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Measuring and managing the carbon footprint of communities: a case study of Southampton, UK

by

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ABSTRACT

A significant proportion of anthropogenic greenhouse gas (GHG) generating activities are concentrated in cities (politically defined geospatial regions). Action to reduce emissions at a local level requires an understanding of emission sources, reduction potentials, and the adequate tools and resources to enable effective decision making. Carbon footprints are increasingly being used as tools for GHG management by organizations in the public and private sectors. However a number of challenges need to be addressed, not least, what does the term ‘carbon footprint’ actually mean? The term needs a universally accepted definition before a consistent, accurate, comparable and transferable methodology can be developed for municipal GHG management. This thesis investigates the concept of the carbon footprint – definitions, applications, boundaries and limitations, arguing the carbon footprint – whilst related to LCA – demands a separate definition, name, and methodology. As collection for CO_2 and CH_4 emissions is relatively straightforward, it is suggested that these two carbon-based gases should be used in the determination of a ‘carbon footprint.’ This allows the carbon footprint to become a cost-effective, practical, repeatable metric; adoptable across the globe as a “baseline” indicator. Where more complete data is available or a more comprehensive investigation is required, a ‘climate footprint’ (Kyoto Basket) is proposed. These metrics are explored considering the concept of urban process flows to develop a framework for application to municipal scenarios. Existing framework guidance often does not include all relevant emissions sources, proposes inconsistent methodologies and fails to consider the significance of limited data. A clear statement of the urban system and its boundary is developed, considering the geographic, economic and political constraints of municipal process flows and governance. Territorial-based models focusing on production emissions within the geo-political boundary and consumption-based assessments considering economy-wide impacts of consumption do not completely reflect the transboundary nature of urban communities. Standardised methods, for example PAS2070, that recognize the transboundary nature of cities have emerged through international collaboration. To consider both the transboundary and in-boundary emissions a dual modelling approach – process and consumption based – is required. Using a case study of Southampton, UK, the data and methods required for a transboundary (PAS2070 – DPSC) and a consumption-based inventory (PAS2070 – CB), are developed. The carbon (DPSC: 3957.12ktCO_2e, CB: 2570.52ktCO_2e) and climate footprints (DPSC: 4222.69ktCO_2e, CB: 2881.86ktCO_2e) are reported. One-at-a-time sensitivity analysis, with tests for linearity, are used to identify high sensitivity variables that require the highest levels of input confidence to inform assignment of effort for GHG management and reduction. Uncertainty is mapped and explored in identified most sensitive variables simultaneously using Monte Carlo methods. The application of the model to a

workable database driven tool is explored. A specification for the development of a computer-based model is provided. In conclusion, a discussion on the progress and challenges remaining in geospatial GHG inventorying at a city level is provided alongside recommendations for future policies/strategies and further research.

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DECLARATION OF AUTHORSHIP

I, Laurence Anthony Wright

declare that the thesis entitled

Measuring and managing the carbon footprint of communities: a case study of Southampton, UK

and the work presented in the thesis are both my own, and have been generated by me as the result of my own original research. I confirm that:

- this work was done wholly or mainly while in candidature for a research degree at this University;
- where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
- where I have consulted the published work of others, this is always clearly attributed;
- where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
- I have acknowledged all main sources of help;
- where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
- parts of this work have been published as: Wright, L.A. *et al* (2011) Carbon footprinting for climate change management in cities, Carbon Management, 2(1), 49-60; Wright, L.A. *et al* (2011) 'Carbon footprinting': towards a universally accepted definition, Carbon Management, 2(1), 61-72; Wright, L.A. *et al* (2012) The role of carbon footprinting in the development of 'low carbon' cities, Proceedings of the LCA XI Conference, Chicago, October 2011, USA

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Definitions and abbreviations

AFOLU	Agriculture, Forestry & Other Land Use
ARTEMIS	Assessment and Reliability of Transport Emission Models and Inventory Systems
BRE	Building Research Establishment
BREDEM	Building Research Establishment Domestic Energy Model
BSI	British Standards Institute
C40	C40 Cities Climate Leadership Group
CB	Consumption Based
CCPC	Cities for Climate Protection Campaign
CF	Carbon Footprint
CFC	Chlorofluorocarbon
CH_4	Methane
CHP	Combined Heat and Power
CO	Carbon monoxide
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COPERT 4	COmputer Programme to calculate Emissions from Road Transport 4
DECADE	Domestic Equipment and Carbon Dioxide Emissions
EEA	European Environment Agency
EEIO	Environmentally Extended Input-Output
EMEP	European Monitoring and Evaluation Programme
EU-ETS	European Union - Emission Trading Scheme
GHG	Greenhouse Gas
GIS	Geographic Information Systems
GTP	Global Temperature Change Potential
GWP	Global Warming Potential
GWP100	Global Warming Potential - 100 year horizon
GWP20	Global Warming Potential - 20 year horizon
GWP50	Global Warming Potential - 50 year horizon
HCFC	Hydrochlorofluorocarbon
HFC	Hydrofluorocarbon
HGV	Heavy Goods Vehicle
ICAO	International Civil Aviation Organisation

ICC	International Code Council
ICLEI	ICLEI-Local Governments for Sustainability (previously known as "*International Council for Local Environmental Initiatives*")
IOA	Input-Output Assessment
IOT	Input-Output Table
IPCC	Intergovernmental Panel on Climate Change
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LA	Local Authority
LCA	Lifecycle Assessment
LGV	Light Goods Vehicle
LiDAR	Light Detection And Ranging
LLSOA	Lower Level Super Output Area
LPG	Liquid Petroleum Gas
LPJ-DGVM	Lund-Potsdam-Jena Dynamic Global Vegetation Model
MLSOA	Medium Level Super Output Area
MRF	Material Reclamation Facility
MRIO	Multi-Region Input-Output
MSW	Municipal Solid Waste
N_2O	Nitrous oxide
NGO	Non-Governmental Organisation
ONS	Office for National Statistics
PA	Process Analysis
PFC	Perfluorocarbon
REAP	Resources and Energy Analysis Program
RF	Radiative Forcing
SEI	Stockholm Environment Institute
SF_6	Sulphur hexafluoride
SOA	Super Output Area
UNEP	United Nations Environment Program
US EPA	U.S. Environmental Protection Agency
VKT	Vehicle Kilometres Travelled
WBCSD	World Business Council for Sustainable Development
WRI	World Resource Institute

1. Introduction

Introduction & original contribution to knowledge

The work covered in this thesis is two-fold. Firstly the work explores the concept of the ‘carbon footprint’ as a mechanism for climate change management. Secondly the work explores how the carbon footprint and related metrics can be used in the context of sub-national GHG management and policy decision making. The work demonstrates a number of distinct original contributions to knowledge – several recognised through peer-reviewed publication. The first of these contributions is the development of an original, pragmatic definition for a ‘carbon footprint’ and a ‘climate footprint’. Secondly, a framework for the application of a carbon footprint to urban environments is developed building on the principles established by urban metabolism. This work was used to inform PAS 2070 “Specification for the assessment of greenhouse gas emissions of a city by direct plus supply chain, and consumption-based approaches” (BSI, 2013, pg. 2). Thirdly, the framework was applied to develop a novel methodology for PAS2070 compliant inventorying of a sub-national carbon and climate footprint through the use of a case study, Southampton, UK. This aspect of the project was further considered to model and estimate the sensitivity and uncertainty associated with the methods developed. This was used to help inform the project recommendations, recognising the components of the model where greatest effort is required – both to maximise community-wide gains, and improve data reliability. The final contribution is provided through a proposed specification of a software tool for use by governments to help inform climate change policy making.

The initial literature review highlighted the confusion surrounding the concept of the ‘carbon footprint’. A simple internet search for the term ‘carbon footprint’ will yield numerous results. A similar case arises when searching the academic literature. The term has proliferated both the grey and academic literature with previously little real consensus as to what it actually means. The term is often used interchangeably with ‘carbon accounting’, ‘GHG inventory’, ‘carbon inventory’, and a plethora of others. ‘Carbon footprint’ has become a buzzword in the politics, boardrooms and the media. The efforts of the academic community have, until recently been focused on more traditional life-cycle assessment (LCA). As a result there was significant confusion regarding what the term actually means. Commonly ‘carbon footprint’ was used as nothing more than a synonym for a certain amount of undefined GHG emission from a given activity. Without a clearly defined, rational definition for a carbon footprint the further development of any methodologies using the carbon footprint are inconsequential and a purely academic exercise. The study reviewed the current literature concerning existing definitions, GHG inclusion, system boundaries and accounting methods. It was clear that whilst numerous references were made to a ‘carbon footprint’; no definition adequately defined the concept and no practitioner (academic, industrial or otherwise) truly understood to what the term referred. The results of this exercise were applied to develop an original, sound and pragmatic definition for a carbon footprint. The resultant concept provides a relatively straightforward, low-cost, and understandable solution that is ideal for uptake in the public and private sectors, with application at a variety of levels: individuals; products; services; activities; events; cites; regions, and beyond. The carbon footprint bridges the gaps between knowledge, theory, rhetoric and practice in climate change management. The study rationale, results and definition were published in the journal Carbon Management. Publication of this definition has caused significant debate in the field, most recently resulting in the publication of an editorial piece discussing the terms used in GHG reporting (Turner *et al*, 2012).

Carbon footprints have applications at multiple scales from assessment of individual lifestyles to cities to regions and beyond. Application to the community level can help interpret the complex nature of climate change management, mitigation and adaptation at the local level. In the context of sub-national emissions management the carbon footprint provides a vehicle for local authority policy and public engagement. The carbon footprint places the magnitude of emissions into a meaningful context, especially for those unfamiliar with the complexities of climate science. A city was chosen as the case study vehicle for this study, however the principles of the subsequent methods, and results could be applied to any scale of community.

Building on the concepts developed by urban metabolism a novel approach for the application of life-cycle orientated carbon footprints and emission inventories to community systems was developed. The framework rationale and definition were published in the journal Carbon Management. The framework was used to inform the development of the PAS2070 “Specification for the assessment of greenhouse gas emissions of a city by direct plus supply chain, and consumption-based approaches” (BSI, 2013).

The development of the community framework and the of a methodology to assess the carbon footprint of a community involved two distinct components: i) develop the methodology, and ii) apply the methodology to a case study area.The first component of the work dismantled the framework developed in the previous phase into ‘modules’, recognising that each element of the framework would require separate calculation methods. This enabled the development of novel methods or the use of existing methods in a novel manner to create an overall methodology for the calculation of all elements of the framework. As proof of concept and to inform the development, the methodology was applied to Southampton, UK. The methodology represents a novel approach, building on established practice to enable the sub-national assessment of carbon footprints in communities. The methodology enables the spatial and temporal reporting of results at a sub-community level to enable effective management and policy development.

Sensitivity analysis - a systematic method for varying model parameters and assessing the impact of those changes - was used to investigate how 'sensitive' the model was to changes in input parameters. The analysis had two uses in the context of this project: i) identification of the variables to which the model is most sensitive, and thus which require the highest levels of input confidence; ii) identification of variables where greatest potential gains could be made through policy and other management options. One of the simplest methods of investigating sensitivity is to vary input parameters One-At-a-Time (OAT), to investigate the impact on output. The logical nature of this approach enabled the identification of the relative significance of input parameters to which output parameters are sensitive.

The OAT sensitivity analysis considered the impact of input variables in isolation - the method cannot fully explore the input space, as it cannot consider multiple simultaneous variation of input parameters. Tests for linearity recognised a number of cases where non-linear relationships exist between input parameters and output dependents. Therefore it was determined that further investigation was required to establish the effect of multi-variable impact on the output parameters. It was reasoned that simple addition of individually varied parameters would be invalid, due to the interlinked and non-linear relationship between a range of input variables. Uncertainty analysis was used to assess the impact of uncertainty on model outputs. The analysis required the nature of the uncertainties be determined - achieved through use of an uncertainty concept map - and the effects of those uncertainties combined simultaneously. The sources of uncertainty in the model were assessed and quantified using published data or expert judgement. A Monte Carlo analysis (multiple defined random simulations to obtain probabilities of a given outcome) was then performed to assess the impact of identified uncertainties on model output. The nature of each uncertainty and how it may relate across parameters was considered.

The project required an investigation into the feasibility of methods being developed into a relational database model for calculation and simulations. The construction of this database, calculations and reporting in a consistent, coherent and transparent manner, will make it easier to perform assessment studies and increase the credibility and acceptance of the results. Quality and user-friendliness were deemed the prerequisites for the establishment of a reliable tool for the assessment of community GHG emissions. Through a detailed exploration of the data requirements, calculation methods and database structure; the specification for a PAS2070 compliant database model was proposed.

Statement of study aims

Aim 1

"*To critically assess the metric of the carbon footprint and develop a sound and pragmatic definition."*

Objectives

a. Critically evaluate the definition of a carbon footprint.
b. Propose a sound and pragmatic definition for a carbon footprint.

Aim 2

"*To develop a methodology for measurement, management and reporting of a carbon footprint of a community*."

Objectives

a. Assess the applicability of the carbon footprint metric in the context of climate change management in cities.
b. Critically assess previous and existing methodologies for the carbon footprinting of cities.
c. Identify the sources, sinks and storage to be included in a city carbon footprint.
d. Apply the carbon footprinting methodology to produce an estimate of the carbon footprint of the case study city, Southampton, UK.

Aim 3

"*To identify and assess areas of sensitivity and uncertainty in the modelling process and investigate how the methods can be developed into data driven tools*."

Objectives

a. Quantify the sensitivity of the modelling process to identify sensitive variables and potential areas for mitigation.
b. Identify and evaluate sources of uncertainty within the model process.
c. Investigate the potential for the development of a data driven tool for the implementation of inventory methods.

Overview of content

This thesis is comprised of a series of papers collected into six chapters with short introductory forewords. Figure 1 provides the general structure of the research project, the outcomes (i – vii) of which have informed the structure of this thesis. The aims and objectives (i) of the project are defined previously in this chapter. Chapter 2 explores the concept of a 'carbon footprint'; previous to this research the concept of the carbon footprint was largely undefined. The paper (Wright *et al*, 2011b) at the core of this chapter explores the concept of carbon footprinting, examining the range of definitions and concluding by building on lessons learnt to develop a pragmatic and universally applicable definition for a carbon footprint (ii), forming the central foundation of the following work.

Chapter 3 comprises two papers (Wright *et al*, 2011a; Wright *et al*, 2012) (the second also presented as a conference contribution (LCA XI, Chicago, USA)) that collectively explore carbon footprints in the context of urban climate change management. The chapter develops a methodological lifecycle framework for the application of carbon footprints to urban communities (outcome iii). Built on the established principles of urban metabolism and material flow, this framework considers the inclusion of sources, sinks and stores of GHG emissions in the urban community. Also considered is the role of government in the use of assessment outcomes, which defines an order of priority for governmental management.

Chapter 4 is presented as a single paper that builds on the lifecycle framework developed in the preceding chapter to develop a methodological approach for the assessment the carbon footprint of urban communities. The methodology develops the approaches necessary to calculate emissions data for a territorial, transboundary, and consumption based footprint. The policy relevance of the three approaches are discussed through application of the methodology to the case study, Southampton, UK for the 2008 elected baseline period (outcome iv).

Chapter 5 describes a sensitivity and uncertainly analysis of the model developed in the previous chapter. A methodology for One-at-a-Time (OAT) sensitivity analysis is described; the results are further tested for linearity. Variable relationships and uncertainty are further explored through use of a Monte Carlo simulation. The results of this paper are used to explore areas of greatest gain in terms of community GHG management and to identify those areas of greatest uncertainty in the modelling process (outcome v).

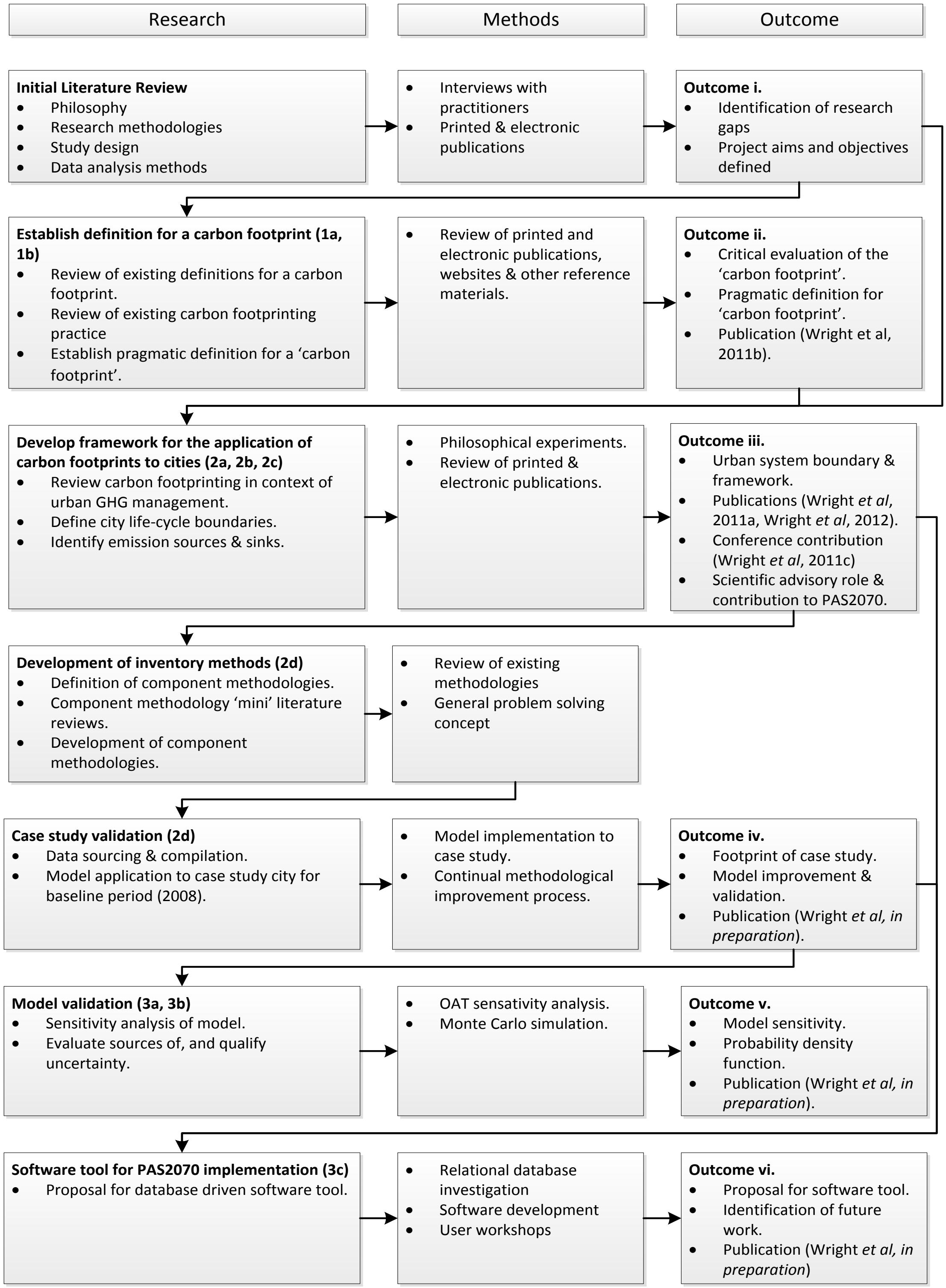

Figure 1. The structure of this thesis and the relationship between research development stages, processes, methods and outcomes including publications.

Chapter 6 (outcome vi) comprises a paper that details the development and specification of a relational database tool currently being developed. Early in the conception of the project it was realised that success ultimately relied on the ability to apply the methodological framework to various cities in a repeatable, efficient and cost effective manner. Therefore the development of a 'tool' was deemed necessary. The initial concept tool developed into *CFM: Cities*, a prototype community GHG assessment package for use by academics, consultants and government for the management of urban GHG emissions. This chapter describes the principle technologies, structures and methods that underpin the tool.

This thesis is closed with a final concluding chapter reviewing the findings of the study, the progress in the field, and the remaining challenges.

Publications Arising from this Thesis

- Wright, L.A. Coello, J. Kemp, S. & Williams, I. (2011a) Carbon footprinting for climate change management in cities, *Carbon Management,* 2(1), 49-60
- Wright, L.A. Kemp, S. & Williams, I. (2011b) Carbon footprinting: towards a universally accepted definition, Carbon Management, 2(1), 61-72
- Wright, L.A. Kemp, S. & Williams, I. (2012) The role of carbon footprinting in the development of global 'low-carbon' cities, Proceedings of the LCA XI Conference, Chicago, October 2011, USA

2. 'Carbon footprinting'; towards a universally accepted definition

Introduction

This chapter provides an examination of the origins, range of definitions, and applications for the term 'carbon footprint'. Subsequently the chapter presents an evaluation of the carbon footprint, considering lessons learnt, to develop a universal and pragmatic definition; to meet aim/objective 1a and 1b of this research project. This chapter was written for this PhD thesis by the author under normal PhD supervision conditions. It was published subsequently in Carbon Management (Wright *et al*, 2011b), with supervisors acknowledged as co-authors.

It is increasingly being recognised that robust approaches are required to measure and mitigate climate change. Climate change is a global issue; however mitigation and impacts are profoundly local. Mitigation relies on community, organisational and individual action to reduce GHG emissions from products, services and processes. The concept of the 'carbon footprint' is increasingly being recognised as a valuable tool for measurement and management of GHG emissions. 'Carbon footprint' is a concept that has become a buzzword in boardrooms, media, policy and most recently in the academic literature. Despite the prolific use of the phrase, a review of the literature indicated that there was little agreement as to an accepted definition. Indeed, the term is often used interchangeably with other terms, for example 'carbon inventory' or 'GHG inventory'.

The proliferation of the phrase through the academic literature and wider media has created a vast range of definitions and understandings. A simple internet search for '*carbon footprint*' will yield numerous hits (circa 17,350,000 results – Google 13th June 2014), with multiple differing definitions – a similar trend can be found in the academic literature. The confusion surrounding the exact use and purpose of the term presents a number of questions, namely:

- What does the carbon footprint measure?
- How should the boundaries of the carbon footprint be set?
- What methodologies should be used to reliably calculate a carbon footprint?

- How should the results of a carbon footprint be presented to create a comparable and understandable metric?

In addition to these fundamental questions is the carbon footprint a 'new' indicator or simply an exercise in re-branding of traditional LCA techniques (i.e. climate impact or GWP)?

The answer to these questions was informed by a review of a range of definitions for a carbon footprint from both the grey and academic literature. Significant variations in range of GHGs included, boundary setting (spatial and temporal), metrics, and purpose – were identified. The only commonality was merely that a 'carbon footprint' represented a synonym that somehow related human activity to an emission of GHGs. Distillation of the various definitions enabled the development of a logical and pragmatic definition which included provision for boundary setting, GHG inclusion and metrics. It is likely that a more comprehensive metric will be required in some circumstances and by some organisations, so a proposal is made for further GHG inclusion in full LCA based assessments, where complete data is obtainable it can be used to provide a 'climate footprint'. This name reflects the addition of non-carbon based gases and encompasses the full range of gases used in the global political community's response in managing climate change.

The development of a workable and consistent definition for a carbon footprint has created a cost-effective, practical and repeatable metric that can be adopted by all types of organisations across the globe as a "baseline" indicator. The carbon footprint provides a tool for governments, sector organisations, and individuals to contextualise their contribution to anthropogenic GHG emissions and promote action, which uniquely has caught the attention of the public, policy makers and the academic community.

'Carbon footprinting'; towards a universally accepted definition

Abstract

As the threat of climate change becomes more acute, so does the need for adequate measures of impact(s), management and mitigation. Although carbon footprints are increasingly being used by organisations in the public and private sectors, a number of challenges and questions need to be addressed, not least, what does the term carbon footprint actually mean? The term needs a universally accepted definition before a consistent, accurate, comparable and transferable methodology can be developed. This paper investigates the range of current definitions proposed for a carbon footprint in the context of inventoried emissions, applications, boundaries and limitations. We argue that to only account for CO_2 emissions would result in the omission of almost one-quarter of greenhouse gases (GHGs) and a significant gap in their global management, whilst inclusion of all GHGs is very time-consuming and expensive, and should be considered only in system specific life cycle assessment (LCA) based assessments; this requires a separate definition, name and methodology. We suggest that as data collection for CO_2 and CH_4 emissions is relatively straightforward, these two carbon based gases should be used in the determination of a 'carbon footprint.' This should allow the carbon footprint to become a cost-effective, practical and repeatable metric that can be adopted by all types of organisations across the globe as a "baseline" indicator. However, it is likely that a more comprehensive metric will be required in some circumstances and by some organisations, so we also propose further GHG inclusion for full LCA based assessments, where complete data is obtainable it can be used to provide a 'climate footprint'. This name reflects the addition of non-carbon based gases and encompasses the full range of gases used in the global political community's response in managing climate change. We conclude by considering lessons learnt with the proposal of sound and pragmatic definitions.

Keywords

Carbon footprint; greenhouse gas; carbon dioxide; carbon management; global climate change; life-cycle assessment; climate footprint

Introduction

The threat posed by global climate change is now widely recognised and the Intergovernmental Panel on Climate Change (IPCC) urges that action must be taken to limit global average temperature rise to 2°C above pre-industrial levels to avoid the worst effects (IPCC, 2007a). Robust approaches for the measurement and management of greenhouse gas emissions (GHGs) are required for target setting and assessing the success of climate change mitigation measures. Carbon footprints are increasingly being recognised as a valuable indicator in the field of GHG and carbon emissions management (Wiedmann & Minx, 2008).

The term 'carbon footprint' has become a commonly recognised phrase frequently used to describe the concept of relating a certain amount of GHG emissions to a certain activity, product or population. The term is also used interchangeably with other terms e.g. carbon accounting, carbon inventory, etc. The use of the term has been driven largely by media, government, industry, and non-governmental organisations, captivating the interest of business, consumers and policy makers (Wiedmann & Minx, 2008), although the term has only recently been adopted by the academic community, where effort has traditionally focused on life-cycle assessment (LCA). As a result, previous reviewers suggest that there is confusion and little consensus over what the term actually means or what the process measures (Wiedmann & Minx, 2008; Finkbeiner, 2009; Weidema *et al*, 2008; Peters, 2010).

Prior to establishing a definition of a carbon footprint, a number of issues demand answers. Firstly, what does the carbon footprint actually measure: is it a complete measure of climate change impact or an indicator of anthropogenic contributions to global GHG concentrations? How can the boundaries of the metric be reliably set to develop a comparable and reliable methodology? Finally, is the carbon footprint a 'new' indicator or simply an exercise in re-branding of traditional LCA techniques? Only once these issues have been addressed can a definition be established. The desired end use of the indicator will help inform the answers to these issues. Indeed, a number of accounting methodologies have been proposed, which depend wholly upon end use and subject. In addition, many emission-generating processes are, by nature, variable spatially and temporally, which increases the difficulty and uncertainties involved in the development of models and selection of input data. In some cases, processes are not fully understood and uncertainties may be high or not considered as an important emission source (Rypdal & Winiwarter, 2001). Even when these challenges are addressed, key questions remain; is the carbon footprint an appropriate, comparable, reliable, accurate, transferable and worthwhile measurement?

This paper addresses these questions and seeks to develop a sound and pragmatic definition for a carbon footprint that could be universally adopted and utilised. The first section of the paper critically reviews current definitions of the carbon footprint, identifying the key components and underlying theory. This critical review informs the investigation of carbon footprint accounting methodologies and approaches before discussing the potential use of a carbon footprint as a tool for the management of carbon and GHG emissions, at a variety of scales.

A clear, workable and universally accepted definition is fundamental to the development of national and international targets, legislation and standards. The final section of the paper summarises the definition issues, methodological challenges and possible uses. It is recognised that carbon footprints are a separate entity to existing impact categories in LCA. Additionally, we recognise that carbon footprints must be kept free of complicated data requirements or high costs, to ensure widespread use and uptake, especially in the private sector. It is suggested this is most easily achieved through the inclusion of carbon dioxide and methane only, which forms the basis of the definition of a carbon footprint. Consequently, a second, full impact indicator for the inclusion of all GHGs and adopting the term 'climate footprint' is proposed.

'Carbon footprint' – a review of current definitions

What does the term 'carbon footprint' actually mean? It appears to have been coined sometime around the millennium in a media article or similar; an exact origin in the scientific literature remains elusive (Wiedmann, 2009); the earliest (reasonably) authoritative etymologies we have been able to find are from 1999 (Collins English Dictionary, 2009). As a consequence, development has been largely driven by corporate, governmental and NGO initiatives, and hence the depth and clarity of definitions varies widely. Previous reviewers identified numerous definitions for a carbon footprint, from which they concluded a baseline that all definitions in some way attempt to relate human activity to an emission of a certain suite of GHG emissions (Weidmann & Minx, 2008). However, that is where the commonality ends. There is little consensus among existing definitions regarding the metrics, methods or life-cycle perspective. Some definitions require a full life-cycle assessment (LCA) of all GHG emissions, while others consider only direct emissions of CO_2. It is possible that the language of the term carbon 'footprint' is rooted in the context of ecological footprinting (Wackneragel & Rees, 1996), whereby the environmental impact of consumption is related to a land area required for its production. Indeed the term carbon footprint is, when considered in ecological footprinting studies, considered to be synonymous with the category "carbon uptake" which represents the biocapacity required to sequester CO_2 emissions from fossil fuel consumption (Global Footprint Network, 2009). However, in contrast to the ecological footprint, the likely key use of carbon footprinting is in the control/limiting of the use of fixed carbon fuels that release CO_2 (and other C-based gases), and convention suggests the carbon 'footprint' relates not to an area of land, but to a mass value representing emissions of CO_2 or a basket of identified GHGs.

Weidman and Minx (2008) summarised the range of definitions for a carbon footprint in the grey literature (i.e. non-peered reviewed). Table 1 explores this range further; presenting a summary of 'grey literature' definitions for a 'Carbon Footprint' identified using the search term '*Carbon Footprint*' in the popular internet search engine, Google. These are compared to definitions that have subsequently been published in peer-reviewed literature. In many cases, a 'carbon footprint' is little more than a synonym for GHG emissions emitted by a particular process. The majority of definitions fail to adequately define the scope and boundaries of a carbon footprint. The term is currently used to mean anything from the direct emissions of carbon dioxide from the activities of an individual (Carbon Footprint, 2009) to the amount of land required to sequester emissions from fossil fuel (Global Footprint Network, 2009).

The UK Carbon Trust attempted to address the issue of ambiguity and proposed a detailed definition of a carbon footprint (Carbon Footprint, 2009). The definition clearly identifies the GHGs to be included, all six 'Kyoto basket' gases, the metric for presenting results ('carbon dioxide equivalents' calculated using GWP100) and outlines the subjects for which a carbon footprint can be used.

The definition explicitly states that the footprint can be used to account emissions from "…a person, organisation, event or product…"; this evidently excludes the use of a carbon footprint for applications such as national and international trade accounting, which have been developed and demonstrated (Minx *et al*, 2009). This restriction is seen in a number of other definitions, either restricting the carbon footprint to a company, an individual's activities, a product, or some combination of these.

The definition proposed by the UK Carbon Trust hints at a life-cycle orientated approach, stating the footprint should include all direct and indirect emissions related to the subject, represented as carbon dioxide equivalents (CO_2e), calculated using global warming potential (GWP) (The Carbon Trust, 2009). The GWP facilitates the representation of a given GHG emission against a CO_2 equivalent (CO_2e). Presenting results as CO_2e using the GWP allows the inclusion of a range of GHGs in a carbon footprint. The GWP provides a simple, straightforward, and widely recognised index for policy makers to rank GHG emissions in terms of equivalence (Shine *et al*, 2005).

A number of definitions in Table 1 include 'all' GHGs; this is evidently too vague as the influence of a number of GHGs on global climate is still highly debated and thus one cannot be certain what to include and what to exclude. Other definitions set the GHG inclusion boundary using legislatively controlled GHGs, such as the six Kyoto gases (United Nations, 1998). Although this sets a clearly defined boundary, reducing misinterpretation, it relies on accurate data being available for all cases to allow comparability, which may not always be true (Weidmann & Minx, 2008). Weidmann and Minx (2008) propose a definition for a carbon footprint, that considers the underlying life-cycle approach, and which addresses the issue of GHG boundaries. They suggest the carbon footprint should 'exclusively' measure CO_2. This approach is consistent with a number of previous definitions in Table 1 (e.g. Weidmann & Minx (2008), Strutt *et al* (2008)), The Energy Saving Trust (2011), and may appear logical as the carbon footprint is a carbon-based metric. As with others they go on to suggest that further inclusion of GHG emissions should perhaps be termed a '*climate footprint*' (Weidmann & Minx, 2008; Hammond, 2007).

Table 1. Examples of definitions for a carbon footprint from the 'grey literature' and the academic literature

Sample Definitions
"a measure of the amount of carbon dioxide released into the atmosphere by a single endeavour or by a company, household, or individual through day-to-day activities over a given period" (Collins English Dictionary, 2011).
"The demand on biocapacity required to sequester (through photosynthesis the carbon dioxide (CO_2) emissions from fossil fuel combustion" (Global Footprint Network, 2011).
"…a gauge of your CO_2 emissions or the impact your activities have on the environment measured in carbon emissions. Your carbon footprint is measured in units, tonnes or kg of CO_2" (Carbon Footprint, 2011a).
"A 'carbon footprint' measures the total greenhouse gas emissions caused directly and indirectly by a person, organisation, event or product. The footprint considers all six of the Kyoto Protocol greenhouse gases…A carbon footprint is measured in tonnes of carbon dioxide equivalent (tCO_2e)...CO_2e is calculated by multiplying the emissions of each of the six greenhouse gases by its 100 year global warming potential (GWP)" (The Carbon Trust, 2011).

Table 2 (cont'). Examples of definitions for a carbon footprint from the 'grey literature' and the academic literature

Sample Definitions
"A carbon footprint (CF) analysis is the sum of the estimated carbon dioxide and other greenhouse gas (GHG) emissions associated with a particular activity or industry..." (Strutt *et al*, 2008).
"The 'carbon footprint' of a functional unit is the climate impact under a specified metric that considers all relevant emissions sources, sinks, and storage in both consumption and production within the specified spatial and temporal system boundary" (Peters, 2010).
"The greenhouse gases (GHGs) CO_2, methane, nitrous oxide, and fluoride emitted in the production of goods and services used for final consumption and GHG emissions occurring during the consumption activities themselves" (Hertwich & Peters, 2009).
"A carbon footprint is equal to the greenhouse gas emissions generated by a person, organization or product" (Johnson, 2008).
"The total set of greenhouse gases (GHG) emissions caused by an organization, event, product or person" (Wikipedia, 2011).
"A carbon footprint is a measure of the impact our activities have on the environment, and in particular climate change. It relates to the amount of greenhouse gases produced in our day-to-day lives through burning fossil fuels for electricity, heating and transportation etc. The carbon footprint is a measurement of all greenhouse gases we individually produce and has units of tonnes (or kg) of carbon dioxide equivalent." (Carbon Footprint, 2011b)
"A measure of the greenhouse gases that are produced by activities of a person, a family, a school or a business that involve burning fossil fuels" (US EPA, 2011).
"Your carbon footprint is the amount of CO_2 that enters the atmosphere because of the electricity and fuel you use. It's measured in tonnes of CO_2." (The Energy Saving Trust, 2011).
"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" (Weidmann & Minx, 2008).
"The total mass of greenhouse gases directly and indirectly emitted by an individual, a company or throughout the lifecycle of a product" (Moss *et al*, 2008)

Measuring the Carbon Footprint

The following sections consider the methodological issues that must be addressed prior to the proposal of a definition e.g. what GHGs should the carbon footprint include and how should it be presented? The answers to these issues are used to inform a discussion of the methodological processes available for the calculation of a carbon footprint.

Selection of GHGs

Different reporting mechanisms require the reporting of different GHGs. The Kyoto Protocol requires the reporting of six gases: CO_2, CH_4, N_2O, hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and SF_6, commonly referred to as the 'Kyoto basket' (United Nations, 1998). The European Emissions Trading Scheme (EUETS) only requires the reporting of CO_2 emissions (Council Directive 2003/87/EC). These reporting mechanisms only consider a limited selection of GHGs; there are many more emissions which affect climate (e.g. black carbon, SO_2, contrails, various aerosols, ozone precursors) (IPCC, 2007b). A full LCA attempts address this issue by reporting the full suite of environmental impacts, including all climate impact emissions to produce a holistic picture of the environmental impacts of a given system. These different reporting and accounting perspectives have generated confusion regarding the inclusion or exclusion of GHGs from the carbon footprint. Some definitions require a complete catalogue of all GHGs arising from the studied process or activity (e.g. The Energy Saving Trust (2011), Moss *et al*, (2008)), whilst others define the footprint as a measurement of only carbon dioxide (e.g. Weidmann & Minx (2008), Strutt *et al*, (2008)). Others require the accounting of legislatively controlled GHGs (e.g. the Kyoto Protocol gases (United Nations, 1998) (e.g. Carbon Footprint (2011a)).

Weidman and Minx (2008) suggest the carbon footprint should be an exclusive measure of CO_2 from the life-cycle stages of the product or activity in question. The use of the 'exclusive' term defines the carbon footprint as purely a CO_2 based assessment. They claim the benefit is twofold; the measurement of CO_2 is relatively straightforward compared with other GHGs and the term carbon footprint refers to a carbon metric. However, one must be aware of the dangers of oversimplification as reliance on the exclusive measurement of CO_2 can result in misleading outputs.

As an illustration of this issue, consider worldwide CH_4 emissions from the waste sector, which constituted approximately 14% of total global anthropogenic CH_4 emissions in 2004 (IPCC, 2008). Many developed countries have implemented technology for the capture and treatment of landfill CH_4 emissions, often combusted for the purpose of energy generation (Spokas *et al*, 2006). Applying the definition of a carbon footprint proposed by Weidman and Minx (2008), exclusive measurement of CO_2 would indicate a large carbon footprint as CH_4 combustion will generate CO_2. A likely policy conclusion from this output is that venting CH_4 is preferable to combustion; this would be an erroneous conclusion since the GWP of CH_4 is significantly higher than CO_2 (CO_2 = 1, CH_4 = 25) (IPCC, 2007b).

To overcome this potential problem, one could suggest that a carbon footprint be defined by carbon flows, whereby the footprint represents the mass or weight of carbon species emissions (e.g. CO, CO_2, CH_4, etc.). This definition would address the issues identified with combustion of other carbon species, but it fails to capture other potentially important GHG emissions and incorporates carbon monoxide (CO), a relatively low impact GHG but important ozone pre-cursor.

Moss *et al* (2008) suggest that the issue of data complication cannot be used to justify exclusion of emissions, claiming that data availability for emissions monitoring will improve over time, and a useful indicator must include all GHGs. Even if one considers the carbon footprint as an indicator for all GHGs, its use in isolation may give a misleading picture of the overall impact(s) in certain cases. To be practically useful, a carbon footprint should be an indicator of the anthropogenic contribution of a named process/product/land area (e.g. city, country) to climate change. Utilised in isolation, it may fail to consider environmental impact categories such as land-use, resulting in 'burden shifting'. For example, biofuels are preferential to fossil fuels when considered in terms of climate impact, but this fails to consider the impacts on land. A carbon footprint is an indicator of a product's, activity's or population's contribution to climate change, rather than a full life-cycle assessment, so it must be treated as such i.e. as, a decision assisting tool, rather than a decision making tool.

In the context of all GHGs, and including negative forcing, the global anthropogenic radiative forcing (RF) is estimated to be $+1.6Wm^{-2}$, (-1.0, +0.8, within 90% confidence intervals) (IPCC, 2007b). Figure 2 shows the total estimated global RF due to anthropogenic emissions of GHGs (IPCC, 2007a). CO_2 is by far the most important GHG, estimated RF of $+1.66Wm^{-2}$, followed by CH_4 at $+0.48Wm^{-2}$, with the remaining anthropogenic GHGs contributing $+0.497Wm^{-2}$ (IPCC, 2007b).

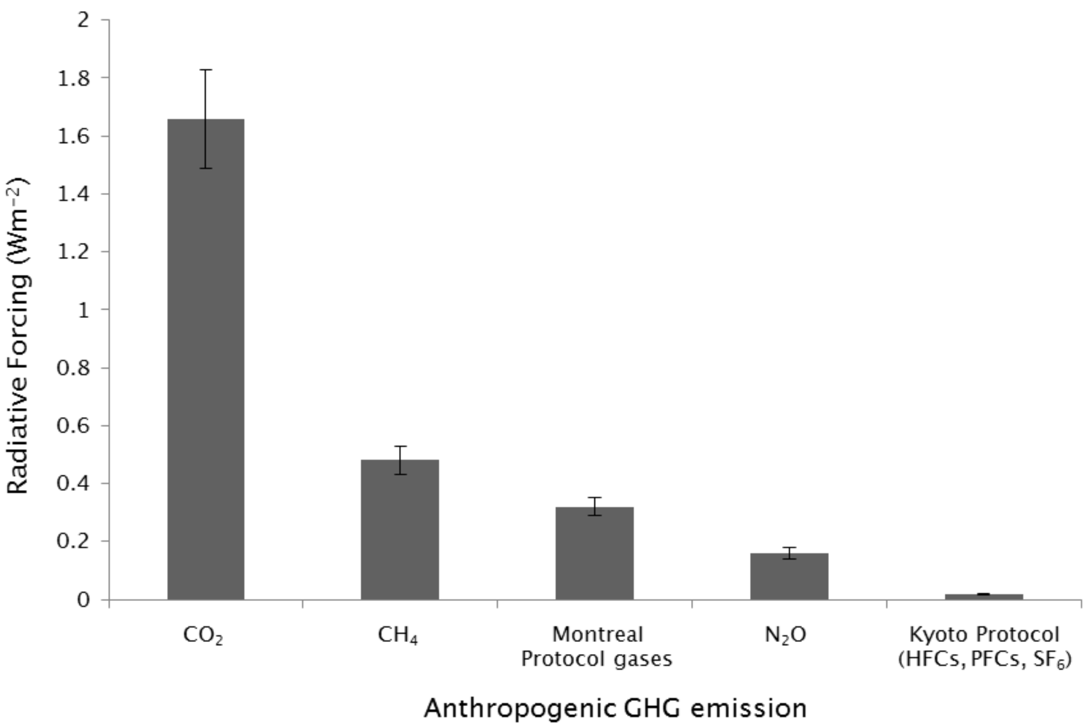

Figure 2. Positive raditive forcing of climate between 1750 and 2005 due to anthropogenic GHG emissions (IPCC, 2007a)

To only account for CO_2 emissions would capture circa two thirds of anthropogenic GHGs based on estimate global RF. However, the omission of CH_4 would result in the exclusion of almost one-fifth of GHGs and a significant gap in their global management. One could argue that exclusion of the Montreal Protocol gases would have a similar effect. However this represents the combined RF of a suite of gases (CFCs, HCFCs, chlorocarbons bromocarbons and halons) for which, since 2003, total RF is decreasing due to restricted use (IPCC, 2007b). We suggest that as data collection for CO_2 and CH_4 emissions when compared to other anthropogenic GHGs is relatively straight forward, these two carbon based gases should be used in the measurement of a carbon footprint. The effort required and uncertainties involved in the collection of data regarding the remaining suite of GHGs may not justify the outcome. Where the estimation of the full suite of GHGs is required, one could suggest this is merely slim-lined LCA, which as previously suggested could be termed a 'Climate Footprint' (Weidmann & Minx, 2008; Hammond, 2007). This is especially true in countries where detailed national reporting is currently not practiced and data must be developed from first principles.

Metrics

The way that carbon footprints are presented to policy decision makers, businesses and consumers is an important issue. Presentation and representation of carbon footprints is a key issue in the development of a pragmatic and usable carbon footprint definition. The most common unit for results is CO_2 equivalents (CO_2e), calculated using the GWP. The GWP indicator is commonly used in LCA to indicate potential climate change impacts (Bauman & Tilman, 2004), is a measure of the extent to which a given GHG contributes to global warming, representing the integrated RF of over a specified time period (e.g. 100 years – "GWP100") against a reference emission of CO_2, for example methane has a GWP of 25 (i.e. 25 times that of the reference CO_2), Therefore a 1 ton emission of CH_4 is equivalent to 25 tons of CH_4 or 25 tons CO_2 (IPCC, 2007b). The choice of time horizon applied to GWP determines the point in time at which the impact is measured. If the target year of interest is 2100 based on a 2000 baseline, application of the GWP100 is adequate. However if the target year is 2050 the application of GWP100 may result in distorted and erroneous comparisons of the relative impacts of GHGs with different lifetimes. The GWP time horizon effectively assigns a weight to time, the shorter the time horizon the greater the weight applied to impacts occurring close to the moment of emission (Moura *et al*, 2013).

Radiative forcing is used to describe the change in net irradiance at the atmospheric boundary between the troposphere and the stratosphere. Net irradiance is the difference between the incoming radiation energy and the outgoing radiation energy in a given climate system and is measured in Wm^{-2} relative to a base period, usually the year 1750 (the commonly agreed start of the industrial era) (IPCC, 2007b). More incoming energy (positive forcing) tends to warm the climate system, while more outgoing energy (negative forcing) tends to cool it. The requirement to represent GHG emissions in a way that enables the formation of targets requires a metric that accounts for different gases, with different RF and different atmospheric lifetimes. Hence the development of the GWP, a metric designed to weight gases in a manner that ensures 'equivalence' in climate impact within the chosen time frame (IPCC, 2007b). Despite its widespread use and proliferation, the GWP is not without critics (e.g. Shine *et al* (2005), Shine (2009)). Perhaps misleadingly, a GWP does not purport to represent the impact of GHG emission on temperature. The GWP relies on time integrated RF and due to the differences in atmospheric lifetimes of GHGs, it cannot provide a true indication of emission on temperature (Shine *et al*, 2005). A number of alternatives for the GWP do exist. Shine *et al* (2005) propose an alternative climate change metric, the global temperature change potential (GTP). The GTP is defined as the ratio between global mean surface temperature change at a given future time horizon following an emission (pulse or sustained) of a compound *x* relative to a reference gas *r* (e.g., CO_2) (for a full description of the GTP see: Shine *et al* (2005). The GTP could be used in a similar way to the GWP where a temperature change metric is required. However, despite the GTPs ability to more explicitly represent the temperature response due to a sustained emission, uptake has not been widespread among policy decision makers who tend to rely on the relatively well-established GWP.

However, other metrics also exist including: CO_2 per monetary unit; CO_2 emissions versus a reference unit; or CO_2 as C. Alternatively, should a carbon footprint be presented as a land area-based impact (e.g. in hectares) or – as the impact is climate change – is it more appropriate to consider a footprint as a quantification of the contribution to the impact (e.g. tonnes CO_2)? Conversion of the total emissions from physical mass (g, kg, t, etc) to area unit (m^2, ha, km^2, etc) would have to be based on a range of assumptions, increasing the uncertainty and errors contained within the metric. Following the arguments of previous authors, unnecessary unit conversions should be avoided (Weidmann & Minx, 2008). Thus a land-based metric is not appropriate; the carbon footprint should be measured in carbon equivalent mass units, reflecting the established metric of GWP.

Methodological Processes

We have established that the carbon footprint is a life-cycle orientated indicator of a process, activity, organisaiton or population's contribution to climate change. There are three principal life-cycle approaches to emissions inventories: environmental input-output analysis (EIOA); process analysis (PA), hybrid environmental input-output life-cycle assessment (hybrid EIO LCA). EIOA uses national Input-Output Tables (IOTs) as the basis for LCA. IOTs are designed for economic use and industries with very different environmental impacts are often aggregated into a single sector for reporting (e.g. paper production and publishing) and represented by average emissions intensities for the sector (Weidmann, 2009; Suh *et al*, 2003).

Top-down EIOA utilises aggregate data at the meso level to attribute emissions based on economic flow. Input-output tables provide detailed accounts of all economic activities at the sector level. Combined with environmental accounting data, they can be used to calculate the carbon footprint, setting the economic system as the boundary (Weidmann & Minx, 2008). EIOA is a top-down approach to LCA that uses national economic and environmental data such as fuel mix and technologies to estimate emissions caused by the activities of industry sectors within an economy

Weidmann, 2009; Minx *et al*, 2009). EIOA is a method for tracking economic flows (inputs and outputs) between sectors within an economy and on to final consumer demand (Minx *et al*, 2009). Tier 1 (activity data available to all countries) and 2 (technology specific) data collection methods are commonly used in EIOA (Larsen & Hertwich, 2009). Emission coefficients for units of activity of the different sectors are then used to generate emissions estimates (Larsen & Hertwich, 2009; Suh *et al*, 2003).

EIOA is effective for LCA of upstream processes and associated emissions. Given that data on a whole economy are used, the footprint generated is a complete picture of upstream GHG emissions embodied in products and services used (Minx *et al,* 2009). It is also less labor intensive than other methods of LCA as it requires no primary data collection (Weidmann & Minx, 2008). One of the main disadvantages of this approach is that GHG emissions associated with delivery, use and end-of-life of products are not calculated (Matthews *et al*, 2008). Calculation of a complete carbon footprint would require additional data and methods to estimate emissions from these missing life-cycle phases. There are also inherent uncertainties as material flows are derived from economic flows, introducing additional sources of error (Hondo & Sakai, 2001).

EIOA falls down when required to assess micro systems, such as an individual product, due to the assumption of homogeneity at the sector level. Process analysis (PA) is a bottom-up LCA approach to analyse the emissions associated with specific processes, leading to a greater level of accuracy than EIOA (Suh *et al*, 2003) but requires more time and resources (Weidmann & Minx, 2008; Larsen & Hertwich, 2009). The use of PA does raise the significant question of where to most effectively set the system boundary. System boundaries are normally set using expert judgment rather than a standard method as a more objective approach would require analysis of process emissions prior to setting the system boundary, making boundary selection redundant (Hondo & Sakai, 2001). PA is a bottom-up LCA methodology originally developed for the environmental assessment of individual products. PA excels when examine micro systems, providing the ability for assessment from cradle to grave. However, the bottom-up nature of PA can cause truncation errors, whereby only most first and some second order impacts are considered (Suh *et al*, 2003).

To overcome the shortfalls of EIOA and PA, Hybrid-EIO-LCA is becoming widely accepted within the academic community as the best approach to carbon footprint calculation (Weidmann & Minx, 2008; Larsen & Hertwich, 2009; Matthews *et al*, 2008). Hybrid-EIO-LCA is a combination of PA and EIOA approaches, where PA is carried out for specific processes and embedded within an EIOA (Larsen & Hertwich, 2009). This approach overcomes the issues of EIOA not being specific or detailed enough to monitor minor changes at an organisational or sub-national scale and omitting use and end-of-life product life-cycle phases, and the issue of incompleteness and truncation errors in PA (Larsen & Hertwich, 2009).

System Boundary and Scope

The majority of carbon footprint definitions and methodological structures consider emissions in terms of overarching "scopes" that relate to the spatial boundary of the activity or process, including the World Resource Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) (Greenhalgh *et al*, 2003). Scope 1 includes all emissions as a direct result of the defined system processes or activities (e.g. the emissions from a company owned vehicle). Scope 2 expands the boundary to consider the upstream emissions of electricity generation. Finally scope 3 further expands the boundary to include other indirect emissions of the system (Greenhalgh *et al*, 2003). This final scope is often cited as 'optional' with little guidance given as to the cut-off procedure for process external upstream process emissions. None of the definitions reviewed previously offer any explanation or guidance regarding upstream and downstream cut-off.

Suh *et al* (2003) outline a method of hybrid-EIO-LCA for estimating an industry sector carbon footprint. The method uses PA to calculate GHG emissions for the main industrial processes undertaken within the sector as well as the main industrial processes in the immediate upstream supply chain. PA is also used to estimate GHG emissions from all downstream processes to the product end-of-life phase. EIOA is used for all upstream supply chain processes for which PA is not performed. This has the benefit of being accurate and specific to the industry sector and would monitor carbon footprint variation due to changes in controlled processes while also including complete upstream emissions embodied in goods.

The temporal boundary of the carbon footprint will generally depend on the subject and the desired end use of the carbon footprint. In applications such as product manufacture, the temporal boundary is largely defined by the time taken for manufacture completion. Conversely for applications such as personal carbon footprints, a defined time period (e.g. one year) is required to facilitate boundary setting and effective measurement.

A tool for climate change management

The different reporting structures, procedures and methodologies of carbon footprinting and GHG inventorying can easily become confusing. The number of procedures originates from the high number of companies, individuals and organisations producing and developing methodologies, standards and protocols of varying complexity and technicalities and for a range of applications. The main difference between carbon footprinting at the various levels discussed is the definition of system boundaries. Whilst national accounting is structured around industrial sectors and large operational groups, the organisational procedures are structured using organisational and financial boundaries, and individual methodologies are structured on individual economic influences.

Carbon footprints of nations, regions and cities

Given the interest in the carbon footprint of products and more recently organisations, there has been comparatively less interest in the carbon footprint of nations and sub-national regions (Weidmann, 2009; Minx *et al*, 2009; Herwich & Peters, 2009). National GHG reporting was initiated by the IPCC in 1988 and is a requirement of the signatories of the United Nations Framework Directive on Climate Change (United Nations, 1994). Participating countries are required to submit an annual declaration of all GHG emissions associated with all activities in their boundaries.

Carbon footprints of nations are closely linked to debates regarding fairness in the allocation of emissions: should production or consumption be used as the basis of for responsibility in national carbon footprint allocation? Production accounting may lead to large underestimates of carbon emissions from products and services as a large proportion of carbon emissions are from supply chain processes. It often conceals issues of 'burden shifting', energy intensive process are exported to less developed countries, imports of products then increase in consumer countries (Suh *et al*, 2003). Consumption is the main driver of environmental degradation, so allocating full responsibility for emissions to producers, and giving no responsibility to consumers seems inequitable (Lenzen *et al*, 2007). This is especially true when both producer and consumer countries have legally binding emissions reduction targets (Hertwich & Peters, 2009).

The widespread use of EIOA to the carbon footprint of nations has led to the development of national carbon (GHG) trade balances where the influence of trade in producer/consumer nations is taken into account for the purposes of calculation, ensuring equitable allocation of emissions (Minx *et al*, 2009).

Carbon footprints can also be used at the sub-national level - most importantly for cities. Cities are by their very nature areas of high consumption and population density, often blamed for a plethora of environmental problems. Arguably per capita emissions in cities are reduced by high population densities; however, more than half the world's population is now urban. Local and regional governments are responsible for many decisions that affect GHG emissions such as transport and land-use planning, zoning and setting building standards as well as the management of their own activities (Greenhalgh *et al*, 2003; Salon, 2010). The development of city carbon footprinting models is relatively new, but of potential importance in providing accurate data to allow evidence based strategic decisions.

Carbon footprints of sectors and organisations

Reporting of GHG emissions in an organisational context (e.g. business units, municipal organisations, etc) is becoming increasingly important. A number of mandatory and voluntary schemes exist, the latter recognising the important role of climate change management as a legitimate business concern. A number of voluntary codes of practice and standards have been developed by industry-led initiatives, sometimes supported by non-governmental organisations. The most commonly accepted methodology is the greenhouse gas protocol, developed by the WRI and the WBCSD (Greenhalgh *et al*, 2003). The protocol can be seen to form the basis of a number of prominent carbon footprint methodologies and calculators used in the organisational context.

The most common approach to organisational carbon footprints uses emissions factors. Emissions factors relate a certain activity to an amount of emission (e.g. the mass of CO_2 released through the combustion of 1 kWh of natural gas). These factors are often based on previous detailed studies of the particular aspect (Moss *et al*, 2008). Rarely is it an efficient use of resources or possible to directly measure emissions, therefore emissions factors are an essential component of the organisational footprint. As previously discussed, EIOA provides a viable alternative to bottom-up type emissions factors; the advantage being that the boundary can be set at the economy of the economic sector, reducing truncation errors. In reality, a hybrid approach is often taken combing bottom-up and top-down analysis.

Demand for an internationally recognised standard and consistency in the calculation of organisational GHG inventory reports has led to the creation of ISO14064 part I (ISO, 2006). The ISO14064 standard provides a standardised framework methodology for the reporting of organisational GHG emissions, however it does not provide an accounting or calculation methodology.

Carbon footprints of products and services

Life-cycle assessment (LCA) was initially developed for product environmental impact analysis in order to assess and compare different methods of production and provision, enabling environmentally sound decision making. It follows that the single impact indicator of carbon footprinting should have received attention regarding the potential assessment and labeling of products.

In the UK, the Carbon Trust worked with manufacturers and businesses to introduce a carbon footprint label in 2007, a direct result of the development of a British Publicly Available Specification (i.e. not a standard), PAS2050 (revised 2011 – PAS 2050:2011 – with provision for individual sectors) (BSI, 2008; BSI, 2011). PAS 2050, Specification for the assessment of the life cycle greenhouse gas emissions of goods and services, was jointly sponsored by the Carbon Trust and the UK Department for Environment, Food and Rural Affairs and was published by the British Standards Institution on 29 October 2008 (Sinden, 2009). The specification attempts to limit the variability in interpretation of the underlying LCA methodology required for the single impact indicator of a carbon footprint. These requirements, including goal setting and life cycle inventory assessment, aspects of system boundary identification and temporal aspects of GHG emissions, clarify the approach to be taken by organisations implementing product carbon footprinting, and simplify the application of LCA procedures in relation to product carbon footprinting (Sinden, 2009).

In principle, the ISO LCA standards 14040 and 14044 (ISO, 2006a; ISO, 2006b) provide a tool for the calculation of GHGs associated with a product (Schmidt, 2009). However the standards do not explicitly document the process or boundaries required to calculate a carbon footprint, in response ISO14067 is currently being developed (ISO, *in development*).

Personal carbon footprints

There is an increasing awareness of an individual's behavior or lifestyle as a source of GHG emissions (Bin & Dowlatabadi, 2005). Individual carbon footprint software (often in the form of a "calculator") provides a tool that enables individuals to calculate their GHG emissions and link them to their lifestyle and activities. These tools play an important role in education and awareness, allowing individuals to relate more easily with climate change and mitigation. Additionally, the individual is empowered by the ability to directly visualise the impact of lifestyle changes. Some calculators go so far as to include recommendations for mitigation or investment in 'carbon offsets'.

Carbon footprint 'calculators' are now widely available on the Internet for the calculation of household or individual carbon footprints. Most models calculate individual emissions based on energy consumed and use of transport, some models exclude various types of public transport such as air travel. Calculators are provided by a range of organisations including governments, NGOs and private companies. Currently, there exist significant differences between these calculators and no standard model or practice. Previous studies have found that outputs from these calculators can vary by up to several metric tonnes (Kenny & Gray, 2009). Despite these inconsistencies, individual carbon calculators may have an important role in promoting behaviour change, and increasing the focus of public pressure on policy officials. Agreement on the definition of a carbon footprint and a method for estimating a personal carbon footprint are essential, if any decision to pursue individual carbon credits or taxation is followed. Without a method of fair, equitable and accurate apportioning of emissions, trading or taxation on carbon emissions will be impossible.

A definition for a carbon footprint

A definitive and unilaterally agreed upon definition for a carbon footprint is vital for the development of national and international targets, legislative agreements and standards. Comparability, accuracy and transferability cannot be ensured until this agreement is reached. It has been established that the carbon footprint has a number of key criteria, from which a definition can be proposed. The carbon footprint builds on the principles established by LCA. The metric uses the concept of the GWP indicator to present anthropogenic climate change impacts. Whilst the GWP has been criticised, it remains widely used and accepted by policy makers. The GWP enables the representation of any number of GHGs in a consistent manner of carbon dioxide equivalence. Emissions of CO_2 account for greater than half of all anthropogenic emissions; however, one cannot ignore CH_4, the second most prominent GHG, due to its direct relationship with CO_2 production. Data are usually readily available or easily gathered for these emissions, whereas complications arise in data acquisition for the remaining GHGs and uncertainties are increased with GWPs. In some cases it is recognised that other GHGs (e.g. N_2O, SF_6, etc) will be important; however, in specific environmental assessments, arguably LCA is preferable.

Taking these key components into consideration, the carbon footprint should act as a proxy indicator of a process's, product's, activity's or population's contribution to anthropogenic climate change, accounting for the most prominent anthropogenic GHGs, CO_2 and CH_4. Further GHG inclusion should perhaps be termed, as a *'Climate Footprint*' conducted in the context of LCA. We propose the following definition for a carbon footprint.

"A measure of the total amount of carbon dioxide (CO_2) and methane (CH_4) emissions of a defined population, system or activity, considering all relevant sources, sinks and storage within the spatial and temporal boundary of the population, system or activity of interest. Calculated as carbon dioxide equivalents (CO_2e) using the relevant 100 year global warming potential (GWP100)."

Thus we define a carbon footprint as a measure of CO_2 and CH_4 emissions. This excludes other GHGs, CO_2 and CH_4 are by far the most prominent GHGs (see Figure 2). Additionally accurate measurements of other GHGs are not always available, thus comparability could not otherwise be insured. In cases where data are available and a wider measure of GHG emissions is needed, we suggest that a wider measure of all GHGs controlled under the Kyoto Protocol caused by a defined population, system or activity is of value in some situations to evaluate climate risk more accurately. This should be defined as a 'climate footprint' (for a detailed discussion see Williams *et al*, 2012).

The use of the term 'population or activity' ensures the definition does not exclude potential subjects from the methodology, the definition can be interpreted to include products, services, events, individuals, companies, sectors, processes, communities, cities and even nations. It must be realised that these scales are not additive due to issue of double counting, and the different measurement perspective. For example, the summation of individuals cannot represent the emissions of a city, as this would not accurately or equitably assign emissions from transboundary sources such as transport. It is recognised that sources of emissions will not always be located within an easily defined geographic boundary (e.g. an organisation's manufacturing site). A proportion of emissions could originate from process chain related sources, for example grid electricity generation or imported products. Whilst one must be aware of the issues of possible double counting inherent in life-cycle based assessments, the definition can include all direct, upstream and downstream emissions, depending on the desired outcome and system boundary setting, acknowledging the life-cycle perspective and origin of carbon footprinting. Lastly, the footprint is presented in CO_2e using a 100 year GWP, ensuring consistent and comparable representation of results. Our definition potentially provides a clear, practical, cost-effective and universally applicable solution to the question of 'what is a carbon footprint?'

Conclusions

The term carbon footprint is widely used and discussed in grey literature and more recently in the academic community, but no commonly agreed definition has yet been established. The development of the carbon footprint can trace its origins to the concept of the ecological footprint and to the principles developed in LCA. This paper has highlighted the myriad methodological and philosophical issues that need to be resolved before a universally accepted definition for the term carbon footprint can be realised.

There is debate regarding the choice of GHGs to be included in the carbon footprint. This depends heavily on the purpose of the measurement, whether it is to be another name for the climate impact category utilised in LCA or a separate indicator of anthropogenic GHG emissions. It has been demonstrated that a significant proportion of anthropogenic GHGs can be captured through measurement of CO_2 and CH_4. These two gases are relatively easy to monitor and data is readily available worldwide. This is not the case for other anthropogenic GHGs for which uncertainty in data is often greater. Where further inclusion of GHGs is required, one should either refer to traditional LCA or consider reference to a '*Climate Footprint'*.

The carbon footprint has developed based on the concepts proposed by LCA, and consequently the carbon footprint has been estimated via a number of different life-cycle methodologies. The choice of methodology depends highly on the subject and desired outcome, although the concept of hybrid accounting is becoming the methodology of choice. However, it is not always abundantly clear where system boundaries are drawn or where cut-offs are created. These issues will need to be addressed on a case-by-case basis as the use of carbon footprints become more widespread. Carbon footprints can be used with a variety of cases and subjects and the methods used for calculation should reflect them.

By addressing the above issues, we propose a sound and pragmatic definition for the carbon footprint, that we feel can be applied universally worldwide to a variety of cases and subjects. The definition is developed for the concept of the carbon footprint as an indicator of anthropogenic contribution to climate change, versus a full climate change impact 'tool', as this has been previously achieved with the development of LCA. Achieving true sustainability requires the evaluation and consideration of all relevant environmental impacts. By definition then a carbon footprint should constitute a tool to aid in GHG mitigation policies and practice, at all levels from the individual to nation. The carbon footprint must be recognised as a tool for assisting decision making, not a definitive answer. If decisions based on the carbon footprint indicator are correct the majority of the time, it is better than no indicator and decision making with ignorance. Carbon footprinting is by no means the only aspect of the LCA process to be simplified for the purpose of communicating complex environmental issues, but uniquely has been able to catch the attention of the public, policy makers and the academic community. As such the carbon footprint has the potential to stimulate improvement in GHG management and climate change mitigation.

3. Carbon footprinting for the development of low carbon communities

Introduction

This chapter explores carbon footprinting as a tool for GHG inventories in communities, specifically cities, and the role of sub-national government in the management of a community footprint, to meet Aim/Objective 2a, 2b and 2c of this research project. Presented in two parts, the first section considers the urban system, processes, and sub processes to develop a novel framework for the application of carbon footprints. The second section explores the role of subnational governance in the management and mitigation of climate change, by exploration of the impacts and the modes of mitigation. The two sections of this chapter were written for this PhD thesis by the author under normal PhD supervision conditions. The two sections were subsequently published in Carbon Management (Wright *et al*, 2011a), and as Proceedings of the LCA XI Conference, Chicago (Wright *et al*, 2012) (with associated conference presentation), with supervisors acknowledged as co-authors.

Climate change is a global issue; however the solutions and impacts are profoundly local. Local governments must play a central role in the management of action to reduce emissions at a local level requires that municipal and local governments have a good understanding of emissions sources and reduction potentials. Local and regional governments are responsible for many decisions that affect climate change management including transport and land-use planning, zoning and setting building standards as well as the management of their own activities (Fleming & Webber, 2004; Salon *et al*, 2010). For this to occur the global issue of climate change must first be localised. Subsequently local authorities must be equipped with the tools and techniques needed to management climate change.

The carbon footprint is increasingly becoming recognised as a valuable tool for the localisation of climate change (Wright *et al*, 2011b). Development has been driven by NGOs, government, organisations and individuals. As explored in the previous chapter this has created significant confusion regarding what the metric measures, the methods and the representation of data. The previous chapter culminated with the proposal of a pragmatic and universally applicable definition of a carbon footprint. The carbon footprint has applications for nations, regions, communities, organisations and individuals. The metric enables the localisation of climate change contributions and mitigation measures.

Significantly it offers sub-national governments, commonly cited as a primary actor in the mitigation and management of climate change (Wright *et al*, 2011a) the ability to measure, manage and mitigate GHG emissions in their community. It offers the opportunity to municipal governments to develop models to inform climate change strategy decision making. Enabling municipal authorities to localise the issue of climate change and promote the benefits of climate change mitigation at the local level.

Action to reduce emissions at the urban level requires an understanding of life-cycle emissions sources and reduction potentials. Policy response(s) by municipal authorities needs to consider all emissions scopes across the urban lifecycle. In many ways a community can be considered similar to a large incumbent organisation, with a series of distinct manageable processes (e.g. transport, domestic energy consumption, industrial processes) occurring across the urban lifecycle. To develop methods for the measurement of the carbon footprint of these processes a clear statement of the community system and boundary is required.

Practitioners and governments are presented with a number of options when considering the community system. Evidently the community is defined in political and geographic extent by the geopolitical boundary of the sub-national government. However this differentiation does not consider the role of communities as transboundary entities. A community does not simply serve residents but has significant cross boundary flows of people, goods, services and resources. It is important to realise the need for complete inclusion of all sources and sinks of emissions including those indirect process emissions occurring outside the geographic boundaries. Exclusion of out-of-boundary emissions risks carbon leakage and burden shifting.

This consideration of source inclusion must be placed within the context of sub-national government influence and control. Inclusion of transboundary processes, for example energy provision is vital to ensure equitable emissions apportioning; whilst inclusion of emissions from products and services are less within the influence of government. However, to achieve holistic management of emissions and avoid implementation of policies that may lead to burden shifting and increased emissions outside communities, inclusion of these emissions is essential.

Carbon footprinting for climate change management in cities

Abstract

A significant proportion of anthropogenic greenhouse gas generating activities are concentrated in cities. As centres of high consumption, wealth and creativity, cities must play a significant role in tackling climate change. Action to reduce emissions at a local level requires that municipal and local governments have a good understanding of emissions sources and reduction potentials. To achieve this municipal governments require adequate tools and resources to enable effective policy decision making. The carbon footprint is becoming an increasingly recognised tool for the management of climate change. The term carbon footprint originated in the grey literature, it is widely recognised in the public arena. It offers the opportunity to municipal governments to develop models to inform climate change strategy decision making. Enabling municipal authorities to localise the issue climate change and promote the benefits of climate change mitigation at the local level. Existing framework guidance often fails to include all relevant emissions or follow widely varying methodologies, limiting comparability. This paper examines the concept of climate change localisation and management. The carbon footprint is explored in the context of a tool for municipal government management of GHG emissions. We conclude by suggesting the carbon footprint become a cost-effective, practical and repeatable metric that can be adopted municipal governments across the globe as a “baseline” indicator.

Keywords

Carbon footprint, GHG management, carbon management, GHG inventory; city metabolism

Introduction

The threat posed by global climate change requires international action that includes commonly agreed approaches for the measurement and management of greenhouse gas emissions, target setting and the development of climate change mitigation measures. Cities are areas of high consumption and population density, often responsible for a plethora of environmental problems (Dodman, 2009); >2/3 of the world’s energy was consumed in cities in 2006 (Bader & Bleischwitz, 2009) and at least half of the world’s population now lives in urban areas, with the numbers of mega cities (populations greater than 10 million) increasing rapidly (UNFPA, 2007). As a consequence, the activities within, and management of cities are disproportionately important in terms of global climate change policy making.

Local and regional governments are responsible for many decisions that affect climate change management in cities, including transport and land-use planning, zoning and setting building standards as well as the management of their own activities (Fleming & Webber, 2004; Salon *et al*, 2010). They must play a central role in reducing greenhouse gas (GHG) emissions. In order for this to happen a better understanding of local responsibilities in relation to national targets must be developed (Carney & Shackley, 2009). This requires the localisation of national climate change legislation and targets. Political emphasis has traditionally focused on global solutions negotiated through agreements such as the Kyoto Protocol (United Nations, 1998) and national legislation such as the UK's Climate Change Act 2008 (Climate Change Act 2008, c.27). A shift to sub-national, regional and municipal climate change governance has emerged in recent years, giving rise to programs such as the 'Cities for Climate Protection' (CCP) (ICELI, 2008).

In pursuit of climate change localisation the concept of the 'carbon footprint' is increasingly being recognised as a valuable tool in GHG emissions management. Despite its proliferating use, there is still little consensus in the academic community to what it actually means (Wright *et al*, 2011b). Following a previous review, we understand the carbon footprint as a life-cycle climate change metric: *"A measure of the total amount of carbon dioxide (CO_2) and methane (CH_4) emissions of a defined population, system or activity, considering all relevant sources, sinks and storage within the spatial and temporal boundary of the population, system or activity of interest. Calculated as carbon dioxide equivalents (CO_2e) using the relevant 100 year global warming potential (GWP100)"* (Wright *et al*, 2011b). The development of the carbon footprint was driven largely by organisations; as a result the majority of research has focused on methods for the calculation of product carbon footprints, organisational carbon footprints, and individual carbon footprints. Interest in the carbon footprints of cities is increasing (Fleming & Webber, 2004; Carney & Shackley, 2009; Kennedy *et al*, 2009; Kennedy *et al*, 2010; Peters, 2010). Carbon footprinting provides policy makers with a mechanism for the localisation of climate change. Carbon footprints can provide an uncomplicated, low cost solution, ideal for widespread uptake in the public and governmental sectors. A more comprehensive metric for the assessment of all GHGs may be required in some circumstances, where complete data is obtainable it can be used to provide a 'climate footprint' (Wright *et al*, 2011b; Williams *et al,* 2012). This name reflects the addition of non-carbon based gases and encompasses the full range of gases used in the global political community's response in managing climate change. As with other forms of carbon footprinting, city carbon footprinting must be as accurate as reasonably possible to track progress and to form the basis of future carbon taxation and pricing schemes (Salon *et al*, 2010), whilst still considering the time and financial constraints often placed on municipal governments.

In many ways a city can be considered similar to a large incumbent organisation, utilising the same principles developed for the carbon footprinting of organisations (Peters, 2010). City based footprints often include direct emissions and emissions from electricity. Less common is the inclusion of further production chain emissions due to data collection complications and difficulties in boundary assignment. The capacity of urban areas for carbon sinks and storage, may be significant, in which case there is a strong argument to additionally report sinks and storage within the system boundary. Boundary definition presents a significant issue; cities do not just serve residents, but have strong involvement at both national and international scales.

There is increasing interest in the carbon footprint of regions, local areas and cities (e.g. New York City (2010), GLA (2007)), although the cataloguing process is yet to mature (Kennedy, 2009). The models vary considerably in complexity, boundary conditions and emissions sources included in an estimation, making comparison of the overall carbon footprint between cities difficult or at worst, invalid. Successful achievement of GHG inventory emission targets depends on the availability of high-quality data, which implies transparency, consistency, completeness and accuracy (Rypdal & Winiwarter, 2001). However, whilst voluntary frameworks have been developed, no internationally accepted standard methodology currently exists providing detailed guidance for the collection of a city based carbon footprint (at time of writing) (Dodman, 2009). The development of a standardised methodology requires consistent boundary selection, driven by clear, defined responsibility and emissions allocation, and methods to be utilised for the calculation of the urban carbon footprint between cities.

This paper aims to: i) critically evaluate the carbon footprint as a potential tool for management of emissions at a municipal level and ii) critically discuss key methodological issues for sub-national community carbon footprinting, including boundary selection, allocating responsibility for emissions and treatment of carbon sinks.

Cities and GHG Management

The 1987 Bruntland Report included a specific chapter on the environmental issues facing cities, arguing that as an increasing proportion of the population live in urban areas cities are key to attaining sustainability objectives (World Commission on Environment and Development, 1987). Not only are cities a major source of GHG emissions, their populations also possess the knowledge, creativity and resources to reduce them. These principles were taken up at the 1992 Rio Earth Summit in Agenda 21 chapter 28 which calls for municipal authorities to establish a Local Agenda 21 (LA21). LA21 promotes the localisation of global issues, including climate change, the so termed “act local, think global”. Through cooperation with their local communities municipal governments establish mechanisms for local action on GHG reduction and climate change management. LA21 additionally includes mechanisms to facilitate and encourage communication with other local authorities. In theory, via this process, cities may learn from each other, developing and adapting suitable strategies for the reduction of GHG emissions (United Nations, 1992).

As demonstrated by the development of LA21, climate change requires policy response at the international level, although the impacts and solutions are local (Collier & Löfstedt, 1997). In any local action, the initial motivator will have an element of global awareness; however, the action involves local actors working with local projects to reduce emissions, contributing to national and international reduction targets. Political rhetoric must frame climate change in a way that makes it solvable at the local level (Betsill & Bulkeley, 2006). Concepts, such as the carbon footprint, provide a mechanism for the measurement and management of emissions on a local scale. Policy makers and constituents can relate and understand the concept more easily than the global nature of climate change. Elements of global climate change policy can mutually reinforce local sustainability agendas, where action can be reconciled with secondary benefits. Some benefits are environmental, emphasising the linkages between global and local environmental problems. Such links include improved local air quality, improvements in public health, reduced traffic congestion and better urban liveability. Additional benefits focus in the social and economic areas, such as reduced costs of municipal operation and local job creation (Lindseth, 2004). Localisation of the global issue of climate change is most likely to occur where appropriate policy responses (to control CO_2 emissions) are already present (e.g. in the UK, local air quality monitoring areas (The Environment Act 1995, s83(1)), rather than providing a justification for policy action in itself (Bulkeley & Betsill, 2003). By publishing the co-benefits of climate change policy, local authorities can localise the issue of climate change for their constituents and justify expenditure on climate protection initiatives (Kousky & Schneider, 2003).

Municipal governments have significant influence over areas crucial to GHG reduction and management. They are pivotal in managing both transport demand and supply, with responsibility for supply and maintenance of infrastructure and traffic control mechanisms (Collier & Löfstedt, 1997). They have considerable buying power, and through the provision of healthcare, education and municipal services, are poised to develop new markets for energy efficient products and services (Larsen & Hertwich, 2009). Furthermore, they have significant influence over land use planning, waste management and energy consumption, all of which have implications for CO_2 emissions (Bulkeley & Kern, 2006).

To establish an effective, workable climate change emissions reduction policy and action, municipal governments must first identify the appropriate means of monitoring and assessment. Many cities have established inventories of CO_2 or wider GHG emissions, using simple pragmatic approaches, such as the approach of the International Council for Local Environmental Initiatives (ICLEI) (ICLEI, 2008). These inventories provide the basis for action on climate change, providing a measurement and method of identification of cities with lower per capita emissions. For strategies to work, it is necessary for cities to have reliable, consistent, and transparent inventories and to understand why emissions totals differ between cities (Kennedy *et al*, 2009). This requires the development and widespread uptake of a common indicator, such as the carbon footprint. This requires cooperation at the national and local levels, even where the initiative exists, the translation of political rhetoric to policy is often hampered. Research on the Cities for Climate Protect (CCP) program identified two obstacles for the development of policy at the local level: the global framing of climate change; and institutional barriers (Betsill, 2001).

The global framing of climate change leaves individuals and policy makers feeling distanced, in both space and time, and are therefore unlikely to see the problem as legitimate local or individual concern. GHG emissions do not respect international boundaries or treaties, effecting climatic systems at a complex global level, over prolonged periods of time, local regions are only affected through impacts on the global-scale climate system. Officials and individuals have a limited understanding of the future implications, and how they might mitigate the problem. At the individual level, the scientific rationale and moral concerns of climate change are often outside the temporal and spatial perspective. People do not define climate change in terms of personal risk because the linkages between cause and effect are perceived to be weak.

Reasoned choice may suggest that it is not good business sense for a municipal authority to expend effort reducing its GHG emissions as it cannot be certain that action in the local environment will make measurable difference to the global threat of climate change (Betsill, 2001).

The Cities for Climate Protection Campaign (CCPC) successfully localised the policy of controlling GHG emissions rather than localising the problem of climate change (ICLEI, 2008). CCPC does not discuss climate change with reference to climate change or the harm it causes nature, but with reference to local benefits and the need to appeal to many diverse interests (Lindseth, 2004). CCPC did also try to frame climate change in a picture of vulnerability for city dwellers (city dwellers are at risk of climate change and therefore should cut emissions). However, using risk as a basis of local climate action is not easy; no clear links have been identified between local action and the effect on global climate change (Lindseth, 2004). Effective action requires the localisation of climate change emissions, impacts, and actions. The carbon footprint provides a mechanism to achieve this localisation.

Boundaries and scope

Cities looking to develop carbon footprints must first examine the issue of boundary selection. In very simple terms, municipal leaders are presented with a series of choices for boundary setting: municipal operations; all emissions directly emitted from the city; all territorial emissions, including process linked emissions (e.g. production, generation), or consumption emissions. The options offer progressively more completeness at the cost of increased complexity.

Cities generate GHG emissions as the result of a range of processes and activities, within and outside their geographic boundaries. Emissions sources can be categorised in two forms: ‘direct’ and ‘indirect’. Direct emissions include those emissions occurring within the geographic boundary of the city, not associated with energy production. Including all emissions associated with in-boundary fuel consumption, industrial processes, and transport. Emissions from the production of energy are traditionally excluded from ‘direct’ sources due to the commonly national nature of energy distribution. Energy produced in a given locale will likely be utilised at a national level through ‘grid’ type transmission networks. Indirect emissions include those associated with sources outside the geopolitical spatial boundary, including emissions from primary energy production, electricity generation, waste, and imports/exports. The World Resource Institute (WRI) differentiates emissions sources in terms of ‘scopes’ (1-3) that relate to the spatial boundary of the activity or process. Scope one is defined as direct emissions within the geographic or administrative boundary of the city, e.g. emissions from the domestic combustion of heating fuels, or industrial process emissions (Greenhalgh *et al*, 2003). Scope two considers emissions from electricity generation at a gird level, apportioned to end-users at the city level. Scope three considers further non-electricity out-of-boundary indirect emissions sources, e.g. emissions associated with energy production chains, or imports and exports. Emissions sources can be further classified by life-cycle perspective (Kennedy *et al*, 2010), either production chain or single process emissions. Upstream emissions occur as the result of the services, energy and products supplied to a city. Downstream emissions result from the outputs of in boundary processes, including waste, and exported products and services. Some transboundary processes, for example shipping and aviation are difficult to categorise. Emissions from these sources can be considered both direct (scope 1) and indirect (scope 3). Some studies completely exclude emissions from aviation, other studies (e.g. Carney & Shackley, 2009) exclude international flights and include emissions from the take-off and landing cycles of domestic flights at the local level. The United Nations Framework Convention on Climate Change suggests emissions from cruising aircraft be excluded from emissions inventories. Other studies, for example London (GLA, 2007) and New York City (New York City, 2007), consider emissions from aircraft in terms of fuel loaded on airports within city boundaries. Whilst technically considering movement away from the host city, this approach begins to consider the economic importance of aviation to the host city. This method does not consider the movement of passengers between flights and the surface movement of passengers from outside city limits, although this could be considered a function of the city (Kennedy *et al*, 2010). The majority of emissions from these processes will be outside the geographic boundary of the city. However, cities must take responsibility for emissions as a direct process due to the social and economic benefits (e.g. local economic growth, employment) gained, and a certain level of control through mechanisms such as tourism and business development.

Figure 3 demonstrates the concepts of emissions sources in cities in relation to scope and life-cycle perspective, from the perspective of the processes occurring within a city. Figure 4 demonstrates the same concept, although in terms of the wider process boundaries.

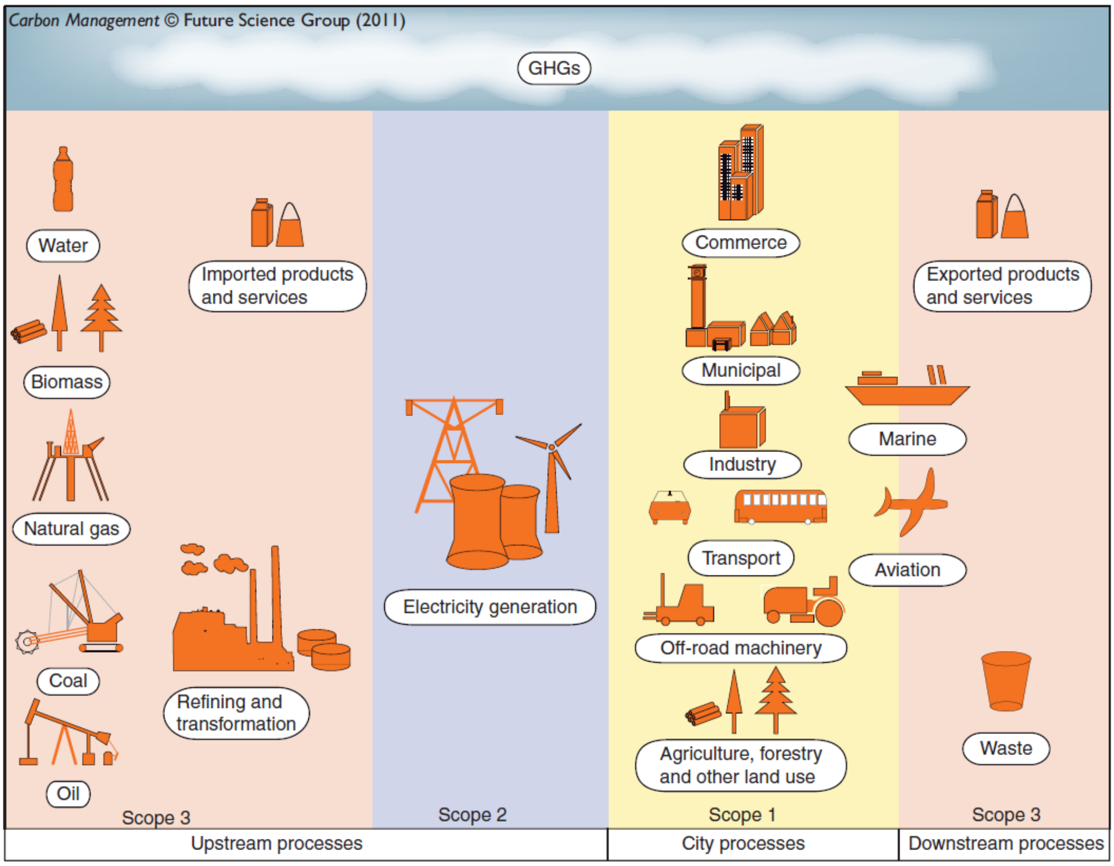

Figure 3. Emissions sources in cities in relation to scope (a way of differentiating emissions sources employed by the World Resource Institute) and life cycle perspective, from the perspective of the processes occurring within a city (Wright *et al*, 2011a)

The inclusion of all scope 1 – 3 emissions sources does raise an issue with regards to equity and fairness. Cities with net exporting energy production industries will be penalised over cities who are net energy importers. Thus a boundary that considers process linked emissions becomes favourable, as process energy emissions can be assigned on an end-user basis to ensure fairness and equity. More difficult to categorise are emissions as a result of transboundary transport processes, for example, shipping and aviation. These processes may well cross international boundaries, let alone city boundaries. The further scope 3 inclusion of all imported and exported products and services is a possibility. However, it is arguable as to how much influence municipal government can exert over consumer purchasing decisions. Thus inclusion of emissions from imported and exported products and services may not represent an effective use of resources in the context of local policy decisions.

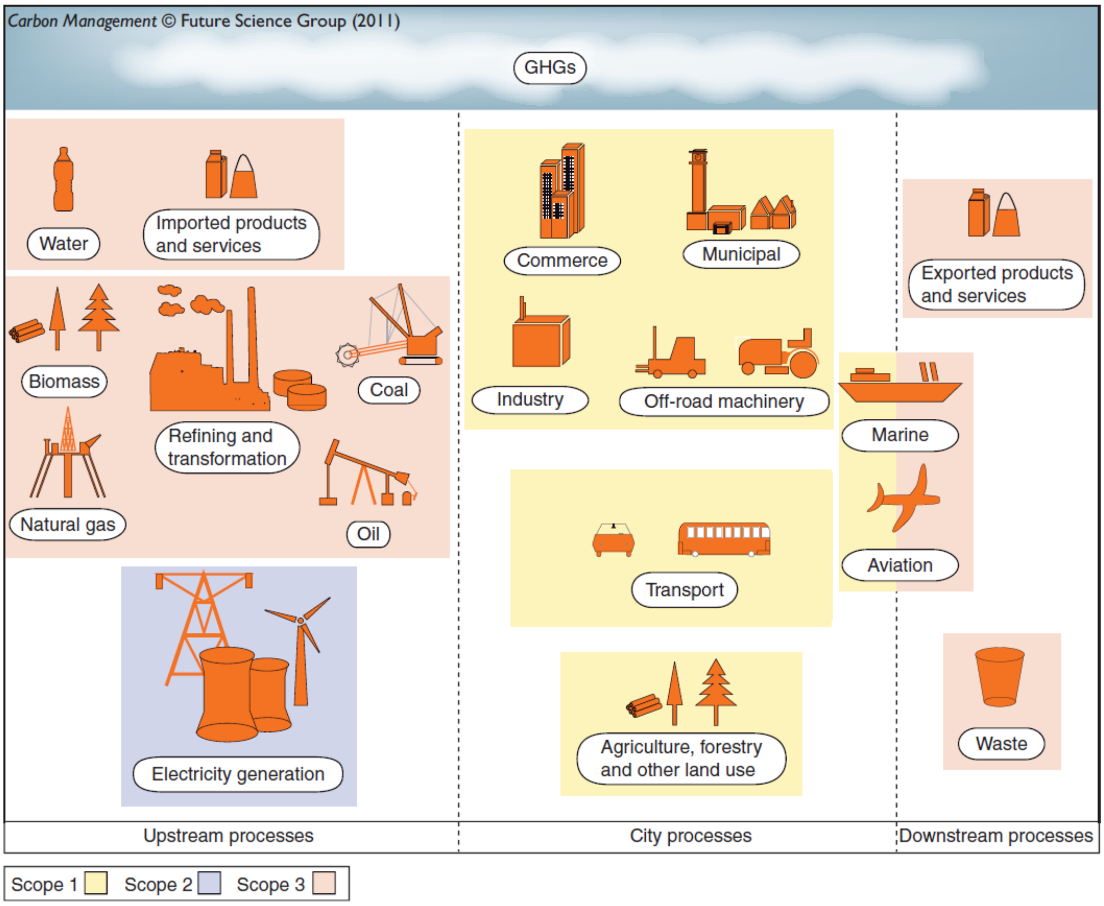

Figure 4. Emissions sources in cities in relation to scope (a way of differentiating emissions sources employed by the World Resource Institute) and life cycle perspective, from the perspective of the wider process boundary (Wright *et al*, 2011a)

Assigning Emissions Responsibility

Most cities in North America and Europe were formed and grew as a result of industrial activities, and still require these activities for employment and revenue generation. The global pattern of industrial activity has shifted; developed countries are increasingly reliant on developing countries for resources, in part due to large transnational organisations seeking lower costs and increased profitability. Legislative changes (although these rarely consider GHG emissions specifically) have also changed the geographic distribution of industry (Dodman, 2009). In turn this has caused a global shift in GHG emissions distributions. For emissions reduction measures to be effective, allocation of responsibility must be consistent to allow change over time to be monitored and avoid double counting (Lenzen *et al*, 2007). Broadly there are two methodologically different accounting principles: production (at source) versus consumption (end user) (Munksgaard & Pedersen, 2001). The distinction is made on the difference between 'emission' and 'responsibility'. Production accounting assigns responsibility to the producer of emissions, in this way emissions are located to the actual site of the emitting process (e.g. to the site of a factory or industrial process). Conversely, consumption accounting locates emissions to the final consumption of goods and services. Emissions associated with the manufacture and transport of all goods and services are appropriated to the point of consumption.

Production accounting is the most commonly used procedure and it is the procedure advised by the IPCC for national emissions accounting (IPCC, 2006). Spatial boundaries can be easily defined in accordance with geographic and political boundaries. Given this perspective, national accounts consider emissions from exported products and services, but exclude those from the life-cycle of imports (Peters, 2008). Upstream emissions are attributed to the producer, not the consumer.

Production accounting may lead to large underestimates of carbon emissions from products and services as a large proportion of carbon emissions are from supply chain processes (Matthews *et al*, 2008). It often conceals issues of 'burden shifting', energy intensive process are exported to less developed countries, imports of products then increase in consumer countries (Druckman *et al*, 2008). The consumer state is thus 'rewarded' with lower absolute emissions, yet a higher consumer driven lifestyle. This issue can also be realised at the city scale, whereby high production regions are penalised with higher absolute emissions over those with high consumerism. This has led to concerns over 'burden shifting' and 'carbon leakage', the association of trade relations in national policy shifting carbon emissions (Munksgaard *et al*, 2009).

The allocation of carbon footprints in this way is also at odds with methods of national economic assessment, such as Gross Domestic Product (GDP). GDP is based on the value added for all institutional units within an economy regardless of the geographical location of activities (Hertwich & Peters, 2009). This means that national GHG inventories cannot be directly compared to economic measures. Consumption is the main driver of environmental degradation (Lenzen *et al*, 2007), so allocating full responsibility for emissions to producers, and giving no responsibility to consumers seems inequitable and unrealistic. This is especially true when both producer and consumer countries have legally binding emissions reduction targets (Hertwich & Peters, 2009).

Consumption-based accounting defines spatial scope on the target economy; emissions from the manufacture, transport and consumption of imports are counted. Due to the macroeconomic spatial scale, uncertainties and complexity are generally higher than production accounting, especially at national and international scales. In this model the end user of a product is responsible for the emissions associated with the life-cycle of the product. End user accounting does not penalise areas with high emissions for exported services, making it more equitable and reducing the risk of 'carbon leakages'. Consumption-based allocation of responsibility also gives a more realistic indication of a nation's total environmental impact than production-based. It also removes some of the barriers that have previously prevented developing nations from becoming involved in international agreement for the reduction of GHG emissions, as they won't be held responsible for GHG emissions embodied in their exports to developed nations (Wiedmann, 2009). The main disadvantage of this approach is that nations are made responsible for GHG emissions generated by processes they do not have any direct control over (Munksgaard & Pedersen, 2001). Conversely, institutions that have direct control over processes that cause GHG emissions are not given a direct incentive to take measures to reduce emissions (Lenzen *et al*, 2007). Peters (2008) discusses these issues in detail, and provides a useful summary, see Table 2.

Taking a consumption-based approach may more accurately represent the GHG emissions caused by the activities of the city. However, there may be equity issues as highly industrialised cities would benefit from business revenues and taxes, whilst not being held responsible for the carbon emissions of their industries. Largely residential cities would be held responsible for large carbon footprints due to their higher density of consumers. Additionally, they have a limited ability to influence consumer decisions. They have a greater ability to influence areas such as industrial process and infrastructure, through local legislative controls and policy. Given these considerations, a form of shared responsibility accounting whereby emissions from some process sources, such as energy and waste are considered on a consumption approach, and other 'direct' emission sources, such as industrial processes and transport and considered using a production approach, would seem appropriate. This has the methodological benefits and equity advantages offered by both perspectives (Table 2) (Peters, 2008).

Emissions estimation

Emissions factors, an emissions rate per unit of fuel or energy consumption, per activity, are the most easily applied and indeed widely used method of conversion. The IPCC guidelines emphasise the requirement to follow standardised, scientifically accepted principles, procedures and processes, collectively referred to as 'good

Criteria	Production based accounting	Consumption based accounting	Shared responsibility accounting
Boundary	Administered territory	Global	Global
Allocation	Domestic production	Domestic consumption	Domestic activity
Allocation of trade	Includes exports, not imports	Includes imports, not exports	Compares imports and exports
Mitigation focus	Domestic activities	Domestic consumption activity	Domestic production and consumption
Comparability	Consistent with GDP	Consistent with international trade	Consistent with international trade
Complexity	Lower	Higher	Higher
Uncertainty	Lower	Higher	Higher
Transparency	Higher	Lower	Higher
Resolution	Higher	Lower	Higher

Table 2. Comparison of production, consumption, and shared responsibility GHG accounting (adapted from Peters, 2008)

practice' (IPCC, 2006). Additionally, the guidelines consider the concept of 'tiers' of data complexity, tier one the most basic, utilising average and/or default emissions factors, tier three the most complex, utilising country, fuel and process specific emissions factors. The tier system allows for extrapolation across spatial levels, from national level inventories to the local level.

The IPCC describes categories, or 'tiers' of methods used for estimating GHG emissions, in the guidelines for the creation of national GHG inventories (IPCC, 2006). Tier 1 methods are simple and quick to apply, but are the least accurate and specific, relying on national and international statistics and default emissions factors. An emissions inventory can be relatively rapidly compiled through the application of national data and assumptions based on regulatory requirements and default values. GHG emissions are calculated by multiplying standard emissions factors by activity data. The method can be applied in the case of emissions removal, storage and capture, whereas one refers to an emissions factor, this could be termed a 'removal coefficient' (IPCC, 2006).

Tier 2 methods are similar to Tier 1 methods but are characterised by the use of more specific data such as country specific emissions factors; engineering estimates of technology efficiency; per capita rates of fuel and electricity use augmented to take into account annual temperature variations; estimates of distances travelled within a community based on vehicle counts and road segment lengths; and quantity of fuel used based on price paid and annual average fuel cost. These methods are likely to require more work but yield more accurate results that are more specific to local conditions. Similar to Tier 1 methods, emissions are calculated from a combination of emissions factors and activity data (IPCC, 2006).

Tier 3 methods are the most complex, time consuming and costly methods, but also the most accurate. Utilising material mass balance calculations for fuel combustion; direct emissions monitoring and further mathematical modelling, estimates can be extremely accurate. In a city context, data from local energy supply can be collated and used for the estimation of emissions from energy consumption. For example, annual collection of vehicle odometer readings can be used to calculate emissions from vehicles within a city (Salon *et al*, 2010). In reality, these methods are often not practical at the city scale or indeed necessary to achieve reasonable levels of data accuracy. IPCC (2006) and ICLEI (2009) guidelines suggest that the highest practicable Tier of methods should be used. In practice, methods of calculation should be determined by the intended use of the footprint and the time and resources available for the calculations (Matthews *et al*, 2008).

In the context of a city footprint, this would indicate that all emissions associated with the upstream supply, direct process and downstream outputs of a city should be accounted. The majority of existing and previous emissions inventories factor in emissions from direct fuel use (Scope 1) and indirect emissions associated with the provision of electricity, heat and steam (Scope 2) (Munksgaard & Pedersen, 2001). However, calculation of indirect emissions embodied in goods and services consumed are often neglected (Bin & Dowlatabadi, 2005). Estimates suggest that as much as 52% of UK household emissions are embodied in goods and services consumed (Druckman & Jackson, 2009). Whilst this may appear significant, inclusion of these emissions is highly dependent on the desired final use and purpose of the footprint. In terms of municipal government management of GHG emissions, their ability to influence consumer purchase decisions is limited, and effort perhaps would be better expended managing infrastructure.

There are three principal approaches to emissions inventories: environmental input-output analysis (EIOA); process analysis (PA), hybrid environmental input-output life-cycle assessment (hybrid EIO LCA). EIOA is a top-down approach to LCA that uses national economic and environmental data such as fuel mix and technologies used to estimate emissions caused by the activities of industry sectors within an economy

(Larsen & Hertwich, 2009). Top-down EIOA utilises aggregate data at the meso level to attribute emissions based on economic flow. Combined with environmental accounting data, they can be used to calculate the carbon footprint, setting the economic system as the boundary (Wright *et al*, 2011b). Process analysis (PA) is a bottom-up LCA approach to analyse the emissions associated with specific processes, leading to a greater level of accuracy than EIOA but requires more time and resources (Larsen & Hertwich, 2009). To overcome the shortfalls of EIOA and PA, Hybrid-EIO-LCA is becoming widely accepted within the academic community as the best approach to carbon footprint calculation (Wright *et al*, 2011b; Larsen & Hertwich, 2009; Matthews *et al*, 2008; Suh *et al*, 2004). Hybrid-EIO-LCA is a combination of PA and EIOA approaches, where PA is carried out for specific processes and embedded within an EIOA (Matthews *et al*, 2008). The merits and limitations of each method are assessed below in the context of municipal government carbon footprints.

Applying the EIOA method for city carbon footprinting is problematic given that input-output data are only available at a national scale, and therefore to apply them to a city scale requires pro-rata scaling of national data (Peters, 2010). The accuracy of the local footprint then becomes dependent on how closely local conditions match national averages. EIOA uses national Input-Output Tables as the basis for LCA. Industries with very different environmental impacts are often aggregated into a single sector for reporting (e.g. paper production and publishing) and represented by average emissions intensity for the sector (Suh *et al*, 2004; Wiedmann, 2009). This is relatively unproblematic in national carbon footprinting, although at a city scale, this approach to carbon footprinting would not be sensitive to many of the strategies that might be employed to reduce emissions, such as site specific energy reduction measures. In the UK, where EIOA has been applied to a sub-national scale, it has been augmented with data regarding household expenditure (Family Expenditure Survey annually produced by the Office of National Statistics), and national census data. Census data include classification of small groups of households called Output Areas by their economic status. This can be matched to the spending patterns of families in that economic category to generate a carbon footprint that varies depending on socio-economic factors (Druckman *et al*, 2008). These expenditure survey data are based on an annual survey of 7000 households. Accuracy will depend on how closely households within a given socio-economic category within a city match to the national average. The survey results could form the basis of a more accurate and specific application of EIOA for use in city carbon footprinting, although this would increase the time and resources needed to carry out the EIOA.

PA is a bottom up methodology originally developed for the environmental assessment of products (Wright *et al*, 2011b). The bottom-up approach ensures accuracy and transparency are high, and uncertainties minimised. PA is process specific leading to a greater level of accuracy than EIOA (Suh *et al*, 2004) but requires more time and resources (Larsen & Hertwich, 2009), not always available to municipal governments. The methodology excels when examining micro systems, enabling the study of impacts from cradle-to-grave. However, the bottom-up nature of PA can lead to truncation and cut-off errors when scaling up to subjects such as cities.

Hybrid-EIO-LCA is rapidly becoming widely accepted as the best approach to the carbon footprint calculation (Wright *et al*, 2011b; Larsen & Hertwich, 2009; Matthews *et al*, 2008; Suh *et al*, 2004). Hybrid-EIO-LCA uses PA to calculate emissions for the main industrial processes undertaken within a sector as well as the main industrial processes in the immediate upstream supply chain, and downstream processes to the product end-of-life phase. EIOA is used for all upstream supply chain processes for which PA is not or cannot be performed (Suh *et al*, 2004). This has the benefit of being accurate and specific to the industry sector and would monitor carbon footprint variation due to changes in controlled processes while also including complete upstream emissions. In the context of city footprinting, hybrid-EIO-LCA energy intensive processes, emissions associated with waste processing and disposal and the carbon footprints of major suppliers of goods and services could be assessed with PA. Less significant and higher order upstream processes could be included in the EIOA.

Treatment of 'Carbon sinks'

Biogenic carbon stocks are recognised as having an important role in GHG management (Ostle *et al*, 2009). Land use and land use change are also recognised as having a complex and significant effect on albedo, nutrient cycling, biodiversity, heat fluxes and carbon sequestration (Robinson *et al*, 2009). With over 50% of the world's population now living in urban areas, there is a potential for changes in soil carbon stocks due to urban land use change (Pouyat *et al*, 2002). However, understanding of urban biogeochemical cycles is still limited; making urban carbon sinks hard to quantify (Lorenz, 2009).

The estimation of GHG removals by sinks is required in national GHG inventories. At a national level, agriculture, forestry and land-use change is the only category to include a form of carbon sink (IPCC, 2006). The majority of existing methodological approaches to inventorying cities neglect to include GHG removals or biogenic processes. Green space management practices have been identified as having considerable impact on the rate of carbon sequestration within cities (Tratalos *et al*, 2007). Lawns and gardens in residential areas can also sequester considerable amounts of carbon (Tratalos *et al*, 2007; Qian *et al*, 2003).

To motivate municipal governments to maximise the favourable management of green space vegetation and soil carbon stocks must be included in city carbon footprints. To prevent emissions 'offsetting' and burden shifting, only green space GHG removals within the administrative boundary of the city should be counted. Municipal governments have little or no influence over green space external to their administrative boundaries. As discussed previously to ensure equitability and fairness accounting must consider all sources within the city boundary. In the same sense municipal governments cannot claim the benefit of offsets outside this defined boundary of emissions sources. The approach outlined by the IPCC for national inventories, to estimate the stock change over time, could feasibly be adapted for use at the local level (IPCC, 2006).

Conclusions

As centres of population and of high concentrations of anthropogenic GHGs, cities are key to the global response to climate change. Action to reduce emissions at a local level requires the localisation of climate change and that municipal and local governments have a good understanding of emissions sources and reduction potentials. In pursuit of climate change localisation the concept of the 'carbon footprint' is increasingly being recognised as a valuable tool in emissions management. This paper has examined the methodological challenges that still abound the development of a model for city carbon footprinting; issues including boundary selection, emissions responsibility and methodological choice.

Municipal governments are presented with several options for boundary selection. It is important to realise the need for complete inclusion of all sources and sinks of emissions including those indirect process emissions occurring outside the geographic boundaries. Whilst also considering the limits of municipal government influence and control. Inclusion of transboundary processes, for example energy provision is vital to ensure equitable emissions apportioning. Whilst inclusion of emissions from products and services is outside the scope of municipal government control, and less can be gleaned from inclusion.

Selection of boundary conditions is informed by issues surrounding emissions responsibility. Action for reduction of these emissions is difficult until responsibility and roles of leadership are established. Equitable apportioning of emissions is also vital to ensure responsibility is taken for emissions management, mitigation and reduction, and to prevent burden shifting. This includes those emissions from transboundary processes, such as shipping and aviation, where equitable apportioning is a subject of much debate. This also raises the issue of fairness and equitability. Emissions must be allocated fairly; to achieve this work is required to establish appropriate methods to apportion both the social and economic benefits and the environmental impacts of emissions.

The accuracy of a model depends on the quality of data. The prerequisite of this is the desired end use of the model. Model developers must remain aware of the financial and time constraints faced by municipal governments, as well as the requirement to produce a decision assisting tool. Tier 1 methods are simple and quick to apply, but are the least accurate and specific. Tier 3 methods are the most complex and accurate, but also the most resource intensive. We suggest tier 2 methods offer the most cost-effective balance, indeed the added accuracy offered by tier 3 methods is unlikely to be required in the context of city level policy decisions. This is not however to rebuke the use of tier three methods where data and resources are available to produce improved accuracy.

Models must be developed to be transferable to enable comparisons and foster a sense of competitive emissions reduction. Without a consistent methodology whereby different cities can be footprinted using the same underlying structure, comparison of models is invalid. The majority of current research agrees that the most successful and inclusive method of carbon footprinting at the sub-national level is achieved through the application of Hybrid-LCA, where EIOA is utilised to assess high level processes, supplemented with PA where possible. This results in increased accuracy, completeness and the ability to capture city and sector level emission reduction measures.

Adequate tools are required for municipal governments and policy decision makers to make informed decisions to improve city level climate change management. A number of tools have already emerged, but for the reasons presented above these tools do not constitute a definitive approach. Further development to develop a comparable, consistent, complete, transferable and accurate carbon footprinting tool is greatly needed.

The role of carbon footprinting in the development of global 'low-carbon' cities

Abstract

A substantial proportion of anthropogenic greenhouse gas generating activities are concentrated in global cities. Action to reduce emissions at the urban level requires an understanding of life-cycle emissions sources and reduction potentials. Policy response(s) by municipal authorities needs to consider all emissions scopes across the urban lifecycle. Existing framework guidance often does not include all relevant emissions sources, proposes inconsistent methodologies and fails to consider the significance of limited data. A clear statement of the urban system and its boundary is required to develop benchmarks and facilitate the development of global 'low-carbon' cities. Carbon footprints are being developed to facilitate the management of anthropogenic GHG emissions and offer the opportunity to develop internationally agreed models to inform decision making and localise the issue of climate change. This paper will explain the theory behind their development and outline how theory can be translated into a practical tool.

Keywords

Carbon Footprint, Life-cycle Assessment, Low Carbon Cities, Climate Footprint, Climate Change Management

Introduction

Global cities are powerful drivers of global economic activities, centers of innovation and the hearts of their regions and countries. More than half the world population, > 3.3 billion, live in cities, with more than 4 per cent living in 'megacities' (a population of 10 million or greater), predicted to increase to 5 per cent (400 million people) by 2015, with a total urban population of 5 billion by 2030 (UNFPA, 2007).

Many cities are situated in low-lying coastal areas, providing regional industry access to the global trade market (De Sherbinin, 2007). As a result, a significant proportion of the global population is at increased risk from current and projected hazards resulting from climate change. Urban areas are disproportionately important in terms of global climate change policy-making; without the commitment, understanding and knowledge to develop effective policy at a local level, global climate change management targets are likely to be unattainable (Wright et al, 2011a). Local mitigation of climate change is required for the stabilisation of global climate, addressing this issue can also have benefits for urban biodiversity, pollution, social wellbeing and economic development.

The complex nature of urban climate change mitigation requires the bridging of the gaps in knowledge theory and rhetoric. Carbon footprints have emerged in recent years as an indicator, to measure and manage anthropogenic emissions of GHGs, which is both understandable and relatable. We have defined the carbon footprint as a life cycle climate change metric: "*A measure of the total amount of CO_2 and CH_4 emissions of a defined population, system or activity, considering all relevant sources, sinks and storage within the spatial and temporal boundary of the population, system or activity of interest. Calculated as CO_2 equivalents using the relevant 100-year global warming potential*" (Wright et al, 2011b).

Carbon footprints provide a tool for the calculation of emissions linked to a process, activity or behaviour. The merits of the carbon footprint are grounded in its simplicity, making the metric economically viable, (potentially) internationally comparable and readily understandable.

Policy Response

The urban politics of climate change has been a key factor in challenging global policy development (Bulkeley, 2010). Climate change affects cities in three ways: i) a significant proportion of GHG emissions occurs in cities; ii) the effects have direct impacts on cities; iii) synergies between climate policy and sustainable development are most obvious at a local level and motivate cites to generate the social and technological innovations that reduce GHG emissions and adapt to the challenges (Alber & Kern, 2008). Additionally, national governments will be unable to reach national greenhouse gas (GHG) reduction targets without the commitment of sub-national and local governments (Bulkeley and Kern, 2006). Recognising these factors, many cities are starting to develop climate change strategies and policies. For these policies to be effective, adequate tools to measure, monitor and manage sources of GHG emissions need to be available (Bastianoni et al, 2004).

Many cities have developed 'low carbon agendas', for example Southampton's (UK) '*Low Carbon City*' (see www.incadesign.co.uk/low-carbon/), many more have become members of networks, for example the Swedish '*Klimatkommunerna*' network

(Klimatkommunerna, 2011); the U.S. Mayors Climate Protection Agreement (The United States Conference of Mayors, 2008); or the C40 Cities (C40 Cities, 2010). Many of these networks are working to develop standardised procedures for the assessment of urban emissions (Alber & Kern, 2008). For example the ICLEI has agreed a GHG emissions measurement protocol, and now provides a multinational tool (the 'Harmonized Emissions Analysis Tool' (HEAT)) (ICLEI, 2006). Problems remain, especially in cases where cities are compared and do not utilise the same methodology (Wright *et al*, 2011a).

Carbon Footprints

Carbon footprints provide a relatively straightforward, low cost, understandable solution that is ideal for uptake in the public, private and governmental sectors. We recognise that in some situations a more comprehensive metric for the assessment of further GHGs may be required; where the need is identified and complete data is available it can be used to provide a 'climate footprint'; this name reflects the inclusion of further Kyoto Basket non-carbon gases, encompassing the full range of gases in the global communities' response to climate change (Wright *et al*, 2011b).

An assessment of a city's carbon or climate footprint requires boundary selection. The following options are available: municipal operations; all emissions directly emitted from the city; all territorial emissions, including process linked emissions (e.g. production, generation), or consumption emissions. The available options offer increased completeness at the cost of greater complexity (Wright *et al*, 2011a). Cities generate emissions across the urban life-cycle: from direct emissions as a result of fuel combustion activity within the geographic boundaries; to life-cycle emissions associated with products and services. The World Resource Institute differentiates emissions sources in terms of 'scopes' (1-3) that relate to the spatial boundary of the activity or process (Greenhalgh, 2003). Scope 1 encompasses direct emissions. Scope 2 is defined as emissions associated with electricity consumption; we expand this definition to include all regionally or nationally produced energy consumption (e.g. regional community heating, regional steam production, etc.). Scope 3 encompasses all 'physical' out of boundary, process linked emissions. Emissions sources can be further differentiated by life-cycle perspective, single process or production chain emissions. Production chain emissions occur upstream (e.g. service provision, energy production) and downstream (e.g. waste) (Kennedy, 2010).

The issue of boundary selection is largely addressed through the consideration of emissions responsibility. Broadly, there are two perspectives, production (at source) and consumption (end-user) accounting. Production accounting assigns emissions responsibility to the producer of emissions, whereas consumption accounting locates emissions associated with production to the final consumption of goods and services. Production accounting methodologies are currently widely used but may lead to 'burden-shifting'; energy intensive process are exported to less-developed countries and imports of products then increase in consumer countries (Larsen & Hertwich, 2009). Also production accounting creates an artificial truncation of emissions from transboundary processes (e.g. aviation, shipping, etc.) at the city boundary (Chavez & Ramaswami, 2011). Thus consumption accounting is a preferred method of emissions accounting; however, it fails to consider the role and ability of a local authority (LA) and the nature of community level emissions. Hybridisation of production and consumption accounting, recognising that cities are not the same as nations, offers a suitable methodology for transboundary emissions selection at the city level. These types of boundary selection methodologies, building on the scope concept at the community level have started to develop (e.g. Chavez & Ramaswami, 2011).

To achieve effective management, carbon footprints must be constructed in a way that reflects the role of LA in climate change management. Breakdown of emissions to sectors, subsectors and fuel types is required to prioritise mitigation. Blackhurst *et al* (2011) suggest that only a limited number of community level footprints are currently reported by scope. The same review identifies that a number of reports are disaggregated by sector, however this breakdown is often too broad and limits the ability to conduct informed decision making.

In the context of LA roles, it is helpful to consider emissions across multiple dimensions: magnitude – the proportional contribution of the emission; measurability – the ability to obtain accurate data; governance – the level of political control; and feasibility of mitigation. The latter two, whilst closely, linked can be distinguished by way of policy analysis tools (e.g. cost-benefit analysis, political feasibility, etc.) (Blackhurst *et al*, 2011).

Alber & Kern (2008) identify four modes of governance applicable to LA climate change management: 's*elf-governance'* – LA management of corporate activity; *'enabling'* – coordinating and facilitating partnerships with the private sector and encouraging community participation; '*provision'* – action through the provision of public services; and '*authority'* – action through regulation or sanction. In reality, these modes often overlap with any individual measure often based on a combination of these modes. This identification defines and limits the action that can be undertaken by a LA across the various scopes and life-cycle emissions. As a consequence, any model must reflect the ability of the user authority to implement the results. Given the limits of LA governance and feasibility of mitigation there are four key areas for climate change policy action by LAs: energy; transport; waste and waste water; and urban planning. The prioritisation of action and mitigation measures depends on the overall proportional contribution of the emissions source. To achieve maximum return on investment LAs must work to develop as comprehensive inventory as possible. To achieve this comprehensive system of management, understanding of emissions in sectors where LAs have limited governance is required, including: industrial processes; land use and land use change; 'other' uncontrolled accidental or fugitive emissions; and, where possible, high order process chain emissions (e.g. products and services). In respect of these further consumption emissions, measurability is limited due to the significant data requirements (Huang *et al*, 2009). A significant number of data streams must be tracked to effectively footprint a single product, multiplication to the numerous products and services consumed at the city level quickly illustrates high uncertainty and restrictive data requirements (Blackhurst *et al*, 2011). Additionally a comprehensive, consistent methodology accounting scope 1/2 emissions will capture all potential scope 3 emissions at source (i.e. scope 3 emissions are scope 1 emissions in a parallel system). We suggest these emissions, where measured, be reported separately for management purposes (e.g. education and awareness campaigns). There are three principal approaches to emissions inventories: environmental input-output analysis (EIOA); process analysis (PA), hybrid environmental input-output life-cycle assessment (hybrid-EIO-LCA).PA based LCA assesses emissions from the bottom-up based on the energy and mass flows in the process flow. EIOA is a top-down assessment that uses national economic and environmental data to estimate emissions caused by the activities of industry sectors within an economy (Larsen & Hertwich, 2009). Top-down EIOA utilises aggregate data at the meso level to attribute emissions based on monetary transactions.

The development of an urban carbon footprint methodology requires the combination of bottom-up PA and top-down EIOA in addition to estimate fuel consumed and emissions (see Table 3). The principal theory governing emissions of CO_2 and CH_4 - through the bottom-up PA models is mass balance. For example, in combustion activities emissions of CO_2 and CH_4 are a direct result of the carbon content of fuel and rate of oxidation during the combustion process; in waste systems the emissions are determined by mass of carbon through the various treatment processes. In some cases it is not possible to construct mass balance models, therefore pre-determined 'emissions factors', consumption or activity data combined with coefficients that quantify emissions or removals per unit of activity are used. The higher order

	Scope	Description	Methods
Energy	Scope 1	Direct emissions from fuel combustion within boundary.	Mass balance based process analysis (PA) of fuel consumption and combustion.
	Scope 2	Emissions associated with electricity consumption and district heating/cooling.	PA of fuel consumption in all connected regional/national generation facilities. Environmental input-output analysis (EIOA) to determine regional/national distribution and local emissions apportioning.
	Scope 3	Life-cycle emissions from fuel production, processing and transport.	PA studies of individual fuel sources. EIOA of fuel supply system to determine emissions apportioning per unit fuel.
(Transport)	Scope 1	Emissions from in-boundary fuel consumption in transport.	PA of fuel consumption and combustion.
	Scope 3	Life-cycle emissions from fuel production, processing and transport.	PA studies of individual fuel sources. EIOA of fuel supply system to determine emissions apportioning per unit fuel.
	Scope 1/3	Shipping and aviation.	PA of fuel consumption process at unit level. EIOA at national/international level with transboundary type emissions apportioning (e.g. Chavez & Ramaswami, 2011).
Industrial Processes	Scope 1	Emissions from industrial processes within city boundary.	PA of industrial processes.
Waste & waste water	Scope 3	Emissions from the collection, transport, treatment and disposal of wastes. Emissions can occur within or outside city boundary.	PA of waste treatment processes. Mass balance of waste production and routing applied to determine emissions.
Land use, agriculture & land use change	Scope 1	Emissions, sequestration and storage from/to/in land and land based processes.	Predetermined emissions/removal coefficients. Soil/plant interaction modeling (e.g Coleman & Jenkinson, 1999).
Products & services (consumption)	Scope 3	Life-cycle emissions from consumption of products and services within city boundary (reported separately, where data is available).	EIOA Reported separately from emissions totals to enable inter-authority comparisons.
Other	Scope 1	Includes all other emissions sources (e.g. accidental fires, small scale waste burning).	Combination of methods, regional & national statistics and pre-determined emissions factors.

Table 3. Description of source/sink categories, scope, and associated emissions calculation methodologies

upstream processes are then calculated using a tiered input-output analysis allocated using a suitable transboundary accounting methodology (e.g. Ramaswami *et al*, 2011).

Assessing the carbon footprint or the underlying drivers is purely an academic exercise unless this information is used to develop effective, locally appropriate models to inform strategy and policy. This theory is being used to inform the development of a model for use by a local authority, currently focusing on the case study city of Southampton, UK, but intended for roll-out to others in due course (Wright *et al*, 2011a). The methodology presented (potentially) offers a globally comparable urban carbon footprint methodology.

Conclusion

The lack of a standard method for city level GHG inventories has produced numerous inconstant and incomparable methodologies across the world. Carbon footprints offer a comparable, readily understandable and relatively easily implemented metric for the management of urban GHG emissions. Urban carbon footprints utilise the methods established in traditional LCA. The majority of current research agrees that the most successful and inclusive method of carbon footprinting at the sub-national level is achieved through the application of hybrid-LCA, where EIOA is utilised to assess high-level processes, supplemented with PA where possible. This results in increased accuracy, completeness and the ability to capture city and sector level emissions reduction measures. Selection of techniques for the calculation of emissions is dependent on the ability of LAs to govern climate change. LAs have significant influence in transport; waste; energy and urban planning, thus emissions models must strive to present these emissions as a priority. Understanding of further process emissions, for example embedded in goods and services, is essential to provide a complete and holistic picture of urban emissions profiles.

The application of these techniques is enabling the development of models that that are transferable between cities, such as those in Southampton. Without an internationally agreed methodology, comparison of models will remain invalid.

4. A sub-national community GHG assessment: A case study of Southampton, UK

Introduction

This chapter investigates and develops the methods required for the development of sub-national community carbon and climate footprints, to meet Aim/Objective(s) 2b and 2d of this research project. This work was developed by the author and written for this PhD thesis under normal PhD supervision conditions.

Carbon (CO_2 and CH_4) and climate footprints (Kyoto basket GHGs) provide a powerful indicator for the quantification of anthropogenic GHG emissions for relativity little investment (Wright *et al*, 2011b; Williams *et al*, 2012). CO_2 and CH_4 constitute circa three quarters of all anthropogenic GHG emissions, with the remainder comprised in majority of N_2O emissions from agriculture and transport, and various trace elements of the other Kyoto Basket GHGs (Wright et al, 2011b). The carbon footprint metric provides a baseline level indicator that can be applied in virtually any circumstance, even with limited data. In some cases the application of a climate footprint (Williams *et al*, 2012) will be desirable. For instance in cases where N_2O is particularly prevalent or simply in cases where data is known to exist – as Wright *et al* (2011) conclude, lack of data or perceived difficultly should not be construed as a barrier to rigorous and complete investigation.

The framework provided in the previous chapters of this thesis provides a mechanism for the application of life-cycle accounting measures, including carbon- or climate-footprints, to sub-national communities (e.g. cities, towns). The development of the framework identified the need for methods that appreciate the nature of cities as both producers and consumers of products and services. Previous methods and frameworks for the inventorying of city GHG emissions tend to territorial based (i.e. production based methods) developed from the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), or the Greenhouse Gas Protocol (Ranganathan *et al*, 2004) (corporate GHG reporting)). Derivatives of these guidelines fail to realise the nature of cities as fundamentally different from nations and organisations. Organisations are defined by financial boundaries, nations by geographic boundaries. Cities present a combination of both geo-political (the extent of municipal government control) boundaries and transboundary flows of energy and materials. Cities are responsible for a range of production based emissions from a range of sources (e.g. in-boundary transport; manufacturing; residential fuel consumption), but also for a demand led transboundary range of emissions (e.g. grid-connected electricity; imported products; water supply). As a result territorial based approaches to sub-national footprints fail to capture the full range of emissions associated with sub-national communities.

Transboundary approaches to sub-national footprints add out-of-boundary emissions associated with economic demand to territorial emissions. Whilst there are concerns regarding comparability of boundary conditions between studies, these methods offer a more accurate picture of sub-national community processes and flows (Chavez and Ramaswami, 2013). Alternatively consumption based footprints include all emissions along the supply-chain of goods and services, with boundary conditions defined by final consumption of households and governments. Consumption based approaches are useful in the informing mitigation of emissions associated with consumption activities, but are dependent on data availability, may not be sensitive to local emissions management strategies, and exclude emissions associated with production activities and exports.

Ultimately transboundary and consumption approaches are complementary, focusing on different aspects of community composition (Ramaswami and Chavez, 2012). This consideration is reflected by recent international standards (e.g. BSI, 2013; Arikan *et al*, 2012) to account both the production and consumption aspects of a community in parallel – ideally moving towards an approach that replicates the process(s) of urban metabolism (Baynes & Wiedmann, 2012).

The following chapter develops the data and methods required for a sub-national territorial, transboundary, and consumption based carbon and climate footprint, using a case study of Southampton. The results and implication of each footprinting perspective, and the role of carbon and climate footprints, are then discussed in the context of emerging international standards.

A sub-national community GHG assessment: A case study of Southampton, UK

Abstract

Sub-national governments are increasingly interested in local-level climate change management. Carbon- (CO_2 and CH_4) and climate-footprints - (Kyoto Basket GHGs) provide an opportunity to develop models to localise the issue of climate change and facilitate effective mitigation. Three approaches are available for the footprinting of sub-national communities. Territorial-based approaches focus on production emissions within the geo-political boundaries. These approaches are useful for highlighting local emissions sources, but do not reflect the transboundary nature of sub-national community infrastructures. Transboundary approaches extend territorial footprints through the inclusion of key cross boundary flows of materials and energy. A transboundary approach is more representative of community structures and processes, however there are concerns regarding comparability between studies. The third option, consumption based, considers global GHG emissions that result from final consumption (households, governments, investment). Using a case study of Southampton UK this paper develops the data and methods required for a sub-national territorial, transboundary, and consumption based carbon and climate footprint. The results and implication of each footprinting perspective are discussed in the context of emerging international standards. Finally, the results explore the differences in GHGs captured by the application of a carbon versus a climate footprint.

Introduction

Increasing population pressure, resource limitations, and GHG emissions concentrations have catalysed urban GHG management as an essential component of sub-national community governance (Hillmer-Pegram *et al.*, 2012; Harris *et al* 2012; Sovacool and Brown, 2010; Ramaswami *et al.*, 2008). Many cities have established sub-national and transnational climate networks, initiatives or management plans (Harris, Chow and Symons, 2012; Wright, *et al* 2012; Kern and Bulkeley, 2009). Academic discussions have primarily focused on the appropriateness of the allocation of emissions to the local level (e.g. Ramaswami *et al*, 2008, Kennedy *et al*, 2009, Ramaswami *et al,* 2011, Wright *et al,* 2011a, Kennedy *et al*, 2011, Baynes & Wiedmann, 2012, BSI, 2013, Chavez and Ramaswami, 2013). The majority of progress in this field of research has been driven by improved understanding of “urban metabolism” – the relationship between material and energy flows through the city system and urban form (Baynes and Wiedmann, 2012). Broadly speaking approaches can be categorised as process led bottom-up approaches, top-down economic led analysis, or more recently top-down "natural laboratory" approaches relying on atmospheric measurement and concentration (e.g. remote sensing) (Gurney *et al*, 2012). Commonly emissions are further categorised using the system of scopes – 1 (direct emissions); 2 (indirect emissions associated with electricity); 3 (indirect and supply chain emissions) – introduced by the GHG Protocol (Ranganathan *et al*, 2004).

Traditionally methods have been "territorial-based" (alternatively termed "in-boundary", "geographically-based", or "production-based") bottom-up approaches, generally adaptations of the IPCC Guidelines for National Greenhouse Gas Inventories, or the Greenhouse Gas Protocol developed for corporate GHG reporting, accounting for emissions within geopolitical boundaries (IPCC, 2006; Ranganathan *et al*, 2004). These methods can successfully identify local emissions patterns and inform local development policy. However, more recently there has been growing recognition of need to account some share of the emissions outside geopolitical boundaries, as urban economies invariably demand resources beyond their geographic locations (Minx *et al*, 2013). Emissions occur across the urban lifecycle, from direct in-boundary emissions, to emissions embodied in consumption (Kennedy *et al*, 2011; Chavez and Ramaswami, 2011; Wright *et al*, 2011a). Holistic management of urban GHG inventories and the prevention of burden shifting necessitates the inclusion of both direct, and indirect emissions (Wright, 2012; Harris *et al*, 2012). In response two main approaches have emerged in the literature – "transboundary" and "consumption" based approaches.

"Transboundary" (alternatively termed "territorial-plus", "geography plus" or "metabolism based") approaches add out-of-boundary emissions associated with economic demand to territorial emissions, with the exact boundary conditions and scope varying between studies (Ramaswami *et al,*2008; Wright *et al*, 2011a; Baynes and Wiedmann, 2012; Chavez and Ramaswami, 2013). Conversely top-down consumption based methods include all emissions along the supply-chain of goods and services, with boundary conditions defined by final consumption of households and governments (Wright *et al*, 2011a). The consumption approach is useful in the informing mitigation of emissions associated with household and government consumption. However the exact origin of embodied emissions cannot normally be delineated and emissions from local production for exports are excluded (Minx *et al*, 2013). Consequently, methods are not sensitive to many local strategies to reduce emissions (e.g. site-specific energy reduction measures) (Wright *et al*, 2011a).

Ultimately both concepts are complementary, focusing on different aspects of community composition (Ramaswami and Chavez, 2012). The primary cause of inconsistency between studies (for a review see: Chavez and Ramaswami, 2011; Kennedy *et al*, 2011) and emerging standards (e.g. BSI, 2013, Arikan *et al*, 2012) is the approach taken to boundary conditions, both spatial and temporal; an issue which is particularly apparent in responsibility-based emissions accounting – i.e. assigning emissions to a particular geopolitical entity to develop policy response and action (Wright *et al,* 2012). Temporal boundaries vary, but typically consider an annual period (e.g. Kennedy *et al*, 2009), however models that operate at finer temporal scales are developing (e.g. Gurney *et al,* 2012). The geographic extent of cities as considered by researchers varies, reflecting both the needs of the organisations and applications involved, and the lack of a singular definition for a city or an urban area (Liu *et al*, 2014). A city may be defined a settlement reaching an arbitrary geographic extent, a population size, or for regional economic or political administrative significance (Krupat & Guild, 1980). The OECD and the European Commission propose for a definition for a city based on the presence of an 'urban centre' defined based on high-density population grid cells (1500 inhabitants per km^2). The definition is further defined to include a 'greater city' where urban centres extend beyond the 'city' for example the City of London (2.89km^2) is part of a collection of municipal boroughs that comprise the area of Greater London (1,623.37km^2) (Pointer, 2005); and a commuting zone based on residents employed within the city area (Dijkstra & Poelman, 2012). However, ultimately a 'city' or and 'urban area' is simply a taxonomic division of a 'community' ("particular area or place considered together with its inhabitants", (Collins English Dictionary, 2014). Geographic boundary extent can be decided on a case-by-case basis, defined by a combination of motivation (e.g. municipal governance) and the ability to develop effective management (Wright *et al*, 2012).

This paper describes the result of a partnership between the University of Southampton's Carbon Management Group and Southampton City Council, England, to develop a carbon, and climate footprint (Wright *et al*, 2011b, Williams *et al*, 2012) of Southampton, UK. Through the use of a case study, Southampton, UK, boundary conditions and methodologies with equations (where appropriate) to obtain activity data and calculate emissions for territorial, transboundary, and consumption based footprints are presented. Methods are developed for stationary sources (e.g. residential energy), and mobile sources (e.g. road transport) utilising a bottom-up process analysis method. Consumption emissions are calculated using a two-region (UK and rest-of-world) environmentally extended input-output model. Unless otherwise stated, all emissions are calculated as per Williams *et al* (2012); from first principles (e.g. carbon content and oxidation during combustion) or from recognised emissions factors. Results are presented for the carbon footprint – emissions of CO_2 and CH_4 across the community life cycle – and the 'climate footprint' –six Kyoto Basket GHGs

(CO_2, CH_4, N_2O, SF_6, HFCs, PFCs) (Wright *et al*, 2011b and Williams *et al*, 2012, respectively). We assess the difference in GHG inclusion between two metrics – carbon and climate footprints. We discuss the results and policy implications of applying a territorial, transboundary, and consumption based footprint and conclude with a consideration of the effectiveness of current practice and ongoing issues.

Case study – City of Southampton

Southampton (pop. 239,428 during study period) is the largest city in Hampshire, England (area: 51.91km^2) based on the geographic extent of the city geo-political boundary (ONS, 2010a). The case study was chosen as Southampton contains the representative components of many cities. The city forms the majority of the Greater Southampton region, and a significant part of the South Hampshire Metropolitan Area. The city is governed by Southampton City Council, a unitary authority(a single tier local government responsible for local government functions); the wider region is within the remit of Hampshire County and multiple district councils (a hierarchical system of governance common to many countries).

Southampton is a commerce hub; a major international cruise terminal, and the UK's second largest container port. A significant proportion of Southampton's workforce (circa 42%), commute from the wider region and surrounding counties (Department for Transport, 2008). The city has two universities with a transitory student population of in excess of 40,000 (Zhang *et al*, 2011). Southampton Airport is a regional domestic and international airport located outside the north of the city's geopolitical boundary, in neighbouring Eastleigh. The airport is included and reported separately for completeness and methodological demonstration.

Residential energy

Energy consumption in domestic properties is a major source of emissions and is highly dependent on both the building structure and the behaviour of residents. Management of residential property is difficult, as domestic buildings tend to have long lives, many are privately owned, and there is wide variation in stock age, type and build form (Firth & Lomas, 2009). The calculation of emissions from residential dwellings necessitates knowledge of domestic energy consumption. This requires an understanding of current housing stock and usage patterns and subsequent changes in the stock. Scale poses an issue at the community level, large communities contain a significant number of dwellings (e.g. Southampton, a small city > 91,000 dwellings) (Office for National Statistics, 2001). There are two possible solutions, model simplification or model generalisation. The second option of generalisation is preferable; model simplification would require extended investigation and expert assumptions, which are not within the scope of this project. Various methods have been developed for this purpose (for detailed review see Natarajan *et al,* 2011; Kavgic *et al*, 2010). Generalisation can be achieved through the assumption that the distribution of energy use in dwellings can be simplified with the application of 'average' performance categories. Energy consumption parameters are derived for categories of dwelling, and applied to individual property build forms with Geographic Information Systems (GIS) (Natarajan *et al,* 2011; Kavgic *et al*, 2010). It should be noted that output is restricted to an aggregation of properties rather than the individual building level. At this level accuracy would be open to significant variation. Additionally output at this level introduces a number of confidentiality issues. The main aim of the model is to provide output to assist policy decision and resource allocation at the city level, rather than provide estimates of energy use for *individual* properties.

A fundamental component of many of these models is the Building Research Establishment Domestic Energy Model (BREDEM) (BREDEM-8 – monthly or BREDEM-12 – annual) (Anderson *et al*, 2002). The model has an established track record of accurately predicting the energy consumption of dwellings (Kavgic *et al*, 2010). The methodology uses building physics-based algorithms coupled with empirical data to derive energy consumption (disaggregated across: space heating; hot water; cooking; lights and appliances). The 'full' version of the model requires physical data relating to a dwelling, all of which is measurable, but not easily obtainable, owing to the high cost of actual site surveys. As consequence GIS tools in conjunction with inference databases have been used to facilitate building specific data, without the need for visual inspection (Rylatt *et al*, 2003). The modular nature of the model enables exportability outside the UK – British assumptions can be substituted for local values (Ward, 2012). Additionally elements can be replaced with more accurate methods or data (e.g. to update lights and appliance use with Domestic Equipment and Carbon Dioxide Emissions (DECADE) project data) (Natarajan *et al*, 2011; Environmental Change Institute, 1997).

In the Southampton case study physical spatial characteristics (build form, location) (Ordnance Survey MasterMap Topography Layer (Ordnance Survey, 2014a)) and Light Detection and Ranging (LiDAR) building height data (Cities Revealed database, accessed using the MIMAS data repository under the EDINA access licence (The GeoInformation Group, 2007)), were merged with local address gazetteer datasets (Southampton City Council, 2010) to create a postal address point database of all dwellings. Individual dwelling physical properties (e.g. stories (assumed 2.15m ceiling height plus 0.25m void); total floor area; wall area; glazed area) were calculated using GIS algorithms (DECC, 2010; International Code Council, 2012). Where floor space of an individual dwelling is unknown (e.g. single building polygon representing a block of apartments), it was assumed floor space was equally distributed between the address points contained in the relevant building. Individual dwellings were assigned a performance category based on age (estimated from cities revealed dataset (The GeoInformation Group, 2007)) and regulatory standard; physical properties, and behavioural assumptions (368 physical, 9 behavioural categories). Assumptions other than those associated with GIS derivation were developed using the English Housing Condition Survey (EHCS) – annual survey (n ≈ 13,300) of English housing circumstances, condition, and energy efficiency (Department for Communities and Local Government (DCLG), 2011), and the UK government Standard Assessment Procedure (SAP2009) for the assessment of the energy performance of dwellings (DEFRA, 2009). The SAP2009 documentation includes a reduced data option (RdSAP), with a number of assumptions and standards to be used in the case of limited data availability (DEFRA, 2009) (see table 4 for full variable list).

Total energy demand was assumed to be met using a combination of electricity, natural gas, and a range of 'residual' fuels (e.g. coal, wood, anthracite, steam). Consumption data for electricity (assumed primarily to meet demand for lights and appliances) and natural gas (assumed to meet primary demand for heating, cooking, and hot water) were available from local metering records (DECC, 2013a; DECC, 2013b). Remaining energy demand, assumed to be met using residual fuels (e.g. coal, anthracite) was calculated equation 1. Consumption of fuel types was allocated using the ratio fuel type to total regional (where available) or national fuel sales (equation 2).

$$ED_{dom} = \sum C_g + \sum C_e + \sum C_{r_i}$$

Equation 1. Domestic fuel demand, where: ED_{dom} = Total domestic energy demand, C_g = Consumption of natural gas (kWh), C_e = Consumption of electricity (kWh), C_{r_i}= Consumption of residual fuel type *i* (kWh)

$$C_r = \sum C_r \left(\frac{C_{nr}}{\sum_r^i C_{nr}}\right)$$

Equation 2 Specific residual fuels consumption, where: C_r = Consumption of fuel type *r* (kWh), C_{nr} = Total national consumption of fuel type *r* (kWh)

Table 4 Summary of data required for the estimation of total domestic energy demand (UK) (data categories adapted from Firth & Lomas, 2009)

Data category	Information required	Data sources and derivation
Site information	Degree days	Monthly degree days (Environmental Change Institute, 2014)
	Height above sea level	GIS Derived
	Number of sides sheltered from wind	GIS Derived
	Wind speed	User input (assumed average)
	Level of overshading	RdSAP 2009 (assumed average 0.77) (DEFRA, 2009)
	Age of dwelling	Cities Revealed (GeoInformation Group, 2007)
Building information	Number of storeys	GIS Derived
	Definition of zones 1, 2	RdSAP 2009 (DEFRA, 2009)
	Total floor area	GIS Derived
	Volume of dwelling	GIS Derived
	Area of roof, external walls, for each zone	Assumption from GIS data
	U-values of roof, external walls, for each zone	RdSAP 2009 (DEFRA, 2009)
	Heat loss floor area for each zone, floor perimeter, type and amount of insulation	RdSAP 2009 (DEFRA, 2009)
	Window area, type, level of leakiness, orientation, zone	GIS/ RdSAP 2009 (DEFRA, 2009)
	Pressure test result	Assumed none
	Number and type of fans and vents	RdSAP 2009 (DEFRA, 2009)
	Type	EHCS (DCLG, 2011)
	Fuel	EHCS (DCLG, 2011)
	Controls fitted (e.g. room thermostat)	RdSAP 2009 (DEFRA, 2009)
	Level of independent control on zone 2	EHCS (DCLG, 2011)
	Secondary heating, type and fuel	EHCS (DCLG, 2011)
	Number of pumps and fans	RdSAP 2009 (DEFRA, 2009)
	Type of hot water heater	EHCS (DCLG, 2011)
	Volume of hot water tank	RdSAP 2009 (DEFRA, 2009)
	Thickness and type of tank insulation	RdSAP 2009 (DEFRA, 2009)
	Whether primary pipework is insulated	RdSAP 2009 (DEFRA, 2009)
	Whether there is a cylinder thermostat	RdSAP 2009 (DEFRA, 2009)
	Location of tank (zone 1 or zone 2)	Assumed 2
	Area of solar panel (if fitted)	Assumed none
	Efficiency of heat recovery (if present)	Assumed none
	Cooking system and fuel used	EHCS (DCLG, 2011)
	Location of kitchen (zone 1 or zone 2)	Assumed 2
	Proportion of low energy bulbs	Assumed 0.75
	Number of occupants	Derived GIS/BREDEM-8 (Anderson *et al*, 2002)
	Demand temperature for each zone	BREDEM-8 default (Anderson *et al*, 2002)
	Heating periods (for each zone if different)	BREDEM-8 default (Anderson *et al*, 2002)
	Usage of hot water, lights, appliances and cooking	BREDEM-8 average level (Anderson *et al*, 2002)
Fuel data	Total natural gas consumed	DECC (DECC, 2013b)
	Total electricity consumed	DECC (DECC, 2013a)
	Total residual fuel consumed	Derived

Commercial & Industrial energy, processes & product use

Emissions from in-boundary commercial and industrial activity occur as a result of energy consumption and emissions occurring in physical or chemical processing for non-energy uses (including: production/consumption of mineral products; metal production; chemical production; and petroleum product end-uses). Complexities exist in the allocation of emissions between the energy and processing sectors (e.g. residual heat may be used in part for the production of electricity).

Actual consumption data from sales records or feedstock records is difficult to obtain, primarily due to the commercially sensitive nature of such data. Point source data from larger facilities may be available from legislative emissions reporting schemes; however this often does not encompass small schemes (Gurney *et al*, 2012). Emissions from commercial and industrial energy consumption can be calculated using local fuel sales data. This method assumes fuel combustion at place of purchase, and may not accurately reflect the source of emissions (e.g. where purchase records for a production site are held at a geographically separate financial office).

Alternatively Gurney *et al* (2012) describe a model to simulate energy demand based on building parameters combined with known local atmospheric emissions. The same study notes that this method is only suitable for large point source emitters. Therefore they describe a proxy estimation, whereby total national emissions are divided by total national floor area by industry, allocated to the local level. Similarly employment by industrial sector can be used to pro-rata known national emissions (Tsagataskis *et al*, 2008). Both methods rely heavily on the correct industrial classification of facilities in the local community, where facilities are incorrectly classified this will produce erroneous results.

Commercial and industrial energy and process related emissions for Southampton were estimated using 'employment-intensity' factors. The ratio of total national employment by sector (to a 4 digit Standard Industry Classification (SIC2007)) ($\mathrm{Emp}_{\mathrm{nat}_j}$) (ONS, 2009), to fuel use by type (C_{nat_i}) (DECC, 2014), was calculated. This was updated with meter point (grid connected natural gas and electricity aggregated to Medium Layer Super Output Area (MLSOA) (DECC, 2013c; DECC, 2013d)). The factors are combined with local employment by sector ($\mathrm{Emp}_{\mathrm{lcl}_j}$), obtained from the Inter-Departmental Business Register (a secure access organisational-level database which uses VAT and PAYE, to develop records of unit-level data including local offices for individual organisations), (Emp_{lcl_j}) (ONS, 2009) to estimate local fuel consumed (C_{lcl_i}), equation 3:

$$C_{lcl_i} = \left(\frac{C_{nat_i}}{Emp_{nat_j}}\right) Emp_{lcl_j}$$

Equation 3. Calculation of commercial & industrial (inc. municipal) energy related emissions.

Electricity, steam and heat

Electricity supplies commonly rely on centralized thermal generation systems, a national high-voltage transmission network, and regional low-voltage distribution networks (Allen, Hammond and Mcmanus, 2008). Commonly emissions related to electricity are calculated using an aggregated emissions factor for a national system of generators and transmission network (Kennedy *et al,* 2010). Emissions from combined heat and power plants (CHP), which produce electricity and locally usable heat are often reported separately due to data collection conventions (e.g. Kennedy *et al*, 2010), and the rationale that composite emissions factors may represent an over- or under-estimate of emission intensity. For example, a community with high CHP utilisation (e.g. Copenhagen) has reduced demand for electrical heating, use of a composite emissions factor (heat and electricity) would penalise the community.

At the municipal level there are three primary considerations: i) management of consumption patterns (i.e. reduce demand to reduce generation); ii) management of emissions from existing in-boundary generation facilities; iii) the creation of favourable conditions for the development of low carbon micro and decentralized generation. Aggregated emissions factors for consumption of nationally generated electricity do not segregate in-boundary generation, or consider low GHG decentralised (generation of energy close to the point of consumption) or micro generation schemes (UK <50kW) - likely to be a component of meeting carbon reduction targets (Allen *et al*, 2008; Energy Act 2004). Unless the generating facility is directly connected to a consumer in the community there is no incentive for community low carbon generation - investment will not show in the aggregate factor.

This problem can be overcome in a number of ways. Firstly, as electricity drawn from national supply grids cannot be traced to a point of generation, one can accept that the electricity generated in-boundary is fed into national generation systems, and therefore the energy consumed is representative of the grid average. In a territorial based assessment emissions associated with in-boundary electricity generation can be reported separately, either as a proportion of total consumed or absolute. Alternatively LCA deals with similar situations in product systems (e.g. a renewable electricity supply contract) by allowing renewable energy credits to be allocated to the system using “green” electricity (Weber *et al*, 2010). This principle can be applied to the community level, emissions can be calculated using an emissions net balance, where in-boundary generation ‘offsets’ grid consumed electricity. Community governments are then presented with information regarding the impact of local renewable generation at both the local and national levels, and are encouraged to deliver favourable conditions for local generation investment. It is suggested therefore that the emissions ‘avoided’ as a result of in-boundary renewables is calculated as the total renewable generation (GWh) multiplied by the equivalent grid connected emissions factor.

Emissions for Southampton’s electricity consumption (E_{elec}) were calculated using a national grid emissions factor (accounting for transmission, transformation and other losses (L) (typically circa 6 – 11%) (Kennedy *et al*, 2010)), estimated from national generation and electricity consumption (C_{nat}) statistics reported in the Digest of UK energy statistics (DUKES) (DECC, 2011). Data for the local consumption (C_{lcl}) of electricity were available for billing, energy balancing and management purposes (DECC, 2013a). Data for the consumption of specific fuels (C_{fuel_i}) used for generation were taken from DUKES (DECC, 2011), combined with fuel-specific emission factors (EF_i) (IPCC, 2014) total emissions were calculated (equation 4). Emissions associated with useful heat/steam energy were separated from emissions associated with electrical energy for both national and local generation using the efficiency method as presented by the Greenhouse Gas Protocol (Ranganathan *et al*, 2004). Emissions were assigned on the basis of steam and electrical energy generation efficiency (DECC, 2011). Note, emissions exclude electricity derived from the combustion of waste, from a data collection perspective it was easier to use a convention of reporting emissions from waste combustion under the waste sector. Emissions from local generation and ‘offset’ by renewable generation were estimated using the same procedure. Point generation source emissions were calculated from facility data presented in DUKES (DECC, 2011), total installed renewable capacity was estimated from sub-regional installation and generation statistics (DECC, 2013e).

$$E_{elec} = \left(\frac{\sum C_{fuel_i} EF_i}{C_{nat}}\right) C_{lcl}$$

Equation 4. Calculation of emissions from consumption of grid connected electricity

Road Transport

Cities could not exist without the supporting systems to transport people and goods. Efficient logistical systems are a perquisite of economic development and a key driver of global GHG emissions. Road transportation in cities is commonly powered by petroleum and diesel with small numbers of LPG, natural gas or electrically powered vehicles (Kennedy et al, 2010). Due to the relatively small spatial scale of cities, commuter road transport emissions are often artificially truncated at the city boundary, but represent a significant transboundary emissions source (Chavez and Ramaswami, 2012).

Estimation of emissions from road transportation has been approached from a number of perspectives. Economic data on fuel sales can be a viable indicator of road transport emissions, where the study area represents a commuter-shed (Kennedy *et al*, 2010). However this method is less effective where significant numbers of commuter trips occur (e.g. Southampton – circa 42% of work related trips are from outside the city (ONS, 2010a)). In these cases the location of fuel purchase is not necessarily representative of fuel consumption. An alternative method is through the use of proxy relationships, with emissions estimated through regression based approaches (e.g. impervious surface area and road density (Brondfield *et al*, 2012) or population density and road density (Shu and Lam, 2011). High temporal and spatial resolutions have been achieved using activity-based approaches, combining vehicle kilometres travelled (VKT) with fleet and fuel data (Gatley *et al,* 2013). This approach requires total distance travelled by all vehicles in the study area, fuel efficiency, and fleet composition. Issues arise in comparability of VKT techniques as the methods for the collection of activity data differ between cities; many cities have their own bespoke model (Kennedy *et al*, 2010). However, models utilised by local authorities are often developed to model traffic flow and intervention. Linking these models to emissions has the advantage of allowing the modelling of spatial and temporal impacts of traffic policy intervention. The restriction on temporal and spatial distribution then becomes a function of the resolution of data available, Gurney *et al* (2012) achieved high temporal distributions using a series of automatic traffic recorders.

The case study uses an activity-based approach. The basic principle of an activity based models is the relationship of the mass of fuel consumed in the distance travelled (note that emissions related to electrical vehicles are captured as electricity emissions). The amount of fuel a vehicle consumes in a given distance is dependent on a number of parameters, including drive cycle, engine temperature, ambient temperature, fuel type, and fuel quality (Boulter and Mccrae, 2007). Drive cycle describes the relationship of vehicle speed against time during a trip, interrogation of this data can describe characteristics of the drive pattern (e.g. how much stopping and starting the vehicle does, how aggressively it is driven, how much load is applied to the engine). Emissions are normally considered in three types: hot-start, cold-start, and evaporative representative of different sections of the drive cycle and losses (Ntziachristos *et al*, 2009). ‘Cold-start’ emissions occur in the early stages of vehicle operation prior to the engine reaching operational temperature. Rates of fuel consumption and emissions are typically higher during cold-start, than during thermally stable (hot-start) operation (typical operation: engine coolant at 70°C – 90°C, exhaust systems at several hundred degrees centigrade) (Boulter and Mccrae, 2007). Evaporative emissions occur as a result of loss from parts of the vehicle other than the exhaust emissions (e.g. fuel tank; fuel injection systems). The approach considered for the case study does not consider evaporative emissions, these are considered to be negligible (Ntziachristos *et al*, 2009).

Hot-start emissions were calculated using experimentally derived emissions factors for vehicle type and pollutant by trip length and velocity from the ARTEMIS (Assessment and Reliability of Transport Emission Models and Inventory Systems) methodology and TRL emission factor database (Boulter and Mccrae, 2007). Which have previously been successfully applied at the urban level for the assessment of an urban toll system in Stockholm, Sweden (André, 2008) and urban CO_2 modelling of Norwich, UK (Nejadkoorki, *et al,* 2008). Emissions factors are a polynomial function of drive cycle and speed (equation 5), categorized by vehicle type. An example of the parameters used in estimating emissions factors is presented in table 5.

$$y = k(a + bx + cx^2 + dx^3 + ex^4 + fx^5 + gx^6)/x$$

Equation 5. CO_2 emission factor (g km^{-1}), where x = average speed (km/h), a, b, c, d, e, f, g, k = vehicle technology coefficients (Boulter and Mccrae, 2007)

Vehicles were categorized by type (e.g. passenger car; light goods vehicle (LGV)), fuel (e.g. petrol; diesel), and installed technologies (e.g. catalytic convertors) – defined by progressive legislative standards (e.g. Directive 98/69/EC; Regulation 715/2007/EC).

Estimates of local fleet composition are comprised of estimates of both national and local vehicle type estimates. Estimated Annual average daily flows (AADFs) produced annually for 51 census points for each junction to junction link on the major road network (DfT, 2013a). The city census points were assumed to represent the most accurate fleet data, however vehicle categories are provided as counts only, with no technology or engine type breakdown. The TRL emissions factors require more

Table 5. Example of TRL database factors for petrol cars, <1400cc, CO_2 emissions (note values rounded to 4SF)

Legislation	a	b	c	d	e	f	g	k	Valid speed (km/h)
Pre-Euro 1	2261	103.1	0.293	0.003	0	0	0	1	5 - 140
Euro 1	2261	87.56	0.293	0.003	0	0	0	1	5 - 140
Euro 2	2261	80.15	0.293	0.003	0	0	0	1	5 - 140
Euro 3	2261	70.18	0.293	0.003	0	0	0	1	5 - 140
Euro 4	2261	59.44	0.293	0.003	0	0	0	1	5 - 140
Euro 5	2261	44.38	0.293	0.003	0	0	0	1	5 - 140
Euro 6	2261	31.58	0.293	0.003	0	0	0	1	5 - 140

detailed modal split, for example proportion of EURO II, <1400cc capacity petrol engine cars. The local vehicle estimates were complimented and adjusted with national data on fleet split (DfT 2014a; DfT 2014b; DfT 2014c; DfT 2014d; DfT 2014e). Under ideal conditions the fleet composition for each road link in the network should be known, however this information did not exist for the case study. Therefore an assumption is made of a homogenous fleet split across the network, this is acknowledged as a potential source of error in the model, which could be improved with more detailed census point counts or other specific local data.

Cold-start emissions are commonly accounted using an excess factor over the hot-start emission rate (Nejadkoorki, *et al,* 2008). Cold emissions occur on average during the first 5.9km of a trip (Joumard, 1999), previous authors suggest it is a reasonable assumption that the majority of journeys would be less than this distance in a city (Nejadkoorki, *et al,* 2008), and thus cold-start factors are applied to all journeys. Cold-start emissions are dependent on ambient air temperature, an average monthly air temperature for the case study was applied (Met Office, 2014).

Road lengths and locations were taken from Ordnance Survey MasterMap Integrated Transport Layer (Ordnance Survey, 2014b), and DfT road length statistics (DfT, 2014f). Roads were categories according to type (e.g. Motorway, Urban A-road), to enable estimates of average vehicle speed. Average speed was taken as the free-flow speed, taken from census data where known, in all other cases the speed was taken as the national average free flow speed for the road type (DfT, 2013; DfT, 2014g; DfT, 2014h). Total distance travelled was estimated from distances between known census points, supplemented with national scaled data of total vehicle kilometres by road type (DfT, 2014i). Distance travelled from regional commuting was estimated using local government regional transport models (MVA Consultancy, 2012).

Rail

Trips by rail transportation typically traverse the geopolitical boundary of a number of communities. A boundary limited (i.e. scope 1 direct emissions from rail in-boundary) methodology does not account the full impact of the transbounsdary nature of these trips. For trips that originate outside the community boundary, only the in-boundary proportion of the trip is accounted, however the community vitality is based on the demand of both the origin and destination (Ramaswami *et al*, 2008). Conversely, pass through trips (i.e. where the passenger does not embark or disembark within the boundary) that are not a result of the study community demand are counted.

When considered as a transboundary emissions source the issue is further confounded by the nature of national and internationally connected rail networks. A commuter trip could begin a significant distance from the community. Furthermore train journeys involve a series of embarkation points between origin and destination. In many communities there are multiple stopping locations within the geopolitical boundary of the community. Furthermore accounting in-boundary and transboundary emissions related to rail commuting does create the potential for double counting between communities. Pass through trips are accounted as a direct emission of the study community, and then accounted again by the destination community on the basis of the commuting trip. In reflection of these difficulties a number of community based GHG inventories do not explicitly define emissions from rail transportation (e.g. Kennedy *et al*, 2010; Ramaswami *et al*, 2008). These issues can be addressed by accounting emission on the basis of proportional commuter distance travelled. Assigning emissions from rail commuter demand as passenger kilometres travelled to total passenger kilometres travelled on the relevant routes offers a mechanism to apportion trips to the local community a demand basis.

In Southampton rail emissions were calculated using a combination of two methods – one to calculate in-boundary emissions, and another to allocate commuter passenger demand. In-boundary emissions were calculated using ARTEMIS technology specific bottom-up algorithms and emissions factors (Boulter and Mccrae, 2007). Where data is available methods can include engine, technology, distance and speed specific factoring. The network in the study area is partially electrified; resultantly there was a combination of electric and diesel locomotive trips. All journeys on non-electrified rail were assumed to be power by diesel. Trips on electrified rail are apportioned to diesel or electric locomotives using operator timetables. Total trips, distance travelled and operation engine time were estimated from train operator time tables (Network Rail, 2010), combined with the Ordnance Survey Integrated Transport Layer (Ordnance

Survey, 2014b). Emissions associated with commuter trips were estimated as a function of rail demand for Southampton, passenger kilometres travelled (ORR, 2014) were estimated as proportional to the total ticketed exits on the national rail network (collected by automated barrier passes) divided by number of ticketed exits at Southampton. It is recognised that this a rather crude estimation of demand and a potential avenue for future research, limitations include unrecorded exists, and an assumption that distances travelled by passengers are nationally equally split.

Other off-road mobile emissions

Mobile off-road emissions sources represent an extremely diverse range of domestic and commercial emissions sources including mobile off-road machinery (e.g. generators; lawnmowers; off-road motorcycles), and machinery which is, in principle, mobile, but remains in-situ for an extended period (e.g. tower cranes). These sources may contribute significantly to overall GHG emissions totals (EEA, 2009). Usage patterns of off-road machinery range from controlled to chaotic. Controlled activities are consistent and follow specific procedures (e.g. grab loader loading dockside vessels); chaotic activities follow no pre-determined procedure and activity patterns are difficult to quantify (e.g. domestic lawn mowers) (Lewis *et al*, 2009).

Fuel sales data may be a viable indicator of emissions, where the operation of off-road machinery are geographically constrained to the location of fuel purchase (e.g. landfill site owned and operated by the site) (Colodner *et al*, 2011). However this method fails where fuel purchase does not represent the location of consumption (e.g. large national or international organisations where financial offices are separate from site operation).

The US EPA (2010) suggest a bottom-up methodology estimated through the application of run-time (t_j(hrs)), engine loading (L_j) and engine power (P_j) factors to total source unit population, equation 6.

$$C_i = n_{unit} P_j L_j t_j$$

Equation 6. Fuel consumed by off-road mobile machinery (Assessment and Standards Division EPA, 2010)

Calculation of unit source population is complex; unlike road transport the majority of off-road machinery units are not registered. Therefore an accurate estimate of total source population is difficult. Proxy estimates of population can be developed on the basis of national unit purchases and expected operational lifetime, however this is subject to significant uncertainty (EPA 2010). National source machinery unit populations can then be proportionally allocated to a community population. This may not give a representative picture of actual machinery population distribution as it assumes a uniform distribution across total national population. Alternative methods of allocation could be utilised, for example population and runtime of grounds maintenance machinery (e.g. lawn mowers) could be considered a function of green space area. However the wide range and chaotic usage patterns of off-road machinery confuse this method – for example cranes, bulldozers, and other construction equipment could be considered as f(growth), lawnmowers f(green space), tractors f(agricultural production), aircraft tugs f(flights). Ideally population data would be provided at the community level, collated from survey and other data. In a number of UK communities, data for future public construction works, and government machinery operation activity are to be recorded which will significantly improve data (Taylor, 2011).

For Southampton the EPA (2010) runtime based emissions factors were used for the calculation of off-road emissions. The source population (n_{unit}) is estimated as national unit population (N_{unit}) (McGinlay, 2004), disaggregated by the ratio of total national population (N_{pop}) (ONS, 2010), to community population (n_{pop}) (ONS, 2010), equation 4.

$$n_{unit} = \left(\frac{N_{pop}}{n_{pop}}\right) N_{unit}$$

Equation 4. Local off-road machinery population

This assumes a uniform distribution across a national population. Domestic GIS datasets were used to supplement unit population data. Dwellings of multiple occupation (e.g. blocks of flats) are assumed to limit garden machinery to a single unit (i.e. communal management).

Shipping

The majority (circa 90 per cent) of global merchandise shipment is seaborne (Harris *et al*, 2012). Communities (e.g. Southampton, Portsmouth, New York) that are international cruise and container terminals, rely heavily on these industries for local economic growth and employment. Exclusion of emissions from these industries would lead to misinterpretation in policy making (Wright *et al*, 2012).

Emissions from shipping can be considered both a direct (port-side emissions) and an indirect (under sail in international waters) emissions source (Wright *et al*, 2011a). Territorial inventories may, depending on the extent of territorial waters in the geopolitical boundary, include port-side operations or entirely exclude shipping operations. A transboundary approach must consider the indirect emissions (movement between ports) of these sources (Wright *et* al, 2011a; Wright *et al*, 2012). Emissions from shipping are a result of engine operation, and can be considered largely a function of fuel consumed. Calculation of fuel consumed has broadly been undertaken using two approaches – 'bunker fuels' and 'engine use models'. A bunker fuels approach considers international bunker fuels loaded at the departure port provide a proxy to estimate emissions from shipping (EEA, 2009). However, shipping companies are likely to source the cheapest available fuel for the route, the result being where fuel cost is low, emissions are overestimated (e.g. Belgium), and where costs are high, emissions are underestimated (e.g. New Zealand) (De Meyer *et al*, 2008; Fitzgerald *et al*, 2011).

Alternatively emissions can be estimated from total fuel consumed, estimated using engine load, power and run-time, by engine and ship type (e.g. ro-ro ferry, liquid bulk, container, dry bulk), in the three phases of operation (hoteling, manoeuvre and cruise) (e.g. tier 3 presented by the European Environment Agency (EEA) EMEP emissions inventory guidebook (EEA, 2009)). The method requires significant data input on vessel characteristics, routes and time spent in operation phases. Detailed data of all ship movements (>250 gross tons) and characteristics are available from the from historic AIS (Automatic Identification System) datasets. However, the majority of these datasets demand a high cost purchase, which excludes some sub-national governments from using the data (e.g. Lloyds List Intelligence, (Informa, 2014)).

Despite the inherent uncertainly, due to restrictions on data availability, emissions related to shipping in Southampton were calculated using a bunker fuel approach. Fuels loaded at Southampton were calculated from national bunker fuels data (DECC, 2011) allocated to Southampton on proportion of total shipping movements at Southampton to national ports (DfT, 2013b).

Aviation

Assigning aviation emissions to communities is complex. Emissions are transboundary; many airports are located outside the community of study; and often cities act as regional or international aviation hubs (e.g. London, New York, Hong Kong) with transit passengers occupying a significant proportion of airline capacity (Harris *et* al, 2012).

Allocation of emissions must address these concerns, so as not to generate political tensions. Some studies exclude emissions on the basis that emissions are almost entirely transboundary and largely beyond the control of local government (e.g. Brown *et al*, 2009; Parshall *et al*, 2010; Scovacool and Brown, 2010). Some studies include domestic emissions and take-off and landing cycles to 1000m altitude for international emissions (e.g. Bush *et al*, 2008). Other studies have reported emissions based on quantities of fuels loaded at airports within city boundaries (technically considering movement away from the host city) (e.g. New York City, 2007; Kennedy, 2009). These methods do not consider the movement of passengers between flights and the surface movement of passengers from outside city limits. However demand from beyond the geographic boundary could be considered a function of the community demand, thus arguably, related aviation emissions should be accounted (Ramaswami *et al*, 2008). Previous authors suggest that regional airport usage by community inhabitants can be estimated as a function of local to regional population (Kennedy *et al*, 2010; Ramaswami *et al*, 2008). Assignment of emissions by community demand offers a truer picture emissions, considering only those emissions associated with the local population. However, this method is fraught with complexity, especially in cases where a number of international airports operate within close proximity (e.g. southern UK – Southampton; Bournemouth; Gatwick; Heathrow; Stansted, London City). Without accurate passenger origin-destination data, subjective judgments must be made to establish the geographic extent of airport demand.

Due to an absence of suitable passenger origin-destination data and close proximity of a number of regional airports, emissions arising from aviation at Southampton airport were estimated using a fuel consumed by departing flights methodology. Total flights by destination and aircraft type were obtained from operational timetables (Southampton Airport, 2008). Flight distances were calculated using as direct flight route between Southampton and destination airport (Google, 2012). Total fuel consumed was estimated using engine technology specific emissions factors (assumed 65% cabin occupancy) (TRL, 1999).

Agriculture, Forestry & Other Land Use (AFOLU)

Some argue AFOLU may be too insignificant for reporting at the urban level and should be excluded (UNEP, 2010). This is based upon the assumption that green space is both, relatively limited in urban centres, and the perception that urban green space has limited value due to human modification (Tratalos *et al*, 2007; Davies *et al*, 2011). This is often untrue (e.g. Southampton Common is 145 hectares; London's Hyde Park is 142 hectares, Beijing's Fragrant Hills Park is 160 hectares, and Vancouver's Stanley Park is

>400 hectares), and fails to consider the importance of public and private land in urban centres (e.g. private gardens, green roofs) which, whilst small compared to per unit area GHG emissions, are potentially important stocks of GHGs (Pataki *et al*, 2006).

Land use and management significantly influences ecosystem processes that effect GHG fluxes, (e.g. photosynthesis, respiration, decomposition). The IPCC (2006) guidelines for national inventories contain significant information for the calculation of AFOLU GHG fluxes. These guidelines suggest two methods: i) net carbon stock change over time, and ii) direct carbon flux rate (more commonly utilized for non-CO_2 species) (IPCC, 2006). AFOLU carbon flux for Southampton was calculated using the first option, to provide consistency with annual reports and promote favourable management of non-urbanized space over an extended time scale.

The methodology utilised established algorithms from the Rothampsted soil carbon model (RothC-26.3) and the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM) (Coleman and Jenkinson, 2008; Smith *et al*, 2001; Sitch *et al*, 2003). Basic climatic inputs (temperature, precipitation, daylight hours) were required (Met Office, 2014), in addition to data on organic matter inputs (obtained from LPJ-DGVM), soil clay content, and atmospheric CO_2 concentrations (Sitch *et al*, 2003).

GIS data (OS MasterMap) of land-cover types were used to create a map of the city area; where available this map was augmented with specific vegetation cover data provided by the municipal authority (Ordnance Survey, 2014; McCulloch, 2012). Land-cover data was classified into 11 broad categories (table 6) appropriate for vegetation modelling using LPJ-DGVM (Coleman and Jenkinson, 2008; Smith *et al*, 2001; Sitch *et al*, 2003). In cases where land-cover types are not complete for an area (e.g. scattered trees), the land-cover was assumed to be divided evenly between land-cover types. Where trees are described as 'scattered' (>30% of surface by canopy extent) 20% of total area is classified as that tree type, the remainder is divided evenly between other represented land-cover types (JNCC, 2010). In the grass (cut) category, data are required for total clippings collected, thus removed from the system, and total clippings left in-situ.

Table 6. Land-cover categories for modelling of vegetation or other land-cover types (adapted from JNCC, 2010)

Land-cover category	Example land cover types
Grass (cut 11 times a year)	Natural surface, Slope
Rough grass (not mown)	Rough Grass, Rough Grass and Other
Other herbaceous Plants	Perennials, Flowers, Roses
Private Gardens	Multiple surfaces in private residence
Broadleaved Summergreen Trees	Non-coniferous Trees, Scattered Non-coniferous Trees, Orchard
Needleleaved Evergreen Trees	Coniferous Trees, Scattered Coniferous Trees
Scrub	Scrub, Shrubs, Hedges, Heath
Marsh	Marsh Reeds or Saltmarsh
Sealed Surface	Road, Made Surface, Paths, Steps, Track, Structure, Traffic Calming, Pylon, Rail, Upper Level of Communications, Building, Glasshouse, Overhead Construction, Unclassified
Water	Inland Water, Foreshore, Tidal Water

Private gardens are representative of multiple land-cover types (e.g. lawn; ornamental planting; patios; tarmac; gravels). Land cover types in private gardens were estimated using expert judgment (6 land use categories (table 7).

Table 7. Assumed proportions of land cover types in private gardens for the southern UK (expert judgment)

Land Cover Type	Proportion of total area (%)
Grass (cut 11 times a year)	
Clippings removed	10%
Clippings left in situ	30%
Shrubs	10%
Temperate Broadleaved Summergreen Trees	10%
Other Herbaceous Plants	10%
Sealed	30%

The model was run across a temporal period of one year, with GHG flux calculated as the change in storage between runs.

Waste

Waste treatment represents a significant transboundary emissions source, with transport and treatment facilities often outside the city boundary (Wright *et al*, 2011a). Waste management activities generate emissions of CO_2, primarily of biogenic origin, with some fossil carbon and CH_4. At the community level local governments have influence over waste collection, treatment, and destination, meaning significant emission savings can be made through system reconfiguration (e.g. reduce landfill by increasing composting) (Gentil *et al*, 2009). Waste treatment emissions occur from waste transportation, biological treatment (composting; anaerobic digestion), mechanical treatment, processing, recycling to point of product production), landfill, and incineration. The waste industry, a number of academic groups and regulatory bodies have developed tools and methods for the accounting of GHG emissions from waste systems (Gentil *et al,* 2010) however there are considerable concerns regarding consistency, accuracy and transferability of these methods at the community level (Turner *et al*, 2012).

Different forms of waste collection, treatment and disposal are used within and between different communities. Each regional or municipal government has their own infrastructure, service provision and socio-economic conditions (Timlett and Williams, 2011). The waste sector has potential overlaps with electricity (e.g. generation from incineration) and AFOLU (changes in biological carbon stocks through disposal on land). To consider the role of the local government as both a manager of and an actor in waste management both in in-boundary, and out-of-boundary waste management needs to be included (Wright *et al*, 2011a; Wright *et al*, 2012).

Knowledge of waste composition and subsequent mass balance of CO_2 and CH_4 throughout the waste system is the key determinate in modelling waste emissions. Material fractions (e.g. cardboard, glass) define the different chemical (e.g. carbon content) and physical (e.g. moisture content) properties of the waste, leading to different treatment processes in the waste management system. The composition of the wastes in the treatment system will affect the mass balance due to the changes in carbon content and subsequent degradation patterns (Gentil *et al*, 2010).

The mass of waste in Southampton was split across household, construction and demolition, and commercial and industrial waste streams. The fractional composition and destination of these streams was determined on available data – twenty six household fractions (e.g. cardboard, glass) (WasteDataFlow, 2013), ten commercial and industrial fractions (DEFRA, 2013), and a single construction and demolition fraction (DEFRA, 2012). A direct linear relationship was assumed between waste quantity arising and waste fraction compositions.

Waste Collection & Transport

Collection of solid waste in the majority of cities is done using truck – ranging from small open “pickup” type vehicles to specialized vehicles with multi-compartment and compaction capabilities. Further transportation between waste treatment facilities (e.g. material reclamation facility residuals to further treatment at an alternative site) is achieved using large bulk transportation (e.g. 25t articulated trailer). A number of variables affect the emissions associated with the collection and transport of waste including: waste composition; the collection area; distance transported; the driver, and density of collection points (Larsen *et al*, 2009). Gentil *et al* (2010) review nine waste system LCA models and identify two main approaches to modelling waste transportation emissions – mechanistic and deterministic modelling. Mechanistic models require the detailed input of waste, vehicle, and environmental parameters to calculate fuel consumption and emissions. Conversely deterministic modelling uses only distance travelled and fuel consumptions as determinates of emissions (Gentil *et al*, 2010). Deterministic models offer the advantage of simplicity and less onerous data requirements.

In the case study waste transportation emissions within the geopolitical boundary are captured in the transport model. Emissions from waste transport to external treatment sites are then considered as a component of the waste system. Emissions related to out-of-boundary waste transport are apportioned using a deterministic approach based on the total mass of a given waste fraction transported (MJ t^{-1} km^{-1}) (Larsen *et al*, 2009). Actual transport distances were calculated from local government data for household waste fractions (WasteDataFlow, 2013), transport distances for other waste fractions were estimated using expert knowledge (Thomas, 2012).

Waste recovery & recycling

The recovery and recycling of wastes is common, recycled products are important raw materials and new products. The main source of emissions from waste processing, recovery and recycling are a result of energy use (e.g. diesel for the operation of plant). Emissions are calculated using emissions factors per unit waste treated by process (European Commission, 2014) (equation 5).

$$E_i^{waste} = \frac{\sum En_k^{Facility}}{M_{jk}^{waste}} EF_{ik}$$

Equation 5 Calculation of emissions from energy use per unit waste, where: E_i^{waste} = emission type *i* per unit waste, $En_k^{Facility}$ = energy use of type *k*, M_{jk}^{waste} = total mass waste treated of type *j* at facility *k*, EF_{ik}= emission factor emission *i*, energy type *k*

Composting

Composting is an aerobic process with high degradable organic carbon content. Material is converted to CO_2, some CH_4 is formed in anaerobic sections of the compost matrix, but is oxidised to a large extent in the aerobic sections (Boldrin *et al*, 2008). Methane production occurs in small quantities even in well aerated piles. The calculation of CO_2 and CH_4 emission from piles is calculated on mass balance of carbon input to carbon lost from the final product, as per equations detailed by Boldrin *et al* (2008).

Incineration

Incineration of waste is an increasingly popular and robust waste management technology for the management of a wide range of waste materials. A significant driver of increased use of the technology is increasing legislative pressure for the diversion of waste to landfill (Astrup *et al*, 2009). Emissions from incineration originate from the almost complete oxidation of waste carbon content to CO_2 (some material may not be fully oxidized and released in other forms (e.g. CO)). Emissions from incineration are calculated using a mass balance approach based on carbon content of input waste materials (equation 6).

$$E_i^{CO_2} = E_i^{CO} - E_i^{CH_4} - E_i^{C} - C_i^{unburnt}$$

Equation 6. Calculation of emission of CO_2 from combustion of waste in incineration using a mass balance approach, where $E_i^{CO_2}$ = emission of CO_2, E_i^{CO} = emission of carbon as CO, $E_i^{CH_4}$ = emission of carbon as CH_4, E_i^{C} = emission of carbon as elemental C, $C_i^{unburnt}$ = unoxidised carbon remaining in waste material

Landfill

Landfill, globally the most popular method of waste disposal, presents a number of complex emissions modelling challenges. Landfill technology has developed significantly, however technology use varies, ranging from open dumps to highly engineered bioreactor landfills with complex gas capture systems. The methodology must recognize the two stages, operational – waste is deposited at site, and closed – storage capacity is reached and the site is capped (Lou and Nair, 2009). Following closure, a landfill continues to emit GHGs, possibly for several hundred years, although some carbon will be indefinitely stored in the landfill (Manfredi *et al.*, 2009). CO_2 and CH_4 are generated and accumulated in landfill under anaerobic conditions. The overall emission from the landfill depends on the degradable organic content, the material in which the carbon is bound and the gas management system capture rate and type.

The IPCC (2007) guidelines advise use of a first order decay model, which assumes variable rate of degradable organic component of waste over time. With this form of reaction the amount of product is always proportional to the amount of reactive material in the landfill. However this modelling method may introduce complications in emissions apportioning at the urban level, as not all waste mass in the site may have come from the same locations. Kennedy *et al* (2010) propose a pragmatic solution adapted from the IPCC guidelines called Total Yield Gas. The approach estimates the long-term emissions for the waste landfilled in the assessment year, recognising that emissions will be released over a period of several years. Emissions arising from Southampton landfilled wastes were calculated using the Total Yield Gas approach.

Water

The provision of water and waste water services are similar to the provision of electricity. In many communities, water is supplied through a national or regional “grid” with centralized treatment and supply facilities. Water represents a transboundary material flow, excluded in territorial emissions modelling, but vital to community infrastructures (Ramaswami *et al,* 2008). Emissions associated with water are calculated on an end-user basis for water processing, treatment and transportation using per unit consumed emissions factors (European Commission, 2014). Commonly, as during this case study, water use is not metered and thus no actual consumption data are available. Currently in the UK significant effort is being directed to the installation of end-user metering; this will provide improved data resolution for future investigations (Walker, 2009). Emissions were calculated using standard estimates of water consumption provided by water suppliers.

Consumption emissions

It is generally accepted that the addition of a consumption based modelling approach extends the research implications and policy potential of a GHG inventory considerably (Baynes and Wiedmann, 2012). Territorial, production based accounts include emissions associated with exports at the point of production; but exclude those associated with supply chains and imports. The upstream impacts of production are allocated to the producer – the tendency is to mask embedded emissions and burden shifting (energy intensive industries are effectively exported). Transboundary approaches add an element of these out-of-boundary emissions, but do not give a full picture of the impact of consumption. Consumption based accounting focuses on the final consumption of households and governments, methods account all GHG emissions upstream of the community but exclude emissions from production within the city (Chavez and Ramaswami, 2011). A consumption based approach compliments a transboundary methodology, capturing emission flows and the driving forces associated with consumption (Wiedmann, 2009). Recognising these drivers is important for the development of policy – locally, regionally, nationally, and internationally – that considers the economic and environmental trade linkages and the wider implications of burden shifting (Peters, 2008).

A consumption based approach requires the of linking supply chain emissions with local consumption activities. Input-Output (IO) models detail the transactions between industries and sectors within the economy. An IOT requires knowledge of all flows of goods and services among intermediate and final sectors in disaggregated form for a given time period. This implicitly implies high volumes of data, which is difficult to obtain at the sub-national level, necessitating some form of scaling from national data (Bonfiglio and Chelli, 2008).The core element of an input-output model is a matrix concerning flows through the economy – sales and purchases from an industrial sector (a producer), to other sectors and the sector itself (consumers) (Miller and Blair, 2009). The basic input-output out model assumes homogeneity in sectors (i.e. each sector produces a single product) and linear production (i.e. proportionality of inputs and outputs which precludes economies of scale) (Weidema *et al,* 2006). The basic IO model can be extended to include material consumption and emissions – an Environmentally Extended Input-Output (EEIO) model. Effectively this creates an 'environment' sector, and the value of each item represents the 'output' of pollution (Yu *et al*, 2007: Leontief, 1986).

Consumption based emissions factors (GHG/£ spent) for the UK were calculated using an EEIO model. The IO data only holds data on final consumption at the national level. A downscaling methodology was therefore required to estimate final consumption at the local level. The model assumes no variation in emissions per £ spent between the national and local levels. The technical coefficient matrix was derived from UK supply and use tables for the year 2008 with 123 products and industry sectors in basic prices (ONS, 2011a). GHG emissions data by industry sector for the period were taken from the UK Environmental Accounts (ONS, 2012). The Environmental Accounts provide data on GHG emissions from 129 industrial sectors and 2 household emissions sources (travel and non-travel). The GHG data is provided at a more disaggregated level than the IO data in some sectors, this was scaled to the 123 sectors of the IO model using the parent sector of the lower level disaggregation according to the UK Standard Industry Classification 2007. A domestic technology assumption is applied to imported goods and services, whereby imports are assumed to have the same GHG intensity as domestic equivalents (Minx *et al*, 2009). It assumes the energy structure and economic structure of the imports can be approximated based on the domestic make-up of the UK. This may be a valid assumption for some regions, but underestimates GHG intensities of imports from emerging and developing regions (Druckman and Jackson, 2009).

Household consumption contributes circa 70% of the UK carbon footprint, the remainder is then attributable to government expenditure and capital investments (Minx *et al*, 2009). Expenditure between regions will vary considerably as a result of differences in incomes and other socio-demographic factors. However, the underlying IO data only provide expenditure data at the national level. Household demand was therefore downscaled to the local level using household expenditure data from the UK Living Costs and Food Survey (LCF) (annual survey of household expenditure on consumer products and services), and derived summary datasets provided in the Family Spending report (ONS, 2008, ONS, 2011b). Government expenditure is downscaled to the local level on an equal per capita basis. This assumes that individuals in the national population benefit equally from all government expenditure, which may not be the case. However it is considered a reasonable assumption in the absence of alternative data. Researchers have downscaled government expenditure using local expenditure statistics, however these data do not exist for the UK (Minx *et al*, 2013). The study does not consider emissions relating to capital investments.

Results and Discussion

As identified previously there are three methods for the GHG footprinting of communities – territorial, transboundary, and consumption based. Building on the discussion started by Wright *et al* (2011a), and continued by Chavez and Ramaswami (2011), who discuss the advantages and disadvantages of transboundary inventory approaches, this section discusses the implications of the three methods using the Southampton case study. Further to this the carbon footprint (CO_2 and CH_4) has been proposed to be suitable as a metric for anthropogenic GHG emissions (Wright *et al*, 2011b). To test this thesis, the carbon footprint (Wright *et al*, 2011b) and climate footprint (Williams *et al*, 2012) results of the Southampton case study are presented for each inventory method, and the policy implication discussed.

Territorial emissions

Geographically constrained territorial methods to GHG accounting are commonly used in national GHG accounting methodologies (e.g. IPCC, 2006). These approaches are largely ‘production-based’ accounts, which include emissions from direct energy use, industrial processing, in-boundary road and other surface transport, and other non-energy in-boundary processes (e.g. waste facilities), although end-use emissions from electricity consumption are often additionally included (e.g. Brown *et al*, 2009; Parshall, et al, 2009; Scovacool and brown, 2010).

The emissions results for Southampton suggest a direct territorial emissions of CO_2 and CH_4 (GHGs included in a carbon footprint (Wright *et al*, 2011b)) and the Kyoto Basket GHGs (climate footprint gases (Williams *et al*, 2012)) respectively of 268ktCO_2e and 273ktCO_2e. Addition of end-use electricity consumption in households and use by commerce and industry substantially increases this figure by 601ktCO_2e and 604ktCO_2e, respectively (figure 5). The minor increase (0.99%) in emissions between the carbon and climate footprints is driven primarily through inclusion of additional GHGs in transport (e.g. N_2O).

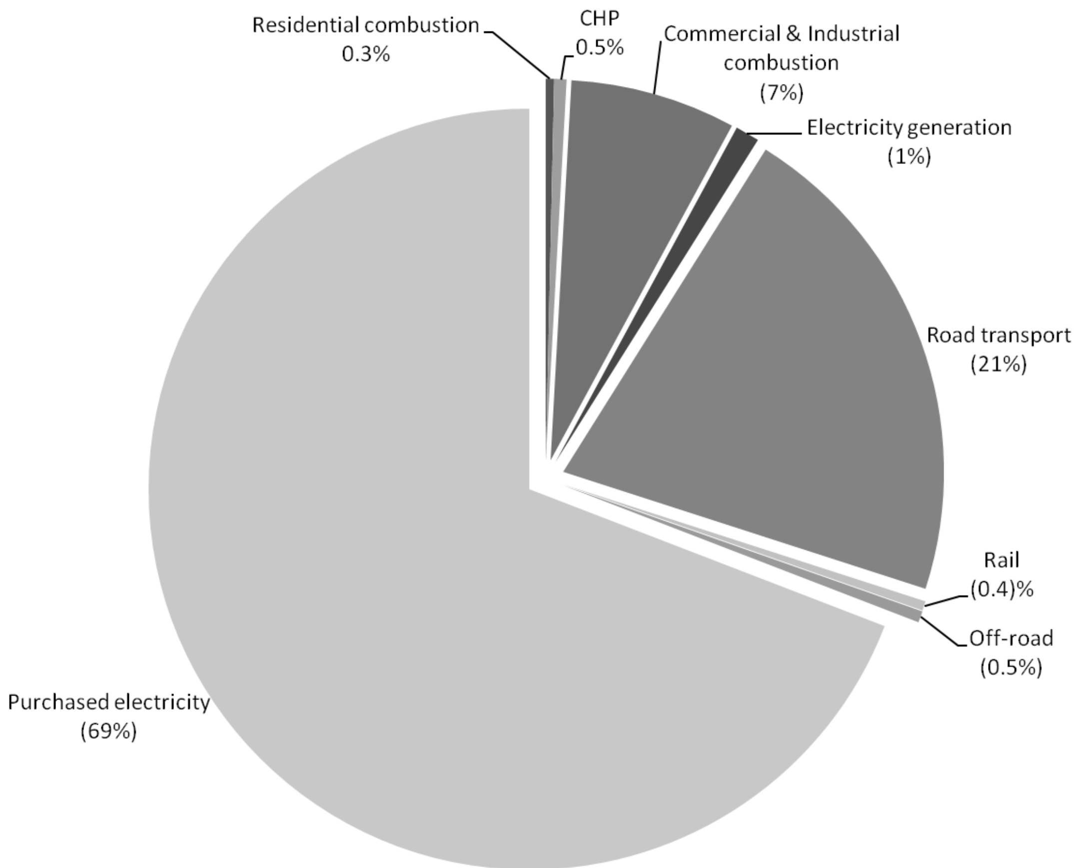

Figure 5. Southampton 2008 territorial (including end-user electricity) carbon footprint (CO_2 and CH_4) emissions (3.7tCO_2e/capita)

National territorial GHG accounts are commonly linked to related metrics of population or productively (e.g. GDP). Calculation of per capita emissions for the case study indicates 3.7 tCO_2e/capita carbon footprint, lower than the equivalent national production-based 10.32tCO_2e per capita estimate for the UK (United Nations Statistics Division, 2010). Whilst strictly geographic based methods can successfully identify local production-based emissions patterns and inform local development policy, they fail to capture the full extent of sub-national community infrastructures which extend beyond the geopolitical boundary (e.g. transport) (Chavez and Ramaswami, 2012; Wright et al, 2011a).

Transboundary emissions

Transboundary community GHG footprints extend the territorial approach to include emissions from transboundary infrastructure processes that extend beyond the geopolitical boundary of the community. Cities are responsible for a range of production based emissions from a range of sources (e.g. in-boundary transport; manufacturing; residential fuel consumption), but also for a demand led transboundary range of emissions (e.g. grid-connected electricity; imported products; water supply) (Wright *et al*, 2011a). Described by Ramaswami *et al* (2008) Denver (CO, USA) represents the first known community to have been inventoried using transboundary methodology. The study accounted all in boundary emissions and identified key community flows defined as: food; water; transport, and building materials (for shelter). Hillman and Ramaswami (2010) suggest, based on a study of eight US cities that these cross-boundary activities contribute on average 47% more than the in-boundary emissions sources. This consideration is reflected in developing international standards (e.g. BSI, 2013, Arikan, 2012) which suggest a transboundary approach to account both the territorial and transboundary aspects of a community – ideally moving towards an approach that replicates the process(s) of urban metabolism (Baynes & Wiedmann, 2012).

The transboundary results for the Southampton case study include all direct emissions with the addition of: commuter road transport; shipping; aviation; out-of-boundary waste emissions; water and wastewater supply/treatment; construction materials, and food and drink. The 2008 results, carbon footprint 2643$ktCO_2e$, and climate footprint 2787$ktCO_2e$, are, as expected, substantiality larger than the comparative territorial results (figure 6). The two footprinting techniques, as per territorial emissions methods produce results within 1%. The increased emissions in the climate footprint stem primarily from transport (i.e. road transport, shipping, aviation).

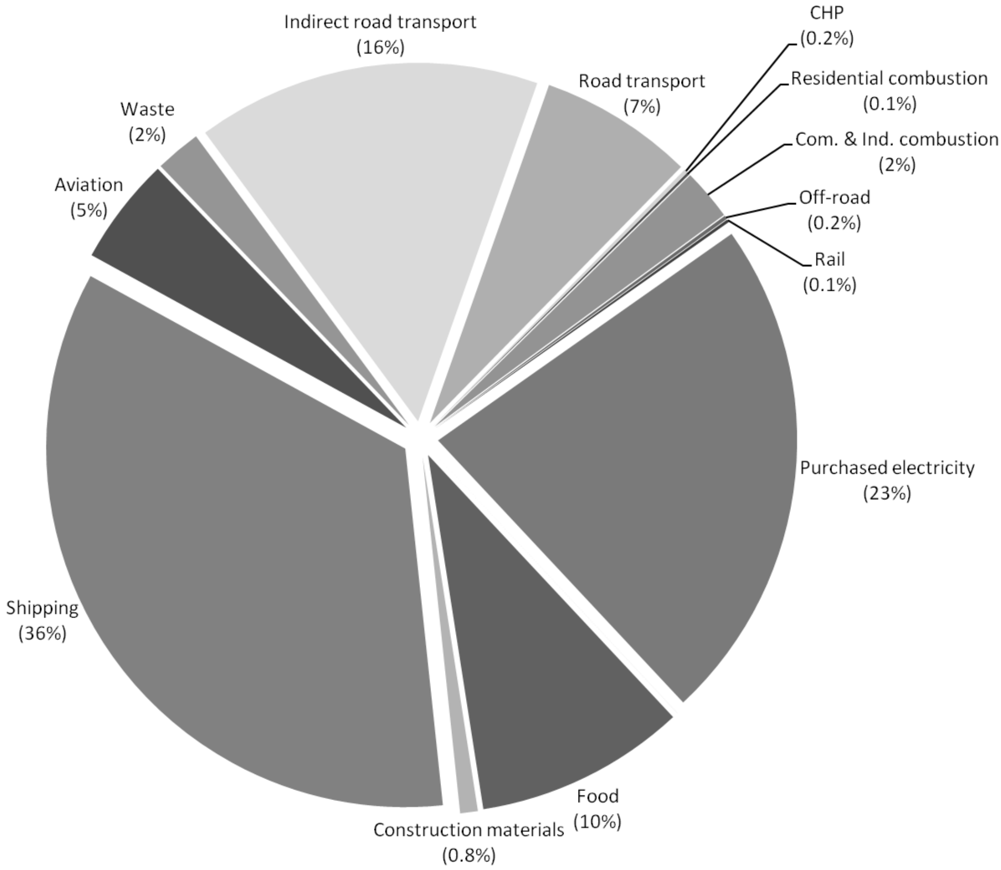

Figure 6 Southampton 2008 transboundary carbon footprint (CO_2 and CH_4) (2643ktCO_2e), note water supply not shown (value >0.01%, 0.04ktCO_2e)

Transport emissions (shipping, road, aviation) represent a significant component of emissions totals (shipping 36%, road 23%, aviation 5%). High shipping emissions are a result of the extended travel distance and subsequent high fuel demands. Whilst sub-national governments may have limited control (i.e. typically only port-side operations) over these emissions sources, inclusion is important due to the strong economic reliance on these industries (Wright *et al*, 2012). However sub-national governments do have access to control to address these emissions through local air quality control. Similarly road transport control can be found through air quality control and additional controls in planning and road management.

Energy related emissions comprise a large component of total emissions, electricity provides the dominant contribution to this sector. The disaggregation of emissions related to heat production from emissions associated with grid connected electricity generation impacted insignificantly (>1%) on emissions per unit electricity consumed. At the local level renewables account for an equivalent grid emission of 3ktCO_2e. Evidently emissions from electricity are mainly dependent on the intensity of supply, highlighting a powerful interlink between local and national policy making. This interlink will become particularly pertinent with the potential advent of locally led energy initiatives (e.g. micro-generation; rail electrification; electric vehicle charging networks) (Weber, 2010).

Emissions from AFLOU are insignificant, however this masks the carbon stored in urban green space in the case study city. Stored carbon amounted to 470.00 ktCO_2e – in public parks, commercial and private land. Sub-national governments must consider green-space in their control when developing GHG management plans. Practice to exclude these emissions assumes green space storage is minimal, the results demonstrate this may not be the case. Careful consideration must be given to development that affects community green space (both negative – e.g. green space urbanisation; and positive – e.g. installation of green roofs), for the creation of carbon sinks, the wider potential social, and economic benefits (Davies *et al*, 2011).

Supply chain and infrastructure related emissions form a significant component of the total transboundary assessment, highlighting the importance of supply chains in community footprinting. The recent PAS2070 (BSI, 2013) suggests further inclusion of all materials making >2% material contribution to the community. This would add a further 1315ktCO_2e and 1435ktCO_2e (carbon and climate footprint respectively) to the Southampton results. However, there are concerns of double counting with the territorial element of the assessment. This double counting is difficult to avoid due to the differences in perspective and resolution of the production and consumption methods used to calculate this data (e.g. food consumption is captured as final household demand, and food production is captured as a territorial emission).

The primary advantage of a transboundary footprint is the level of completeness created through inclusion of in-boundary emissions sources and transboundary infrastructures that supply these activities (Chavez and Ramaswami, 2013). Given this completeness, transboundary based footprints can be utilised to inform a broad range of mitigation and management strategies at the local, regional, and national scales (Bayes *et al*, 2011). Additionally transboundary footprints are more relevant and easier to communicate to residents due to the inclusion of major activities included in personal and home carbon calculators (Chavez and Ramaswami, 2011).

The main shortcoming of the transboundary method is the inconsistency in approach and application of metrics between studies. For example Kennedy *et al* (2009) do not include emissions related to food, construction materials, or water. Standards (e.g. PAS2070 (BSI, 2013), GHG Protocol for Community Reporting (Arikan, 2012)) are emerging that attempt to clarify and develop consistency in reporting structures.

Comparability is also difficult; results require normalisation to enable inter-community comparisons. The majority of territorial inventories are normalised using a per capita metric, however this may not be appropriate for transboundary approaches. Communities with net exporting energy production will be penalized over communities who are net energy importers (Wright *et al*, 2011a). As Chavez and Ramaswami (2011) highlight, metrics for the representation and comparison of transboundary approaches require further research.

Consumption emissions

A consumption based approach accounts for global GHG emissions that result from final consumption (i.e. households, government, and capital investment). Results for Southampton (carbon footprint 3160ktCO_2e, climate footprint 3590ktCO_2e) (table 8) are consistent with previous studies where consumption based estimates are higher than production based emissions, with the majority of emissions ultimately driven by households (Peters, 2008; Wiedmann *et al*, 2010; Minx *et al*, 2009; Druckman and Jackson, 2009). The disparity between the carbon and climate footprint is higher (circa 12%), this is primarily driven by high emissions of N_2O in agriculture. Highlighting the need for a climate footprint approach in certain situations where high emission of GHGs other than CO_2 and CH_4 are probable (Wright *et al*, 2011b).

Table 8. Southampton 2008 consumption based carbon and climate footprints

	Carbon footprint (ktCO2e)[1]	**Climate footprint (ktCO2e)[2]**
Food & drink	516	750
Clothes & apparel	58	65
Mining & fuel processing	186	188
Utility services	742	753
Construction	6	6
Transport services	344	351
Education	39	44
Recreation	48	55
Health	78	84
Other products & services	1034	1142
Direct fuels consumption by households	123	125
Final consumption by national & municipal government	92	132
Total	3264	3695

1. Includes CO_2 & CH_4 as per Wright *et al*, 2011b.
2. Includes CO_2, CH_4, N_2O, SF_6, HFC, PFC as per Wright et al 2011b & Williams *et al* 2012.

It is now generally accepted that the addition of a consumption based account extends the policy implications of local GHG account (Baynes & Wiedmann, 2012). The approach provides value for the assessment of household consumer lifestyle on GHG emissions, making the consumption impact of households and government visible (Chavez and Ramaswami, 2011). Arguably a consumption based approach provides for the most rigorous method for per capita GHG comparison, as consumption is driven by the residents of a community. Additionally a consumption based approach can inform local policy to reduce supply chain emissions as, when accurate local data are available, imports/exports can be traced. Recognising these advantageous policy implications, emerging standard PAS2070 requires the separate completion of both a transboundary inventory, and a consumption based inventory (BSI, 2013).

However consumption based methods are data intensive, and are only truly valuable where accurate IO data are available. Misallocation of emissions can occur where physical flows do not match monetary flows represented in local IO tables (e.g. a large corporate headquarters in the community reporting economic activity beyond the city boundary) (Chavez and Ramaswami, 2011). Additionally the consumption method effectively divides the community into two, with activities for exports not included in the unit of analysis. This can exclude a large element of a local economy (e.g. resorts, industrial communities) which could be managed through local policy.

In this particular study there are several important limitations to note. The assumption of a homogenous technology mix in the EEIO model presents a level of inherent uncertainty – imports come from a range of countries using a range of different emission and resource intensities. This may be a valid assumption for some regions, but underestimates GHG intensities of imports from emerging and developing regions (Druckman and Jackson, 2009). The accepted solution is to employ a Multi-Region Input-Output (MRIO) model. MRIO models represent the interactions between any number of regions with potentially differing technology mixes, by internalising trade flows within internal demand (Minx *et al*, 2009). The method of downscaling presents two important limitations. Firstly expenditure can only be estimated for broad categories (e.g. only a product is recorded not a brand or origin). Partially a result of the homogeneity assumption of the underlying IO model, this assumes common per unit emissions in these categories, which may not be entirely representative. Secondly, this generalisation may misrepresent the quantity of product purchased. For example the same expenditure on a high cost product variant would provide less quantity of product and potentially lower emissions, than a high quantity low cost product.

Conclusions

This paper has presented several important developments to the assessment of community carbon footprints. Methods have been developed to assess emissions at a spatial, sector, scope, and technology disaggregation, suitable for use by policy makers at the community level. The methods have been applied to real data, for Southampton, UK. The methods have been presented to show the policy implications of territorial, transboundary, and consumption based accounting procedures. Only a limited difference in emissions totals was observed between the carbon and climate footprints for the case study city. Adding to the argument that the carbon footprint (CO_2 and CH_4) offers a low cost, low data, universal metric of anthropogenic GHG emission and subsequent management.

Territorial accounts may be suitable for national GHG inventories, but cannot represent the transboundary infrastructures of sub-national communities. Transboundary approaches successfully extend the territorial approach to include emissions from key infrastructures essentially to sub-national communities. The addition of a consumption based account further extends the policy relevance and research applications of community accounting. Consumption based approaches show the impact of household consumer lifestyle on GHG emissions, and making the supply chain impact of households and government visible.

Recognising the advantages of transboundary and the simultaneous application of a consumption based approach, emerging standards, such as PAS2070, advocate a full lifecycle approach, combining a transboundary approach – sensitive to local production factors; and consumption based approach – sensitive to drivers of consumption – providing a comprehensive report.

Finally, the establishment of the goal of a global network of low carbon cities requires good measurement and the appropriate tools. PAS2070 and related standards represent a significant step towards the development of a comparative assessment of urban community GHGs. Barriers still exist – comparable metrics need to be further developed and local governments often do not possess the resources and skills required to complete an inventory assessment. This research demonstrates the effectiveness of a successful academic-public synergy. The partnership capitalized on the strengths and unique skills present in a local academic institution, to deliver a cutting-edge joint working solution to a local issue, at manageable cost within the context of tightly confined financial resources. The success of the project has led to the further proposal development and partnership working, with tangible benefits for both parties.

5. Sensitivity and uncertainty analysis of a sub-national GHG inventory model

Introduction

This chapter investigates the input sensitivity of the model using a local One-At-a-Time (OAT) sensitivity analysis, and explores associated uncertainty using a Monte Carlo simulation, to meet Aim/Objective(s) 3a and 3b of this research project. This work was developed by the author and written for this PhD thesis under normal PhD supervision conditions.

PAS2070 has provided a mechanism for the standardisation of city GHG inventories. The previous chapter developed the methods required for the realisation of a PAS2070 compliant inventory for the case study of Southampton, UK. This development is a significant step towards establishment of a consistent inventory technique and the development of standardised tools for use by academics, politicians and decisions makers. However, often in large life-cycle impact studies of this nature sensitivities and uncertainties of the modelling process are ignored. Systems, such as the urban environment are so complex that models are developed to approximate real-world conditions, empirical evidence gathering is precluded by complexity, resource, or time constraints. Models are defined by a series of equations, input factors, variables, parameters and variables representing an arbitrary enclosure of real-world, often open and un-curtailed systems. Inherently the model constraints will not exactly replicate the 'ideal' constraints of the system being studied. As a result models cannot be empirically proven; instead it must be tested to provide qualification and evidence to support the position that the model is a true approximation of real-conditions. In the case of PAS2070 the primary users are likely to be politicians, other policy decision makers, and academics. It is also likely that the results of such studies will be used to support GHG emissions management policies and decisions. Users of PAS2070 models must realise that such models are inherently subject to multiple sources of uncertainty.

This chapter forms a single paper which builds on the methods for the realisation of PAS2070 and the case study developed in the previous chapter. The methods for the development of a PAS2070 inventory are briefly reviewed. Sensitivity of the model is explored using a local One-At-a-Time (OAT) sensitivity analysis (a single input variable is subject to perturbation whilst all others are held constant). Sensitivity is normalised using sensitivity coefficients to enable comparison of the effects of perturbation in different input variables with differing units. A simple linear regression, Pearson's rho

(ρ), is used to test the linearity of input variable perturbation and change in output. Non-linear correlation within variables may serve to indicate that normalised sensitivity coefficients do not provide accurate representation of the effect of individual input parameters. Additionally, superposition was considered; whether the scaled variation in two or more parameters results in the same scaled variation in output. Where superposition does not hold true it indicates that the variation in output cannot be explained by addition of individual sensitivities.

Whilst sensitivity provides important information regarding the model response, it cannot fully explore the input space. The OAT sensitivity analysis considers one input variable in isolation. A number of input variables were identified as exhibiting non-linear relationships, and superposition was proven to be false. For these reasons a suitable method of assessing uncertainty in the model across multiple input variables was required. Sources of error and uncertainty within the model are explored, and stochastic variation determined using Monte Carlo techniques. Five thousand simulation runs were conducted to determine the model probability density function and predicted output. Monte Carlo simulation relies on repeated random sampling to compute the probability of a given result. A statistical distribution is identified for each of the input parameters in the model, random samples are drawn from these variables which represent a 'set' of input values. For each set of input variables a set of output variables are created. The sampling statistics of the output variables are used to characterise the output variation.

The work presented demonstrates an important development, highlighting areas for particular consideration by policy makers, and other users of PAS2070. Whilst simultaneously informing areas for future research. Importantly failure to acknowledge these uncertainties can lead to incorrect assumption and interpretation, rather than informed decision making.

Sensitivity and uncertainty analysis of a sub-national GHG inventory model

Abstract

PAS2070 has emerged through international collaboration to offer an international harmonised standard for the inventorying of greenhouse gas emissions from cities. Uncertainty and sensitivity are often ignored in large LCA-type studies of this nature. This paper reviews and demonstrates methods for estimating the sensitivity (parameter sensitivity) and uncertainty (parameter importance) for the DPSC approach of a PAS2070 community GHG model for Southampton, UK. Local sensitivity analysis was undertaken to investigate the effect of one-at-a-time changes to inputs on predicted emissions. Sources of error and uncertainty within the model are explored, and stochastic variation determined using Monte Carlo techniques. Five thousand simulation runs were conducted to determine the model probability density function and predicted output. The actual estimated emissions profile 3957ktCO_2e deviates 36ktCO_2e (>1%) from the predicted mean (3845ktCO_2e), with a 95% confidence interval, we suggest the actual emissions from the modelled case study are 3957 ±6.25ktCO_2e. The work demonstrates areas of particular consideration for policy and decision makers; and the uncertainties in urban GHG emissions models. A failure to acknowledge these uncertainties can lead, at best, to a false sense of reliability and, at worst, to incorrect assumption and interpretation.

Key Words

City GHG Inventory; Monte Carlo Simulation; Uncertainty analysis; Carbon Footprint; sensitivity analysis

Introduction

Cites are increasingly being recognised as significant contributors to emissions of GHG emissions (Kennedy *et al,* 2011, Wright *et al*, 2011b). City authorities are responding to the challenge of carbon emissions through the development of policy initiatives, partnerships and management plans (Harris, Chow and Symons, 2012; Wright *et al*, 2012; Kern and Bulkeley, 2009). A Publically Available Standard[1] – PAS2070 (BSI, 2013)

[1] A Publicly Available Specification (PAS) is a sponsored fast-track standard directed by the requirements of a client organisation. Selected stakeholders develop a PAS that is endorsed by the British Standards Institution. The PAS is reviewed after 2 years and a decision is made as to whether it should progress to become a formal British Standard.

has emerged through international collaboration, to offer a standardised process for the inventorying of GHG emissions from cities. Publically Available Standard (PAS) 2070 'Specification for the assessment of greenhouse gas emissions of a city' recognises the need for a holistic approach to city GHG inventories (BSI, 2013). The specification details a dual approach to the inventorying: i) a 'Direct-Plus Supply Chain' (DPSC) approach - an extended territorial approach with the addition of supply chain emissions, providing the advantages of CB and TB methods) (Chavez and Ramaswami, 2011; Chavez and Ramaswami, 2012); and ii) a consumption based (CB) approach. The results of the two methods are reported separately to avoid double counting and misinterpretation (BSI, 2013). In a parallel paper, Wright *et al* (in preparation) develop the methods to achieve the PAS2070 specification using the case study of Southampton, UK. This estimation and the underlying model is the subject of this paper. PAS2070 doesn't specify a particular method for uncertainty analysis in inventory claims, instead preferring to provide a series of data quality rules.

Many systems, such as a city GHG inventory, are so complex that experimental measurement is impossible due to resource, time or other constraints. Mathematical models are often developed to approximate real-world conditions, defined by a series of equations, input factors, parameters and variables designed to provide approximation and characterisation of the systems being studied. Essentially models represent an arbitrary enclosure of open and un-curtailed real-world systems and conditions. Inherently the 'rules' applied through the model equations will not capture all the 'rules' of the true system. The model cannot be validated in the sense of being 'proven true'; rather it must be subject to a series of tests and qualification to provide corroborating evidence to support the position that the model is true approximation of real-conditions (Saltelli *et al*, 2008). In the case of PAS2070, the primary users are likely to be municipal governments and other policy decision makers. The use of models such as this to inform and support decision-making is becoming increasingly common. Uncertainty and sensitivity analysis are important steps in the model development process in order to obtain credible and reliable information (Campolongo, 2007). Policy-makers should recognise that such models are inherently subject to multiple potential sources of uncertainty. A model is an abstraction, simplification and interpretation of reality. Whilst no model is a perfect representation, they become especially useful when limitations and assumptions are known so the user can make informed decisions on the practical results (Cheng & Steemers, 2011). Additionally, qualification of sensitivity and uncertainty is not only critical to the validation of the model, but may also serve to guide future research (Hamby, 1994).

The sensitivity and uncertainty of the DPSC approach developed in the previous chapter is estimated. The background to the case study briefly is briefly reviewed. The sensitivity of the DPSC input components of the PAS2070 model are considered. Sensitivities for the CB component of the PAS2070 model are considered at the aggregated emission factor per unit expenditure level, rather than at the EEIO input variable level due to complexities involved in this form of modelling (Lenzen *et al*, 2010). This is followed by a review of the local sensitivity analysis methods. Sensitivity analysis is used to assign the uncertainty in the output of the model to different sources of uncertainty in the model's inputs and how the model responds to changes in input data (Saltelli *et al*, 2008). Whilst this method provides a good indicator of variables with high impact on the model, it does not provide qualification of uncertainty. Related to sensitivity analysis, uncertainty analysis is a means of quantifying the uncertainty (doubt) in a measurement. Ideally, to provide holistic model validation, sensitivity and uncertainty analysis should be run in tandem (Saltelli *et al*, 2008). Sources of uncertainty associated with the PAS2070 model are philosophically explored. A differentiation is made between error and uncertainty within the model. Sources of error (recognisable inaccuracy not due to a lack of knowledge); and uncertainty – either aleatory (inherent variation associated with the system of study) or epistemic (inaccuracy due to a lack of knowledge) uncertainty (Oberkampf *et al*, 2002) are evaluated and philosophically mapped. The magnitude of uncertainties are estimated through use of a Monte Carlo analysis (repeated random sampling to generate a probability distribution), to provide a confidence limit (95%) for the model. We conclude through discussion of the model confidence, and future methods that may be applied to improve uncertainty estimates, and address input variables identified as having high sensitivity and/or uncertainty.

PAS2070 Case Study – Southampton, UK

Wright *et al* (2011a) developed a framework for the application of LCA-type inventories, specifically carbon- or climate-footprints (Williams *et al,* 2012), to cities. PAS2070 builds on this framework to specify the requirements for an urban community GHG inventory. Chapter 4 of this thesis developed the methods and techniques required for the achievement of PAS2070, using the case study of Southampton.

The city is a part of a wider metropolitan region, the most densely populated part of the county of Hampshire on the south coast of England. The city serves as a representative, manageable case study and proof of concept, with all of the possible components (e.g. road transport, waste, shipping, aviation) including in PAS2070 present. The city is an international cruise and container terminal; has two universities with a combined transitory population in excess of 40,000 (Zhang *et al*, 2008); is a regional business and commercial destination; and has Southampton Airport in close proximity in neighbouring Eastleigh.

The methods developed specify the requirement for both activity data collection and GHG inventory calculation. A range of techniques are proposed on a sector-by-sector basis. For reference in the following sections a summary of the methods proposed is provided in Table 1. Based on the methods proposed, the Inventory for Southampton 2008 is calculated as 3808.17ktCO_2e across all sectors (for a detailed breakdown see reporting tables in the previous chapter).

Sensitivity Analysis

There are a number of situations when the stability of model results in terms of sensitivity to perturbations of input data are of interest, including a need to determine: i) which variables the model is most sensitive to and subsequently which parameters require additional research to improve knowledge and reduce uncertainty. ii) which parameters are insignificant, and may be excluded; iii) which inputs contribute most to output variability; iv) which parameters are most correlated with output; v) the consequences of changes in a given input parameter (Hamby, 1994). It is also useful in model validation (i.e. is the model doing what it is supposed to?); if a model responds unacceptably, efforts can be focused on the source of the problem relatively easily (Frey & Patil, 2002). In the context of policy- and decision-making, identification of these variables is valuable for the assignment of resources for GHG management and reduction.

Sensitivity analysis provides a systematic method for varying model parameters and assessing the impact of those changes. Sensitivity analysis is extremely useful in determining how a particular dependent variable will change based on a variation in the value of an input variable from what was previously assumed (Heijungs, 2010). Methods include probabilistic analysis, using either frequentist or Bayesian framework statistical-based analysis, and deterministic analysis, using mathematical based approaches (e.g. nominal range) (Frey & Patil, 2002). One of the simplest approaches to sensitivity analysis is one-at-a-time (OAT) (sometimes referred to as local or perturbation analysis) screening techniques to examine the effect this has on the output (Campolongo *et al*, 2007). In an OAT method, a small change is applied to a single input variable (e.g. in x, hence Δx) whilst all other variables remain constant, producing a change in the result (z, hence Δz) (Heijungs, 2010). This logical approach allows the identification of relative significance of input parameters to which output parameters are sensitive (Saltelli et al, 2000). In general, this effect can be defined as

Table 9. Summary of the methods proposed in chapter 4 of this thesis for the calculation of a PAS2070 inventory

Sector	**Methods proposed**
Material supply	Single-region Environmentally-Extended-Input-Output (EEIO) adjusted to local coefficients using location quotients developed from Gross Value Added (GVA).
Water supply	Per unit activity emissions factor.
Electricity, heat/steam	Per unit consumed emissions factor.
Residential stationary combustion	GIS based bottom-up energy demand model - BREDEM-8 (Building Research Establishment Domestic Energy Model) (Anderson *et al*, 2001) & SAP 2009 (DECC, 2009).
Commercial stationary combustion	Employment/energy intensity fuel consumption factors.
IPPU	Employment/energy intensity fuel consumption factors.
AFOLU	Adapted Rothampsted soil carbon model (RothC-26.3) and the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM) (Coleman and Jenkinson, 2008; Smith, Prentice and Sykes, 2001; Sitch et al., 2003).
Road transport	Algorithms and emissions factors adapted from the ARTEMIS (Assessment and Reliability of Transport Emission Models and Inventory Systems) methodology.
Rail	ARTEMIS (Assessment and Reliability of Transport Emission Models and Inventory Systems) methods, GIS, passenger movements.
Other off-road	Activity, use, run-time unit population model.
Waterborne navigation	Bunker fuels per unit consumed emissions factor (movement activity data preferred where possible)
Aviation	Unit activity to estimate fuel and emissions.
Waste disposal	Fractional waste composition and treatment LCA-type emissions model
Waste water treatment	Per unit treatment emissions factor.

the supposition that an output variable y depends on the the input variable x and that dependence can be expressed by a function f:

$$y = f(x)$$

Equation 7. Dependency of output variable on input

The element that determines sensitivity is the change in the result of the dependent (Δy) caused by the change in the input (Δx). This can be expressed using the partial derivatives:

$$\Delta y = \frac{\partial y}{\partial x} \Delta x$$

Equation 8. Partial derivate expression of sensitivity

Coefficients such as $\frac{\partial y}{\partial x}$ are referred to as sensitivity coefficients, whose evaluation requires a specification of the function f. Therefore the sensitivity coefficient of f with repect to input parameter x can be expressed as:

$$\frac{\partial y}{\partial x} = \frac{\partial f(x)}{\partial x}$$

Equation 9. Sensitivity coefficients

Sensitivity coefficients are important for both sensitivity and uncertainty analysis (Heijungs, 2010). In the context of sensitivity, they serve to provide the multipliers that establish sensitive from non-sensitive parameters (i.e. key parameters during data collection). In the context of uncertainty, they provide essential base information (e.g. identification of high sensitivity variables where uncertainty estimates are important) for expansion and exploration. Sensitivity coefficients are usually normalised to enable comparison of the effects of perturbation in different input variables with differing units. The normalised sensitivity coefficient (S_i) can be calculated as a percentage change in the output parameter given a 1% change in input parameter:

$$S_i = \frac{\partial y}{\partial x_i} \frac{x_i}{y}$$

Equation 10. Normalised sensitivity coefficients

An OAT analysis was conducted using estimated x values based on the actual values from the Southampton 2008 case study and normalised sensitivity coefficients were calculated. A total of 474 primary input variables were identified in the model; a number of these input variables represent multiple data input values utilised for different internal scenarios within the model (e.g. vehicle technology installed proportion in fleet – a single variable, relevant to multiple technology types). These variables were considered as a singular input and varied using a scaling factor applied to all data items simultaneously, thereby demonstrating the sensitivity of the input variable, rather than a data item which may vary between studies. Each input parameter is subject to a small perturbation Δx whilst all other parameters remain constant at the determined nominal value. Accuracy of the calculated sensitivity depends on the size of the perturbation Δx. Too greater variation potentially violates the assumption of linearity required by OAT sensitivity analysis, too small will result in a high round-off error (Saltelli *et al*, 2000). A 1% perturbation is used in this study; this is consistent with recommendations in the literature and previous similar studies (e.g. Hughes *et al*, 2013; Cheng & Steemers, 2011; Firth *et al*, 2010).

A simple linear regression, Pearson's rho (ρ), was used to test the linearity of input variable perturbation and change in output. Non-linear correlation within variables may serve to indicate that normalised sensitivity coefficients do not provide accurate representation of the effect of individual input parameters. To test linearity, parameters were varied over a wide range (up to 200%). The principle of linearity dictates that a variation in an input factor by a scaling factor will yield the same scaling in the output (i.e. Pearson's ρ = 1). In the model used, a number of variables are known to exhibit non-linear relationships with outputs, due to the polynomial nature of the input, for example in the transport emissions factors.

Additionally superposition was considered; superposition holds true if the scaled variation in two or more parameters results in the same scaled variation in output. Superposition was modelled through comparison of the variation predicted by addition of the individual effects of scaled variation, against the actual simultaneous variation in output. The results (Table 10) suggest that the combined effect of multiple input parameters cannot be predicted through addition of individual effect (Hughes *et al*, 2013). It must be noted that local sensitivity methods cannot be used for a full comprehensive evaluation of uncertainty. The OAT approach does not fully explore the input space, as it cannot consider the simultaneous variation of input variables, meaning one cannot explore input variable interactions (Saltelli, 2010).

Uncertainty Analysis

The OAT sensitivity analysis considers the impact on emissions totals due to variation in input parameters in isolation. In a number of cases, non-linear relationships were observed and thus further investigation was required to establish the effect(s) of multi-variable impact on the output parameters. It would not be valid to assess the impact of multiple variables through addition of the effect of individually varied input parameters. Uncertainty analysis investigates the impact of uncertainty on model outputs. It necessitates that the nature and magnitude of the uncertainties in the modelling process be determined and the subsequent effects be combined simultaneously. Uncertainty is commonly not considered in large LCA-type studies, which retracts from both usability and transparency. Common reasoning is a lack of method (Ciroth, 2004) or prohibitive computational requirement and processing times (Heijungs, 2010). Further to this typically there is limited sound information on which to base estimates of uncertainty or correlations between uncertainties (Hughes *et al*, 2013).

Probabilistic uncertainty methods, such as Monte Carlo sampling are the most frequently applied methods for studying the propagation of uncertainty through deterministic models (Oberkampf *et* al, 2002; Heijungs, 2010). Monte Carlo simulation relies on repeated random sampling to compute results and is very similar to random experiments where specific results are not known in advance. Monte Carlo simulation can thus be considered a methodological way of compiling 'what-if' scenarios. A statistical distribution is identified for each of the input parameters in the model, random samples are drawn from these variables which represent a 'set' of input values. For each set of input variables a set of output variables are created. The sampling statistics of the output variables are used to characterise the output variation. In this study, we use traditional Monte Carlo methods, however the case study, Southampton, is proportionally a smaller dataset than larger, especially mega-cities (10+ population). This should not preclude quantification of uncertainty, the solution to the greater data requirements may lie in smarter algorithms (Peters, 2007), or more sophisticated sampling strategies than the native Monte Carlo method (e.g. Latin Hypercube) (Heijungs, 2010).

Sources of uncertainty

The calculations in the model are approximations of real-world values, and there may be errors in both the calculations and the execution of them. There are numerous potential sources of uncertainty in the input data and emissions calculations; furthermore it is impossible to ascertain whether they truly represent the relationships between the variables. The model suffers from various uncertainties at the structural, parameter and input levels. The key sources of uncertainty in model output are input data; parameter values, and model structure (i.e. the conceptual model) (Refsgaard *et al*, 2006), explored in the uncertainty map, Figure 5.

At the structural level, uncertainty primarily arises from oversimplifications in the modelled processes, or simply as a result of the non-inclusion of an unknown variable. Additionally expert judgment is inherent in model development as part of the transition from a perceptual (systematic structuring and qualitative understanding of the system) – to a formal (set of equations to describe this understanding) model. At each stage, the abstraction and assumptions naturally increase. The perceptual model is a constrained and subjective perspective a natural system; this is further constrained by mathematical and computational limits. Inherently this knowledge is influenced by social and political process (e.g. through personal motivation or a result of institutional agendas and funding) (Krueger *et al*, 2012).

At the input and parameters levels, uncertainties are concerned with a combination of assumptive errors: through simplification; expert judgment or otherwise; a particular concern; missing data (Bhander *et al*, 2008). Data may not exist to reflect specific local conditions or technologies, for example local vehicle fleet statistics, resulting in the use of national fleet statistics as a proxy for local conditions. Missing data uncertainties can become particularly problematic if not recognised and noted in the final report. The model requires the input of a range of activity data across a range of sectors (e.g. fuel combustion in domestic property; vehicle speed). The discussions in the previous chapter developed a number of sector-based methods to obtain the necessary activity input data (see table 1). Inevitably this requires the use of expert knowledge and opinion in the modelling process (Krueger *et al*, 2012). Boundary setting may present a significant source of uncertainty, and has been identified by previous authors as important in the establishment of the true emissions profile (Matthews, 2008). The model utilises a number of GIS input sources, and indeed one of the primary concerns of the modelling process is the setting of the geopolitical boundary of the study city. There are known geographical uncertainty tolerances within many of these elements. A specific requirement of PAS2070 is the definition of a single annual period in the setting of the temporal boundaries of the study (BSI, 2013). Boundaries must be set with consideration to the specific case study, so as not to curtail or truncate data. Additionally, the (necessary) imposition of a temporal boundary forces careful consideration of emissions sources (e.g. landfill emissions) that occur over far greater time scales.

The PAS2070 modelling process is potentially subject to a range of uncertainties and errors. The system of study is complex and will include non-deterministic features. Non-deterministic features refer to the response of the system that is not predicable because of uncertainty within the system itself. Qualification and assessment of these uncertainties is important for both model validation and reliability. Within this study a Monte Carlo approach is applied to assess the combined effect of uncertainty in input variables.

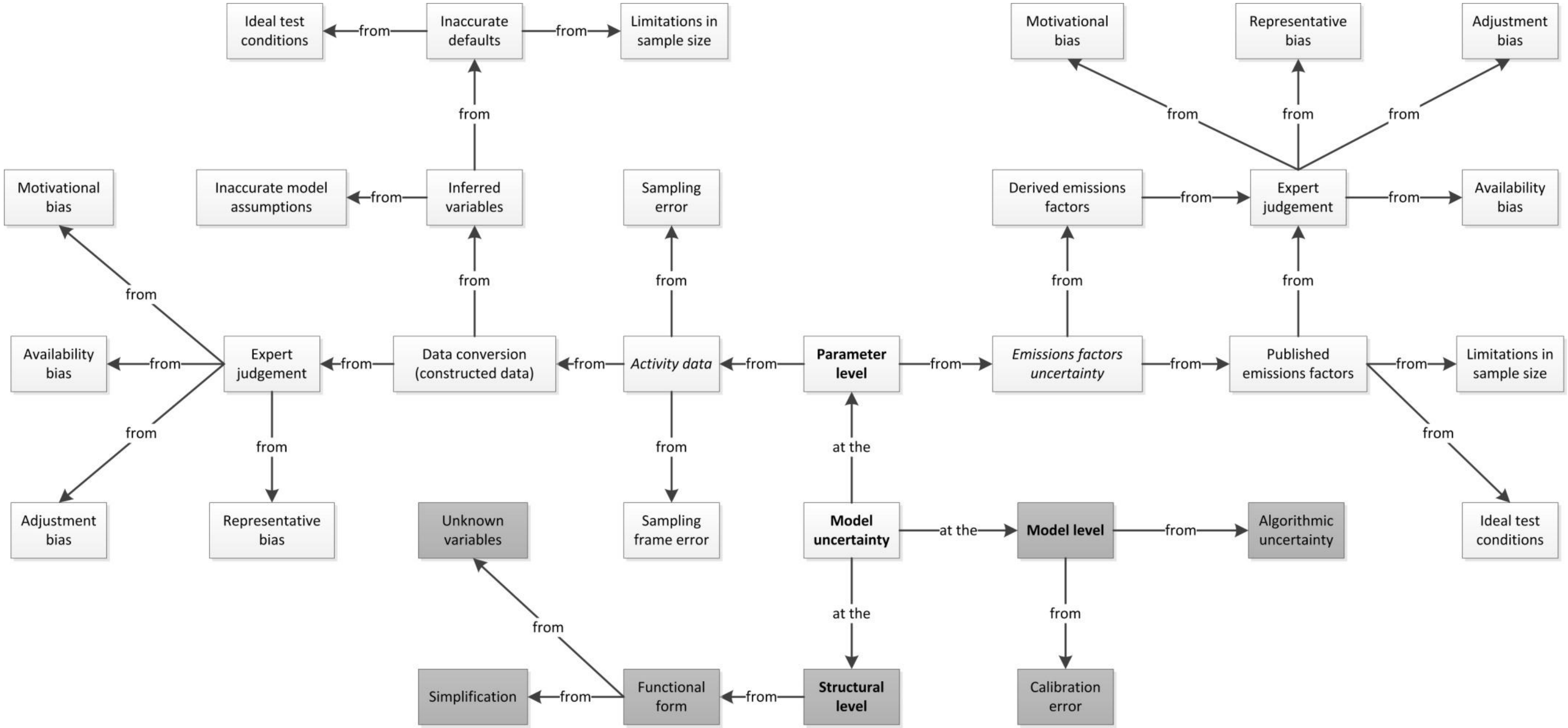

Figure 7. Exploration of the uncertainty sources associated with a PAS2070 GHG inventory (adapted from Hughes *et al*, 2013). Sources in white are considered in this analysis.

Results of OAT Sensitivity Analysis

Table 10 shows the normalised sensitivity coefficients for the top 33 ($S_i \geq 0.01$) most sensitive variables. Sources of sensitivity identified can be broadly categorised into supply chain effects; transport emissions; power generation; and waste disposal. A number of input parameters exhibit a non-linear correlation with total city emissions, suggesting the normalised sensitivity coefficient alone does not provide an accurate representation of the effect of the individual parameters. Further to this supposition, the additive variation suggested from the OAT does not align with actual total variation. If all perturbations are implemented simultaneously (i.e. increase of 1% in all variables) total ktCO_2e at the city level varies by +1.02%. This differs from the predicted variation achieved through addition of individual variable effects, +1.76%. Demonstrating the combined effect of multiple input parameters cannot be accurately predicted through addition of individual effects.

Table 10. Results of OAT sensitivity analysis and test for linearity for top 33 ($S_i \geq 0.01$) sensitive parameters

Input variable	x	Δx	**Δ Total ktCO_2e**	**% variation**	S_i	r
Supply chain CO_2/£ (scaling factor)	1.00	1.01	12.51	0.33	0.33	1.00
HHFC (scaling factor)	1.00	1.01	10.94	0.29	0.29	1.00
Carbon dioxide GWP	1.00	1.01	9.15	0.24	0.24	1.00
Road transport average Speed	79.38	80.17	7.14	0.19	0.19	0.99
Road transport km travelled	1155878603.52	1167437389.55	4.08	0.11	0.11	0.99
HHFC food (scaling factor)	269.97	272.67	3.31	0.09	0.09	0.98
Coal generation installed capacity	23.07	23.30	2.57	0.07	0.07	0.94
Coal generation load factor	0.45	0.45	2.57	0.07	0.07	0.94
Coal Power stations CO_2 emission factor	2251.33	2273.85	2.57	0.07	0.07	0.94
Supply chain CH_4/£ (scaling factor)	1.00	1.01	1.97	0.05	0.05	0.98
Local Com/Ind electricity consumption	715.78	722.94	1.79	0.05	0.05	0.96
Road transport factor bCO_2 (scaling factor)	1.00	1.01	1.73	0.05	0.05	0.96
Supply chain CO_2/£ food	0.55	0.55	1.48	0.04	0.04	0.93
Road transport factor kCO_2 (scaling factor)	1.00	1.01	1.40	0.04	0.04	0.92
Aviation spirit N_2O emission factor	0.10	0.10	1.32	0.03	0.03	0.90
Aviation 'a' factor Climb Cruise Descent (3000ft)	1.00	1.01	1.12	0.03	0.03	0.84

Input variable	x	Δx	Δ Total $ktCO_2e$	% variation	S_i	r
Supply chain CH_4/£ food	0.38	0.38	1.12	0.03	0.03	0.84
Local Domestic electricity consumption	428.22	432.50	1.07	0.03	0.03	0.82
Supply chain N_2O/£ (scaling factor)	1.00	1.01	0.93	0.02	0.02	0.99
Supply chain N_2O/£ food	0.29	0.29	0.85	0.02	0.02	0.81
Total waste collected	112667.00	113793.67	0.54	0.01	0.01	0.07
Net CV (scaling factor)	1.00	1.01	0.52	0.01	0.01	-0.04
Incineration ash transfer coefficient	0.10	0.10	0.52	0.01	0.01	-0.04
Waste bio-carbon content (scaling factor)	1.00	1.01	0.52	0.01	0.01	-0.04
Waste prop. routed to incineration	0.40	0.40	0.49	0.01	0.01	-0.14
Road transport factor aCO_2 (scaling factor)	1.00	1.01	0.33	0.01	0.01	1.00
NIPSH expenditure (scaling factor)	1.00	1.01	0.32	0.01	0.01	1.00
Supply chain HFCs/£ (scaling factor)	1.00	1.01	0.30	0.01	0.01	1.00
Road transport factor dCO_2 (scaling factor)	1.00	1.01	0.29	0.01	0.01	1.00
HHFC construction (scaling factor)	24.02	24.26	0.28	0.01	0.01	1.00
Oil generation installed capacity	3.64	3.67	0.27	0.01	0.01	1.00
Oil generation load factor	0.39	0.40	0.27	0.01	0.01	1.00
Oil CO_2 emission factor	2190.83	2212.74	0.27	0.01	0.01	1.00

Results of Uncertainty Analysis

The OAT analysis considers the effect of input variable perturbation in isolation and cannot fully explore the input space. Neither can the OAT analysis provide an estimate of the combined effect of input variable uncertainties. A Monte Carlo analysis was performed using random sampling of input variables, based on defined uncertainty probability distributions. The output from each individual model run is a single estimate of total city GHG emissions. Therefore the output from the multiple combined runs represents the potential distribution of modelled total emissions. The analysis consisted of 5,000 model runs, completed for the model as a whole and for three of the broad category areas identified in the OAT transport emissions; power generation; and waste disposal. Supply chain emissions uncertainty is excluded due a need for further investigation and modelling.

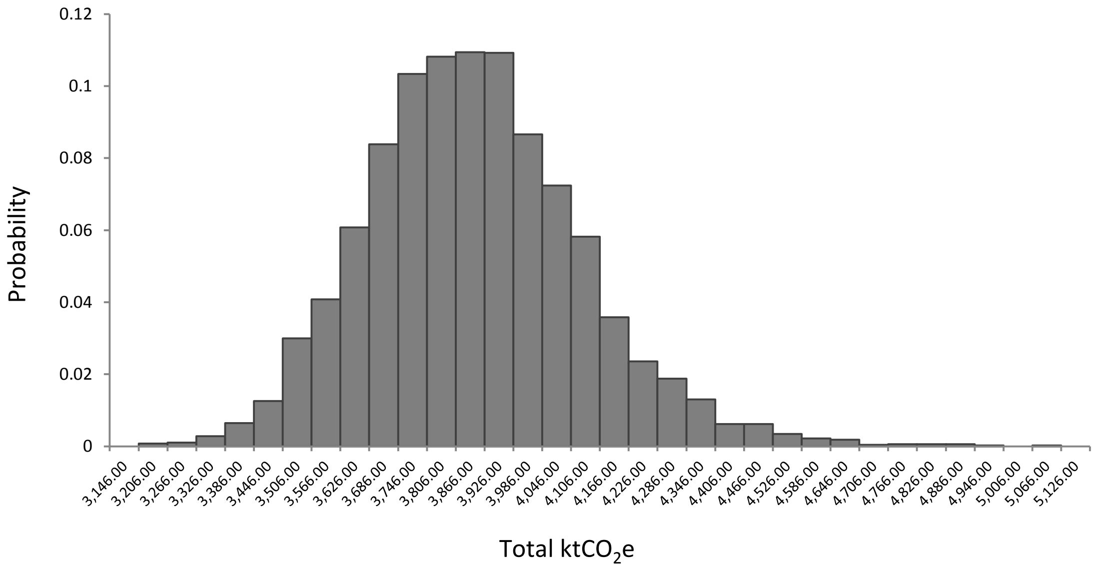

Figure 8. Estimates of total ktCO2e emissions from DPSC methodology for Southampton, UK, 2008 using 5000 model runs

The mean total emissions for the whole model (all variables) for the simulated period (2008) are 3845$ktCO_2e$ (Figure 11), with a standard deviation of 225$ktCO_2e$. Compared with the 'measured' figure of 3957$ktCO_2e$, this represents an underestimate of 112$ktCO_2e$ (2.83%). The results present a 95% confidence interval 95% (i.e. p = 0.05) of ± 6.25.

Further to the exploration of the model as a whole, the areas identified in the OAT transport emissions; power generation; and waste disposal were explored. Estimates of uncertainty associated with energy generation, based on 5,000 model runs against energy input parameters, suggest a 95% confidence of ± 0.42kt. Transport simulations suggest a higher level of uncertainty of ± 3.86kt at the 95% confidence interval. .Within waste whilst input variable uncertainty is high, the relative contribution of the sector to total emissions infers a low variability at the city level ± 1.51kt.

Discussion

A number of types and sources of uncertainty have been identified in the model, representing uncertainties and possible error at the structural, parameter, and input levels. A Monte Carlo simulation on the whole and on sections of, the model was performed to estimate the magnitude and nature of these uncertainties. It is evident that individual sectorial models cannot be taken as reprehensive of wider model uncertainty. Additionally if a parameter value is well known (e.g. electricity) (low input uncertainty) it will not have significant impact on the model even if the model has a high sensitivity to that parameter (Hughes *et al,* 2013).

Further investigation of and modelling of identified uncertainty to enable distinction and separate modelling for aleatory and epistemic uncertainty is important to develop understanding. Whilst a cursory examination of this distinction is made in this study, formal recognition will allow development of mathematic models for each – probabilistic representations for aleatory uncertainty, and other information theories for epistemic uncertainty. One possible method in the estimation of epistemic uncertainties is Bayesian probability (Oberkampf, *et al*, 2002). A winding stairs approach to sensitivity analysis could be implemented to overcome the localised nature of the OAT analysis (Chan *et al*, 2000). Furthermore a variance-based sensitivity could be implemented to develop sensitivity indices to quantify the impact of individual inputs in the context of input multiple changes (Saltelli & Annoni, 2010). The analysis of uncertainty associated with the PAS2070 emissions model attempts to capture all possible variations of underlying data and calculation procedures relevant to the DPSC (non-IO) component. For the majority of data a number of stochastic variations are considered, however there may also be sources of systematic errors not captured using the techniques presented. Including structural change of the make-up of the underlying case study which cannot be captured due to lack of time series data.

The PAS2070 model requires and is likely to be used to inform predictions beyond the range of temporally or spatially available observations, the potential for structural uncertainty to be a major source of model uncertainty is high (Refsgaard *et al,* 2007). In the case of the PAS2070 model the large spatial scale of the model will often prohibit the development to actual measurements for the purpose of calibration. Some calibration has previously been achieved through observation, notably in the waste emissions modelling algorithms (e.g. landfill gas production (Kirkeby *et al*, 2007)). The uncertainty from these observations can be captured in part, through variation in input uncertainty. However this fails to capture the complete modelling process. A possible solution would be the use of expert elicitation – a subjective process to elicit judgment and opinion – from experts as a supporting method to the uncertainty analysis (Refsgaard *et al*, 2007). This is possibly achievable through the use of Delphi type techniques, whereby a panel of experts is asked to provide answers and comment in a series of two or more rounds, with a summary of responses provided for revision of thought in the following rounds.

The uncertainty analysis undertaken in this paper specifically excludes the model uncertainties associated with the consumption based component, due to the inherent complexities (Lenzen *et al*, 2010). It should, however, serve as a basis for further assessment of uncertainties, e.g. with improved uncertainty data or temporal series data.

Conclusions

The OAT sensitivity and Monte Carlo analysis undertaken was designed to provide insight into the model sensitivity and uncertainty. Assessment of sensitivity and uncertainty in these types of life-cycle study are often not done, due to perceived high resource constraints and methodological complexity. An OAT sensitivity analysis was conducted on a PAS2070 DPSC simulation model for Southampton 2008. The analysis indicates that supply chain effects; road transport; power generation; and waste disposal are by far the most sensitive parameters. Consequently, determining accurate estimates for these key parameters is likely to offer the optimum balance of investment and return in developing accurate models for PAS2070 assessments and local government emissions reductions. The model responds in a non-linear manner to a number of input variables and variation in input variables is generally not additive, suggesting the normalised sensitivity coefficient alone does not provide an accurate representation of the effect of the individual parameters.

To explore the uncertainties associated with the model a Monte Carlo simulation (5000 runs) was constructed. The uncertainty analysis investigates the nature and magnitude of the uncertainties in the modelling process and the subsequent effects simultaneously. The model suffers from uncertainties at the structural, parameter and input levels. The key sources of uncertainty in model output are input data; parameter values, and model structure (i.e. the conceptual model). The simulation suggests a high confidence in the model outcome. The model is subject to a range of uncertainties in both the input and computational stages.

PAS2070 has made an important contribution to the standardisation and realisation of a comprehensive city GHG inventory. However, users of PAS2070 models must realise and appreciate the limitations of such models, only then can informed judgments and decisions be made on the results.

The effort required for a comprehensive uncertainty analysis of this type is considerable, the alternative however, is to provide decision makers with incomplete information. At best this will lead to a false sense of reliability, at worse incorrect assumptions and decision making. We strongly recommend that as more studies become available continuous effort to identify and improve uncertainty be applied;

leading to a better communication of information to policy makers and a better underpinning of their decisions.

6. Specification for a tool for the assessment of sub-national community carbon footprints

Introduction

This chapter reviews the methods required for the attainment of PAS2070 and describes the requirements of these methods in a database context. A general description of the database model is provided, and future developments are discussed, to meet Aim/Objective 3c of this research programme. This work was developed by the author and written for this PhD thesis under normal PhD supervision conditions.

International standard PAS2070 'Specification for the assessment of greenhouse gas emissions of a city' has developed through international collaboration to provide a framework for the measurement of city GHG emissions. As GHG management becomes more integral to sub-national community governance quantitative tools are required for the assessment of community GHG inventories. The primary users of the PAS are likely to be policy decision makers in municipal government. The application of the methods developed in the previous chapters of this thesis for the attainment of PAS2070 are likely to be remain academic, unless suitable tools are developed to enable implementation. This chapter describes a proposal for the development of a database driven software tool for the attainment of the PAS2070 standard.

Specification for a tool for the assessment of sub-national community carbon footprints

Abstract

As GHG management becomes more integral to sub-national community governance quantitative tools are required for the assessment of community GHG inventories and mitigation measures. PAS2070 'Specification for the assessment of greenhouse gas emissions of a city' has emerged through international collaboration to offer a harmonised standard for the GHG inventorying of cities. This paper proposes a specification for a decision support, database driven tool to model GHG inventories and scenarios at the community, regional or wider levels in accordance with PAS2070. The tool comprises of two methodological approaches in accordance with PAS2070 requirements – a Direct Plus Supply Chain (DPSC), and a Consumption Based (CB) approach. The proposed database tool consists of a series of input models, an emissions factor database relevant to each of the two PAS2070 specification methods, and a reporting module. A number of sub-models (modules) are proposed, each modeling a separate community process required to provide input data to the primary emissions model. These modules combine to provide input data for the reporting of a complete PAS2070 GHG inventory for the chosen community. The model is described through a general description of the emissions factor database and modelling process and a more detailed exploration of the input methods required by the two PAS2070 specified approaches. The specification described successfully addresses a number of challenges in the delivery of the tools accomplish standardised GHG inventorying of communities.

Introduction

Interest in the management of GHG emissions from communities – regions, towns, cities – is increasing in the academic community, government, and among policy decision makers. The concentration of population, consumption activities, and production processes has firmly positioned GHG management as an essential component of local community governance (Kennedy *et al*, 2011; Wright *et al*, 2012). The increasing interest in the governmental and academic sectors has led to concerted effort for the establishment of methods for the inventorying of community GHGs (Ramaswami *et al*, 2008; Kennedy *et al*, 2009; Kennedy *et al*, 2011; Wright *et al* , 2011a; Wright *et al*, 2012). Publically Available Standard (PAS) 2070 'Specification for the assessment of greenhouse gas emissions of a city' has been developed through international cooperation to offer a harmonised standard for the assessment of city GHG emissions (BSI, 2013).

Previously, development of city GHG inventory methods was based upon the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) or the Greenhouse Gas protocol developed for corporate accounting (Ranganathan *et al*, 2004). However, cities have markedly different boundary conditions and process flows to organisations and nations, the original purpose of these guidelines. As a result assessments of community GHGs commonly focused on the geopolitical boundary of the community (usually defined by the political extent of sub-national local government) to form the assessment boundary. Effectively accounting all production based emissions within the geopolitical boundary of the community – a production based assessment (PB) (Williams *et al*, 2012). The bottom-up PB approach enables detailed assessment of specific local conditions and processes, but fails to capture supply-chain or other linked processes outside the geopolitical boundary. This assessment type fails to consider the nature of communities as producers and consumers of resources, and the role these have in cross-boundary flows of materials, services, and emissions (Baynes & Weidmann, 2012). Alternatively, in recognition of this issue consumption based (CB) methodologies can be applied, setting the boundary economically, opposed to geographically. CB methods commonly utilise environmentally extended input-output (EEIO) models, which set the economy as the boundary, considering the flow of materials, goods and services across a theoretically unlimited geographic area (Minx *et al*., 2009). The nature of CB assessment necessities the use of national or international datasets, often pro-rata scaled for use at the local level (Wright *et al*, 2011). As a result the effectiveness of CB assessments are often limited once modelling is attempted below the meso scale.

Recognising the strengths and weaknesses of PB ad CB accounting, PAS2070 suggests the use of a dual approach (reported separately to avoid double counting) to a GHG inventories – a consumption based (CB) and a transboundary approach, termed ‘Direct Plus Supply Chain’ (DPSC) (a PB assessment with additional elements to capture supply-chain emissions). Chapter 4 of this thesis develops the methods required for the calculation of the transboundary methods, through the use of the case study of Southampton, UK. This paper considers the methods proposed and describes a specification for the development of a data driven tool for use by the academic, governmental, business and public sectors. For the purpose of context setting an introduction to the wider project goal and scope is provided. The general structure of PAS2070 and the data model are described. Subsequently we explore the two aspects of the specification – DPSC and CB –briefly introduce the underlying theory and the methods proposed in chapter 4 of this thesis in order to describe the detailed structure of the various data modules used in the model. Following the preliminary specification of the database, we conclude with a brief assessment of possible advancements.

Goal, scope and background of the project

Researchers from the Carbon Management Group at the University of Southampton in partnership with Southampton City Council commenced a project in 2009 to develop a methodology for the carbon footprinting of a city or other similar sub-national community. Southampton, the largest city in Hampshire, England, was chosen as a case study and 'proof-of-concept'. The city was chosen as a suitable due to its role as a commerce hub; a major international cruise terminal, and the UK's second largest container port. Additionally Southampton Airport is located a short distance outside the geopolitical boundary in neighboring Eastliegh. Southampton presented the opportunity to develop a case study with methods for the inventorying of a range of components found in many other cities – helping develop and test a consistent, transferable methodology.

The release of PAS2070 has provided the definition for the specification of a sub-national community (BSI, 2013). The previous chapter provides the specification for methods to obtain the input data and calculate emissions in accordance with the standard using the Southampton case study. Consistent, coherent and transparent datasets, calculations and reporting are needed to make it easier to perform assessment studies and increase the credibility and acceptance of the results. Quality and user-friendliness are the perquisites for the establishment of a reliable tool for the assessment of community GHG emissions.

General description of the proposed database model

The dual methodological approach (DPSC and CB approaches) specified by PAS2070 necessities a parallel calculation and report in the compilation of a compliant GHG inventory. The CB methods specified by the PAS2070 require the use of an environmentally extended input-output analysis (EEIO) (BSI, 2013). The DPSC methodology is essentially an transboundary based approach to emission accounting. Traditional territorial based accounts inventory community emissions based on the process emissions occurring within the geopolitical boundary of the city. The DPSC extends this territorial approach through the inclusion of significant transboundary process-linked emissions (e.g. grid-connected electricity, construction materials). Input from a CB method in the compilation of the supply chain emissions inventory is required. As a result there are significant cross-over between the two methods.

Territorial, production based accounts include emissions associated with exports at the point of production, whereas consumption based allocate these emissions to final demand. Effectively the production emissions are captured by both methods, but allocated on a different basis.

The results of the two methods must be reported separately to prevent double counting. This necessitates a parallel calculation in the database model. Due to the cross-over in calculation and data requirement the database operates a 'quasi-dual' approach to the modelling process, with separate reports for each method. The database model (figure 7) operates on the basis of an emission factor database, subdivided into two databases (with a further subdivision required for transport emissions in the DPSC approach, due to the polynomial nature of emissions factors) relevant to each method – process and consumption – with a series of input models. Process emissions factors with activity module inputs, and an EEIO input model, for a per unit expenditure consumption emissions factors. The approach enables an output for both methods and a subsequent PAS2070 compliant report. The model provides full flexibility to redefine data at any level in a scenario. Changes can be made at any level within the scenario, modifying the default supplied data, allowing the user to define new emissions factors and activities within the scenario. The model has been structured so that it can use data specific to an actual case study, thereby improving the model applicability and accuracy. In order to enable the use of the model at an early stage, where data may be limited, default data are provided. Figure 6 illustrates the general structure and significant inputs/outputs of the model.

The emissions factor database contains a series of per unit activity and per unit expenditure emissions factors, relevant to activities identified as pertinent to the wider model, for example tCO_2e per unit of Gas Oil combusted. All emissions factors are taken from peer-reviewed literature, recognised databases, calculated from basic principles, or input models (e.g. EEIO model) (Williams *et al*, 2012). The process activity based element of the database can be updated with new processes, fuels and factors enabling modification by the user to better reflect local values, or accommodate new variables and processes. The transport section of the EFDB is held separately. Transport emissions factors account for technology, speed and drive cycle and are taken from the ARTEMIS methodology, and are polynomial in nature (Boulter and Mccrae, 2007). Whilst it is proposed that the user have the ability to amend these emissions factors, the reality is that alternative emission factors will likely be unavailable.

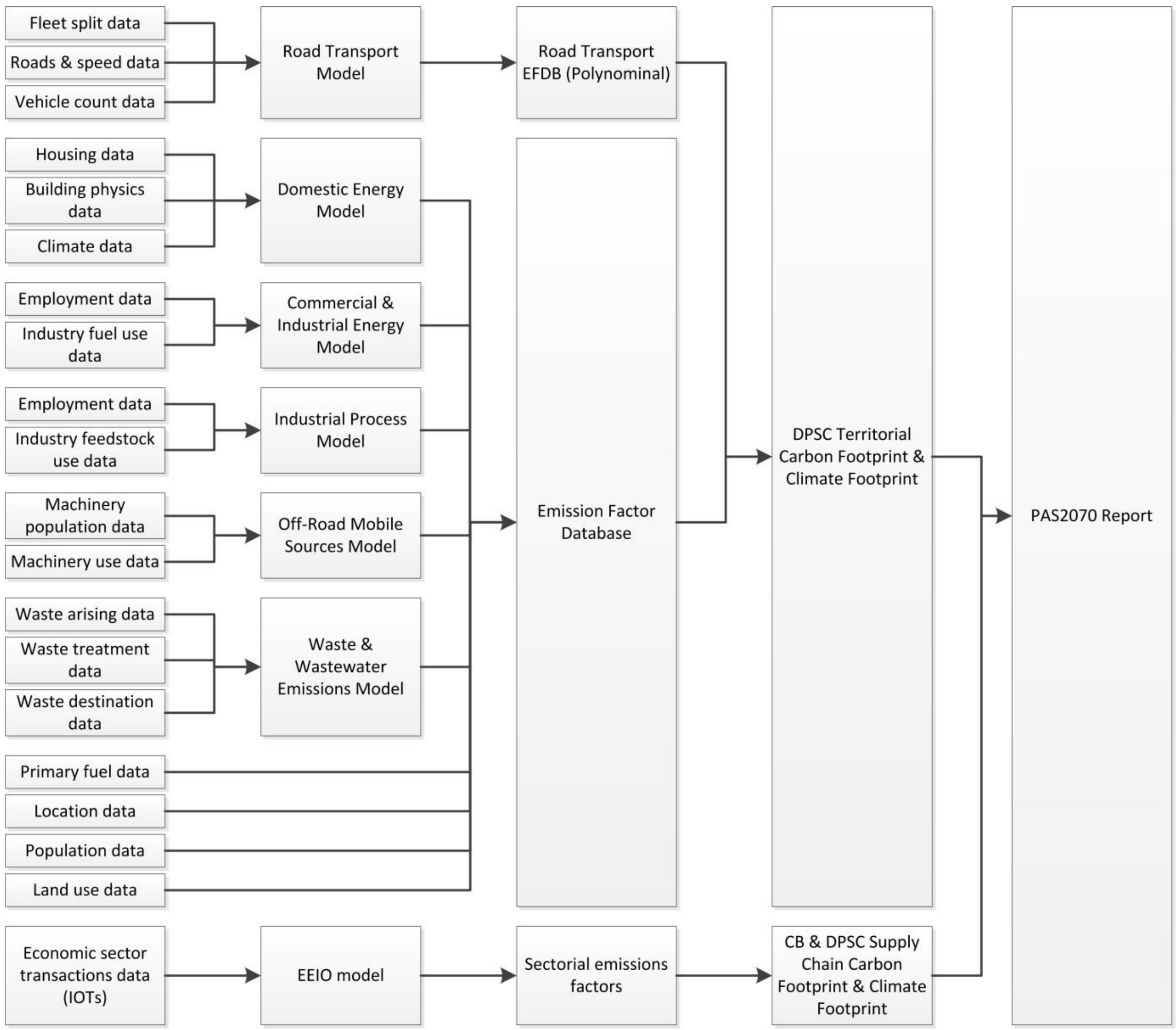

Figure 9. Significant inputs, processes, and outputs of the model

The emissions factors relevant to the CB and supply-chain section of the DPSC methodology are additionally presented separately from the process EFDB. The CB emissions factors are calculated using an EEIO input model, the proposed model structure is based on UK default data, however the basic structure of the model is applicable outside the UK. The input model is proposed to be made available to the user to calculate new up-to-date emissions factors as required.

Functional unit

The model is proposed to operate on a functional unit of a given amount (mass) of carbon dioxide equivalents (tCO_2e). CO_2e are calculated using Global Warming Potential – the equivalent mass of CO_2 to achieve the same warming effect of a given GHG over a given timescale – over a 100 year time horizon (GWP100) (IPCC, 2006). Total mass of

CO_2e can be normalised for comparison and metric development for a given geopolitical area, unit of activity, population, or unit of expenditure.

Input variables are developed to use a common unitisation to avoid errors and confusion with user input and to ensure consistency in the use of emissions factors (e.g. mass inputs required in metric tons). All energy and fuels are to be converted to energy units (based on calorific value (CV)) (GJ), to ensure consistency in mass comparison (e.g. coal from Wales has a different chemical composition and CV to coal from China). Default calorific values for the input fuel types are provided for the user, in addition to a data conversion model to provide consistent conversion. The user is able to update the default calorific values if required.

Boundaries

The model boundaries are to be developed to cover the two required approaches of PAS2070. The boundaries are developed temporally, spatially and economically in the case of the CB assessment. Temporal boundaries are set in accordance with PAS2070 requirements for annual period defined by a 12 month consecutive period of time (BSI, 2013). Default spatial boundaries are defined by the geo-political boundary of the community government involved in the study. Users can define alternative spatial boundaries if required, however this will not necessarily be PAS2070 compliant, therefore must be highlighted to the user at the reporting and data entry stages.

Reporting

PAS2070 presents stringent minimum reporting standards that must be met in fulfilment of the standard (Table 3. PAS2070, 2013). Due to the dual methodological nature of the standard two separate reports – DPSC and CB – are necessary to prevent double counting. PAS2070 specifies minimum standards for reporting of both methods. The database model is proposed to include the two options for PAS2070 as specified, with additional further reporting to enable sector (e.g. residential dwelling) or geographic (e.g. to super-output area[1]) reporting (where required GIS data is available). For example data are commonly available to point source (e.g. grid electricity at meter point) this enables the modelling of emissions to sub community boroughs.

[1] UK statistically significant population area.

Limitations

The model limitations arise from uncertainties within the model. Uncertainties arise from two sources within the model: software programming errors (coding mathematical equations errors) and data uncertainties (collections and modelling). Software errors can be further disaggregated by coding errors – compilation; run-time; logic – and arithmetic errors.

Compilation errors are captured by the software complier, as they would otherwise prevent the program build. Run-time errors occur when the program is run, typically when the program attempts an impossible task (e.g. division by zero). These errors are often captured by the compiler or during the software testing phase. Logic errors cause the software to operate incorrectly, but not so as to cause abnormal termination, and arithmetic errors in mathematical code are higher risk as there is no method of automatic identification. This risk is especially prevalent during the early stages of development – the solution is through, logical and complete checking of all calculations. However the risk remains that mathematical error will only be discovered through use of the model (Bhander *et al*, 2008).

Qualification of these uncertainties has been discussed in detail in chapter 5, which considered the sources of error within the model. The study quantified the sensitivity – the impact of variation in independent on dependent variables – and the uncertainty, due to a range of conditions within the model. In summary errors occur at the model level, structural level or parameter level. Uncertainties may occur in the modelling process for the following reasons:

- Model level

Errors may result from uncertainty associated with arithmetic processes; inaccuracies in measurement; unrepresentative technologies.

- Structural level

Uncertainty may result from over simplification of modelled processes, or simply as a result of the non-inclusion of an unknown variable.

- Parameter level

Incorrect assumptions, through simplification, expert judgment or otherwise can present issues of uncertainty. Missing data is a particular area of concern, and a common problem (Bhander *et al*, 2008). Data may not exist to reflect specific local conditions or technologies, for example local vehicle fleet statistics resulting in the use of national fleet statistics as a proxy for local conditions. Missing data uncertainties can become particularly problematic if not recognised and noted in the final report, potentially resulting in erroneous interpretation and recommendations.

DPSC Assessment

The DPSC approach specification encompasses direct process emissions from activities within the city and indirect emissions from the consumption of grid-connected electricity, heating/cooling, transboundary travel, and key materials in supply chains (Chavez and Ramaswami, 2013).The DPSC methods specified by PAS2070 are an extended territorial based approach. Traditional territorial methods are extended to include emissions from transboundary processes and supply chains. Wright *et al* (in preparation) develop methods for the calculation of emissions related to the DPSC specification, using the case study of Southampton, UK. The key relationship in these methods is the concept of emissions factors – per unit activity emission.

$$e_i = ef_{ij}A_{\mathrm{j}}$$

Equation 11. Per unit activity emission factor. Where e_i = emission of GHG *i*, ef_i = emission factor GHG *i* activity *j*, A_{j} = rate of activity *j*

This relationship is sufficient for the calculation of a significant proportion of activity emissions. All emissions factors detailed in the database are to be sourced as per the methods detailed by Williams *et* al (2012). However, some further calculations are required to obtain the data necessary for the calculation of emissions and emissions factors. For example calculation of input data and emissions from solid fuel combustion in residential dwellings utilises a process based energy demand model (see chapter 4). Accounting procedures for AFOLU require input from complex 3rd party vegetation and climate interaction models (Adapted Rothampsted soil carbon model (RothC-26.3) and the Lund-Potsdam-Jena Dynamic Global Vegetation Model (LPJ-DGVM) (Coleman and Jenkinson, 2008; Smith *et al*, 2001; Sitch *et al*, 2003)). Due to this complexity and third party nature it is proposed that the model enables the user to input annual stored carbon budgets from AFOLU and includes reference to the procedure for the calculation of this data.

Additionally transport is calculated separately within the database – both on the input, and emissions factor level. The transport emissions models require the use of complex polynomial equations relating engine factors and driving factors to fuel consumption and emissions (Boulter and Mccrae, 2007)). Opposed to the use of traditional LCA type methods, which would provide fixed emissions factors per unit consumption (or distance travelled). The use of these methods provides a more detailed and accurate assessment of the associated GHG emissions.

Residential dwellings

The calculation of emissions associated with residential property requires a two phase implementation. The model calculates emissions on the basis of fuel combusted or otherwise consumed. This requires input data regarding the amount of fuel consumed in domestic property. This necessitates the calculation of the energy requirement of the residential stock in the area of study. A method that presents a number of issues, namely the large number of individual properties that may be present in the community of study. The solution to this problem is a separate model that considers every dwelling in the community, through assignment of an average performance category (insulation, glazing property, etc) based on age and build type. The model is constructed using the Building Research Establishment BREDEM-12 model – a physics based building energy model (Anderson *et al*, 2002). Primarily designed for UK properties the modular nature of the model enables extension or compatibility with non-UK communities, provided input data are updated to reflect local variables (Ward, 2012).

Dwellings are assigned a performance category based on age, regulatory standards, physical properties, and behavioural assumptions (368 physical, 9 behavioural categories). The input data are divided to modules accounting for elements required by the modelling process, including, location physical determinants (climate and weather data); building physical data (e.g. heat transmission coefficients); physical behavioural detriments (e.g. heat and hot water demand). The input data can be combined to form dwelling archetypes to reflect the input data required by the user. Individual parameters are proposed to be updatable to enable the user to define new technologies (e.g. heating technologies, insulation technologies).

This input system translates in to the database as a series of input models presented to the user. A series of UK default values are proposed to be included, the user can create additional new input data to reflect local conditions or new technologies

Road Transport

The transport section of the model requires data regarding fuel consumption within the target community and as a result of commuter demand activity. This data can then be used to calculate emissions either in a linear fashion using traditional emission factors, or more accurately using polynomial emissions factors (Boulter and Mccrae, 2007). Emissions of CO_2 from road transport are calculated on the balance of fuel consumed by vehicles in the study area. Emissions of other non-CO_2 GHG emissions are dependent on a variety of variables, including vehicle after treatment technology, fuel type, and drive-cycle. Polynomial emissions factors are used to account for the variation in these technologies and driving styles.

Vehicles are categorised by type (e.g. passenger car; light goods vehicle (LGV)), fuel (e.g. petrol; diesel), and installed technologies (e.g. catalytic convertors) – defined by progressive legislative standards (e.g. Directive 98/69/EC; Regulation 715/2007/EC). Input data are designed to reflect this, the user can specify fleet compositions at either the local or national level, and this depends on availability of data. Technology standards can be specified but should represent a stepped change in the emissions regulation of vehicles, to enable comparison within fleet composition.

CB Assessment

Input-output model

The consumption based approach necessities the use of an economic input-output model to examine transactions between industries and sectors within the economy. Whilst IO has been explored extensively in the literature it is useful to consider the structure of the model here to understand the structure of the database model. The downscaling of expenditure data to the local level is explored in chapter 4.

The core element of an input-output model is a matrix concerning flows through a national economy. The economic activity is divided into various homogeneous (i.e. single product) producing sectors (e.g. steel production). The key component of the matrix is then sales and purchases from an industrial sector (a producer), to other sectors and the sector itself (consumers) (Miller and Blair, 2009). The rows of the table describe the distribution of a sectors output across an economy, the columns describe the inputs required by each sector to produce its output. The model is based on a series of *n* linear equations, each denoting the observed monetary value of the flow from sector *i* to sector *j* by z_{ji}. The basic model assumes linear production (i.e. proportionality of inputs and outputs which precludes effects economies of scale). Demand by sector *j* from other sectors is linked to the quantities of goods produced by sector *j* (Miller and Blair, 2009). Additionally there are sales of products to purchasers (households, government, exports) external to the industrial producers, generally known as *Y*. The demands of these purchasers are normally not linked to sector quantities being produced (Miller and Blair, 2009). It is important to note that even where there is no external product demand, there may still be production due to inter and intra sector demand for an outputted product (e.g. steel to manufacture cars). Therefore total output of sector *i,* denoted by X_i, is defined as:

$$X_i = z_{i1} + z_{i2} \dots + z_{in} + Y_i$$

Equation 11. total output of sector *i* in economy of *n* sectors

In an economy of *n* sectors there will be a set of *n* linear equations representing each of the *n* sectors.

Each sector additionally makes a series of payments for employee compensation (i.e. wage costs) and various other value added items (e.g. taxes, capital (interest payments, rental). These payments are normally combined to form a ‘payments sector’, denoted as V_i (Miller and Blair, 2009). Combining the various interindustry flows, final demand, and value added, as previously mentioned a tabular output (input-output table) can be formed (table 12).

Table 12 Example basic input-output table (note database representation of from/to table, payments, and final demands tables) (Miller and Blair, 2009)

		Purchasing sector				Final Demand	Total Output
		1	2	…	*n*		
Processing sector	1	z_{11}	z_{12}		z_{ni}	Y_1	X_1
	2	z_{12}	z_{22}		z_{ni}	Y_2	X_2
	…						
	n	z_{in}	z_{in}		z_{nn}	Y_n	X_n
Payments Sectors	Value Added	V_1	V_2		V_n		V
Total Outlay		X_1	X_2		X_n		X

The IO model assumes that flows of sector *i* to *j* depend on the total output of sector *j*. Therefore a technical coefficient, a_{ij}, defining a fixed relationship between a sector and its output, can be derived through division of inter-sectorial flows from *i* to *j* (z_{ij}) with the total output of *j* (*Xi*), equation 12.

$$a_{ij} = z_{ij}/X_i$$

Equation 12 derivation of technical coefficient a_{ij}

Therefore accepting the fixed technical coefficient, equation x.x. can be rewritten as:

$$X_i = a_{ij}X_i + a_{i2}X_2 \ldots + a_{in}X_n + Y_i$$

Equation 13 application of technical coefficient a_{ij} to compute total output of sector *i* in economy of *n* sectors

These equations demonstrate the dependence of inter-industry flows on the total output of each sector. If all *X* terms are then moved to the left, equation 14,

$$X_i - a_{ij}X_i - a_{i2}X_2 \ldots - a_{in}X_n = Y_i$$

Equation 14

and, grouping the X_1 together in the first sector equation, the X_2 in the second and so on, equation 15,

$$\left(1 - a_{ij}\right) \quad - a_{ij}X_i - a_{i2}X_2 \ldots - a_{in}X_n = Y_i$$

Equation 15

the relationships can be represented in matrix notation as, equation 16:

$$A = \begin{bmatrix} a_{11} & a_{12} & a_{2n} \\ a_{21} & a_{22} & a_{2n} \\ a_{n1} & a_{n2} & a_{nn} \end{bmatrix}, X = \begin{bmatrix} X_1 \\ X_2 \\ X_n \end{bmatrix}, Y = \begin{bmatrix} Y_1 \\ Y_2 \\ Y_n \end{bmatrix}$$

Equation 16 coefficients represented in matrix notation

If *I* is considered as the *n* by *n* identity matrix, then (*I-A*) will represent (1–a_{11})…(1–a_{nn}), thus equation 16 can be written in matrix notation as, equation 17:

$$(I - A)X = Y$$

Equation 17 inter-industry flows as matrix notation

$(1\text{–A})^{-1}$ is termed the Leontief inverse matrix and represents the production of each sector required to meet final demand in the economy, representing the dependence of each of the outputs on each of the final demands (Miller and Blair, 2009).

$$X = (I - A)^{-1}$$

Equation 18 Calculation of the Leontief inverse matrix

Extension to GHG emissions

The combination of economic input-output and environmental information is based on the work of Leontief (1986), who first suggested the extension of traditional economic data with environmental data. Leontief extended the input-output model to analyse environmental issues by adding rows or columns corresponding to new input or outputs originating from production (Yu *et al*, 2007: Leontief 1986). Effectively this creates an 'environment' sector, and the value of each item represents the 'output' of pollution.

GHG emissions are treated as a primary input to economic flows, therefore a GHG coefficient (f_j) is calculated by dividing the total GHG consumed (emitted) by sector *j* by total output X_i. To calculate both direct and indirect GHG emissions, GHG multipliers are created through multiplication of direct GHG coefficients with the Leontief inverse $(I\text{-}A)^{-1}$.

It is important to note the model uses a domestic technology assumption, whereby imports are assumed to have the same GHG intensity as domestic equivalents. It assumes the energy structure and economic structure of the imports can be approximated based on the domestic make-up of the UK. This may be a valid assumption for some regions, but underestimates GHG intensities of imports from emerging and developing regions (Druckman and Jackson, 2009).

Conclusions

The advent of PAS2070 demonstrates that the development of large-scale sub-national community GHG inventories are possible and in-demand. The proposed database model and preceding project work recognises this demand and demonstrates the ability to develop the methods and tools required for to achieve high-quality reliable, consistent and transferable inventories. PAS2070 requires the application of two approaches – DPSC and CB. The nature of the dual approach in the standard necessities a 'quasi-dual' approach to modelling in the database specification. The CB approach requires the utilisation of an EEIO model. The results of this model are additionally applied in the DPSC specification.

The primary concern at this stage is the completion of a marketable and usable tool for the completion of PAS2070 compliant inventories. However it is envisaged that future iterations will include additional features. These may include: uncertainty consideration through the use of basic uncertainty factors to provide users with a reliable indicator of input and output. Implementation of a Monte Carlo simulation model to the areas of the model identified as of causing significant uncertainty. Upon completion the database model will provide a tool for decision and policy makers, academics and other users to compile a PAS2070 compliant inventory consistently and accurately.

Thesis Conclusions

The urban population is predicted to continue to increase, as cities grow and expand, their role in the mitigation of climate change will proportionally rise in prominence. Municipal leaders and local policy makers will need access to the appropriate methods and tools if they are to make effective and workable decisions. Such tools must be developed in a way as to make them useable with the often limited time and resources available to local governments. Carbon- and climate-footprints, and GHG inventories offer a suite of tools and metrics required for the measurement and management of GHGs in cities and other systems.

The term ‘carbon footprint’ has rapidly risen in prominence in recent years. It is likely to have emerged at some point in the late 1990s in the media or other grey literature. The language is a derivation of the previously developed ‘ecological footprint’. With the basic principles being developed on the basis of established LCA methods. Despite the relative popularity of the term ‘carbon footprint’ a literature search suggested there was limited consistency in the application of a definition (objective 1a). Definitions ranged from inclusion of solely CO_2 emissions, to an undefined inclusion of ‘all GHGs’, with varying methods and boundary conditions. In many cases the term was used simply as a synonym to relate some amount of GHG emission to a given activity. Distillation of the various definitions enabled the development of a logical and pragmatic definition for a carbon footprint (CO_2 and CH_4 emissions) which included provision for boundary setting, GHG inclusion and metrics (objectives 1a, 1b). To account only for CO_2 would result in a significant gap in the global management of GHG emissions. However the inclusion of all GHGs is potentially extremely resource intensive and arguably should only be considered in specific LCA based assessments. The accounting of CO_2 and CH_4 is relatively straight forward, and encompass circa three quarters of global anthropogenic GHG emissions, therefore these two carbon based gases are suggested for inclusion in the carbon footprint. In cases where wider GHG emissions inclusion are required the use of a ‘climate footprint’ – all Kyoto Basket gases (CO_2, CH_4, N_2O, SF_6, HFCs, PFCs) – is suggested. Beyond this the concept of a full emissions inventory could be applied, or arguably one should consider the application of a full LCA. The metric has developed on the basis of established life-cycle principles, and can subsequently be estimated via a range of life-cycle based methodologies, including process and consumption based approaches. The choice and application of method will need to be addressed on a case-by-case basis as the use of carbon and climate footprints become more widespread.

Many cities have developed GHG inventories, however previously methodologies have been variable, encompassing different scopes and having important methodological variations. The application of the previously defined carbon or climate footprint to cities and communities offers a universal and repeatable metric for the assessment of city GHG inventories. Practitioners of city GHG inventories are presented with a number of options when considering the community system. Not all cities are identical; they have different industries, locations, topographies, levels of commercial activity and demographic profiles. A community can evidently be defined in the simplest form by the extent of the geopolitical boundary. However a city does not only serve residents, nor do the residents only utilise resources from within the geopolitical boundary; significant cross boundary flows of people, goods, services and resources occur. Models for local government GHG management in cities must be developed in such a way as to fairly and accurately assign emissions, encompassing all relevant sinks and sources, whilst simultaneously considering the role of and ability of municipal governments. Equitable assignment of emissions is vital to establish appropriate methods to apportion both the social and economic benefits and the environmental impacts of emissions.

The establishment of the system boundary both defines and is partially dictated by the selection of methods. Primarily two methods exist – production (at source) and consumption (end user) – both methods provide tools for the accounting of effective accounting of differing aspects. These have developed into three approaches for community footprints, territorial (direct in-boundary), transboundary (addition of infrastructure, material and energy flows), and consumption based. Territorial methods are easily defined by the geopolitical boundary, accounting all in-boundary emissions, but excluding those from the life-cycle of imports. Upstream emissions are attributed to the producer, not the consumer. Territorial accounting may lead to large underestimates of carbon emissions from products and services as a large proportion of carbon emissions are potentially from supply chain processes. A transboundary approach extends the territorial method to include emissions from transboundary energy and material flows, more effectively representing sub-national community processes.

A consumption based approach allocates emissions to final consumption (households and government) emissions from the manufacture; transport, and consumption of imports are accounted, in this manner areas with high production and high exportation are not penalised. However emissions associated with exported products and services are excluded, potentially missing a large component of the local economy. Whilst consumption based accounts can model the macro emissions, methods fall-down below the meso scale. At the meso scale process-based methods enable the modelling of specific processes.

Given these considerations, a form of shared responsibility accounting whereby emissions from some transbounday process sources are considered on a consumption approach, and other 'direct' emission sources are considered using a production approach, is used to develop a framework for the application of life-cycle emissions modelling to cities (objectives 2a, 2b, 2c). The development and publication of the framework led to the invitation of expert members of the Southampton University Carbon Management Group to a scientific advisory role on the steering committee for the development of the PAS2070 'Specification for the assessment of greenhouse gas emissions of a city'. The framework approach developed in this thesis was adopted as a fundamental building block of the standard (PAS2070, pg. 2), forming a significant component of the source identification and boundary setting within the standard.

The development of PAS2070 provides a standardised framework for the inventorying of GHG emissions from cities. Recognising the strengths and weaknesses of transboundary and consumption accounting, PAS2070 suggests the use of a dual approach (reported separately to avoid double counting) to a GHG inventories – a consumption based (CB) and a 'Direct Plus Supply Chain' (DPSC) (a PB assessment with additional elements to capture supply-chain emissions). Chapter 4 presents several important developments to the assessment of city carbon footprints. Methods are developed to assess emissions local emissions using a territorial, transboundary, and consumption based approach. The methods were applied to real data, for Southampton, UK (objective 2d). The city was chosen as suitable due to its role as a commerce hub; a major international cruise terminal, and the UK's second largest container port. Additionally Southampton Airport is located a short distance outside the geopolitical boundary in neighbouring Eastleigh. Southampton presented the opportunity to develop a case study with methods for the inventorying of a range of components found in many other cities – helping develop and test a consistent, transferable methodology. The results and implications of each method were discussed, ultimately a transboundary and consumption methods are complementary.

The methods developed are inherently complex, and impossible to prove empirically. Essentially models represent an arbitrary enclosure of real-world systems and conditions. As a result inventories cannot be proven true, but must instead be subject to a series of 'tests' to build a portfolio of supporting evidence. In the case of PAS2070 the primary users are likely to be municipal government and other policy decision makers. Policy-makers should recognise that such models are inherently subject to multiple potential sources of uncertainty. As a result uncertainty and sensitivity analysis are important steps in the model development process.

Often Uncertainty and sensitivity are ignored in large LCA-type studies of this nature, due to perceived complexities and resource requirements; however only once limitations and assumptions are known the user can make informed decisions on practical results. A One-at-A-Time (OAT) sensitivity – a variation in a single input variable whilst all others are held constant – was applied to the PAS2070 model developed in the previous chapter. Normalised sensitivity coefficients were calculated to enable comparison between variables of differing magnitudes and units. Tests for linearity and additivity indicated that simple summation of variable sensitivities cannot fully explain variation in model output. Additionally whilst sensitivity analysis provides important information it cannot fully explore the input space. To explore uncertainties associated with the model Monte Carlo simulation was constructed. The simulation provided a probabilistic distribution of model output due to uncertainty considerations. The simulation demonstrated a high confidence (<1% variation) in model outputs.

The final part of this thesis proposes the development of a data-driven tool for the implementation of a sub-national community inventory. The proposed dual methodology reflects the complementary nature of transboundary and consumption based accounting and the PAS2070. It requires that the database contain both a process emission factors and consumption based emissions factors. The proposed database model and preceding project work recognises this demand and demonstrates the ability to develop the methods and tools required for to achieve high-quality reliable, consistent and transferable inventories (objective 3c).

In conclusion this thesis has presented several novel and original contributions to knowledge, including: an original and pragmatic definition for a carbon footprint; a framework for the application of carbon footprints to cites – adopted as part of the international PAS2070; novel methods for the achievement of a territorial, transboundary, and consumption based sub-national GHG footprint via a case study of Southampton, UK; sensitivity and uncertainty analysis of PAS2070 methods; and a proposal for a sub-national community footprinting database driven tool.

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