback, it is assumed that the electricity offset by the device will be the average of the UK grid, with a CO_2 intensity of 0.499 kg/kWh [9].

4.2 Life Cycle Inventory Analysis (LCI)

The Life Cycle Inventory (LCI) involves detailing all resource use and pollutant emissions at each life cycle stage (Figure 4). Where data is not readily available, justifiable assumptions are made. Previous studies on other renewable energy converters have shown that the most significant impacts arise during the manufacturing stage. Care was therefore taken to gather the most comprehensive and accurate data available for this stage of the life cycle.

The current study builds upon the work carried out in 2006 by Parker *et al.*, and therefore all base data for quantities of raw materials, processing and manufacturing methods, and transportation were sourced from the same original data [12]. This was based on figures derived from PWP's own records, particularly that pertaining to the P1 device under production at the time.

Materials & Manufacture

The main structure of the Pelamis is formed from four cylindrical tube sections which increase in length from fore to aft (nose to tail). Sand ballast is placed within the tubes to optimise the buoyancy. The nose tube is tapered at one end to allow the WEC to cut through waves in rough conditions, and also houses the switchgear and transformer to collect and transform the power from the generators for export to shore. Three Power Conversion Modules (PCMs) sit between the tube sections and house the hydraulic power take-off, generators and control equipment. The Pelamis is connected to the mooring and cabling system via the Yoke, a Y-shaped element connected to the nose tube. This has a quick-release tethering system to allow for rapid attachment and detachment.

Stock Material	*Mass (kg)*
Steel	561954
Sand	475722
Stainless Steel	550
Nylon 6	416
Polyurethane	343
Glass Reinforced Plastic (GRP)	90
PVC Pipe	55

Table 1: Material quantities in the Pelamis P1

All data for the structure, hydraulic system and mooring components was based on the mass and materials of major components provided by PWP, as used Parker *et al.* [12]. A full breakdown of the materials used in the Pelamis is shown in Table 1.

Data for the resource use and pollutant emissions was sourced from the Ecoinvent database where possible [2]. This Swiss dataset provides comprehensive European average data, with UK specific data being selected where available. Data not available within Ecoinvent was sourced from alternative datasets or available literature. One example of this was sand-casting of steel components. Comprehensive data was not available within the Ecoinvent database, so data was applied from a mass balance on the British foundry manufacturing sector, carried out by Donohoe *et al.* as part of the wider Mass Balance Project [6].

In addition to the materials detailed above, over 170 different pre-fabricated components and devices are included in the Pelamis, such as fixings and electrical items. Sourcing detailed LCI data for such devices is very time-consuming, so published guidance allows for cut-off criteria to be defined so that inputs that do not have a significant environmental impact can be excluded from the study [1]. A preliminary analysis of carbon emissions and energy consumption was carried out, using cost-based analysis of the pre-fabricated components. This found that the transformer, main generators and switchboard should be included in the study, but the other pre-fabricated components combined contribute less than 1 per cent to the total impacts.

The carbon dioxide emissions and energy consumption for this life cycle stage were found to be **17 gCO_2/kWh** and **348 kJ/kWh** respectively.

Assembly and Installation

Assembly and installation processes mostly comprise transport of components from assembly plant to the dockyard, and sea vessel operations for installation of the moorings and power cabling, sea trials, initial tow to site and latching to the moorings. The analysis was based on process information provided by PWP.

In this stage the analysis method applied for transportation was different from that used by Parker *et al* [12]. Data was taken from the Ecoinvent database, with manufacturer's data being applied where appropriate. This will have introduced some variation in the results, although the base data was the same. Assembly and installation processes were found to contribute only **3 gCO_2/kWh** to the life cycle carbon dioxide emissions and require **11 kJ/kWh** of energy.

Operations and Maintenance

Annual maintenance operations will mostly involve the use of sea vessels. To date a complete picture of real operation and regular maintenance has not been registered, so data for this stage was based on estimates provided by PWP. These are understood to be conservative estimates with the key aim of confirming and ensuring survivability.

The device itself has very few operational requirements. Remote monitoring and control is entirely computer-based, onshore, so no allowance has been made for the environmental impacts of this, as it is likely to be very small.

The inventory results for this life cycle stage were higher than for assembly and installation, due to the long design life, and resulted in emissions of **7 gCO_2/kWh** and consumption of **19 kJ/kWh.**

Decommissioning and Disposal

As no Pelamis devices have yet been fully decommissioned, assumptions were made about the decommissioning and disposal processes. In line with Parker *et al.* [12] it has been assumed that decommissioning procedures will include sea vessel operations associated with the final unlatching, tow to a disposal yard and recovery of all mooring hardware.

The current study assumes that the waste will be split into two streams, with the majority of the metals (90 per cent) going on to recycling plant, and the remainder of the waste going to landfill. SimaPro contains a number of databases with information about the environmental impacts of waste treatment, but none of this is UK specific. Where available, average European data for landfill of materials was selected from the European Life Cycle Database (ELCD, v2.0), but where this was not available the figures were approximated using the Swiss data published within Ecoinvent.

The potential to recycle components can have a significant effect on the environmental impact of a device, as recycling provides the opportunity for both avoiding the environmental impacts of waste treatment and also the impacts that are associated with primary material extraction. Care must be taken to avoid double-counting that can arise when credit for recycling is assigned to both the waste material and the resulting product.

There are several different methods that can be employed for dealing with recycling within Life Cycle Assessment [8]. The current study has been carried out based on the recycled content method, as this is one of the most commonly used methods in existing published LCAs. This involves simply allocating the waste that goes to recycling to an empty process, thus removing it from the landfill waste stream. Most of the credit will actually appear in reducing the impacts associated with the materials and manufacturing stage. This is different from the method used by Parker *et al*, where recycling credit was allocated to the waste stream [12]. It is likely that this will introduce significant variations in the results.

The carbon and energy intensities at this stage are **1 gCO_2/kWh** and **3 kJ/kWh**.

4.3 Life Cycle Impact Assessment (LCIA)

The final stage of an LCA, the Life Cycle Impact Assessment (LCIA), involves classifying all of the data from the LCI and characterising it into a set of impact potentials. Although it is possible to define a proprietary impact assessment method, there are many published methods available. The key selection criteria for an impact assessment method are to ensure that it includes all relevant impact potentials, and that the number of mismatches between the inventory results and characterisation factors is minimised.

The current study applies the EDIP 2003 impact assessment method. This includes a very broad range of impact categories, in line with the goal of this study, including presenting the global warming potential in terms of mass of carbon dioxide equivalent.

5 Results

All of the results are presented per unit of energy generated by the Pelamis WEC (see section 3) in order to facilitate comparison with other generating technologies.

5.4 Inventory Results

The life cycle inventory analysis produced a list of over 1600 different types of resource use and pollutant emission. The pollutants are examined in more detail with regards to their environmental impact in the next section. Table 2 includes details of the most significant raw material consumption. (Note that gravel is a raw material used in upstream processes, but does not have significant environmental impacts.)

Raw Material	*Quantity (g/kWh)*
Gravel	13.31
Coal	8.50
Iron ore	7.56
Crude oil	4.38
Fresh water	2.97
Calcite	2.77

Table 2: Significant raw materials

The inventory also details the energy consumption associated with the life cycle of the device, and found the energy intensity to be **381 kJ/kWh** (Figure 5). This corresponds to a payback time of 25 months. Over 90 per cent of this embodied energy is associated with the manufacturing stage, mostly due to the steelmaking process.

This figure agrees well with the results presented by Parker *et al.* [12], although the increase would merit further investigation. It is likely to be due to practitioner assumptions, in particular with regards to the treatment of recycling credits.

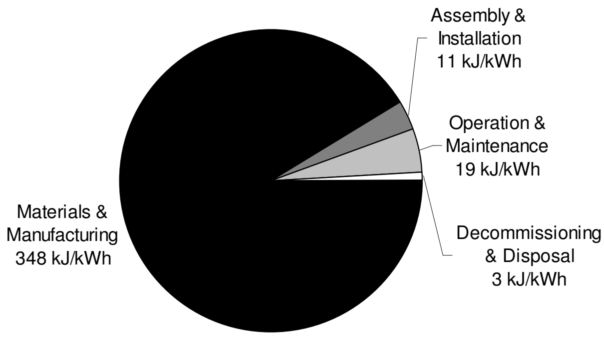

Figure 5: Embodied energy of the Pelamis WEC

In order to enable a true comparison with the figures published in Parker *et al*, the carbon dioxide emissions have also been examined at the inventory stage. Note that this does not take into account all greenhouse gases. The carbon intensity for the Pelamis is **28 gCO_2/kWh**. This is a 27 per cent increase on the earlier study, again most likely due to practitioner assumptions. Over 60 per cent of these carbon dioxide emissions are due to the manufacturing of the device, particularly in the manufacturing of the steel.

5.5 Impact Assessment

The environmental impacts of the Pelamis WEC are summarised in Table 3. It can be seen that the global warming potential (over a time horizon of 100 years) rises to **30 gCO_2e/kWh** when all greenhouse gases are included. Assuming that the carbon intensity of the offset grid electricity is 0.499 kgCO_2/kWh (see section 3), full carbon payback will be achieved in 14 months.

Impact potential	*Total*
Global warming 100a	29.8 gCO_2e/kWh
Ozone depletion	2.3 µgCFC-11e/kWh
Ozone formation (Vegetation)	0.42 m^2.ppm.h/kWh
Ozone formation (Human)	2.83E-05 person.ppm.h/kWh
Acidification	2.88E-03 m^2/kWh
Terrestrial eutrophication	5.32E-03 m^2/kWh
Aquatic eutrophication EP(N)	21.0 mgN/kWh
Aquatic eutrophication EP(P)	9.84 mgP/kWh
Human toxicity air	638.9 m^3/kWh
Human toxicity water	1.59 m^3/kWh
Human toxicity soil	5.51E-03 m^3/kWh
Ecotoxicity water chronic	10.3 m^3/kWh
Ecotoxicity water acute	1.90 m^3/kWh
Ecotoxicity soil chronic	2.87E-03 m^3/kWh
Hazardous waste	2.26 mg/kWh
Slags/ashes	3.66 mg/kWh
Bulk waste	7.90 g/kWh
Radioactive waste	468.1 µg/kWh
Resources (all)	61.6 mg/kWh

Table 3: Results of life cycle impact assessment

The relative contributions of the different life cycle stages are illustrated in Figure 6. It can be seen that the manufacturing stage is a significant contributor across all categories, again mostly due to steelmaking processes, with the shipping operations associated with maintenance also contributing significantly in some categories.

An item of interest is the radioactive waste impact category. This is as a result of the nuclear energy content of electricity. An examination of the impact flow shows that 50 per cent of this is from electricity generated in France being used in the production of European steel.

5.6 Comparison with other studies

The results for carbon and energy intensity have been compared to a number of other studies, as shown in Figure 7, demonstrating that the Pelamis performs well in comparison with other technologies.

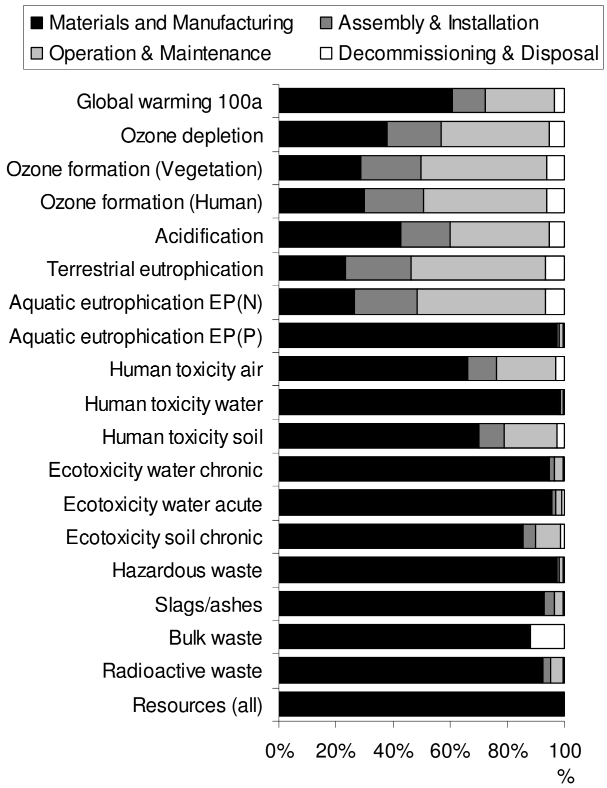

Figure 6: Life cycle stage analysis of impact potentials

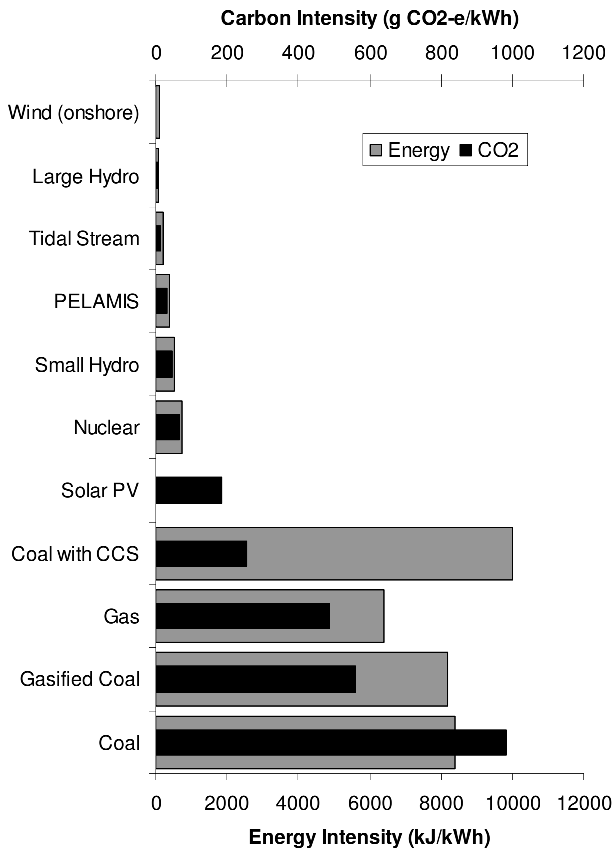

Figure 7: Comparison with other studies [7, 10, 11, 16, 17]

The results for the other impact categories have also been compared to published studies, finding that the Pelamis performs well across all environmental impacts. One such example is given in Table 4. It can be seen that the Pelamis performs significantly better than fossil-fuelled power stations with regards to pollutant emissions to the air.

Pollutant emission (g/kWh)	**Pelamis**	**Natural gas**	**Coal**
SO_2	0.0563	0.22	6.7
NOx	0.2052	0.61	3.35
CH_4	0.0555	2.6	0.91

Table 4: Comparison of life cycle emissions [14, 15]

5.7 Further Work

Further examination of the differences between the current study and that published in 2006 should be carried out, to identify where the variations in the results arise [12]. One priority will be to examine the effect of changing the recycling method applied in the analysis. The study could also be repeated with different impact assessment methods, to examine how these affect the results, and to expand the range of existing studies that can be compared.

6 Conclusions

The current paper presents a detailed full Life Cycle Assessment (LCA) of the first generation of the Pelamis. This builds upon work published in 2006 by Parker *et al.* [12], expanding the carbon and energy audit to a full assessment of the life cycle environmental impacts and considering emissions of all greenhouse gases. The resulting carbon intensity of 30 gCO_2e/kWh and energy intensity of 381 kJ/kWh compares well with the earlier study and published figures for other renewable energy technologies. The broader environmental impacts associated with the Pelamis also compare well with published studies for other power generating technologies.

The study also found that the most significant contributors to environmental impacts are in the steel structure and the sea vessel operations required for maintenance of the device.

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Life Cycle Assessment in the Marine Renewable Energy Sector

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Abstract

Reliable figures for the life cycle impacts of power generation are needed to inform developments of the energy system and enable market trading of environmental credits. Marine energy is likely to form a significant part of the future energy mix in the UK and the Pelamis wave energy converter is emerging as one of the most promising devices in this sector. This study examines the environmental impacts of the Pelamis. By comparison with the results of an earlier carbon and energy audit for the same device, the implications of practitioner decisions on LCA results are investigated, specifically with regards to the allocation method for dealing with materials recycling.

Keywords: Life cycle assessment, wave energy, recycling methods, renewable energy, carbon emissions

Introduction

The drive to reduce Greenhouse Gas (GHG) emissions has led to the development of new technologies to harness renewable energy. In the UK marine energy has the potential to supply around 20% of electricity demand, so significant developments are occurring in the marine renewables sector (Callaghan and Boud, 2006). However, while marine energy sources are themselves 'carbon-free', there are wider environmental impacts associated with the process of converting this energy into electrical power. In order to make informed decisions for future developments of the energy system, and to confidently evaluate environmental impacts for market trading, it is important to develop a detailed understanding of the life cycle impacts that arise indirectly due to the manufacture, operation and decommissioning of generators and network infrastructure.

Unlike conventional power generation and wind power there is little consensus on the general design of wave and tidal energy converters. New technologies are constantly emerging and few full Life Cycle Assessments (LCAs) have been carried out to date. Some high-level analyses, however, have been published, assessing the embodied carbon and energy of the material content of marine devices (Banerjee *et al.*, 2006; Woollcombe-Adams *et al.*, 2009). This paper details a full LCA of the Pelamis, one of the most promising devices in this sector (Figure 1). The analysis follows the framework described in the ISO 14040 series of standards, which allows a number of practitioner assumptions (ISO, 2006).

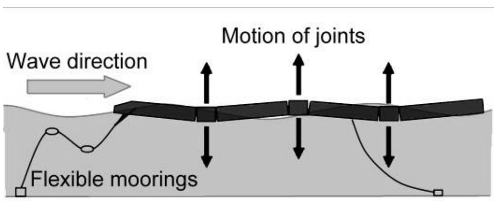
Figure 1: The Pelamis (PWP, 2011)

The results are compared to an earlier carbon and energy audit of the same device to examine how variations in assumptions and methodology, specifically that of recycling allocation, affect LCA results (Parker *et al.*, 2007). The findings will be used to better inform comparisons of the environmental impacts of different marine energy technologies.

Developed by Pelamis Wave Power Ltd, the P1 version of the Pelamis wave energy converter was successfully installed at the world's first commercial wave farm at Aguaçadoura, Portugal, in 2008. The experience has been fed into the second-generation P2 device currently on test at the European Marine Energy Centre. Several commercial projects for the P2 are under development, and lease agreements have been agreed for two Scottish farms comprising around 70 devices (PWP, 2011).

The Pelamis is a semi-submerged snake-like offshore wave energy converter. The P1 version is 120 m long, 3.5 m in diameter and rated at 750 kW. It has four cylindrical sections linked by three Power Conversion Modules (PCMs) at the hinged joints. The moorings allow the Pelamis to face into the oncoming waves and the joints flex vertically and horizontally as the wave front passes (Figure 2). This motion is resisted by hydraulic rams that pump high-pressure oil into hydraulic motors, in turn driving generators. The resistance of the rams can be tuned to maximise power capture in small sea states while protecting the device from potentially damaging storm waves.

Figure 2: Side view of Pelamis (Parker *et al.*, 2007)

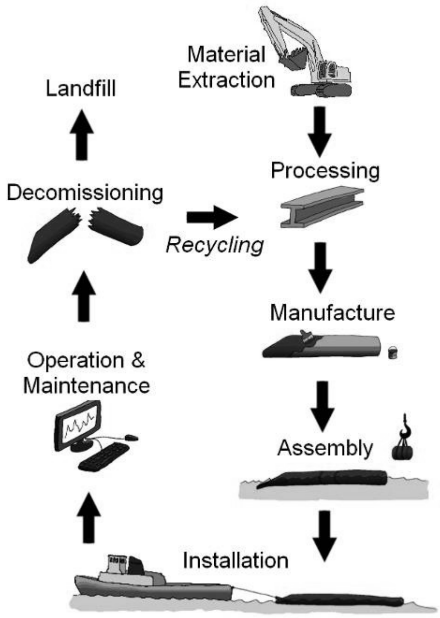


Goal and Scope

In 2007 an in-depth life cycle carbon and energy audit of the Pelamis P1 device was published by Parker *et al.* (2007). It found that the energy and carbon intensities of the generated energy were 293 kJ/kWh and 23 gCO_2/kWh. This paper expands the analysis to a full life cycle inventory and impact assessment. The results of the two studies are compared to investigate the effect of practitioner assumptions, and the comprehensive results also highlight the components, materials or life cycle stages with the largest environmental impacts.

The system boundary of the current study encompasses the entire life cycle from "cradle-to-grave" (Figure 3). Physically this includes the device, its moorings and sub-sea connecting cable, but excludes all downstream electrical components. The functional unit is one kilowatt-hour of output power (1 kWh), with a calculation reference flow of 1 Pelamis device.

Figure 3: Pelamis Life Cycle



To facilitate comparison with the analysis carried out by Parker *et al*. (2007), the fundamental assumptions and base data have been retained. In line with this it is estimated that the power output of a single device installed at a typical site off the coast of Scotland will average 2.97 GWh/year over the 20-year design life. The successful installation at Aquaçadoura found that the Pelamis performed as expected, so this assumption is still considered valid (PWP, 2011). The study assumes that all major components and sub-components are manufactured in the UK and subject to UK energy statistics and transport distances. The typical wave farm is within 200 miles of a commercial port.

The study was carried out with SimaPro (version 7.2 PhD). Life cycle inventory data was mostly sourced from the Ecoinvent database, which is recognised as one of the most comprehensive sources of such data in Europe (Ecoinvent, 2010). Data not available within Ecoinvent was sourced from alternative datasets or literature. The EDIP 2003 impact assessment method was applied, as it includes a broad range of impact categories and was developed for use with Ecoinvent data, minimising inaccuracies caused by mismatches.

Life Cycle Inventory Analysis (LCI)

The quantities of raw materials, processing and manufacturing methods, and transportation were based on figures derived from the manufacturer's own records (Parker *et al.*, 2007).

The main structure of the Pelamis is formed from cylindrical steel tube sections with sand used as ballast. The mooring and cabling system includes several plastic components. Electrical equipment, housed in the nose tube, collects and transforms the power to high voltage for export to shore. The hydraulic power-take-off, generators and control equipment are located in the PCMs.

A mass-based analysis was carried out for the structure, hydraulic system and mooring components (Table 1). Such data was not available for the pre-fabricated components, such as fixings and electrical items, and sourcing detailed LCI data for these is time-consuming, so cut-off criteria were defined to exclude inputs without a significant environmental impact (ISO, 2006). These criteria were applied to a preliminary cost-based analysis of carbon emissions and energy consumption, finding that the transformer, main generators and switchboard should be included in the study. Other pre-fabricated components were excluded as they contributed less than 1% to the total impacts.

Table 1: Material quantities in the Pelamis P1

Stock Material	Mass (kg)
Steel	561954
Sand	475722
Stainless Steel	550
Nylon 6	416
Polyurethane	343
Glass Reinforced Plastic (GRP)	90
PVC Pipe	55

The next life cycle stage involves the transportation of components from the manufacturing plant to the dockyard for final assembly. A range of sea vessels are then used for installation of the moorings and power cabling, sea trials, tow to site and latching to the moorings. Annual maintenance operations also involve the use of sea vessels. Data for this stage was based on manufacturer estimates, as a complete picture of real operation and regular maintenance has not been registered to date. These estimates are understood to be conservative, with the key aim of confirming and ensuring survivability. The device itself has very few operational requirements, as remote monitoring and control is entirely computer-based, onshore, so no allowance has been made for the small environmental impacts of this. Ecoinvent includes mass-distance data for freight transport. Other processes and sea vessel operations were approximated from fuel consumption data.

It is expected that decommissioning will involve sea vessel operations associated with the recovery of all hardware. The waste will be split into two streams, with the majority of metals being recycled (90%), and the remainder of the waste going to landfill. UK-specific LCI data for landfill is not readily available so average European data was selected from the European Life Cycle Database (v2.0). Where this was not available the emissions were approximated using Ecoinvent data for Switzerland.

Recycling of waste materials has a significant effect on the environmental impact of a device, as the use of recycled materials avoids the greater impact of primary material production. This results in an environmental credit. Marine energy converters may be responsible for both the consumption and creation of recycled materials, so it is not immediately clear where this environmental credit should be applied. Currently there is no consensus on the most appropriate methodology for allocating the benefits of recycling, as it can be applied to the product that *uses* the recycled material, the product that *produces* the recyclable scrap, or *both* products (Jones, 2009).

The recycled content approach is one of the most commonly applied allocation methods, as it is used in the assessment of cradle-to-gate impacts of materials for LCI datasets. All credit is allocated to the product that uses the recycled material, as recycling is of no benefit without the resulting material being consumed. However, recycled materials could not exist without a primary product to generate them, and therefore it could be argued that the recycling credit should be allocated to the product that is recycled. This can be calculated using closed loop substitution, the method recommended by the International Iron and Steel Institute (IISI, 2002). This was the method applied by Parker *et al.* (2007).

The 50:50 method is a compromise that recognises that both the upstream and downstream products are necessary for recycling, and assumes that half of the benefit goes to each product (Jones, 2009). The 50% figure is fairly arbitrary and open to discussion, but it does ensure that the results of different studies can be combined without double-counting. This is the only method that achieves the goal of promoting sustainable design that minimises primary material use and maximises recyclability of materials at the end-of-life.

In order to examine the effects of applying these different methods, the results are presented for all three: The Recycled Content (RC) method, the substitution method (Sub) and the 50:50 method (50:50).

Table 2: Results of life cycle impact assessment

Impact potential	**Total**			**Impact potential**	**Total**		
	RC	**Sub**	**50:50**		**RC**	**Sub**	**50:50**
Global warming (gCO_2e/kWh)	30	23	27	Hazardous waste (mg/kWh)	2.0	1.2	1.6
Acidification ($x10^{-3}$ m^2/kWh)	2.9	2.4	2.7	Slags/ashes (mg/kWh)	3.3	3.4	3.4
Ozone depletion (μgCFC-11e/kWh)	2.3	2.2	2.3	Human toxicity			
Ozone formation				*Air (m^3/kWh)*	670	450	560
Vegetation (m^2.ppm.h/kWh)	0.41	0.37	0.39	*Water (m^3/kWh)*	1.6	0.5	1.0
Human ($x10^{-5}$ pers.ppm.h/kWh)	2.8	2.5	2.7	*Soil ($x10^{-3}$ m^3/kWh)*	5.5	4.3	4.9
Eutrophication				Ecotoxicity			
Terrestrial ($x10^{-3}$ m^2/kWh)	5.3	4.9	5.1	*Water chronic (m^3/kWh)*	10.5	7.9	9.2
Aquatic (N) (mgN/kWh)	21	19	20	*Water acute (m^3/kWh)*	2.0	1.5	1.8
Aquatic (P) (mgP/kWh)	9.9	6.8	8.3	*Soil chronic ($x10^{-3}$ m^3/kWh)*	2.8	2.1	2.5
Radioactive waste (μg/kWh)	470	384	430	Bulk waste (g/kWh)	16	18	17

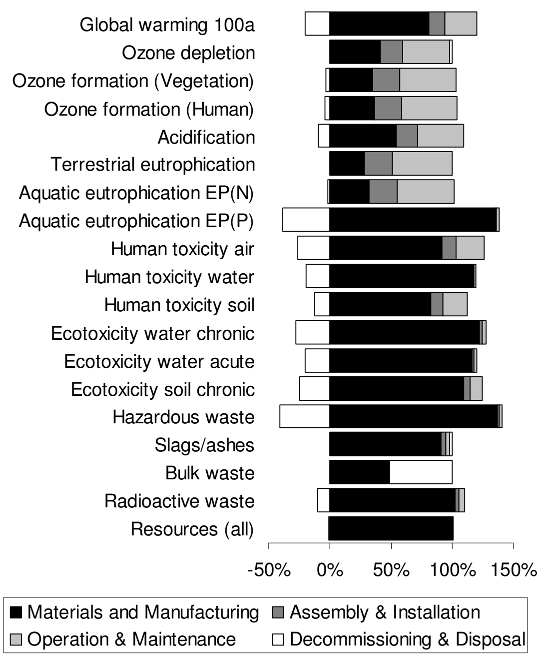
Results

All results are presented in relation to the functional unit of 1 kWh of output power. The life cycle inventory produces data for the energy consumption associated with the device (Table 3), giving an energy intensity of 310-404 kJ/kWh, which corresponds to a payback time of 21-27 months. Over 90% of this embodied energy is associated with the manufacturing stage, mostly due to the steelmaking process.

Table 3: Energy intensity

Life Cycle Stage	Energy Intensity (kJ/kWh)		
	RC	**Sub**	**50:50**
Materials & Manufacture	247	311	279
Assembly & Installation	47	47	47
Operations & Maintenance	95	95	95
Decomg. & Disposal	15	-142	-63
TOTAL	*404*	*310*	*357*

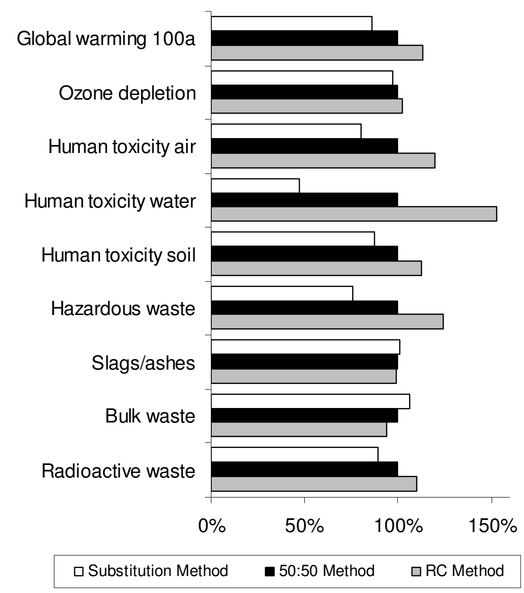
Figure 4: Life cycle stage analysis (50:50 method)



The full environmental impacts of the Pelamis are summarised in Table 2 on the preceding page. The GWP is of particular interest: 23-30 gCO_2e/kWh. Taking the carbon intensity of the offset grid electricity as the 5-year average of 0.491 $kgCO_2$/kWh, in accordance with Defra/DECC guidelines (Hill, 2009), carbon payback will be achieved in 11-15 months. This will be shorter if the device offsets only marginal carbon intensive generation.

Manufacturing and maintenance shipping operations are significant contributors across all categories (Figure 4). It can be seen in Figure 5 that the substitution method generally gives the most optimistic results, due to the average recycled content of European steel being around 40% (Classen *et al.*, 2009), and the assumed recycling rate for waste being 90%.

Figure 5: Effect of recycling method on results



Effect of Practitioner Assumptions

This study was carried out to a higher level of detail than that published by Parker *et al.* (2007), with a different software tool and different LCI datasets. In particular freight transport and waste treatment were dealt with more comprehensively. However, a comparison of the analysis results shows that it is the recycling method that has the most significant effect. If the recycled content method is applied, the results of the current study are 22% and 38% greater than those found by Parker *et al* for carbon and energy respectively. However, by applying the same substitution method as that used in the earlier study, the differences are reduced to 4% and 6% (Table 4). The inclusion of all greenhouse gases increases the carbon intensity by 5-7%.

Table 4: Comparison of results

Impact	Current study	Parker *et al.*
GWP (g CO_2-e/kWh)	23	-
CO_2 Emissions (g/kWh)	22	23
Embodied Energy (kJ/kWh)	310	293

Conclusions

This paper presents a detailed Life Cycle Assessment of the Pelamis wave energy converter. It expands an earlier carbon and energy audit to a full assessment of environmental impacts. The resulting carbon intensity of 23-30 gCO_2e/kWh generated and energy intensity of 310-404 kJ/kWh generated compares well with the earlier study. It highlights that the choice of recycling method can significantly affect the LCA results so it is important that assumptions about recycling credit are clearly stated for future studies in this sector. As the 50:50 method provides an average of both figures it is considered to be the most appropriate for marine energy converters.

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MARGINAL GREENHOUSE GAS OFFSET FOR RENEWABLE ENERGY IN THE UK

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ABSTRACT:

The reduction in Greenhouse Gas (GHG) emissions associated with the generation of electricity from intermittent renewable sources such as wind, wave and tidal power is typically estimated from the average annual emissions associated with the entire power system [1-5]. However, as the UK government continues to encourage development of renewable energy generation, negative headlines are appearing in the media concerning wind farms being paid not to produce as a result of power network constraints and questions are being raised on the future level of backup fossil generation required to handle the variability of renewable energy. This raises doubts over the accuracy of carbon payback calculations and the greenhouse gas intensity of the electricity that is offset by intermittent renewable energy.

In order to develop a real picture of the GHG offset associated with renewable energy generation this paper examines historic power generation data from the UK grid to identify the effect on the network GHG emissions of the current small penetrations of wind, wave and tidal power output. The resulting marginal GHG offset is then combined with life cycle emissions data for renewable energy converters to calculate the carbon paybacks. The model will build upon the work of Hawkes [6] that examined the marginal CO_2 emissions from demand-side interventions to produce an estimate of 0.69 $kgCO_2$/kWh. This is significantly higher than the figures recommended for use in carbon payback calculations by the UK Department for Energy and Climate Change, most recently quoted as 0.50 $kgCO_2$-e/kWh [7]. A recent study of the Pelamis wave energy converter found the global warming potential to be 27 gCO_2-e/kWh (assuming a design life of 20 years) [2], which would correspond to a carbon payback of either 9 or 13 months with the above figures.

The model developed here will feed into further work to develop more accurate estimates of the true carbon footprint intermittent renewable energy. This will enable future renewable energy scenarios to be modelled to inform government policy and energy industry plans for renewable generation and network development.

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DO WIND FARMS REALLY REDUCE CO_2 EMISSIONS?

Greenhouse Gas Emissions Savings from Wind Power

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Motivation

Wind farms are responsible for the emission of Greenhouse Gases (GHGs) over their lifetime, primarily from turbine manufacture, and therefore it is necessary to demonstrate that they are responsible for greater emissions reductions through replacing fossil-fuelled generation. Current guidance recommends estimating the emissions savings as the average emissions of UK electricity[1]. However, not all generators respond to changes in wind power output (Figure 1), and those that do operate at a lower fuel efficiency, which increases the GHG intensity of any power they generate (Figure 2).

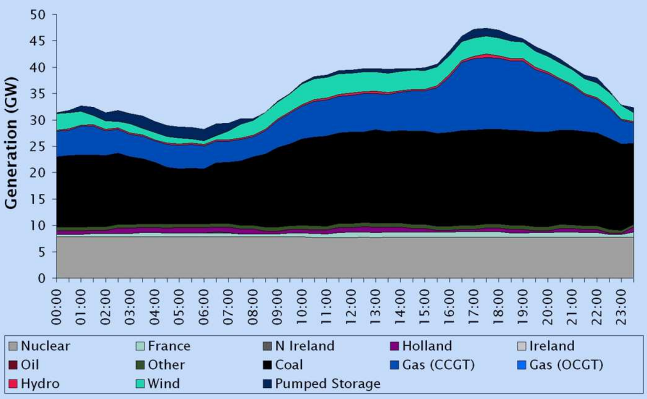

Fig 1: Total generation by fuel type, 9th December 2012

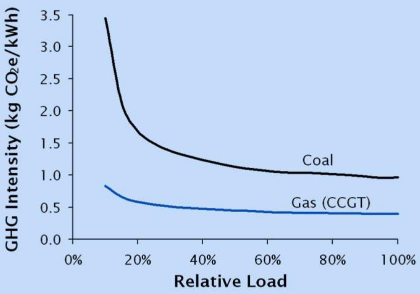

Fig 2: Effect of efficiency penalties on GHG intensity

Method

A short-term marginal analysis of real data from the National Grid for 2012 was carried out to identify the true GHG emissions savings of wind power. The method was based on that developed by Hawkes[3] and examined the relationship between the half-hourly changes in wind power output and GHG emissions across the network, taking into account the effect of changes in total power generation. A linear relationship was found (Figure 3), with the gradient representing the change in GHG emissions as a result of a change in wind output.

The initial analysis was based upon historic generation data aggregated by fuel type, and used fixed values for the GHG intensities. Further work then analysed Balancing Mechanism data[2] for every individual generator on the network so that the GHG intensity curves shown in Figure 2 be applied to the coal and gas-fired plant.

Findings

- The preliminary results show that increases in wind power result in a decrease in GHG emissions.
- When efficiency penalties aren't considered the emissions savings are 0.72 ± 0.03 kg CO_2e/kWh, showing that wind is mostly replacing gas and coal as an energy source.
- The inclusion of efficiency penalties for gas and coal plant reduces the emissions savings significantly, to 0.49 ± 0.03 kg CO_2e/kWh.

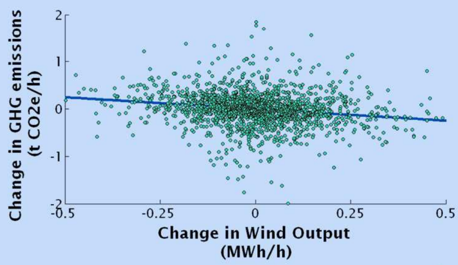

Fig 3: Linear fit to estimate marginal emissions savings savings (with efficiency penalties)

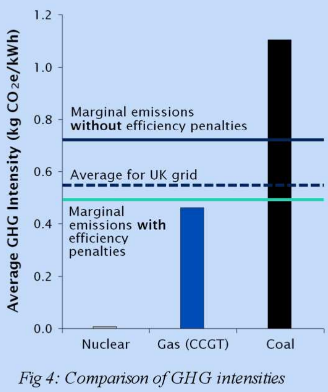

Fig 4: Comparison of GHG intensities

Conclusions

Although wind power mostly replaces electricity from gas and coal plants, thus avoiding significant GHG emissions, the actual savings are not as high as might be expected because these power stations have significantly lower fuel efficiency at reduced power outputs. These preliminary findings show that the actual emissions are close to the average for UK electricity, the value currently used to calculate GHG payback, suggesting that current estimates of payback times are valid and that wind power does reduce CO_2 emissions.

Further work will extend and refine this analysis, include data for a longer time period and consider the effect on emissions of starting and stopping conventional plant and running back-up generation.

Images courtesy of www.sixdegrees.org.au and Leaflet at commons.wikimedia.org

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DO WIND FARMS REALLY REDUCE CO_2 EMISSIONS?

Greenhouse Gas Emissions Savings from Wind Power

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Background

There is currently no reliable estimate for the reduction in Greenhouse Gas (GHG) emissions attributable to wind power generation. It is normally taken to be the average emissions of UK electricity, despite acknowledgements that this is decreased by low-carbon nuclear energy (Fig. 1) while wind mostly replaces marginal carbon-intensive coal and gas-fired generation[1] (Fig. 2). This work estimates the true emissions savings for 2012 by carrying out a short-term marginal analysis of real data from the National Grid, taking into account the efficiency penalties for operating conventional plant at part load (Fig. 3). The method, based on that developed by Hawkes[2], examines the relationship between the half-hourly changes in power outputs and GHG emissions to extract the relationship between changing emissions and wind generation (Fig. 4).

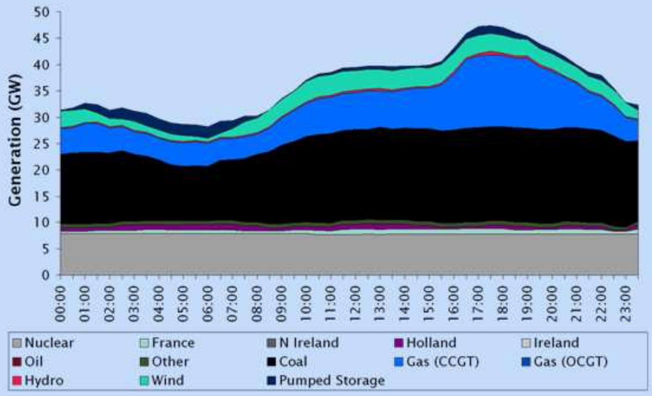

Fig. 2 – Total generation by fuel type, 9th December 2012

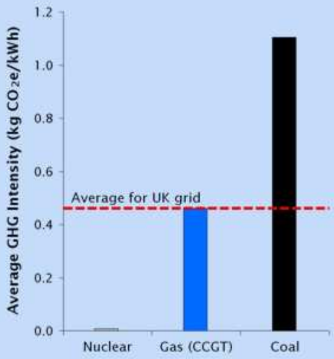

Fig. 1 – Comparison of GHG Intensities

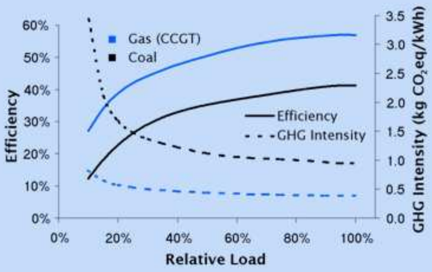

Fig. 3 – Effect of efficiency penalties on GHG intensity

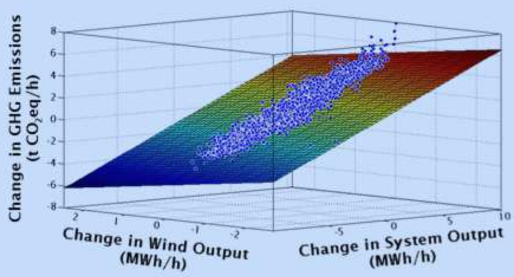

Fig. 4 – Planar fit to estimate marginal emissions savings

Results

- This analysis was based on historic generation data aggregated by fuel type, with part load curves for coal and CCGT plant developed from data from the Balancing Mechanism[3].
- With fixed values applied for the GHG intensities, the average emissions savings were **0.72 kg CO_2e/kWh** for 2012 (Fig. 5)
- When efficiency penalties are considered (Fig. 3) the average emissions savings are found to be **0.51 kg CO_2e/kWh** for 2012.
- This work does not yet include a reliable model for start-up and shut-down situations, and thus may overestimate the efficiency penalties.

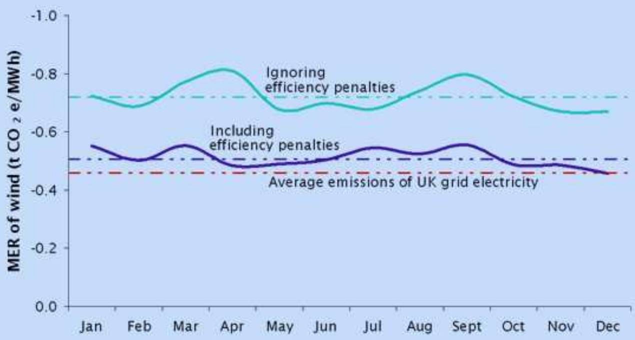

Fig. 5 – Comparing monthly and average emissions savings

Conclusions

- Reliable figures for the actual emissions savings of wind power will support carbon payback calculations and inform policy decisions.
- Wind power is mostly replacing carbon-intensive coal and gas-fired generation, but the efficiency penalties of operating these at part load mean that the GHG emissions savings are not as high as might be expected.
- The actual emissions savings are higher than the value currently used to calculate carbon paybacks, suggesting that current estimates are valid and wind farms really do reduce CO_2 emissions.

Further Work

Models of the GHG emissions during start-up and shut-down need to be developed to further refine the results. The analysis will then be extended to consider data over a longer time period, to identify whether there is a relationship between the increase in wind capacity and the GHG emissions savings.

Images courtesy of www.sixdegrees.org.au and Leaflet at commons.wikimedia.org

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