RETHINKING ECONOMICS:
ACCOUNTING FOR ENVIRONMENTAL IMPACT AT THE LOCAL LEVEL

by

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"If a man does not keep pace with his companions, perhaps it is because he hears a different drummer."

Henry David Thoreau

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Abstract

The quality of the human experience depends on a dramatic change in how we think about economics and, more specifically, about the relationship between human economic activity and the natural world. The continued pursuit of a growth agenda threatens the health and stability of global ecological systems, jeopardizes the wellbeing of many people, and undermines opportunities for future generations. In an era of sustainability challenges, we must measure the impacts of economic activity and use that information toward designing more sustainable human systems. This dissertation supports an ecological economic worldview by extending biophysical based measures to local scale applications to improve understanding of environmental impact at the urban and sub-regional scale. To account for environmental impact, I test two calculation approaches: one to estimate municipal ecological footprint values and one to measure environmental impact at a neighbourhood level. The novel calculation approaches account for environmental impact at finer scales of resolution than has traditionally been applied.

I also explore drivers of environmental impact using Halifax Regional Municipality as a case study. I examine the relationship between direct GHG emissions and socio-economic and wellbeing variables using a multivariate model. Those reporting to be married, young, low income, and living in households with more people have correspondingly lower direct GHG emissions than other categories in respective groupings. Respondents with lifestyles that generate higher GHG emissions did not report to be healthier, happier or more connected to their communities, suggesting that individuals can experience similar degrees of wellbeing largely independent of their GHG emissions. I explored whether where we live influences direct GHG emissions. Findings indicate that individuals living in the suburbs generate similar GHG emissions to those living in the inner city, challenging a widely held assumption that living in the inner city is better for sustainability. These results underscore the importance of understanding the spatial distribution of GHG emissions at the sub-regional scale. The research offers new insights to measure and understand environmental impact at the local level toward supporting ecologically informed decision-making.

List of Abbreviations Used

CATI	computer-assisted telephone interview
CIW	Canadian Index of Wellbeing
CO_2	carbon dioxide
CO_2-e	carbon dioxide equivalent
DA	dissemination area
GDP	gross domestic product
GFN	Global Footprint Network
GHG	greenhouse gas
GIS	geographic information software
GNP	gross national product
GPI	genuine progress indicator
GPS	geographic positioning system
GWP	global warming potential
HANPP	Human Appropriated Net Primary Productivity
HRM	Halifax Regional Municipality
ICB	inner commuter belt
ISEW	Index of Sustainable Economic Welfare
ISO	International Organization for Standardization
IPAT	impact = population * affluence * technology
IQR	inter-quartile range
kg	kilogram
km	kilometer
LCA	Life Cycle Assessment
m^3	cubic metres
MIPS	Material Input per Unit of Service
MNW	Measuring National Wellbeing
NPP	Net Primary Productivity
OCB	outer commuter belt
OCED	Organization for Economic Cooperation and Development

ON	Ontario
PDA	personal digital assistant
REAP	Resource Energy Analysis Program
S.D.	standard deviation
STAR	Space Time Activity Research
SUV	sport-utility vehicle
UNDP	United Nations Development Program

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I am a lucky man in a crazy world, after all these years. I look forward to new wonderful adventures ahead.

(Jeff, March 2013)

Chapter 1: Introduction

"Only the mountain has lived long enough to listen objectively to the howl of the wolf."
- Aldo Leopold

1.1 Dissertation Overview

My dissertation starts from the premise that the energy and material throughput driving our global economy is ultimately constrained by ecological limits. The quality of the human experience depends on a dramatic change in how we think about economics and, more explicitly, how we think about the relationship between human economic activity and the natural world. The continued pursuit of an unlimited economic growth agenda threatens the health and stability of global ecological systems; jeopardizes the wellbeing of many people; and, undermines opportunities of future generations. An economic growth paradigm, however, continues to shape economic and political agendas, define development policies, and influence socio-cultural expectations.

A growing body of scientific evidence suggests that at a global level the magnitude of economic activity is disrupting critical ecosystem function and services (Borucke et al., 2013; Ehrlich and Ehrlich, 2013; IPCC, 2007; Rockström et al., 2009; Running, 2012; Tedesco and Monaghan, 2009; UNEP, 2005; Vitousek et al., 1997). The economic system has reached a size where the intended welfare benefits of throughput economic growth are increasingly negated by the costs in terms of deteriorating natural capital and increasing pollution (Foley et al., 2005; Milesi et al., 2005; Thomas et al., 2004; WWF, 2008). Fundamental flaws in the assumptions underlying the conventional economic model contribute to ecological problems threatening the health of ecosystems and ultimately the human experience. A disconnect between an economic growth agenda and lived experience motivates the need for new economic ideas and models that work better for people and the planet. In an era of ecological urgency, a new economic vision must emerge.

1.1.1 Ecological Economics

Ecological economics originated in response to conventional economics failing to consider adequately the essential role ecological goods and services play in sustaining human lives and society (Costanza, 1989). An ecological economic view contends that natural capital and healthy ecological systems underscore all economic activity. The macro goal of ecological economics as articulated in the defining vision of the field is the sustainability of the combined ecological economic system (Costanza et al., 1991). The field attempts to understand the dynamic relationship between economic activity, human welfare and ecological systems. A critical focus is identifying levels of economic activity and forms of economic activity that are consistent with ecological health and human wellbeing (Costanza, 1989; Costanza et al., 1991; Ekins, 1992; van den Bergh, 1996).

In this emerging field, there are no formal 'ecological economic' methodologies, tools, or structures. Ecological economics is transdisciplinary and methodologically open, inviting contributions and ideas from several disciplines (Daly and Farley, 2004; Norgaard, 1989). The binding thread is a pre-analytical vision of the economy as an open subsystem of a finite, non-growing, and materially closed biosphere. The economy functions on flows of matter, energy, and services from the larger ecological system. As part of a materially closed system, the economic subsystem necessarily depends on the availability of inputs (resources) and waste assimilative capacities of the larger ecological system (Daly 1991, Daly, 1999; Daly and Farley, 2004). The ecological economic pre-analytic view is based on the physical Laws of Thermodynamics. The first law (conservation of energy) states that energy can neither be created nor destroyed but only changed from one form to another. The second law (law of entropy) states that every transformation of energy results in the degradation of energy into a lower quality form (high entropy) (Daly and Farley, 2004). All economic activity requires inputs of low entropy matter-energy and produces outputs of high entropy matter-energy (Daly, 1991; Georgescu-Roegen, 1971). Inputs and outputs come from somewhere and must go somewhere. The materially closed biosphere is a source of inputs and a sink for waste products. As Hardin (1991) noted, humans simply rearrange stuff from nature with varying consequences in the forms of degraded energy and lost inputs. Ecological economics advances a vision of an economy dependent upon and supported by ecological systems.

The pre-analytical vision goes a step further, asserting that the current magnitude of economic activity is disrupting function and capacities of supportive ecological systems (Daly and Farley, 2004; Lenton et al., 2008; Milesi et al., 2005; Running, 2012; Vitousek et al., 1997). The volume of waste products produced is stressing the assimilative capacities of the biosphere (Gruber and Galloway, 2008; IPCC, 2007; Tedesco and Monaghan, 2009). Natural capital inputs (both renewable and non-renewable) are being depleted at rates jeopardizing future use of these stocks (Foley, 2005; Trembley-Boyer et al., 2011; Worm, 2006). The high rate of use in some cases disrupts stability of larger ecological systems (Barnosky, 2012; Ehrlich and Ehrlich, 2013; GFN, 2008; Rockström et al., 2009; Thomas et al., 2004; WWF, 2008). Systems upon which human survival ultimately depend. Paramount to the ecological economic worldview is a sense of urgency.

A world-view based on ecological limits fundamentally disagrees with the conventional economic vision of an economy that is not bounded by resource scarcity (Daly, 1991; Rees, 1996). In the conventional economic view, the resource base driving economic activity is essentially limitless due to technological progress and substitutability (Costanza et al., 1991). Daly and Farley note, “The difference [in views] could not be more basic, more elementary, or more irreconcilable” (2004, 23). Sachs refers to the divide between proponents of an economic growth model in a limitless world and proponents of an economic model bounded by ecological limits as the greatest clash of our generation (Wackernagel, 2003). The debate may not be as dramatic as other ideological clashes of the 20th century; however, the consequences are global, potentially irreversible, and surely will affect more people.

1.1.2 Sustainable scale

The concept of a sustainable scale to economic activity relative to biocapacity is unique to ecological economics and distinguishes ecological economic theory and discourse from other branches of economics. Ecological limits imply a restriction on the useable flow of energy and material to support the human system. The conundrum is how humans satisfy their collective demand for energy and resources given a finite supply. The other two principles of ecological economics, just distribution and efficient allocation, flow from this conundrum and focus on how we best manage the supply (energy and resources) to maximize use and ensure fair distribution (Daly and Farley, 2004). Just distribution considers ethical questions about how to ensure that all humans have equitable access to economic goods and services and that we account for needs of future generations (Daly and Farley, 2004; Farley, 2008). Efficient allocation addresses the challenge of how to allocate limited resources to satisfy household wants and support socio-cultural structures given the boundaries implied by sustainable scale and the importance of equitable distribution. Of the three key principles, sustainable scale is a priority, in part because it implies parameters governing the other two principles. More so, the concept dramatically alters how we conceptualize the economy-natural world relationship. It explicitly rejects the conventional economic view of nature as an unlimited resource base and sink for wastes products.

A human economy restricted by ecological factors does not mean an end to economic development; it implies a limit to economic throughput. The emphasis shifts from increasing the amount of resources consumed to improving quality of life (Anielski, 2007; Costanza et al., 1991). Scale means the throughput energy and matter to support the human system cannot grow indefinitely without interfering with larger system functions. Determining an appropriate or maximum scale of our economy requires a profound understanding of supporting ecosystems, which we arguably do not have. Ecosystems are complex and dynamic and despite our best science, humans have a poor understanding of ecological support functions. The challenge that ecological economists face is how to know what an optimal or sustainable economic scale is. The concept of sustainable scale does not imply a finite target or size. Limits will change depending on population, consumption levels, ecosystem health, technological innovation and social decisions concerning the type of world in which we would like to live.

Acknowledging an ecologically informed sustainable scale to economic activity changes the conditions for allocating resources and their distribution. To maximize the equitable distribution of wealth within a limited system necessitates efficient allocation of resources. Increasing resource throughput to expand the economic pie as a means to reduce poverty, and improve human welfare characteristic of the conventional economic worldview is not an option. The concept of sustainable scale introduces critical ethical questions into economic discourse. If one group consumes more than their share does it necessitate that others live in poverty elsewhere? If we use resources unsustainably are we jeopardizing the prospects of future generations to enjoy healthy lives? Further, an ecological economic view defines the ends of economics differently. The conventional economic view has become obsessed with chrematistics, or moneymaking to maximize consumption (Anielski, 2007; Daly and Cobb, 1989). The model is built on a myth that more growth, more production, and more consumption are good for our lives (Twist, 2003). The focus within ecological economics is on leading healthy, high quality lives within sustainable means (Anielski, 2007; Costanza et al., 1991; Daly and Cobb, 1989). Ecological economics attempts to include ecological principles in economic decision-making bridging the gap between the predictive demands of economics and the complexity of ecological science (Christensen, 1989).

1.1.3 Measurement

For ecological economics to emerge as a replacement to a growth-based economic model requires operationalizing the concept of sustainable scale into economic doctrine, policy, and decision-making. Measuring the ecological costs of human activity and more explicitly understanding the environmental impacts associated with our lifestyles, policy decisions, and economic system is a critical step. Ecological economists argue for the adoption of measurement tools that quantify the biophysical foundations of economic activity by accounting for the depletion of natural capital and the role of ecosystem goods and services in supporting the human system (Christensen, 1989; Daly and Cobb, 1989).

Understanding what an appropriate economic scale is and managing our economies to thrive within those limits requires a biophysically informed understanding of throughput flows driving economic activity. Many biophysical measurement tools have been designed for and applied at the macro level, providing evidence that the magnitude of material and energy throughput supporting the human economy is straining ecological systems (Borucke, 2013; Haberl et al., 2007, Schmidt-Bleek., 1993; Zhao and Running, 2010). Global and national biophysical assessments have been critical in fostering an acknowledgement that the human economy comes with costs to supporting ecological systems potentially jeopardizing the wellbeing of current and future generations (IPCC, 2007; Rockström et al., 2009; Running, 2012; Tedesco and Monaghan, 2009; UNEP, 2005). Biophysical measurement tools have been successful at communicating the concept of scale at the macro-level; they do not, however, extend well to local applications (Baynes and Wiedmann, 2012; Browne et al., 2012; Sahely et al., 2003). Consequently, local decision-makers lack tools to support ecologically informed economic decisions. The gap is largely a function of inadequate data to populate biophysical models. Harmonized datasets are almost exclusively available for the national level only. Those wishing to undertake regional or community-specific analyses struggle with incomplete or incommensurable data. Direct data collection is costly and often not an option (Graymore et al., 2008; Klinsky et al., 2009; McManus and Haughton, 2006; Satterthwaite, 2009; Wilson and Grant, 2009). In addition, the premise that aggregate economic activity may be jeopardizing the health and function of ecological systems is conceptually difficult to connect with day-to-day understandings of the economy at the local and household level. Linking household economic choices and local economic policies to ecological economic concepts such as sustainable economic scale is challenging, especially considering the lack of decision support tools to help formulate such a connection. Adopting an ecological economics framework, however, requires acknowledging the concept of sustainable scale at different experiences of the economy from the household, to the community, to the global level. Furthermore, the local and regional level have been argued to be the most appropriate scale for advancing sustainability issues (Clark and Dickson, 2003; Graymore et al. 2008; United Nations, 1992). Developing local scale approaches to measure environmental impact offers a basis for ecologically informed economic decision-making.

1.2 Research Purpose

My research agenda focuses on rethinking how we measure and account for the impacts of human activity on natural systems in a world with ecological limits to economic throughput. Specifically, my dissertation explores new approaches to account for and better understand drivers and distribution of environmental impact at the urban and sub-regional scale. My dissertation purpose reflects three complementary areas of motivation:

- *Motivation 1- concept:* Rethink measurement from an ecological economics perspective to ground sustainable scale as a criterion of economic measurement.

- *Motivation 2 – method:* Advance biophysically-based approaches to operationalize ecologically informed decision-making at the local level to support sustainability assessments.

- *Motivation 3 – practice:* Improve our understanding of environmental impact at the local level.

My dissertation is organized into three parts, which speak to the above motivations. It is designed around five substantive articles, which either have been published in peer-reviewed journals or are currently under review. The articles were co-authored. In all cases, I was the lead author responsible for research, analysis, and drafting the articles. In addition, two shorter papers contribute context and support the thematic flow of the dissertation. The second of these papers is published as a review in the Encyclopedia of Quality of Life Research.

1.3 Rethinking Measurement - Accounting for an Ecological Economic Worldview

Proponents of an ecological economics worldview argue that the continued expansion of economic throughput threatens ecological sustainability and undermines human wellbeing. Advancing a new societal trajectory requires tools and indicators designed to better assess the societal and environmental impacts of economic activity. An awareness that increased economic throughput is not advancing human welfare and undermining ecological systems has motivated efforts to improve how we account for the benefits and costs of economic activity. Not always explicit, new accounting frameworks are a response to the failure of the economic growth imperative as a pathway to human wellbeing. Part 1 includes three review chapters on rethinking measurement from an ecological economics perspective. The first chapter reviews the role of quality of life and genuine progress indicators within the larger effort to reconsider how we account for the impacts associated with economic activity. The chapter also discusses the suitability of wellbeing models and metrics for local scale applications, recognizing growing interest in these techniques at the urban and local level. The chapter closes with a reflection on the uptake of GPI and wellbeing measures highlighting the Canadian experience. In terms of rethinking measurement, the chapter concludes that GPI and related metrics improve upon the GDP as a surrogate of human welfare; biophysical based metrics, however, are needed to account for the ecological impacts associated with economic growth. The chapter has been published as an article in the journal *Sustainability*.

Chapter 3, the second chapter in Part 1, reviews biophysical measurement tools as a subset of ecological economic accounting. The chapter highlights prevalent accounting techniques used in urban and regional sustainability assessments and discusses measurement challenges. The chapter argues that human carrying capacity approaches are useful to communicate sustainable scale as a basis of economic decision-making.

Chapter 4, the third chapter in Part 1, reviews the ecological footprint, an example of a human carrying capacity-based measurement technique. The chapter has been published in the Encyclopedia of Quality of Life Research. The review supports my use of the tool as a broad measure of environmental impact in subsequent chapters suggesting approaches to extend biophysical assessment tools to the municipal and sub-city level. The three chapters consider measurement from an ecological economics perspective grounding the concept of sustainable scale as an economic decision-making criterion.

1.4 Accounting for Environmental Impact at the Local Level

Part 2 starts from the premise that operationalizing ecologically informed decision-making at the local level requires the adoption of biophysically based measurement tools to support sustainability assessments. As a researcher and practitioner, over the past decade I have witnessed growing interest and excitement at the local level to measure and understand the environmental impacts of consumption. At the local level, frustration often follows the excitement primarily because the tools available to quantify environmental impact are sparse. Measuring the ecological costs of economic activity at fine scales proves challenging. The political and organizational context often exacerbates the challenge. Many proponents of sustainability measurement face opposition either directly within their organizations or indirectly by society's adherence to the current economic growth agenda. By not offering robust, accessible measurement tools, we fail those wishing to push for sustainability policies and practices. Failing to do so undermines efforts to support decisions based on the premise that there are ecological limits to economic activity.

Part 2 includes two chapters that explore different approaches to account for environmental impact of human consumption at the local level. Chapter 5 tests a calculation strategy using the ecological footprint method to account for environmental impact that could be widely adopted by municipalities across Canada and has been published in the journal *Local Environment*. As a measurement tool, the ecological footprint enjoys a high level of awareness and application within Canada. Further, the metaphor of the footprint has captured the attention of the general public and policy makers like no other biophysical measurement tool. This has happened in a very short time span. Mathis Wackernagel and his Ph.D. supervisor William Rees published their book *Our Ecological Footprint* in 1996. The metaphor of the ecological footprint is critical because it conveys clearly that we have a finite amount of ecological productivity or natural capital to support human activity. More so, the metaphor evokes powerful messages. If there is only so much space and I over-consume, does my overconsumption impact ecological sustainability? What are the impacts to future generations? What are the impacts to other people living on the planet now? Does overconsumption in one region necessitate poverty elsewhere? The suggested approach in Chapter 5 recognizes that many communities may not have access to detailed resource and energy flow data, expertise in sustainability modeling, or resources to undertake a comprehensive analysis. Because the calculation approach is consistent with the Global Footprint Network standardized methodology, it permits meaningful comparisons between communities and with global and national footprint estimates. The municipal calculation approach offers planners, policy makers, and community leaders an accessible, straight-forward and cost effective strategy for estimating the ecological footprint at the community and municipal level.

Chapter 6 responds to concerns that municipal-wide results are useful for education and awareness regarding environmental impact but lack specificity to inform policy and planning decisions. Chapter 6 is published as an article in the *Journal of Environmental Planning and Management*. The chapter proposes that community assessments of environmental impact are increasingly relevant to planners and policy makers when reported at finer scales of analysis. Using the Town of Oakville, Ontario, as an example, I report environmental impact by neighbourhood. The analysis tests our ability to measure environmental impact at finer and finer scales of resolution. The results highlight variability in environmental impact within a community, providing planners and policy makers detailed information to prioritise programme delivery, allocate limited resources, and support policy development. Further, the results expand our understanding of the distribution of environmental impact at the sub-city level. The research proposes the concept of a 'footprint floor' dictated by physical and social structural factors and an upper footprint range determined largely by available income. The proposition of a footprint floor has implications for setting community footprint targets and understanding the magnitude of change needed for significant ecological footprint reductions. An important take away for leaders, planners and policy makers from this research is that changing urban form, infrastructure, and resource use patterns may be critical in many settings to achieve large-scale ecological footprint reductions. As major infrastructure and planning decisions made in the past influence a city's ecological footprint, current infrastructure and planning decisions will lock a community into consumption patterns that are difficult to overcome. The long term influence that planning decisions can have over a jurisdiction's ecological footprint highlights the importance of making sure that new development projects, major infrastructure decisions, and city planning and policy documents foster a lower footprint future (Rees, 1997; Rees, 1999).

1.5 Understanding Drivers and Distribution of Environmental Impact Across an Urban Region

In the absence of ecologically informed data at the local level, unfounded assumptions regarding drivers of environmental impacts can direct planning and policy decisions in potentially problematic directions. Part 3 presents two chapters that aim to improve our understanding of environmental impact at the local level. Using results from the Halifax Space Time Activity Research (STAR) project, I estimated direct greenhouse gas (GHG) emissions with a high level of granularity at the sub-regional scale. The GHG estimates are based on household energy-use survey data and GPS-verified travel data. The robust data set provides a platform to identify drivers of GHG emissions at the local level and to explore the spatial distribution of GHG emissions across a community.

In Chapter 7, I conduct a multivariate analysis examining the relationship between direct GHG emissions and 19 socio-economic and wellbeing variables. The analysis confirmed findings from national studies identifying household size, age, income, and marital status as drivers of direct GHG emissions. Among the predictor variables, those reporting to be married, young, have low income, and live in households with more people have correspondingly lower direct GHG emissions than other categories in respective groupings. The chapter has been accepted for publication in the *Journal of Ecological Indicators* pending review of revisions.

The analysis is a first to include several wellbeing variables in efforts to understand better the relationship between subjective wellbeing and environmental impact. Interestingly, all wellbeing variables were dropped from the model. Degree of happiness, life satisfaction, health, sense of community belonging and civic engagement were not associated with GHG emissions, suggesting that lower GHG emission lifestyles do not compromise wellbeing. The research highlights the need for future research in several critical areas including: understanding of the connection between subjective well being and environmental impact; finding opportunities to decouple income from environmental impact; and understanding psychological drivers of the consumer lifestyle. Redefining personal aspirations independent of affluence and high consumption is essential for long-term sustainability.

Chapter 8 investigates whether where we live matters in terms of contributions to GHG emissions. Influencing where people live is, increasingly, considered a strategy to help municipalities meet GHG reduction targets and become a more ‘sustainable’ city. Chapter 8 reports results and statistical differences in greenhouse gas emissions for Halifax Regional Municipality between communities and urban-rural zones (inner city, suburban, and inner/outer rural commuter). Results reveal considerable spatial variability in direct GHG emissions across the study area. My finding that individuals living in the inner city generate similar amounts of direct GHG emissions to those living in the suburbs challenges a widely held assumption that inner city living is more sustainable. Policy and planning decisions to support GHG reductions must consider energy infrastructure, historical design of the city, urban form, development patterns, and household characteristics such as number of people, income, and age. The study underscores the importance of understanding the spatial distribution of GHG emissions at a sub-regional scale.

1.6 Conclusion

Many of the most significant social and political challenges of my time are sustainability based. Introducing and supporting a new economic vision requires that we collectively rethink the relationship between economic activity and the natural world. It is not simply a matter of connecting the economy to ecological systems or changing what we measure. Advancing an ecological economic worldview requires a dramatic shift in philosophical approach. This dissertation aims to support that effort by improving our understanding of material and energy throughput at the urban and sub-regional level. My research, I hope, offers a small contribution within a vast effort to change economics so it works better for people and the planet. In an era of sustainability challenges, it is imperative that we account for the impacts of economic activity and use that information toward designing more sustainable human systems.

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PART 1 – RETHINKING MEASUREMENT - ACCOUNTING FOR AN ECOLOGICAL ECONOMIC WORLDVIEW

"The day is not far off when the economic problem will take the back seat where it belongs, and the arena of the heart and the head will be occupied or reoccupied, by our real problems - the problems of life and of human relations, of creation and behavior and religion."

- John Maynard Keynes

Overview:

Part 1 includes three review chapters on rethinking measurement from an ecological economics perspective. The first chapter reviews the role of quality of life and genuine progress indicators within the larger effort to reconsider how we account for the impacts associated with economic activity. The second chapter reviews biophysical measurement tools as a subset of ecological economic accounting. The chapter highlights prevalent accounting techniques used in urban and regional sustainability assessments and discusses measurement challenges. The third chapter reviews the ecological footprint, an example of a human carrying capacity-based measurement technique. The review supports my use of the tool as an environmental impact indicator in subsequent chapters. The three chapters included in Part 1 consider measurement from an ecological economics perspective grounding the concept of sustainable scale as an economic decision-making criterion.

Chapter 2: Rethinking What Counts. Perspectives on Wellbeing and Genuine Progress Indicator Metrics from a Canadian Viewpoint

2.1 Publication Information

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2.2 Abstract

A prevailing undercurrent of doubt regarding the merits of economic growth has motivated efforts to rethink how we measure the success of economic policy and societal wellbeing. This article comments on efforts to better account for impacts of economic activity emphasizing genuine progress indicator (GPI) and wellbeing metrics from a Canadian viewpoint. The authors caution that GPI and related metrics are measures of human and social welfare and not adequate to account for the ecological costs associated with economic growth. In addition, the article discusses the suitability of wellbeing models and metrics for local scale applications, recognizing growing interest in these techniques at the urban and local level. The article concludes with a reflection on the uptake of GPI and wellbeing measures highlighting the Canadian experience.

2.3 Introduction

A growing body of scientific evidence suggests that at a global level the magnitude of economic activity is disrupting critical ecosystem function and services and contributing to social decline (Global Food Network, 2008; IPCC, 2007; Tedesco and Monaghan, 2009; UNEP, 2005). The intended welfare benefits of economic growth are negated by the costs in terms of deteriorating natural capital, increasing pollution, and contributing to social problems. A disconnect between an economic growth agenda and lived experience has motivated efforts to reconsider how we direct our economies so they work better for people and the planet. A critical starting point has been rethinking how we measure the impacts of economic activity. What we count, measure, and track matters. Basing development decisions solely on economic indicators promotes policies, priorities and investment decisions that may make sense from a Gross Domestic Product (GDP) perspective but often come with human and environmental costs sending misleading signals to planners, policy makers and the public about what fosters wellbeing. Critics argue that growth-based economic policies have failed “the development project” (Banerjee and Duflo, 2011; Sachs, 1999; Sen, 1999; Shiva, 2005; Stiglitz, 2002).

Efforts to design more relevant metrics have focused on integrating social, environmental and human-welfare criteria as a way to ensure potential costs of economic growth are properly factored into decision-making processes. In 2007, the European Commission, the Organization for Economic Cooperation and Development, the Organization of the Islamic Conference, the United Nations, the United Nations Development Programme and the World Bank signed the Istanbul Declaration (OECD, 2007). This important symbolic gesture confirms the growing recognition among global institutions that measurement of societal wellbeing must go beyond traditional economic measures.

This paper comments on efforts to improve macroeconomic metrics and indicators from a Canadian viewpoint in the context of better counting the social and environmental costs associated with economic growth. The Istanbul Declaration and recent traction in

Europe to advance new measures of wellbeing stands in sharp contrast to stalled efforts in Canada, an early pioneer in advancing alternative frameworks of progress and wellbeing. As much of the motivation to design new metrics centres on the failings of GDP as a measure of societal welfare, section two begins with a brief overview of the widely acknowledged shortcomings of GDP. Section 3 reviews efforts to better account for societal welfare by providing historical context and citing Canadian examples. Section 4 identifies limitations of genuine progress indicators (GPI) noted in the literature and cautions that GPI initiatives as currently constructed are not suitable proxies of ecological sustainability. Section 5 discusses the suitability of wellbeing models and metrics for local scale applications highlighted in the literature recognizing growing interest at the urban and local level within Canada. Section 6 comments on the Canadian experience and limited uptake of GPI type metrics. The paper concludes more broadly with reflections on efforts to go beyond GDP and the need to complement GPI metrics with biophysical based metrics to account for the ecological impacts associated with economic activity.

2.4 Growth Metrics and Community Wellbeing

Community wellbeing or progress has traditionally been inferred from a narrow set of economic indicators. The underlying premise assumes that a strong economy signals a society is doing well. In part, the premise is logical. Data demonstrates a correlation between economic indicators such as GDP per capita and employment, educational attainment, health outcomes, life expectancy and crime rates, to name a few (Drolet, 2005; Ellis et al., 2009; Romanow, 2002; UNDP, 2011). At the macro-level, GDP per capita is the principle indicator of 'economic health' and by extension, societal welfare (Costanza et al, 2004; Daly and Cobb, 1989; Stiglitz et al., 2008). Having one 'all encompassing metric' offers policy makers a simplistic directional tool to inform decision-making and communicate the apparent success or failings of an economy.

The GDP, however, measures progress solely in terms of what is being bought and sold. Cobb, Halstead and Rowe state in their landmark article, *if the GDP is Up, Why is America Down, "The GDP is simply a gross measure of market activity, of money* *changing hands. It makes no distinction whatsoever between desirables and undesirables, or costs and gains"* (1995 p.160). While the GDP guides major societal policy direction and development choices, it is an extremely poor proxy for social welfare and environmental sustainability. Simon Kuznets, the architect of GDP/GNP (Gross National Product) accounting, devised the national accounts system to better direct the economy following the Great Depression (Kuznets, 1934). The system was never intended as a measure of societal wellbeing. Kuznets, whose efforts would evolve into the current framework for measuring GDP, even cautioned against using it as an overall barometer of welfare, arguing that it includes no criteria of social productivity (Kuznets, 1941; Kuznets, 1962).

Shortcomings of the GDP as a wellbeing indicator are well documented (Anielski, 2007; Daly and Cobb, 1989; Posner and Costanza, 2011; van den Bergh, 2009) and include:

- GDP regards all expenditures as contributing to wellbeing regardless of what that expenditure is for and its effects. For example, money spent on pollution clean-up, addictions treatment, crime and car accidents contribute to the GDP and therefore are counted as contributing to wellbeing.
- GDP devalues goods and services that do not involve monetary exchange. For example, taking care of a child or a parent, housework and volunteer work are not factored in.
- The GDP is not forward thinking. It excludes the inherent value of natural resource capital and ecosystem services. For example, a forest has no value unless you cut it down. Furthermore, the GDP minimizes the value of expenditures on education, preventive healthcare, and environmental protection because it counts only the immediate expenditure and not the potential return on investment.
- The GDP does not account for the distribution of income within a society. GDP counts any increase in income as positive, overlooking the social costs of income inequality and poverty.

2.5 Alternative Measures of Wellbeing

2.5.1 A Note of Caution

The terminology and methods underlying proposed metrics such as the genuine progress indicator (GPI), community wellbeing measures, and quality of life indicators are loosely applied, particularly in the grey literature. Concepts such as wellbeing, health, good life, progress and societal welfare are frequently used interchangeably and are rarely defined. Michalos and colleagues (2011) suggest that the inherent fuzziness of many of these terms reflects their long history (some dating back to antiquity) of being used freely in discourse. Consequently, since GPI, quality of life and community wellbeing metrics are new and evolving concepts, the lines that circumscribe their terminology and application tend to be blurred. GPI measures, when compared to quality of life and community wellbeing measures have a more formal methodological and theoretical underpinning (Lawn, 2003). Measures of community wellbeing and quality of life are frequently a collection of indicators organized around domains, which are believed to contribute to wellbeing or life satisfaction. A review by Cummins (1996) of over 1,500 articles providing data on life satisfaction found little consensus in the literature around domains, and an even wider range of suggested indicators to consider. A further challenge is to decipher between a group of indicators loosely organized under a theme for a specific project and frameworks to be reported in place of or along-side traditional economic measures. For that reason, this paper focuses more heavily on GPI initiatives, recognizing that most of the measurement challenges and limitations extend generally to other quality of life and wellbeing metrics.

2.5.2 Genuine Progress Indicators

Efforts to develop alternative measures of wellbeing are, in part, a direct response to the misuse of GDP as a barometer of welfare. More profoundly, however, these measures challenge the validity of an economic growth agenda in an era of increasing social and environmental concerns. Critics argue that basing the direction of society on

GDP fosters a socio-economic climate that emphasizes economic growth often at the expense of the environment, health and wellbeing of many people (Costanza et al, 2004; Daly and Cobb, 1989; Max-Neef, 1995; Sachs, 2008; Stiglitz et al., 2008). Proposed alternative metrics attempt to measure a broader range of environmental, social and economic criteria. However, these efforts are not intended to simply represent a shift to improve our accounting, but also to alter perceptions about human values and what makes life worthwhile. Twist (2003) argues that the movement to redefine what indicators we track and use is part of a global effort to change the myth that more growth, more production, and more consumption are good for our lives.

Daly and Cobb’s pioneering index of Sustainable Economic Welfare (1989) and the suite of related genuine progress indicators are a category of measurement tools that adjust the GDP model so to better account for the benefits and costs of economic growth. While methods vary slightly between applications and have been refined over time, major adjustments include: deducting social and environmental costs and other ‘regrettable’ expenditures normally included in GDP estimations, adding the value of goods and services rendered outside the marketplace, and factoring in the costs of income inequality. Proponents of various GPI initiatives argue their results offer a more complete measure of societal wellbeing and qualify the impacts of economic growth (Anielski, 2007; Costanza et al, 2004; Daly and Cobb, 1989; Posner and Costanza, 2011).

Efforts to improve macroeconomic indicators date back to the 1970s. Nordhaus and Tobin (1973) proposed the Measure of Economic Welfare to evaluate if the benefits of growth as a policy directive to improve social welfare were obsolete. In short, based on research covering the period 1929–1965, they observed that while GNP and other national income aggregates are imperfect measures of welfare, the broad message they convey holds correct. Nordhaus and Tobin’s results affirmed the economic growth doctrine and existing means of measuring welfare. The impetus of their research, however, highlighted increasing skepticism regarding the merits of simply counting market-based economic activity. Given Nordhaus and Tobin’s conclusion, efforts to revise measures of aggregate output waned. By the late 1980s, however, Daly and Cobb

(1989) reinvigorated the debate, questioning the validity of Nordhaus and Tobin's earlier findings. Daly and Cobb observed that while Nordhaus and Tobin's conclusions held true for the early years of their research period, increases in welfare per increase in GDP declined significantly over the latter period. Daly and Cobb undertook a new analysis for the years 1950–1985. They suggested additional adjustments reflecting a broader set of social and environmental criteria. Their proposed model, referred to as the Index of Sustainable Economic Welfare (ISEW), indicated that starting in the 1970s the benefits of economic growth were offset by rising social and environmental costs (Daly and Cobb, 1989).

Cobb and colleagues (1995) working for an economic think tank based in California, used Daly and Cobb's research to advance public policy efforts challenging a growth based economic agenda. In addition to revising the ISEW methodology slightly, they renamed the model the Genuine Progress Indicator (GPI). Since 1995, well over 40 GPI initiatives have been conducted following their general framework. Specific calculation methods have varied slightly among studies due to several factors: limited data availability; place specific adjustments; novel adjustment categories, and differing valuation methods of non-market goods and services (Posner and Costanza, 2011). Similar to calculating GDP, Cobb and colleagues (Cobb et al., 1995) started with personal consumption expenditures and factored in 23 adjustments to derive a GPI value. Table 2.1 summarizes adjustments made by Cobb and colleagues (Cobb et al., 1995). What is critical throughout their GPI analyses is that all values are fully commensurate and fully expressible in monetary units. The resulting metrics report an aggregated monetary value similar to GDP.

Table 2.1: Genuine progress indicator (GPI) adjustment categories as implemented by Cobb and colleagues (1995)

Additions	Subtractions
value of volunteer work	cost of crime
value of non-paid household work	cost of family breakdown
services of consumer durables	cost of automobile accidents
services of highways and streets	cost of consumer durables
net capital investment	cost of household pollution abatement
net foreign lending and borrowing	loss of leisure time
income distribution adjustment	cost of underemployment
	cost of commuting
	cost of water pollution
	cost of air pollution
	cost of noise pollution
	loss of wetlands
	loss of farmland
	cost of resource depletion
	cost of long-term environmental damage
	cost of ozone depletion
	loss of old-growth forests

A consistent finding from GPI studies is that while GPI measures parallel GDP over earlier time periods, at some point the indices diverge and GPI either levels off, increases at a much slower rate, or in some cases declines (Posner and Costanza, 2011). The Alberta, Canada GPI per capita, for example, rose in parallel with real GDP per capita until about 1960 when the GPI leveled and began to decline slightly. Between 1961 and 2003, Alberta's GPI per capita decreased by 19% while Alberta's GDP per capita over that same period increaseed by almost 500% (Taylor, 2005). Posner and Costanza (2011), in a review of over 40 national and sub-national GPI studies, confirm that in all cases, while GDP rises over the course of decades, there is a leveling, falling or slow rise of the GPI. Earlier reviews of GPI and related studies support these findings (Costanza et al., 2004; Lawn, 2003). The general relationships between GPI and GDP curves support the “threshold hypothesis” (Max-Neef, 1995) or “uneconomic growth” (Daly and Cobb, 1989), terms used to suggest that beyond a certain point, the resulting environmental and social costs of economic growth outweigh the benefits. In these cases, additional growth in the scale of the economy no longer improves quality of life and may undermine environmental sustainability and societal welfare (Lawn, 2003; Max-Neef, 1995). Measures of GPI affirm that expanding economic capital beyond a certain point can erode social capital and natural capital. The concept of a threshold is context specific and surpassing that threshold depends on total aggregate economic activity but also on what aspect of the economy is growing and how the benefits are distributed.

2.6 Challenges and Limitations of Using a GPI Approach

GPI and related metrics have some limitations. The most significant critiques focus on inconsistent and questionable monetary valuation methods used to estimate the value of non-market goods (Posner and Costanza, 2011; Lawn, 2003; Neumayer, 2000). Neumayer (2000) contends that the widening gap between GPI related metrics and GDP is not a result of the threshold hypothesis but is due to questionable valuation techniques especially in regard to valuing the depletion of non-renewable resources and the costs of long-term environmental damage. Neumayer’s criticism is less applicable within a study if the focus is on comparing trend change in GDP per capita and GPI per capita over time as opposed to absolute values. To avoid challenges of converting costs and benefits into monetary values and the associated criticisms, a growing number of wellbeing metrics avoid monetization altogether. Results typically communicate progress (or lack thereof) against time, a target value, or by comparing results against other jurisdictions. Notable Canadian examples include the Genuine Wealth Model (Anielski, 2007) and the Canadian Index of Wellbeing (CIW) (Michalos et al., 2011). Both models index indicators against a base year. Evaluating progress over time is still problematic, however, as perceptions of progress (or lack thereof) depend heavily on the base year selected.

Critics also suggest the lack of a standard calculation methodology and differing adjustment categories limits comparability and consistency in results across studies, weakening the approach (Posner and Costanza, 2011; Lawn, 2003). For this reason, Posner and Costanza (2011) argue that any hope of mainstream political adoption of the GPI depends upon an accepted standardized methodology. Anielski (Personal communication, 2012) cites the lack of a standard set of guidelines for both the selection of indicators in the GPI framework and full cost accounting protocols as a barrier preventing formal government uptake of GPI in Canada.

Another critique focuses on using personal consumption expenditures as the basis of GPI calculations. Personal consumption expenditures include several questionable categories that count positively toward the GPI including, tobacco, alcohol products, and processed foods, which arguably do not contribute to wellbeing (Lawn, 2003). It is possible for studies to omit these categories. An argument can made however, that moderate use of products in these categories may contribute some wellbeing suggesting that only partial allocation of expenditure should be omitted. In defense of starting with personal consumption, GPI accounting adds undesirable effects of consumption such as defensive and rehabilitative health expenditures.

A common critique of the GPI approach lies in the concern that many diverse aspects of wellbeing are lost in reporting a single aggregate value (Kuznets, 1941; Neumayer, 1999; Wen et al., 2007). For example, the CIW which reports a single aggregate result noted an 11% rise in the index between 1994 and 2008. Two domains, however, saw a decrease in wellbeing. Further, 25 of the 64 indicators reported that individuals were worse off. Also, reporting a single value masks that indicators can potentially conflict with another. While this may seem obvious between domains (living standards and environment), conflict occurs within the same domains. For example, in the environment domain, increases in absolute greenhouse gas emissions count negatively whereas increases in primary energy production counts positively (Michalos et al., 2011).

Several authors take a stronger stance arguing that reporting diverse, complex, and significant amounts of information in a single metric is absurd to begin with (Morse and Fraser, 2005; Salles, 2011; Spangenberg and Settele, 2010; Udo de Haes, 2006). The

Alberta GPI, Nova Scotia GPI and CIW report an aggregate finding but also report results by domain and indicator. Users of the data have access to the information at different levels of aggregation to support decision-making. Calculating a complex measure presents practical challenges. For example, it requires significant time, expertise and resource demands, which clearly limits uptake for resource-constrained jurisdictions. Given the reliance of the GPI on a few categories, Bleys (2008) proposes a simplified calculation approach to enable wider adoption of the metric. Bleys (2008) notes that a few categories: value of household work, depletion of non-renewable resources, cost of income inequality, and the cost of consumer durables dominate the index. Posner and Costanza (2011), however, argue a major strength of the GPI methodology is the underlying detail and that simplifying the metric would undermine its value.

Increasingly, studies estimate GPI values in combination with sustainability or other human welfare based metrics to highlight issues such as ecological sustainability, and happiness. For example, the Alberta GPI and Nova Scotia GPI both reported the ecological footprint as an indicator (Taylor, 2005; Wilson et al., 2001). Studies using multi-tool approaches suggest the increased information supports decision-making and highlights the multiple dimensions of what constitutes sustainability and wellbeing (Browne et al., 2012, Hanley et al., 1999; Wilson et al., 2013). In some cases, the various metrics convey conflicting messages reinforcing that what we count and how we count matters (Wilson et al., 2007). A pluralistic approach corrects for the deficiencies inherent in using a single metric (Browne et al., 2012).

From a sustainability perspective, the GPI ignores the spatial and societal distribution of costs and benefits. Those receiving the benefits of economic growth may not be those who bear the costs. The GPI, therefore creates an indicator bias favouring jurisdictions which export the costs of economic growth to other regions (Clarke and Lawn, 2008; Posner and Costanza, 2011). Makino (2008), for example, notes the likely overestimation of Japan's GPI because of Japan's significant reliance on imports of raw material and energy resources. Makino calls for an "open economy GPI" to capture the environmental costs of Japan's consumption outside of its borders (2008).

Lawn (2003), Wen and colleagues (2007)and Dietz and Neumayer (2007) argue that GPI measures are a weak sustainability indicator. The GPI does not indicate if natural capital is being used or substituted at an unsustainable rate or if rates of economic activity jeopardize critical ecosystem functions. Genuine progress indicator metrics account for ecological costs in terms of monetizing natural resources and pollution costs. They provide no indication if natural capital and critical ecosystems are declining in quantity or quality. Relying on monetary valuation to compare and aggregate adjustment categories into a single monetary amount implies trade-offs between the various indicator categories and a high level of substitutability between forms of capital (Dietz and Neumayer, 2007; Wen et al., 2007). Further, monetization of ecosystems goods and services reveals nothing about the sustainability of energy and material throughput driving the economic system. More generally, valuation techniques to price ecosystem goods and services are inherently limited (de Groot et al., 2012; Farley, 2012; Salles, 2011; Vatn and Bromley, 1994). GPI and community wellbeing metrics tackle one facet of improving how we measure the impacts of economic activity. To account for environmental sustainability, GPI measures need to be supplemented with natural capital stock based indicators or measures of energy and material throughput. GPI metrics look to correct for flaws in current income based approaches to account for societal welfare. GPI metrics may endorse a weak sustainability view; the pertinent point is that they are not proxies of sustainability, especially ecological sustainability (Dietz and Neumayer, 2007). In terms of rethinking measurement, GPI and related metrics improve upon the GDP as a surrogate of human welfare; biophysical metrics, however, are needed to account for the ecological impacts associated with economic growth.

2.7 Adapting Metrics to Local Scale Applications

GPI and related wellbeing metrics have been designed primarily for macro-scale applications. Demand and interest, however, to rethink measurement has surged at the local level (Smale, personal communication, 2012; Wilson and Grant, 2009). Rightly so, the signals conveyed by macro-economic metrics do not resonate with many people's day-to-day experience. In addition, a growing number of community leaders, planners and policy makers are looking to explore different development pathways and projects and want evidence to validate their positions (Wilson and Grant, 2009). Despite the demand, few GPI and community wellbeing metrics have been adapted to the local level. Wilson and Grant (2009) overview major barriers including lack of data, high resource demands to complete studies (time, money, skill set), limited ability of local authority to influence results and difficulties translating results into action plans.

A review of GPI studies conducted by Posner and Costanza identify six urban level GPI analyzes with results for ten cities (2011). With the exception of the GPI of four Chinese cities by Wen and colleagues (2007), the remaining studies were conducted as part of larger regional studies or drew on previous regional work. The calculation method for all studies, with the exception of the Edmonton, Canada study, follow the framework suggested by Cobb, Halstead and Rowe (1995) with consideration of potential adjustments made in subsequent GPI analyzes (see for example, Anielski and Rowe, 1999; Hamilton, 1999; Talberth et al., 2006). The Edmonton study (Anielski and Johannessen, 2009) reports results by wellbeing domain and does not attempt to monetize indicators. The study may be better labeled a wellbeing assessment, raising the question of whether monetization is a requirement of the GPI methodology. The City of Edmonton GPI reports 51 wellbeing indicators for the period 1981–2007. Edmonton is using the comprehensive indicator system to inform the City's long-range strategic plan, budgeting and decision-making (Anielski, personal communication, 2012). Anielski (personal communication, 2012) notes the strength of the Edmonton GPI is the city’s ability to compare long-term trends in well-being conditions relative to GDP growth. According to the Edmonton GPI report, real GDP per capita increased by 22% between 1981 and 2008 while the GPI index decreased by 5% (Anielski and Johannessen, 2009).

Adapting macro-measures of wellbeing to local scale applications raises several challenges. The availability of local data is a significant concern. Studies typically rely on extrapolating regional and national data, and adopting proxy methods given large data gaps at the local level (Anielski and Johannessen, 2009; Bagstad and Ceroni, 2007;

Costanza et al., 2004). Using regional and national data compromises the ability of results to inform specific planning decisions bringing into question the utility of results (Wilson and Grant, 2009). Boundary issues pose a further challenge, limiting local scale calculations. For example, it is difficult at the city level to track imports and exports of market goods and services, and even more so resource and waste flows. Urban centres rely on vast regions outside of their physical borders for material and energy inputs and waste outputs (Rees, 1997). Local GPI studies are intended to support decisions around community wellbeing. Many factors that drive the GPI, however, do not fall under local jurisdictional authority (Posner and Costanza, 2011; Wilson et al., 2012). Real change in GPI values require a coalescing in policy changes at different levels of government. Given the challenges, Posner and Costanza (2011) conclude in a review of GPI studies at multiple scales that the national level is the most useful and reliable spatial scale of analysis.

The strength of local studies is that they stimulate discussion about what constitutes community wellbeing and how to achieve it. More importantly, such studies highlight the broad importance of basing public policy and community decision-makingon social, environmental and economic criteria (Anielski and Wilson, 2006; Constanza et al., 2004). Local context can be incorporated by inviting community input into the process. For example, Anielski and Wilson (2006), in an assessment of community wellbeing for Leduc, Alberta, engaged community members to discuss wellbeing and community values as a basis for defining what indicators to track. As a result of the engagement process, the project identified 120 indicators organized into 23 different wellbeing domains. The domains are grouped and reported by capital accounts: human, social, natural, built and financial assets. The model offers an approach to bridge GPI type metrics with community concerns (Anielski, 2007). The City of Guelph has adopted the CIW framework to advance a Social Wellbeing Plan. The wellbeing domains and priorities were informed by community engagement events (Smale, personal communication, 2012). Any success, however, in widely reporting GPI and other community wellbeing metrics at the local scale depends on access to community-derived data. Communities must begin tracking information related to sustainability and wellbeing themselves (Posner and Costanza, 2011; Wilson and Grant, 2009). Adapting existing metrics will not be perfect for local applications. Communities must understand limitations of approaches and fit efforts within a larger decision-making support framework. Including the GPI or related metrics as part of a basket of indices, however, will contribute to a more nuanced understanding of community wellbeing (Shmelev and Rodriguez-Labajos, 2009; Wilson and Grant, 2009).

2.8 Reflections on the Canadian Experience

The pioneering work of GPI Atlantic, and the Alberta GPI established Canada as a leader in advancing early GPI accounting and indicator frameworks at the turn of the millennium. The climate was optimistic. In 2000, former Finance Minister, Paul Martin, dedicated nine million dollars to the National Round Table on the Environment and the Economy to develop national sustainable development indicators. Mr. Martin noted that this initiative might well be one of the most important elements of his 2000 budget (Colman, 2002). The Pembina Institute released the Alberta GPI in April 2001: it included a replication of the original US GPI monetary measure of progress as an alternative to the GDP as well as non-monetary indicators of wellbeing that comprised a 51-indicator composite index (Anielski et al., 2001). In the autumn of that year, the Atkinson Foundation formed the Canadian Index of Wellbeing, which brought together Canada's leading quality of life, wellbeing and sustainable development indicator experts to construct the world's first index of wellbeing. The process championed by the Honourable Roy Romanow, former Premier of Saskatchewan, brought substantial political influence. The hope of the respective pioneering organizations from the onset was formal government adoption of a GPI type framework leading to a standardized approach for use by federal and provincial governments across the country.

Over a decade later, no government (either federal or provincial) has adopted GPI related metrics, nor appear to be engaged to do so at any point in the immediate future. The National Round Table on the Environment and the Economy will be wrapping up operations in 2013 after having had its federal funding cut. Efforts to advance an Alberta

GPI framework have not resulted in any formal adoption of GPI by the Provincial government. Pembina, the lead organization behind the Alberta GPI, has no plans to update the metric. In Nova Scotia, the New Democratic Party (NDP) government has not signaled a commitment to report GPI despite calling for the adoption of GPI measures as opposition party prior to the last election in 2009 (Colman, 2009). The CIW currently maintained by the University of Waterloo was formally released in 2011, an important success. With the current focus on the economy in Canada, however, the index has not been able to initiate the necessary national dialogue to support political adoption (Anielski, personal communication, 2012; Smale, personal communication, 2012). The architects of the CIW do not see any immediate uptake by the federal government (Smale, personal communication, 2012). Statistics Canada has discontinued many environmental surveys and the mandatory long form census, key data sources to support wellbeing indicator work.

Within Canada, non-government organizations and academics led the development of early GPI efforts. Their motivations were, in part, a response to doubts regarding the contribution of an economic growth agenda to human wellbeing (Anielski, 2001; Colman, 2000). GPI metrics were put forth to replace the GDP as a measure of societal welfare. The federal government and provincial governments appear reluctant to support non-economic growth based measures based on the assumption that they may convey messages that undermine economic growth and job creation or raise doubts regarding key economic priorities. Federal examples include oil sands development, the Keystone Pipeline and resource development more generally. Disbanding the NRTEE and ending reporting on key environmental indicators imply a strategic interest to restrict tracking and reporting data that may contradict the current federal Conservative government's economic priorities.

The CIW has attempted to shift the focus from challenging the economic growth paradigm toward better accounting for wellbeing. It explicitly states that the CIW composite index and GDP (proxy for economic growth) are not in conflict and encourages reporting the metrics in tandem. Further, the notion that CIW should replace

GDP outright as Canada’s benchmark indicator of progress is absent in their messaging (Michalos et al., 2011). The position follows what is happening in the United Kingdom, where the David Cameron Conservative government is explicit that new wellbeing measures will not replace traditional economic growth indicators (Cameron, 2010a; Cameron 2010b). The focus has changed from using alternative metrics to question failings of the GDP and economic growth toward promoting a growth platform with fewer associated environmental and social costs.

Speaking about the Alberta GPI, Anielski (personal communication, 2012) notes government economists supported the idea of correcting for some societal and environmental costs that were otherwise counted as additions to the GDP. They were more reluctant, however, to endorse the GPI as an alternate composite index or replace GDP. Costanza and colleagues (2004) and Anielski (personal communication, 2012) cite an entrenched familiarity and reliance on traditional economic indicators among established organizations as a barrier preventing wider adoption of GPI metrics. The System of National Accounts has been in use and evolving since World War II. Coming up with an international standard for GPI-type accounting and protocols is a daunting task. Critics of the Alberta GPI accused the work of indicator selection bias (i.e. picking indicators that would make Alberta's progress look poor) and giving equal weight to each of the 51 indicators in the Alberta GPI. The sentiment was that economic indicators should be prioritized (Anielski, personal communication, 2012). Stiglitz argues that in the United States (we presume this applies to Canada as well) political interference from lobby groups undermines the development and adoption of new metrics. Many industries feel threatened that changing the emphasis of what we measure will lead to public sentiment and policies that ultimately affect how they operate (Stiglitz, 2008). Whether it is political interference, society’s general malaise for change, or the difficulty in designing new metrics and models, meaningful progress towards suitable alternatives has been disturbingly slow. Critics have been questioning the merits of economic growth for over half a century.

2.9 Conclusion – Moving Beyond GDP

GPI type measures have been produced for over 20 years with limited uptake in mainstream policy arenas. Most calculations have been one-off or short-duration exercises largely led by the academic community and non-governmental organizations. With the exception of the State of Maryland, no jurisdiction currently calculates a GPI in lieu of GDP or alongside GDP as part of regular statistical reporting. In 2009, The State of Maryland calculated their GPI and developed a GPI forecasting tool to support policy and planning decisions. Under current leadership, the State appears committed to updating GPI estimates on a regular basis, although they have not yet done so or outlined a plan forward (State of Maryland, 2011). The State Governor has been an important political champion of the GPI. This is perhaps the critical story; adoption of GPI metrics require strong political champions combined with the capacity to undertake these complicated calculations in a robust fashion.

Several countries in Europe are considering adopting a "Green GDP" or measure of wellbeing. In 2009, the European Parliamentary Commission issued a roadmap outlining a five-year process for moving beyond GDP (Commission of the European Communities, 2009). Nicholas Sarkozy, former President of the French Republic, established The Commission on the Measurement of Economic Performance and Social Progress led by Joseph Stiglitz, Amartya Sen and Paul Fitoussi to identify limits of GDP and consider additional information required for the production of more relevant indicators (Stiglitz et al., 2008). The United Kingdom Office of National Statistics released the country's first set of wellbeing indicators in November, 2012 as part of their Measuring National Wellbeing (MNW) programme. The programme, which began with a six month national debate about 'what matters' aims to report annually a trusted set of national statistics to help citizens understand and monitor wellbeing (Self et al., 2012).

The traction in Europe to go beyond the GDP is positive. From a Canadian viewpoint, the wider debate and discussion seem to reiterate 20 years of effort. In terms of progress, are governments really moving forward? In Canada, much of the early momentum has stalled. Even in Europe, the language supporting new metrics is clearly as an addition to existing economic growth metrics. No evidence suggests that these are replacement measures or signal a new economic direction. GPI metrics broaden what we measure to include the contribution of social and environmental factors toward human welfare. The current emphasis has not been on redefining a new economic worldview reflective of sustainability principles but on tweaking what we measure to correct for perceived flaws in the GDP as a surrogate of wellbeing. Adopting a GPI does not necessarily change the underlying structure of the economy and its imperatives. GPI can, however, be used to argue for that. Increasingly, however, GPI and related metrics are advanced within an economic growth framework. In an era of sustainability urgency, GPI metrics are insufficient on their own to change dramatically the underlying framework of economic decision-making. Biophysical metrics based on physical flows, for example, are necessary to capture the throughput impacts, which drive economic activity. Changing resource use, consumption patterns and development pathways requires a suite of robust tools that challenge the economic growth agenda.

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Chapter 3: Ecological Economic Accounting – a Review of Biophysical Measurement Tools

3.1 Introduction

Within conventional and ecological economics, we count, measure and model to allocate resources efficiently. While efficient allocation is critical to both economic views, the underlying assumptions governing how to allocate resources efficiently differ. The objective within conventional economics is to advance human welfare (or, depending on the level of cynicism, to maximize profits for the owners of capital). Emphasizing human welfare as a basis of economic decision-making, while vitally important, fundamentally neglects to recognize that all economic activity relies on the environment as a source of inputs and sink for wastes (Georgescu-Roegen, 1971; Farley 2008). Ecological economics acknowledges a maximum scale to the economy dictated by ecological limits. Within ecological economics the emphasis of measurement becomes not just how to allocate resources to improve human welfare but how to allocate resources efficiently within ecologically defined parameters.

To direct society on a more sustainable trajectory, ecological economists argue for the use of accounting tools that quantify the biophysical impacts of economic activity as a basis for resource allocation decisions. Chapter 3, a bridge chapter, briefly reviews biophysical measurement tools as a subset of ecological economic accounting. Section 3.2 expands on the rationale supporting use of biophysical accounting tools. Section 3.3 reviews select biophysical accounting tools used in urban and regional sustainability assessments. Section 3.4 highlights challenges of biophysical accounting. Section 3.5 differentiates between the use of biophysical tools to support eco-efficiency objectives and to support the principle of sustainable scale. The chapter concludes by stressing the importance of biophysical measurement tools to advance scale as fundamental decision-making criterion.

3.2 Accounting for Biophysical Impacts

Ecological economics criticizes conventional economics for failing to consider adequately the depletion of natural capital and role of ecosystem goods and services in supporting the production function (Costanza et al., 1991; Daly 1991; Hardin, 1991; Daly, 2011). By considering the economy separate from and independent of the environment, conventional economic decision-making tools are incapable of addressing sustainability challenges (Daly, 1991; Daly and Farley, 2004; Rees, 1995). Conventional economics attempts to integrate environmental sustainability into economic decision-making by assigning a price to ecosystem goods and services. By assigning prices, ecosystem goods and services become commodities allowing markets to determine their allocation. Markets under ideal market conditions (competition, complete information, lack of interference to enter and exit the market, and inclusion of externalities) maximize the monetary value of both inputs and outputs creating a pareto-optimal situation (Farley, 2008, Norgaard and Howarth, 1991). Assigning prices to ecosystem goods and services assumes that we have the capacity to assign meaningfully monetary values to complex ecosystem functions and services. It also assumes prices are an appropriate indicator of scarcity and will signal when to conserve critical natural capital and ecosystem services (Farley, 2008). Prices reflect scarcity of a good in relation to demand (Limnios et al., 2009). They inadequately consider the ‘use’ value of a good or service, the impact of consumption on the environment, and the overall state of natural capital (Daly, 1990; Farley, 2008; Norgaard and Howarth, 1991). Further, prices reveal little about the underlying flows of energy and material and associated environmental impacts required to support economic activity (Brown and Herendeen, 1996; Farley, 2012; Gómez-Baggethun et al., 2010; Norton and Noonan, 2007; Pelletier and Tyedmers, 2011; Vatn and Bromely, 1994). Prices poorly account for the environmental dimensions of economic activity.

Biophysical accounting tools measure the biophysical dimensions of economic activity. The various tools focus on quantifying energy and material flows and environmental impacts as a means to advance sustainability recognizing that no economic activity can occur without energy conversion and entropy production (Hall and Klitgaard, 2006). Impacts are characterized independent of markets and reflect the thermodynamic reality that all economic activity requires inputs of energy and material from nature. A vision of the human economy as a subset of a finite biosphere explicitly acknowledges the importance of natural capital accounting (Daly, 1990; Daly, 1991; Ekins, 1992; Rees and Wackernagel, 1997). The biosphere is the source of inputs driving the economy and the sink for wastes. Issues of scarcity, value, and allocation necessarily have an environmental dimension (Hall and Klitgaard, 2006). Biophysical accounting tools provide an ecological basis to help inform decision-making critical for addressing sustainability challenges (Haberl et al., 2007; Herendeen, 1999; Holmberg et al., 1999; Wackernegal and Rees, 1996).

3.3 Techniques

Various sustainability assessment tools have been developed or adapted from other disciplines to account for the biophysical dimensions of the economy. The predominant approaches to measure regional and urban environmental sustainability can be categorized into three main approaches: energy and material flow accounting, metabolism based accounting, and consumption based accounting (Baynes and Wiedmann, 2012; Browne et al., 2012; Herzi and Davis, 2006; Barles, 2010). It is important to recognize overlap between the groupings. The language characteristic of the different approaches is often used interchangeably in practice (Barles, 2010; Baynes and Wiedmann, 2012). Greenhouse gas reporting, for example, may use an energy flow accounting framework but be referred to as a carbon footprint. Furthermore, reporting can be defined by geographic boundaries characteristic of urban metabolism studies or factor in embodied energy associated with the consumption of goods and services. The suggested categories are congruent with Nees and colleague's (2007) sustainability assessment framework, which makes an important distinction between tools by focus (indicators, integrated assessment, and product related assessment). Indicator assessments include regional flow indicators (energy and material analysis) and integrated indicators.

Integrated indicators encompass GPI and wellbeing indices (discussed in Chapter 2) as well as the ecological footprint. Metabolism based approaches are categorized as integrated assessments. The remaining category, product related assessments includes techniques such as life cycle assessment and exergy analysis.

3.3.1 Energy and Material Analysis

Energy and material analysis quantify resource inputs to support an economy, population, geographic area, or system for a defined period of time (Browne et al., 2012). Energy flow analysis has a long history within economics given the importance of energy as a driver of economic activity (Georgescu-Roegen, 1975; Nordhaus et al., 1975; Odum, 1973; Odum and Odum, 1976). Energy analysis is defined as the process of quantifying the direct and indirect energy requirements to support a system (Brown and Herendeen, 1996). Energy use was adopted as an indicator of environmental impact starting in the 1970s because of the increased focus on the energy-economy dependency brought to attention during the oil embargos (Brown and Herendeen, 1996). The original concern was largely around issues of energy supply. As a sustainability indicator, energy use regained prominence in the 1990s due to the greenhouse gas implications of energy consumption (Brown and Herendeen, 1996). Tracking energy use and greenhouse gas emissions has been an integral component of sustainability reporting in the post- Kyoto era (Hertwich and Peters, 2009; Kok et al. 2006; Lenzen and Peters 2009; Moll et al. 2005; Reinders et al., 2003).

Material flow analysis quantifies the material inputs supporting economic activity for a defined region and often includes energy as a category. Material flow analysis has been widely undertaken, especially in Europe (Eurostat 2002; Weisz et al., 2006). In attempt to standardize approaches, Eurostat (2001) defined an economy-wide material flow methodology and identified reporting indicators such as Total Material Input and Direct Material Input. Reducing the material and energy intensities of economic activity has been a government strategy to achieve sustainable development objectives since the Bruntland Report (Commission of the European Communities, 2001; World Commission on Environment and Development, 1987). Schmidt-Bleek (1993) developed Material Input per Unit of Service (MIPS) analysis to measure the direct and indirect material inputs needed to generate an economic good or support an economic process. The MIPS methodology underlies the Ecological Rucksack model and the Factor 10 economy, a concept to describe the need for an improvement in resource productivity by a Factor of 10 in Germany and other Western style economies (Schmidt Bleek, 1993). Factor 10 focuses on delinking energy and resource inputs from GDP growth.

3.3.2 Metabolism-Based Accounting

Metabolism studies account for system wide impacts of production and consumption within geo-political boundaries (Kennedy et al., 2007). Wolman first spoke of the metabolic requirements of a city as 'all the materials and commodities needed to sustain the city's inhabitants at home, at work and at play" (1965: 179). Wolman's (1965) use of the metabolic image reflects his view of the earth as a closed ecological system and the countless inputs and outputs required to sustain a city (1965). The premise behind urban metabolism accounting is cities depend on large linear material and energy flows from regions outside their geo-political borders (Brunner, 2007; Folke et al., 1997; Kennedy et al., 2007; Rees, 1997; Wackernegal and Rees, 1996). The reliance on hinterlands for material and energy inputs and disposal for wastes make cities highly dependent on the ecological health of global ecological systems from which inhabitants are largely isolated (Brunner, 2007).

Urban metabolism methods are similar to energy and material flow analysis, but often include water and waste flows. While more recent studies emphasize energy and greenhouse gas emissions, Wolman's seminal paper focused on water, air pollution, and sewage (Kennedy et al., 2011; Wolman, 1965). Urban metabolism accounting waned in the 1980s but regained prominence in the 1990s with growing interest in sustainability reporting (Barles, 2010; Kennedy et al., 2011). Urban Metabolism studies provide a useful framework for sustainability indicators and organizing data essential for GHG reporting (Newman, 1999; Kennedy, 2009; Kennedy, 2010; Sahely et al., 2003). Studies usually are hierarchical with results comparable at different scales: household, neighbourhood, community, and metropolitan scale (Baynes and Wiedmann, 2012). For a comprehensive review of urban metabolism studies, see Kennedy and colleagues (2011). Metabolism studies are traditionally defined by geopolitical borders. Increasingly, however, they include important cross boundary impacts associated with production and consumption. See Baynes and Wiedmann (2012) for a detailed list of extended territorial studies.

3.3.3 Consumption-Based Accounting

Consumption based accounting quantifies the impacts of consumption by a defined population (Baynes and Wiedmann, 2012). The predominant consumption based tool used for regional and urban environmental sustainability assessments is the ecological footprint. Given the prevalence, Barles (2010) and Brown and colleagues (2012) use ecological footprint as an urban sustainable assessment category and avoid the term consumption based accounting altogether. The more general category nomenclature allows for the inclusion of hybrid techniques such as carbon footprints, environmental footprints, and consumption based GHG indicators. Within the consumption-based accounting category, I suggest techniques that quantify human carrying capacity as a distinct sub-category. These include the ecological footprint and human appropriated net primary productivity (HANPP) measure. The distinction hinges on tools that, implicit in their design, compare consumption against ecological capacity. Human carrying capacity-based indicators highlight the connection between the economy and the biosphere. The tools relate overall material and energy throughput of human activity in relation to ecological capacity or planetary thresholds.

Ecological Footprint

The ecological footprint is the most widely applied indicator of consumption used in regional and urban sustainability assessments. The ecological footprint is defined as a measure of the demand populations and activities place on the biosphere in a given year, given the prevailing technology and resource management of that year (Borucke et al., 2013). Rees, building on Borgstrom's 'ghost acreage' concept to describe imported agriculture capacity, discussed early variations of the ecological footprint starting in the 1970s. Rees's intention was to communicate that humans can exceed the resource base of their immediate geographic area by importing ecological capacity from elsewhere (Wackernagel and Rees, 1996). The concept was popularized with the publishing of Wackernagel and Rees's (1996) book, *Our Ecological Footprint, Reducing Human Impact on the Earth,* leading to wider use of the concept as a sustainability assessment tool. The Global Footprint Network calculates national ecological footprint estimates for more than 200 countries biannually using a standardized calculation framework (Borucke et al., 2013). Ecological footprints are increasingly calculated at the sub-national and regional level as well. See Wilson and Grant (2009) for a review of sub-national ecological footprint studies and calculation approaches. For a detailed description of the Ecological footprint including strengths and limitations see Chapters 4 and 5.

Human Appropriated Net Primary Productivity

Human Appropriated Net Primary Productivity (HANPP) measures the amount of available solar energy used to support human demand (Vitousek, 1986). The biologically based indicator developed by Vitousek and colleagues (1986) and refined by Wright (1990) and Haberl (1997) measures the human domination of ecosystems by tracking human use of ecological energy flows within a spatial area. HANPP is the net amount of carbon assimilated in a given period by vegetation being altered by humans or harvested for human use (Haberl et al., 1997). As a measure of human carrying capacity, HANPP can be used to argue that human activity is restricted by the accumulated biomass generated by solar energy. HANPP is rarely considered in regional and urban sustainability assessment frameworks (Baynes and Wiedmann, 2012; Browne et al., 2012; Nees et al., 2007). The technique has been primarily calculated and referenced at the global level communicating human dominance of biospheric production (Haberl et al., 2007; Milesi et al., 2005; Vitousek et al., 1986; Vitousek et al., 1997; Zhao and Running, 2010). HANPP can be applied at the regional level. A notable Canadian study is O'Neill, Tyedmers and Beazley's (2007) estimate of HANPP in Nova Scotia. With the increasing sophistication of satellite imagery HANPP can be calculated more easily at the regional level. Milesi and colleagues (2005) have calculated global NPP at 0.5° resolution. Anielski and Wilson (2009) used satellite images to derive NPP estimates in a study estimating the value of the Canadian boreal forest. Stechbart and Wilson (2010) similarly used satellite image data to estimate biocapacity for Ontario.

3.3.4 Methodological Pluralism

The different biophysical-based approaches for assessing regional and urban environmental sustainability share a similar end goal (accounting for environmental sustainability). The techniques reflect different purposes, philosophical underpinnings, and methodological assumptions. Clearly, some techniques will be better suited for certain contexts. Even when using the same technique, researchers can adopt different calculation strategies depending on purpose, scope of study, impacts or indicators included, and data availability. For example, sub-national ecological footprint estimates can be calculated using a top-down, bottom-up or mixed approach (Wilson and Grant, 2009). Findings will differ reflecting the approach used, supporting a call for methodological pluralism (Baynes et al., 2012; Browne et al., 2012; Niccolucci et al., 2007; Wilson and Grant, 2009). The goal is building a repertoire of information based on well-executed studies to support decisions that direct society down an environmentally sustainable path.

3.4 Challenges

Several challenges constrain the wider adoption of biophysical accounting tools. Challenges discussed here are general limitations and do not necessarily apply to all biophysical accounting tools. A paramount concern continues to be a lack of data to populate models and questionable quality of data. In cases where data are not available, studies often extrapolate data from other sources introducing potential error. (Barles, 2010; Baynes and Wiedmann, 2012; Browne et al., 2012; Sahely et al., 2003; Wilson and Grant, 2009). For example, many efforts to estimate sub national ecological footprints adjust national data based on population, expenditure or other variables. Trying to measure, quantify and understand the functions, services, and capacities of ecosystems is difficult. Biophysical measurement tools attempt to simplify complex, dynamic information about ecological systems. Bjorklund (2002) describes an epistemological uncertainty inherent in biophysical measurement. Imperfect knowledge of ecosystems and the complexity of trying to understand the dynamics of ecological sustainability hinder the robustness and integrity of modeling efforts. Uncertainty enters all phases of analysis due to problems such as ignorance about parts of ecological systems, oversight of impact categories, and the uncertainty that characterizes attempts to understand future systems and impacts (Bjorklund, 2012; Giampietro, 2002).

Problems arise when deciding how to analyze, aggregate and present data. Meaning is lost in integrating complex information into impact categories or in transforming results into dimensional indicators (Morse and Fraser, 2005; Spangenberg, 2007; Udo de Haas, 2006; Wilson and Grant, 2009). Trying to interpret results is challenging. Impact categories are often given the same weight even if severity of impacts are quite different (van den Bergh and Verbruggen, 1999). Several tools emphasize one dimension or a subset of environmental factors. Results provide a partial picture of impacts and effects associated with a specific population, activity, system, or process (Baynes and Wiedmann, 2012; Browne et al., 2012). Most of the prevalent biophysical measurement tools present results that are static in time making them inadequate as forecasting tools. Further, many biophysical accounting tools still integrate prices as a means to allocate resource flows and associated impacts to different end uses and consumption categories. Results lose meaning when integrating monetary dimensions (Brown and Herendeen, 1996; Pelletier and Tyedmers, 2011). Monetary values are a poor indicator of ecological scarcity or ecosystem value (Salles, 2011; Spangenberg and Settele, 2010). Furthermore, the intent of focusing on biophysical dimensions of economic activity is to generate an understanding of impacts and flows independent of monetary factors.

Non-experts are often confused regarding the purpose and methods of different instruments making their use vulnerable to oversimplification and misrepresentation (van den Bergh and Verbruggen, 1999; Wilson and Grant, 2009). Further, inconsistency in methodological approaches, boundaries, and assumptions are not comparable between studies. The inconsistency undermines confidence in tools (Neumayer, 1999; Posner and Costanza, 2011; Wilson and Grant, 2009). Efforts to standardize approaches and advance best practices offer an opportunity to create conformity in analyses. Standardization of the LCA methodology, for example, has increased the legitimacy of life cycle assessment and improved the credibility of studies. Critically evaluating tools to address methodological concerns as knowledge of ecosystems and measurement increases is vital for wider use and adoption of biophysical measurement tools.

Biophysical accounting techniques are emergent tools. Efforts to advance and refine methods must continue. Acknowledging limitations and applying tools appropriately are critical to foster wider adoption. Researchers and practitioners have a responsibility as well to be transparent in approach, articulate methods clearly, state assumptions, and use best science. Biophysical accounting tools capture throughput demands and impacts associated with economic activity. They are integral support tools within a larger effort to advance sustainability objectives. Sustainable decision-making takes place in a dynamic milieu of economic, cultural, and social values. Biophysical quantification tools are needed to guide society toward a more ecologically sensitive and sustainable path.

3.5 Eco-Efficiency Versus Scale

Biophysical accounting tools are often used to support eco-efficiency considerations as a means to reduce the energy and material inputs driving economic activity. The emphasis focuses on reducing the impact of consumption per unit of output by becoming more efficient and relying on technology and substitution of capital to expand biophysical limits. The logic follows that improving energy and material efficiency allows us to enjoy the social and human welfare benefits of economic growth without jeopardizing sustainability thresholds. Emphasizing eco-efficiency as a strategy to reduce material and energy throughput associated with economic activity raises concerns that gains in eco-efficiency are lessened or negated by the rebound effect. The rebound effect, also called Jevons' Paradox, describes the phenomenon where greater energy efficiency triggers additional energy use so the net effect on total energy use over time is unclear (van den Bergh, 2011).[1] Alcott (2010) explains the rebound phenomenon by noting that increased efficiency relieves limits that constrain the physical dimensions of the economy. Limits relate to money, time, scarce resources, production factors and space. Daly (2011) argues that in a growth economy, overemphasizing efficiency may undermine sustainability. A resource we use more efficiently becomes cheaper. As a result, Daly argues we use more of it or reallocate spending to other goods and services. While eco-efficiency gains may result in improved social welfare temporarily by making products and services more readily available, increases in total energy-material throughput negate environmental benefits. Eco-efficiency gains are irrelevant as a sustainability objective in absence of quantitative limits on the scale of resource throughput (Daly, 2011). Eco-efficiency will only support sustainability if associated with a cultural shift toward sufficiency or an absolute decoupling of economic growth from material and energy throughput (Alcott, 2010; Sachs, 2008; Schneider, 2009). To include scale as a decision-making criterion for sustainability, biophysical accounting tools must relate impacts to global ecological thresholds or available biocapacity (Borucke et al., 2013; Rockström et al., 2009; Running, 2012).

3.6 Conclusion

The concept of a scale to human activities defines the ecological economics worldview. Genuine Progress Indicators (GPI) and related metrics are useful in accounting for the social and human costs and benefits of economic growth (Chapter 2). They fail, however, to account adequately for the impact of economic activity on

[1] Jevons' reflections (1865) noted that improved efficiency of coal fired steam engines would mean a lower cost of coal which would in turn stimulate the diffusion of coal using technologies contributing to greater coal use and subsequent more rapid depletion of British coal reserves.

supporting biophysical systems. Using GPI or a related metric and a biophysical measure in tandem would better capture the full impacts of economic activity than relying on one approach (Niccolucci et al., 2007; Sahely et al., 2003). Biophysical measures, for example, account for the impacts of economic growth on the health and integrity of ecological systems. GPI related metrics account for the societal wellbeing impacts associated with economic growth (Lawn, 2003; Max-Neef, 1995; Niccolucci et al., 2007). Among biophysical accounting tools, those that relate throughput to available biocapacity advance the concept of sustainable scale. Improving eco-efficiency of economic activity as a strategic approach to support the principle of sustainable scale is important but can be problematic given the rebound effect. The strength of carrying capacity tools is they quantify human impact in relation to ecological capacity. The concept of scale is implicit in their design.

The biophysical impacts associated with economic activity are paramount considerations poorly integrated into conventional economic models. In an era of sustainability challenges, we need decision support tools that reflect the environmental dimensions of economic activity. Biophysical accounting tools shift the emphasis away from markets and prices as a basis of decision making toward measures that explicitly account for flows of energy and material. It is not simply a matter of connecting the economy to ecological systems or changing what we measure, and how. Rethinking measurement toward an ecological economic world-view requires a dramatic shift in philosophical approach. The biosphere supports life. The economy is a subsystem of nature and not vice versa. Measurement tools must reflect a world-view where ecological goods and services underpin all economic activity.

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Chapter 4: A Review of the Ecological Footprint

4.1 Publication Information

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4.2 Introduction

The ecological footprint is a biophysical accounting tool that accounts for the environmental impact of consumption. In more technical terms, the ecological footprint provides a snapshot in time and the trajectory over time of how much nature--expressed in a common unit of bioproductive space--is used exclusively for producing all the resources (food, energy, materials) a given population consumes and absorbing the wastes they produce, using prevailing technologies (Chambers et al., 2000). The ecological footprint provides a comprehensive aggregate indicator of human pressure on ecosystems (Holmberg et al., 1999). At the macro level, if the human footprint exceeds the productive capacity of the biosphere then consumption patterns are clearly not sustainable.

While most sustainability models focus on production, the ecological footprint evaluates consumption: it highlights the role of the consumer as a driver of environmental impact. The ecological footprint is unique in that it accounts for the costs of consumption regardless of where associated environmental burden falls. For example, through trade, consumers may enjoy the benefits of consumption without experiencing the impacts in the local region. While the ecological footprint is an indicator of consumption, important factors other than consumption habits influence the ecological footprint. These include population size, technology, and gains or losses in eco-efficiency (Wackernagel and Rees, 1996).

4.3 Description

William Rees and Mathis Wackernagel developed the ecological footprint concept was developed (1996). Several modifications to the original model have been adopted since that time, notably by Kitzes et al. (2008), Kitzes et al. (2007), Wackernagel et al. (2005), and Wackernagel et al. (1999). The Global Footprint Network, the leading authority on ecological footprint analysis, maintains the National Ecological Footprint Accounts, bi-annually updating ecological footprint values for 241 countries (Ewing et al., 2010). In efforts to ensure continuity and consistency in calculation methodology, a team of ecological footprint experts began developing calculation standards in 2006 under the umbrella of the Global Footprint Network. The current version (2009) includes a national calculation standard and a standard for sub-national footprint studies (sub-national populations, organizations, and products).

The ecological footprint methodology has been extended to account for several missing components not considered in the standard calculation approach. These are pollutants (Bai et al., 2008; Peters et al. 2006); water (Chapagain and Orr, 2009; Hoekstra and Chapagain, 2007); disturbed land (Lenzen and Murray, 2001); non-CO_2 greenhouse gas emissions (Hanafiah et al,. 2010; Holden and Hoyer, 2005; Walsh et al., 2009); nutrient emissions (Hanafiah et al., 2010); and non-renewable resource consumption (Nguyen and Yamamoto, 2007).

In addition, several researchers have proposed promising methodological advancements. Erb et al. (2009), Venetoulis and Talberth (2008) and Haberl et al. (2007) recommend using net primary productivity (NPP) or human appropriated net primary productivity (HANPP) as a means to better account for biocapacity. Siche et al. (2010), Liu et al. (2008), Chen and Chen (2006) and Zhao et al. (2005) have advanced an ecological footprint calculation approach based on the emergy concept. Li et al. (2007), Collins et al. (2006), Hoekstra and van den Bergh (2006), Wiedmann et al. (2006), and Bicknell et al. (1998) allocated ecological footprint impacts to consumption activities using input-output modelling. More recently, Wiedmann (2009) and Wiedmann et al. (2007) have recommended using a multi-regional input-output (MRIO) method to account for changes in production from region to region dramatically improving the ability of the ecological footprint tool to account for the embodied costs of trade flows. For a more detailed discussion of methodological advancements and research needs to further enhance the ecological footprint, see Wiedmann and Barret (2010) and Kitzes et al. (2009).

A major criticism of the ecological footprint concerns how to account for energy consumption. The standardized methodology measures the hypothetical forest land needed to sequester the associated CO_2 emissions. The creation of hypothetical land disconnects the ecological footprint from actual ecological systems and overstates the true land area required to support a given population (Hueting and Reijnders, 2004; van den Bergh and Verbruggen, 1999). The ecological footprint has also been widely criticized for not distinguishing between sustainable and unsustainable yields (Lenzen et al., 2007; Ferng, 2005). In addition, land use is associated with single functions, ignoring that different land use categories may provide multiple services or functions (van den Bergh and Verbruggen, 1999).

4.4 Quality of Life

The ecological footprint, as a key determinant of quality of life, reflects the importance of sustainability in expanding people's choices. The ecological footprint measures the amount of natural capital required to support human consumption (Rees and Wackernagel, 1997). The concept supports a strong sustainability position, which argues that natural capital underpins all economic activity and is the foundation of social and economic well-being. Other indicators of quality of life (income, happiness, health) are secondary and depend on maintaining critical levels of natural capital.

Governments, communities, and organizations increasingly report the ecological footprint as a macro indicator of sustainable resource use in environmental reporting and sustainability indicator studies (see for example, Collins et al., 2006; Dawkins et al., 2008; Sustainable Sonoma County and Redefining Progress, 2002). Well-known quality of life metrics like the Calvert Henderson Quality of Life Indicators and the Environmental Sustainability Index include the ecological footprint as a sub-indicator (Calvert Group and Henderson, 2006; World Economic Forum, 2005). The ecological footprint has also been included as part of genuine progress reporting and health indicator reporting (see for example, Anielski, 2007; Rainham and McDowell, 2005).

The United Nations Development Program (UNDP) identified the ecological footprint as an indicator to consider when evaluating human development. In *Human Development and Sustainability,* Neumayer (2010) argued for the need to report the ecological footprint as an external sustainability qualification in reporting the Human Development Index (HDI). Several authors previously suggested combining the ecological footprint with the HDI. Morse (2003) and Hermele (2006) proposed developing a green or sustainable HDI respectively by adding the ecological footprint as a component of the index. Wilson, Pelot, and Tyedmers (2007) suggested including the ecological footprint as an external qualifier or trump variable. Linking the ecological footprint with the HDI consistently demonstrates that almost all countries with high human development also report large ecological footprints highlighting the need to break the connection between advancing human development and depleting critical stocks of natural capital.[1] Combining development metrics with an ecological threshold would prevent these metrics from promoting development trajectories that neglect or potentially jeopardize environmental sustainability at the expense of social and economic aspirations.

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[1] Cuba is an exception.

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PART 2: ACCOUNTING FOR ENVIRONMENTAL IMPACT AT THE LOCAL LEVEL

"Cease being intimidated by the argument that a right action is impossible because it does not yield maximum profits, or that a wrong action is to be condoned because it pays."

- Aldo Leopold

Overview:

Part 2 starts from the premise that operationalizing ecologically informed decision-making at the local level requires the adoption of biophysically based measurement tools to support sustainability assessments. Part 2 includes two chapters that explore different approaches to account for environmental impact of human consumption at the local level. Chapter 5 tests a calculation strategy using the ecological footprint method to account for environmental impact that could be widely adopted by municipalities across Canada. Chapter 6 responds to concerns that municipal-wide results are useful for education and awareness regarding environmental impact but lack specificity to inform policy and planning decisions. Using the Town of Oakville, Ontario, as an example, the chapter reports environmental impact by neighbourhood to confirm if finer scale analysis make assessments more relevant to planners and policy makers.

Chapter 5: Calculating Ecological Footprints at the Municipal Level: What is a Reasonable Approach for Canada?

5.1 Publication Information

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5.2 Abstract

No clear ecological footprint calculation strategy is available for small and mid-sized communities within Canada. By adjusting provincial or national footprint findings using data sets available in the public domain we develop and test a calculation strategy to estimate municipal ecological footprints. Because the calculation approach is consistent with the Global Footprint Network Standardized methodology it permits meaningful comparisons between communities and with global and national footprint estimates. It offers planners, policy makers, and community leaders an accessible, straight forward and cost effective strategy for estimating the ecological footprint at the community and municipal level. The suggested approach is best suited for using the ecological footprint as an awareness and education tool. The large number of limitations associated with calculating the municipal approach and limitations associated with calculating ecological footprints in general at the local level make it an unsuitable tool to inform community planning and policy development.

5.3 Introduction

The ecological footprint enjoys a high level of application and awareness within Canada. As the concept permeates the consciousness of Canadians, communities express increasing interest in calculating their ecological footprints. Community planners, policy makers, and leaders see the ecological footprint as a tool to measure the state of sustainability in their communities, raise awareness around sustainability issues and assist them with community planning and development to achieve sustainability goals. Despite the optimistic interest, potential adopters of the ecological footprint often experience frustration because of the lack of a consistent and accessible footprint methodology to use at the municipal level and the unavailability of data to undertake a robust footprint calculation.

Central to calculating the ecological footprint are data to understand the flows of energy and material into and out of a study region. At the country level, Canada does an excellent job of tracking domestic production, imports and exports. Below the national level, however, it becomes problematic to account for the flows of material and energy. Typically, the smaller the political jurisdiction the more challenging the calculation becomes. Imagine trying to account for all the movements of goods, services and the associated material and energy inputs within the political borders of a municipality. As the scale of analysis becomes more refined from the national to the community level, the availability of data to calculate the ecological footprint becomes increasingly limited. In most cases, community level footprints rely on significant assumptions, which reduce the accuracy of findings, thereby restricting potential uses of the tool. Accordingly, applications of the ecological footprint at smaller scales need to be interpreted and used with caution. While the ecological footprint can be a valuable tool for municipalities, potential adopters must understand the strengths and limitations to avoid potential disappointment and improper use of the findings.

The Global Footprint Network (referred to hereafter as GFN) endorses and updates a national ecological footprint calculation methodology (GFN, 2005). The calculation method underlying the National Footprint Accounts offers a rigorous, peer reviewed approach that provides a consistent methodological framework to estimate ecological footprints. No accepted calculation methodology to derive sub-national ecological footprints is currently available. GFN has issued "Standard 3" for sub-national population calculations (GFN, 2006). The standard is not a specific calculation methodology but a set of requirements to make sub-national footprint results consistent with the GFN National Footprint Accounts and to ensure calculations apply good research reporting practices.[1] The Standard does not specify a detailed calculation approach on how to adapt the national footprint results to a sub-population. Within Canada, sub-national ecological footprints have been estimated without any continuity in the calculation approach.

Communities face several challenges when considering how to develop an ecological footprint calculation strategy. Ecological footprint projects are time consuming and data intensive. Because communities generally lack staff expertise to estimate their ecological footprint, they must hire consultants for the work. Projects prove costly to undertake for jurisdictions with limited budgets. Data availability poses a major challenge. Detailed material and energy consumption data are rarely available at the community level. As such, communities need to develop a modified calculation approach when estimating their ecological footprints.

The lack of a common ecological footprint calculation strategy has led to a variety of calculation approaches across Canada resulting in inconsistent and even conflicting findings. Inconsistent results among studies can potentially undermine the credibility of the tool and derail interest in the ecological footprint. Furthermore, using different

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[1] The 2006 Standard specifies three key points (GFN, 2006: 8-9). 1) The study calculates sub national footprints by adapting the national results for the population under consideration. 2) The method applied is consistent with the National Footprint Accounts so when applied to all non-overlapping sub-national regions, the sum of regional results equals the National Footprint Accounts national results. 3) The study needs to make explicitly clear what methods are used to construct sub-national accounts.

calculation approaches limits comparability of results across jurisdictions. A consistent calculation framework is essential to convince potential municipal adopters that the ecological footprint is a credible measurement tool.

General misconceptions about how the ecological footprint can help communities achieve sustainability goals are commonplace. The potential uses of the ecological footprint, as with many sustainability tools, are being oversold (McManus and Haughton, 2006; Barrett et al., 2004). Potential adopters must understand how the ecological footprint can best serve their communities to avoid potential disappointment when using the results.

In light of these challenges, we developed and tested an ecological footprint calculation strategy that offers a consistent and accessible calculation approach for Canadian municipalities. Our proposed approach follows a simplified calculation strategy using proxy indicators and drawing solely on data available in the public domain. The approach is specific to Canada, recognizing that the type of data tracked and available at the municipal level differs by country. While we focus on developing a calculation approach for Canadian municipalities, the general philosophy, lessons, limitations and discussion apply to municipal and community based ecological footprint efforts in other countries.

We organize the paper into four major sections. First, we review methodological frameworks for calculating the ecological footprint of municipalities with an emphasis on approaches applied in Canada. Next, we propose a calculation strategy accessible for municipalities using data sets in the public domain. The third section tests the calculation approach on thirteen municipalities in Alberta. We conclude by discussing the potential and limitations of the proposed calculation approach and the ecological footprint in general as a tool for community education and planning.

5.4 Municipal Ecological Footprint Calculation Strategies

Although the ecological footprint has been estimated for numerous cities and sub-national populations, no consistent calculation framework has emerged. We review prevalent calculation approaches used for small urban centres and municipalities emphasizing ecological footprint studies of Canadian communities. Our analysis highlights the latest direction and most prevalent community level calculation strategies, rather than providing an exhaustive review of sub-national ecological footprint studies. Lewan and Simmons (2001) conducted a review of ecological footprint analyses of European cities. Their research and follow up work with practitioners across Europe informed the Global Footprint Network sub-national calculation standard (Lewis, personal communication, 2009).

Prevalent calculation strategies examined include indicators as proxies to adjust national data (indicators approach), resource flow models, community based questionnaires, and the household expenditure survey approach. While each approach has notable strengths and supported key ecological footprint projects, each presents challenges for widespread application in Canada.

5.4.1 Indicators Approach

The indicator approach estimates the ecological footprint for sub-national populations by adapting national ecological footprint estimates based on comparing indicators of consumption between the sub-national population and the national average. This widely used approach compares common data sets between the study region and nation to create proxies for various footprint categories. National ecological footprint results are then adjusted to determine the sub-population ecological footprint. For example, data used to adjust the United States ecological footprint to determine Sonoma County's footprint included: population; vehicle miles traveled; average house size; electricity usage; sales of general merchandise, clothing, electronics, appliances, and building material and supplies; paper consumption; and income (Redefining Progress, 2002). Where relevant indicators are not available, national results are scaled to the community by apportioning the per capita impact based on population.

Rees and Wackernagel (1996) first used the indicator approach to estimate the ecological footprint of the Lower Fraser Valley British Columbia. Onisto et al. (1998) applied the strategy to estimate Toronto's ecological footprint. Wackernagel (1998) advocated indicators as proxies to adjust national footprint results in evaluating the ecological footprint of Santiago, Chile. *Sharing Nature's Interest*, which discussed approaches to calculating the ecological footprint, also endorsed using indicators as a basis of analysis (Chambers et al., 2000). Redefining Progress (2002), a non-profit organization in San Francisco, engaged the community through stakeholder workshops to provide input into selecting indicators in its Sonoma County study.

Indicators used to adjust the national data may miss valuable information. Inherent assumptions associated with the approach can undermine accuracy and confidence in the findings. When data are limited, national per capita results have to substitute for local data: this assumes that the region mirrors national consumption patterns, but if it does not the reliability and applicability of the results will suffer. As a calculation strategy, the approach offers a good starting framework, but results have been project specific. The various projects could not ensure that the assumptions and regional data used to adjust national data would be relevant across jurisdictions or widely available. The lack of consistency in the proxies used across projects makes the comparison of results meaningless.

5.4.2 Resource Flow Models

The Stockholm Environment Institute (York University) and Best Foot Forward (UK) have developed comprehensive resource flow models to estimate the ecological footprints of sub-national populations. Both organizations are leaders in ecological footprint work, especially at the sub-national level. The Stockholm Environment Institute uses the Resource and Energy Analysis software program (REAP); Best Foot Forward uses Regional Stepwise Software. The respective models allocate national material and energy flows to sub-populations based on various data including input-output analysis, expenditure data, and life cycle analysis (Chambers et al., 2005; Wiedmann and Barrett, 2005). The REAP program, for example, combines material flow accounts and ecological footprint accounts by economic sector with monetary input-output analysis. The total material flow and ecological footprint are then reallocated to final demand categories based on expenditure data (Wiedmann and Barrett, 2005). The REAP program and Stepwise Software designed for the UK have been adapted for other countries in Europe and Australia.

Projects using the resource flow models supported by the Stockholm Environment Institute and Best Foot Forward have been part of well resourced sustainability initiatives. Noteworthy projects include the Ecological Footprint of Victoria, Australia (Wiedmann et al., 2008); the Ecological Footprint of the United Kingdom by region and devolved county (Barrett et al., 2006); the Ecological Footprint of Cardiff (Collins et al., 2005); the Ecological Footprint of Scotland (Chambers et al., 2004); and the Ecological Footprint of Greater London (Chambers et al., 2002). [2]

The resource flow models promoted by the Stockholm Environment Institute and Best Foot Forward cannot currently work for Canadian cities and municipalities because the national statistical agency does not track the necessary input-output data required by the model. Neither are adequate specific life cycle analysis data nor detailed expenditure data at the municipal level available in Canada (except for select large metropolitan areas).

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[2] Regions that have been examined using Regional Stepwise software according to the Best Footprint Forward website (accessed January 14, 2009) include: North and North East Lincolnshire, Essex, Angus, Brechin, Scotland, Northern Ireland, English regions, Herefordshire, Oxfordshire, London, Aberdeen, Glasgow, Edinburgh, Dundee, Inverness and Wales.

5.4.3 Community Based Questionnaires

Ecological footprint projects are typically top down, computer based exercises examining datasets to better understand material and energy consumption of the target population. Community based questionnaires, by contrast, actively solicit data from households as the basis for estimating the ecological footprint. Brechin (in Angus, Scotland) and Kings County (Nova Scotia, Canada) used community based questionnaires to collect ecological footprint data (Angus Council, 2003; GPI Atlantic, 2002).

For smaller communities with access to limited statistical data, questionnaires collect primary data otherwise unavailable. Actively having households respond to the questionnaire encourages them to reflect on their environmental habits and behaviours. The approach fosters education and awareness during project promotion and data collection. The questionnaire formats for Brechin and Kings County were not novel calculation strategies. The questionnaires provided a means to collect the necessary local data to adjust national footprint findings following the indicators approach.

Community questionnaires are time consuming and resource intensive. The Kings County study, part of a larger Genuine Progress Index project, ran out of resources before the ecological footprint survey results could be analyzed. In both the Angus Ecological Footprint Study and the Kings County Study, the questionnaire design failed to capture necessary data to estimate a robust ecological footprint. The Angus council noted as a limitation of the study that its questionnaire included vague questions that did not reveal useful data (Barrett et al., 2004). The Kings County questionnaire proved similarly problematic because it was developed by the project's volunteer steering committee, which had limited knowledge of the ecological footprint.

Survey participation can also be problematic. In both Brechin and Kings County completing the questionnaire was voluntary. Participants may not have represented the wider community. The studies may have indirectly targeted homogenous populations.

For instance, to encourage participation the Brechin project targeted schools in the community (Angus Council, 2003). While survey design and protocol challenges can be mitigated, they reflect the human resource demands associated with using questionnaires. Using self reported surveys to collect environmental consumption data presents issues of response reliability. Studies by Gatersleben and colleagues (2002) and Fuj and colleagues (1985) note that respondents in self reported surveys often under report their electricity and household energy use. Respondents may not answer the questions honestly, especially if there is a preconceived notion that excessive consumption is negative. Self reported data of socially unacceptable behaviours such as alcohol use and smoking note significant underreporting (Hatziandreu et al., 1989; Stockwell et al., 2004). Using a household survey questionnaire is not a viable approach for widespread application for communities in Canada.

5.4.4 Household Expenditure Survey Approach (Canada)

The household expenditure survey approach follows the logic of the indicator approach. National footprint data are adjusted based on differences in detailed per capita expenditure data between the study population and the Canadian average. Expenditure data are assumed to provide a good proxy of consumption. Most projects use the expenditure data to estimate the non-energy component of the ecological footprint. The energy footprint (or portions of it) is calculated directly. Several projects using household expenditure survey data modify the expenditure data to better reflect household consumption. The Alberta Ecological Footprint Study (Wilson and Anielski, 2008), for example, attempted to adjust for cost differentials throughout the country by adjusting price data with the Inter City Consumer Price Index, which reflects cost differentials between major urban areas in Canada.

Several Canadian provinces and municipalities have used household expenditure data to derive ecological footprint estimates. Wilson and Colleagues (2001) and Wilson (2001) employed household expenditure data to estimate the ecological footprint for Nova Scotia and Alberta respectively. The Alberta Ecological Footprint Study included ecological footprint estimates for the province's two largest cities: Calgary and Edmonton. Other provincial ecological footprint studies using variations of the household expenditure survey approach include key projects by the International Institute of Sustainable Development for the Province of Manitoba (Manitoba Government, 2005) and the Atkinson Foundation. The Atkinson Foundation (2006) conducted preliminary ecological footprint estimates of all Canadian provinces for potential inclusion in the Canadian Wellbeing Index. The Federation of Canadian Municipalities' Ecological Footprint study, the first major study below the provincial level in Canada, used household expenditure data to derive ecological footprint estimates of 18 municipalities (Wilson and Anielski, 2004).

The most immediate limitation of the household expenditure survey approach is that Statistics Canada reports the household expenditure survey only for Canada, the provinces and selected Census Metropolitan Areas.[31]The data needed to follow the calculation strategy are not available for most municipalities and communities in Canada.

5.5 A Proposed Calculation Strategy

No clear ecological footprint calculation strategy is available for small urban centres and municipalities within Canada. The local and regional level, however, are arguably the most appropriate scale for advancing sustainability issues (Clark and Dickson, 2003; Graymore et al., 2008; United Nations, 1992). The local level is the scale where individuals interact with their surroundings, engage with others, relate to environmental issues and have the most direct influence over decisions that affect their quality of life. Most sustainability assessment tools, though, have been designed for national or provincial scales and are not relevant measures at the regional level. In an extensive review of sustainability assessment methods, Graymore et al. (2008) concluded

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[3] Statistics Canada reports Household expenditure data for the largest Census Metropolitan Areas with national geographical representation to ensure that the survey includes at least one Census Metropolitan Area per province.

that none of the prevalent assessment tools, including the ecological footprint, proved effective for measuring progress toward sustainability at the regional scale.[14] According to Graymore et al. (2008), the ecological footprint method applied at the regional scale was hard to use, proved time intensive, required too many simplifications and assumptions, demanded an unmanageable data set, and was not clear or well documented. Moreover, the findings did not relate well to policy, strategic planning, management action or decision making.[25] An immediate conclusion emanates from their assessment: the standard ecological footprint calculation is not an effective approach for the regional scale. We need a simpler, easier to use, less time and data intensive method. In the next section we address these concerns by developing and testing an ecological footprint calculation strategy that is simple and accessible for small urban centres and municipalities.

5.5.1 The Canadian Municipal Calculation Strategy

The goal of proposing the municipal footprint calculation is to advance a consistent, transparent calculation strategy that can be widely adopted by municipalities in Canada regardless of population size. An accessible strategy requires that the calculation approach draw on common data sets available in the public domain. It recognizes that many communities may not have access to detailed resource and energy flow data, expertise in sustainability modeling, or resources (time, money) to undertake a comprehensive analysis. A consistent framework will permit comparability between jurisdictions and strengthen the credibility of ecological footprint results.

The municipal calculation strategy for Canadian communities follows the Global Footprint Network Ecological Footprint Standard 3 (2006). Adapting the national results to sub-national populations offers a logical starting point and provides results comparable

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[4]Sustainability assessment methods addressed in Graymore and colleague's analysis include: ecological footprint, well-being assessment, quality of life, ecosystem health, and natural resource availability.

[5]For a detailed critique of the ecological footprint methodology refer to van den Bergh and Verbruggen (1999); McManus and Haughton (2006), and Fiala (2008).

with the GFN National Footprint Accounts. The municipal calculation strategy proposes six proxies to adjust national or provincial ecological footprint results as a means to derive local ecological footprint estimates. The proxies reflect major consumption categories of the ecological footprint: food, shelter, mobility, goods, services, and government.[6] The municipal approach elaborates on previous sub-national ecological footprint work completed by Wilson and Anielski (2004; 2008). The data required to complete the calculation are available for communities at the Census subdivision level and above. Statistics Canada (2007a) designation of census subdivision is the general term for municipalities or areas treated as municipal equivalents for statistical purposes (e.g., Indian reserves, unorganized territories). The calculation draws on three main data sources: the Census of Population (Statistics Canada), the Natural Resources Canada Energy Use Database and the Natural Resources Canada Survey of Household Energy Use. These data sources are freely available in the public domain over the internet.

Consumer goods and services

To adjust the goods and services component of the ecological footprint we propose using available income as a proxy. We assume that households spend their available income and that expenditure mirrors the consumption of goods and services. The calculation involves median after-tax household income minus shelter expenses, tax expenses, and savings. Shelter expenses include gross rent or mortgage payment, and the costs of electricity, heat and municipal services. Removing shelter expenses assumes that income allocated to mortgages, rent and municipal services is not available for spending on consumer goods and services. Statistics Canada 2006 Census reports after-tax household income and shelter expense. The Statistics Canada Survey of Household Spending reports expenditures on taxes and savings at the provincial level.

Shelter – energy

[6]The government category accounts for the environmental impact associated with the provision of government services serving the general population such as health care and roads.

To adjust the shelter component of the ecological footprint we propose two proxies: one to adjust energy use and the other to adjust the non-energy component of shelter. The energy use component refers to direct energy demands of the household. For energy we estimate the average energy use of the community's housing stock. The calculation factors into consideration the type of dwellings, age of dwellings, and energy use by dwelling type and age. The 2006 Census reports housing stock data. The Natural Resources Canada online energy use database provides energy use data by housing type. The Natural Resources Canada 2003 Survey of Household Energy Use publishes energy intensity data for dwellings by period of construction.

Shelter – non energy

The non-energy component of the shelter footprint refers to the construction, maintenance, and other material inputs to support shelter. To adjust the shelter-non energy component we use dwelling size as a proxy of the resource inputs of the shelter. The calculation estimates dwelling space occupied per person by dividing the number of rooms per dwelling by the average household size. Both datasets are available from the 2006 Census.

Mobility

To adjust the mobility or transportation component of the ecological footprint we use the average commuting footprint of the community as a proxy. We assume that commuting is a significant portion of household transportation use that reflects overall dependency on the automobile. The calculation to estimate the commuting footprint is the median commuting distance to work multiplied by the footprint of the different commuting transportation modes. Rees and Wackernagel (1996) evaluated the ecological footprint of different transportation modes. While their data is thirteen years old, the energy intensity per kilometre of the passenger vehicle fleet in Canada has changed by less than five percent between 1996 and 2006 (NRCAN 2008). The 2006 Census reports on mode of transportation to work data and average commuting distances.

Food

To adjust the food footprint we use provincial expenditure on food as a proxy of food consumption. We adjust the expenditure data by the Inter-City Consumer Price Index to account for potential food cost differentials throughout the country. We extend the results to all communities within the province as food expenditure data are not available at the community level; thus we must assume that eating habits, dietary preferences, and supply chains are similar across each province. Statistics Canada reports food expenditure in the Survey of Household Spending and Inter-City Consumer Price Index in the Consumer Price Index Survey.

Government

To adjust the government component of the ecological footprint we propose using expenditure on provincial government services as a proxy. As with the food category we assume that the provincial government footprint is similar for all communities within the province. While government expenditures may vary by region within a province, government services such as roads, schools and health care serve all provincial citizens regardless of community. Statistics Canada reports provincial government expenditure in the Provincial and Territorial Economic Accounts.

Starting point

Either the national or provincial ecological footprint data can be used as a starting point to derive municipal ecological footprint estimates. Starting with the provincial ecological footprint saves one step of data analysis.[7] In Canada, electrical energy grids are province specific. The carbon intensity of the electrical supply differs by province. The magnitude of difference can prove significant. Some provinces rely almost entirely on hydro-power (which has a relatively low carbon intensity) while other provinces depend primarily on oil and coal (which have correspondingly higher carbon intensities). Municipalities need to ensure that their ecological footprint calculation reflects the energy mix of their provincial energy grid. These data are available in the public domain from Environment Canada in the Greenhouse Gas National Inventory Report (2008). We encourage municipalities to use provincial ecological footprint estimates as a starting point, if available, recognizing that many infrastructure and policy decisions influencing consumption patterns are provincially based. In addition, supply and distribution chains are more similar at the provincial level than at the national level.

5.6 A Test: Alberta Communities

To test the utility of our approach we applied the municipal footprint calculation strategy to 13 communities in Alberta. The analysis uses the Alberta provincial ecological footprint as the starting point (Wilson and Anielski, 2008). The Alberta ecological footprint was based on the Global Footprint Network 2006 national ecological footprint accounts (2007a) and the Canadian Land Use Matrix (2007b). Table 5.1 provides a breakdown of the Alberta ecological footprint by category. Because we are testing the municipal approach for communities within the same province, we hold the government consumption category and food consumption category common for all communities. For this reason, approximately 35% of the total ecological footprint is uniform across communities within a province.

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[7] In Canada, Alberta and Quebec are the only provinces with ecological footprint estimates following the Global Footprint Network Standard. Ontario is in the process of completing a report. The Alberta Ecological Footprint is currently an unreleased draft version (Wilson and Anielski, 2008).

Table 5.1: Alberta Ecological Footprint by Major Consumption Category in 2005

Categories	Global Hectares per person	Percent of total Ecological Footprint
Goods and services	2.1	24%
Shelter – energy	2.3	26%
Shelter – non energy	0.4	5%
Mobility	0.8	9%
Food	1.9	21%
Government	1.2	14%
Total	**8.8**	**100%**

Source: Wilson and Anielski, 2008.

Table 5.2 estimates the ecological footprint by community including a summary of the ratios used to adjust consumption categories. We report the ecological footprint value in global hectares per capita. The regional ecological footprint estimates range from a low of 7.5 hectares per person in Wetaskiwin to a high of 11.2 hectares per person in Okotoks. The regional differences reflect primarily the goods and services proxy and the mobility proxy. For example, the high number of commuters traveling long distances, above average spending on consumer goods and services, and a slightly higher energy footprint of the housing stock explain Okotoks' large footprint compared to the Alberta average. Wetaskiwin, on the other hand, has the lowest average spending on consumer goods and services and the lowest average commuting distance of all communities in the study. Figure 5.3 depicts the regional footprints by consumption category from lowest to highest; the dotted line is the average Alberta ecological footprint of 8.8 hectares per capita.

Table 5.2: Summary of the ratios used to adjust consumption categories and ecological footprint result by community

	Goods and services	Shelter - energy	Shelter - non energy	Mobility	Food	Govt	Ecological Footprint (gha/ capita)
ALBERTA	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**1.00**	**8.8**
Brooks	1.02	1.04	1.01	0.38	1.00	1.00	8.4
Calgary	1.05	0.95	1.00	1.02	1.00	1.00	8.8
Camrose	0.77	1.01	0.99	0.28	1.00	1.00	7.7
Canmore	1.02	0.91	0.96	0.25	1.00	1.00	8.0
Cold Lake	1.22	1.02	1.04	0.74	1.00	1.00	9.1
Edmonton	0.99	0.98	0.97	1.04	1.00	1.00	8.7
Grand Prairie	1.19	1.01	1.01	0.56	1.00	1.00	8.9
Lethbridge	0.85	1.02	1.03	0.61	1.00	1.00	8.2
Medicine Hat	0.93	1.03	1.04	0.58	1.00	1.00	8.4
Okotoks	1.14	1.03	1.10	3.57	1.00	1.00	11.2
Red Deer	0.99	0.94	0.97	0.55	1.00	1.00	8.2
Wetaskiwin	0.72	0.98	0.97	0.25	1.00	1.00	7.5
Wood Buffalo	1.77	0.97	0.96	0.99	1.00	1.00	10.3

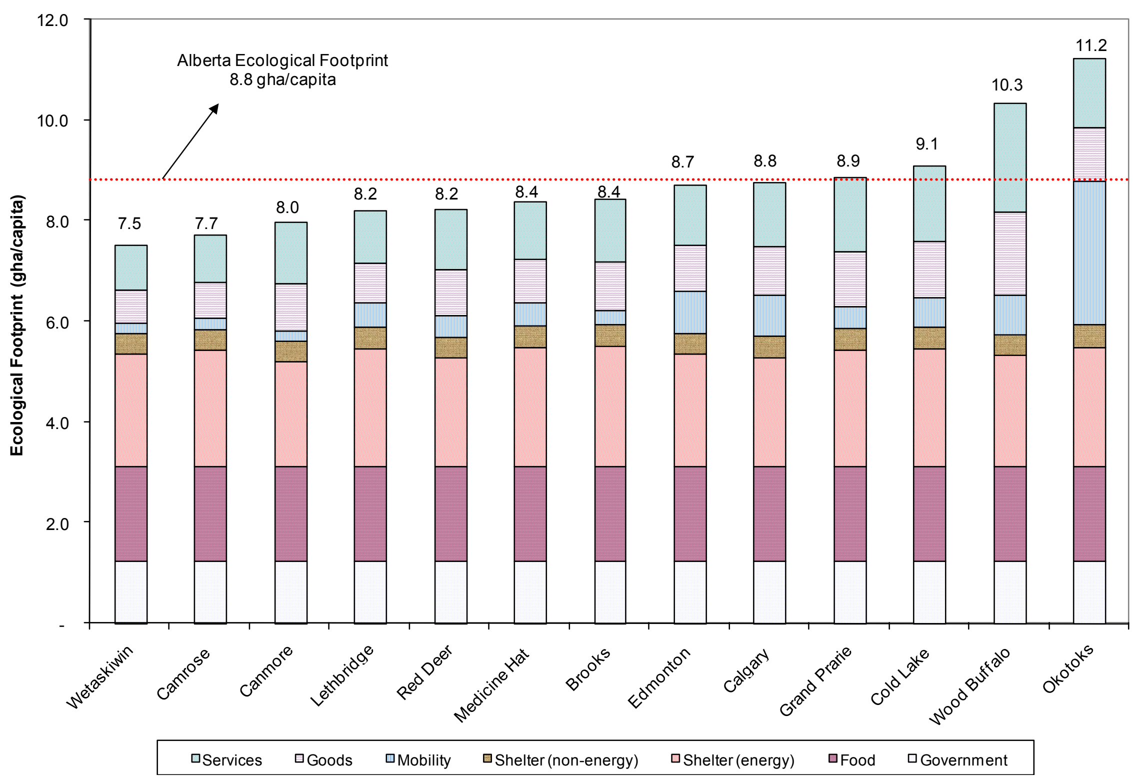

Figure 5.1: Regional Ecological Footprints by Consumption Category

5.7 Discussion

When calculating the ecological footprint of municipalities we need to consider several challenges and limitations. Our analysis discusses limitations of the proposed municipal calculation approach and limitations that pertain to calculating municipal ecological footprints in general.

5.7.1 Limitations of the Municipal Approach

We derived the proxies selected to estimate municipal ecological footprints using data available in the public domain to ensure accessibility. The calculations are not based on direct assessment of the communities. We did not collect primary data or purchase data sets as the point of this study was to use public domain data. We recognize, however, that more robust proxies could be developed using direct data collection or by purchasing data sets.

The approach estimates ecological footprint without requiring particular knowledge of the local situation or specific conditions within a community. Using selected indicators as proxies of sustainability or consumption may generate misleading results. The community of Yellowknife is a good example. Yellowknife was ranked as the most sustainable small city by the Corporate Knights (2008). The sustainability rankings reflected selected indicators, including median commuting distance. As with the transportation proxy proposed for the municipal approach, commuting distance provided a proxy for household transportation energy use. While commuting distances in Yellowknife are small, selecting the indicator as a proxy of household transportation energy use misses important information relevant to a northern community where people burn a lot of fuel warming up their vehicles. When the weather is below -40 C, some people leave vehicles running all the time to keep engines warm.[8] Furthermore, fuel efficiency is low in cold weather. In this case median commuting distance poorly reflects household transportation energy use, and underestimates the local ecological footprint.

The approach entails several critical assumptions. The calculation for the goods and services, food and government components assumes that expenditure is a proxy of consumption. Higher expenditure is equated with higher aggregate volume of consumption which is linked to a higher footprint value. Expenditure does not reveal any detail about what is consumed. For example, expenditure does not indicate where goods or services come from, the quality of the goods or services, or what the goods and services are. The proxy for goods and services fails to consider that geographic differences may impact how unallocated income is spent. The proxy for the transportation component assumes that the footprint of the municipalities' commuting profile reflects the overall transportation footprint. However, commuting is only one component of transportation consumption. Other important components include: vehicle type, fuel efficiency, and non commuting vehicle trips (for example, shopping, schools, sport, weekend trips, and vacations). By assuming that food consumption habits are similar across the province we ignore significant central-remote community differences. For instance, rural regions may have a higher percentage of households with gardens, access to farmers markets, and rates of hunting than urban centres. Urban centres tend to have larger immigrant populations which may influence the dietary profile. Proxies for the food and government categories assume that the food and government footprints are uniform across a province, which may not be the case.

Source of electricity influences a community's ecological footprint value. For example, electricity derived from coal has a larger energy footprint than electricity derived from hydro-power. In Canada, communities access energy through a provincial

[8] Yellowknife had 14 days with temperatures below -40 Celsius during the 2008 Winter (January 1 to March 31 2008). Environment Canada. 2008. Weather Office. National Climate Data and Information Archive. Online: www.climate.weatheroffice.ec.gc.ca. Accessed December 2008.

energy grid over which they have limited influence. Inter provincial ecological footprint comparisons among communities can be misconstrued due to differences in the source of electricity. Instead of revealing local lifestyle choices, the community's ecological footprint may reflect provincial energy choices that residents cannot influence.

5.7.2 General Limitations

Communities face several general challenges and limitations when calculating municipal ecological footprints. Data sets are typically reported in aggregate form or average results per household or per capita. Aggregate and average data fail to highlight disparities in resource and energy use among different community segments and regions within a study area.

Community planning efforts to improve sustainability may not translate into a lower footprint. A significant factor contributing to a community's ecological footprint is the capacity of community members to consume (which is largely driven by income). This can frustrate users of the ecological footprint. A multi city report estimating ecological footprints for 16 communities in Canada led to several communities dismissing the project in a dispute over how to assess sustainability. In some cases, communities could demonstrate compelling evidence that they should score better than the rankings suggested: more green space, a better public transportation system, more resources dedicated to sustainability. Planning efforts to be more sustainable, however, did not offset the fact that some communities were wealthier: in the footprint model, they consequently had larger footprints of consumption (more cars per household, bigger houses). The community of Okotoks exemplifies the situation. Okotoks has won several awards for being a leader in sustainability (Sustainable Okotoks, 2009) yet our analysis indicates that it has the highest ecological footprint of the communities tested. Another example is the City of Calgary which has a per capita ecological footprint well above the Canadian average (City of Calgary, 2007). It is also a leader in ecological footprint efforts with a department of five full time positions dedicated entirely to advancing ecological footprint related issues. The City Council of Calgary has adopted reducing the ecological footprint as a priority and integrated it with the ImagineCALGARY long term planning initiative (Harvey, 2007).

The ecological footprint measures the impact of consumption and does not capture planning initiatives to be more sustainable unless they influence consumption patterns. For example, sustainability actions such as adding green space or supporting rooftop gardens are not directly captured by the tool. If the additional green space deters people from traveling further distances to access green space or if the rooftop gardens offset food imports, those actions will in theory reduce the footprint. In all likelihood, however, these efforts will not be significant enough to produce noticeable changes in the footprint.[9] Moreover, major infrastructure and planning decisions made in the past influence a city's current ecological footprint. Project sponsors and users often want something immediate to lower their community's ecological footprint, but large scale city design and infrastructure changes are not easily or quickly implemented.

It can be difficult for planners and policy makers to relate the ecological footprint to what they perceive as the major environmental concerns of their community. City planners think of sustainability largely in terms of how it relates to direct environmental challenges. They focus sustainability actions on local issues such as energy use, air quality, water quality, waste, congestion, transportation modes, and green space. The ecological footprint of a given population, however, reflects the amount of bio-productive space used or occupied to support the consumption of goods and services by that population regardless of where in the world the environmental impact incurred. In industrialized countries, which typically rely heavily on imported goods and services, the burden of consumption largely falls outside of local political borders. Communities do not see the negative feedback their consumption may be causing on the supportive ecosystems which could be half a world away (Rees 2008). The macro understanding of

[9] The impact that efforts such as these have on the overall footprint will also depend on the scale of the assessment (small community vs. large community) and the scope of the initiative.

sustainability built into the ecological footprint model does not always translate well at the city level.

Many factors influencing a community's ecological footprint fall outside local jurisdictional authority limiting municipal planners' and policy makers' capacity to lower their community's ecological footprint. A notable example is the energy source of electricity generation. Electrical generation falls under the jurisdiction of the province yet embodied carbon content of electricity contributes significantly to a community's footprint. McManus and Haughton (2006) identify jurisdictional responsibility as a key problem limiting the use of the ecological footprint as a policy tool at the sub-national level.

The ecological footprint is an additive model compiling complex information into a single, functional score. While we see merits to providing an aggregate result for communication purposes, adding indicators means that information is lost. Relating an aggregated score to specific planning and policy choices is difficult. McManus and Haughton (2006) argue that presenting ecological footprint results in a single aggregated value limits the tool's ability to shape policy interventions: the reason municipalities are keen to adopt the tool in the first place.

5.7.3 How the Ecological Footprint Can Best Serve Municipalities

Although we have demonstrated that communities can develop local ecological footprint calculations using our method, we conclude that the municipal approach is best suited for local education and raising awareness about the environmental impacts associated with consumption patterns and lifestyle choices. At the municipal level, the tool is not accurate enough, information is lost in aggregation, jurisdictional authority prevents local communities from influencing several key factors impacting their ecological footprint, the ecological footprint calculation can be made without local knowledge about the community nor does the tool reveal enough about the differences in ecological footprint at the local community. McManus and Haughton (2006) and

Graymore et al. (2008) argue that the ecological footprint applied at the regional level is not adequate to inform planning and policy decisions. If municipalities wish to use the ecological footprint for planning and policy purposes, they require a more sophisticated calculation approach nested within a larger sustainability planning effort.

Results from a questionnaire conducted by the Stockholm Environment Institute presented to communities that estimated their ecological footprint support our conclusion. Local authorities responded that it was difficult to identify concrete policy outcomes as a result of their ecological footprint study: "it is safe to state, though, that there is no evidence that the ecological footprint has been systematically used to help construct, analyze, and measure and then monitor the effect of a specific policy within a local authority" (Barrett et al., 2004, p. 239). As Barrett et al., (2004) note, however, estimating ecological footprints was effective in raising awareness of key sustainability issues among elected officials. While the ecological footprint studies have not directly translated into policy outcomes, they played a critical role in motivating later environmental policy. Many communities put important environmental initiatives into practice following ecological footprint studies (Barrett et al., 2004). Thus the educational function of local ecological footprint analyses can prove significant.

Calculating the ecological footprint at the municipal level is most useful for educating and raising awareness about the environmental impacts associated with consumption patterns and lifestyle choices. The municipal approach offers an accessible, easy to use calculation strategy. While the suggested calculation approach reflects several assumptions which limit the accuracy and variability of ecological footprint findings, the integrity of the results is sufficient for advancing the educational function of the ecological footprint tool.

5.8 Conclusion

The development and application of sustainability tools and indicators has become big business. Consulting firms, non-profit organizations, and even educational institutions have vested interests in leading projects. Stakeholders (who in general have good intentions) are busy trying to sell the merits of their tools to different users. Unfortunately, though, practitioners often oversell the functions and accuracy of sustainability measurement tools. Ecological footprint practitioners are no exception. The ecological footprint is increasingly described as a comprehensive sustainability indicator when it is not. McManus and Haughton (2006) and Barrett et al. (2004) suggest that many of the misconceptions surrounding the ecological footprint derive from it being oversold as a comprehensive indicator. Overselling the tool creates misunderstandings among policy makers, planners and municipal leaders about what the tool achieves, how it can be applied and what the results mean. Users often fail to recognize that the ecological footprint is simply a model. Models, especially sustainability models, help us understand complex systems, issues or problems. Regardless of the effort invested in applying them, they do not provide perfect information or complete clarity. Models like the ecological footprint reflect a certain world-view and inherently incorporate varying degrees of assumptions. They have flaws, biases and limitations.

An honest portrayal of what the ecological footprint can achieve and an emphasis on transparency can mitigate potential frustration among users and project sponsors. The municipal approach offers municipalities an accessible calculation strategy which emphasizes the educational benefits of the footprint. Advocating an accessible simplified calculation approach does not imply that communities should avoid undertaking more detailed ecological footprint analyses or sustainability modeling exercises. If the resources and commitment are there, more robust analyzes will help communities better understand sustainability needs and foster the necessary changes to advance community sustainability objectives. Moreover, the ecological footprint results have little value if they are not part of a larger process to advance sustainability at the municipal level.

This paper advances the cause of developing a consistent ecological footprint calculation approach accessible to municipalities in Canada. The Canadian municipal footprint calculation approach is best suited for promoting education and raising awareness. The limitations of the municipal approach make it unsuitable as a general tool to inform community planning and policy development. This is not to say that we dismiss the role that the municipal approach can play in shifting behaviour patterns and motivating action to foster more sustainable communities. In fact, accepting the strength of the ecological footprint as a metaphor for more sustainable lifestyles and embracing the simplicity of its message will help community planners and policy makers to use the tool for what it does best: raise awareness, educate, inspire and promote dialogue. Adopting the ecological footprint can change how planners, policy makers and community leaders see their communities to encourage sustainability thinking. Future research should explore adapting the ecological footprint and other biophysical measurement tools so they can better inform planning and policy decisions at the municipal level.

Reflection, planning, action, and change require inspiration. A simplified ecological footprint methodology that can be applied widely at the community level makes it accessible for community leaders and planners across Canada to educate and engage the citizenry in their communities to live more sustainable lives. If a citizenry is inspired, community planning tools, decision-makingand agendas may change.

5.9 Works Cited

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Chapter 6: Measuring Environmental Impact at the Neighbourhood Level

6.1 Publication Information

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6.2 Abstract

We propose that community assessments of environmental impact are increasingly more relevant to planners and policy makers when reported at finer scales of analysis. Using the Town of Oakville, Ontario, as an example, we calculate neighbourhood level ecological footprint values for 241 neighbourhoods. Ecological footprint results range from 5.4 global hectares per capita to15.2 global hectares per capita with an average ecological footprint for Oakville of 9.0 global hectares per capita. Our results highlight variability in energy and material flows within a community providing planners and policy makers detailed information to prioritize program delivery, allocate limited resources, and support policy development. The lower range of neighbourhood ecological footprint values suggests a potential footprint floor for Oakville around five hectares per capita. The notion of a footprint floor has implications for setting community footprint targets and understanding the magnitude of change needed for significant ecological footprint reductions.

6.3 Introduction

Agenda 21 articulated the need to develop indicators and models of sustainability to support decision-making at the local level (United Nations, 1992). Community leaders, planners, and policy makers want evidence to support sustainable development pathways and policy reform (Benders et al., 2006; Hamdouch and Zuindeau, 2010; Wilson and Grant, 2009). In the 20 years since Rio, however, the majority of sustainability measurement and support tools target macro-scale applications (Bell and Morse, 2004; Herzi and Hasan, 2004; Parris and Kates, 2003; Scipioni et al., 2009). With the exception of greenhouse gas analysis tools, championed largely by ICLEI-Local Governments for Sustainability, planners and policy makers have few tools at their disposal to support sustainability related decision-making at the community and sub-community level (Betsill and Bulkeley, 2007; ICLEI, 2009; Scipioni et al., 2009; Yli-Viikari, 2009).

The ecological footprint is an example of one tool increasingly used by communities to quantify the environmental impacts of consumption to inform decision-making (Chambers et al., 2000; Klinsky et al., 2009; Scotti, 2009; Wackernagel, 1998; Wiedmann and Barrett, 2005; Wilson and Grant, 2009). The ecological footprint is a biophysical accounting tool that provides a comprehensive aggregate indicator of human impact on ecosystems (Holmberg et al., 1999). The tool converts the consumption of energy and resources by a defined population into the hectares of land and water required to support that level of consumption. At the macro level, if the human footprint exceeds the productive capacity of the biosphere then consumption patterns are clearly not sustainable. Several authors, however, question the effectiveness of the ecological footprint as a tool to inform sustainability when applied at the local level (Ayres, 2000; Barrett et al., 2004; Graymore et al., 2008; McManus and Haughton, 2006; Moffat, 2000; Scipioni et al., 2008). While results may be suitable to support general education and awareness regarding the environmental impact of lifestyle choices, community-wide results are simply not specific enough to inform local decision-making towards sustainability (Wilson and Grant, 2009). Community-wide analyses typically do not distinguish between high and low impact sub-populations within a community limiting opportunities to develop tailored policy and planning strategies to address elevated material and energy resource use (Satterthwaite, 2008; Weber and Mathews, 2008b; Wilson and Grant, 2009). We contend, however, that the limited capacity of community-wide measures of environmental impact to support policy and planning efforts is not a failure of the tools themselves but is largely an issue of reporting scale and data availability. Given the importance of addressing sustainability issues at all levels of human organization, overcoming these challenges and developing a robust, easily reproducible technique for reporting resource use and environmental impact at fine geographic scales is of pressing importance.

To illustrate the feasibility and utility of measuring environmental impact at a sub-community level (here after referred to as the neighbourhood level), we calculate average per capita ecological footprints by neighbourhood using the Town of Oakville, Ontario, Canada as an example.[1] Our neighbourhood-level analysis, a first within the Canadian context, uses a combination top down, bottom up calculation approach that can be replicated by other Canadian and North American communities. Our decision to use the ecological footprint as a demonstration tool to account for environmental impact at the neighbourhood level reflects the comprehensive nature of the tool and strong interest by Canadian municipalities to calculate their ecological footprint. In addition, the support data collected to estimate ecological footprints can be used to report other measures of environmental impact such as a neighbourhood carbon footprints or household greenhouse gas emissions. While we use the ecological footprint in our analysis, our focus is less on the importance of this metric and the specific results of the analysis but more to demonstrate that reporting environmental impact at finer scales can be done robustly and that such results better support local policy and planning decisions than community-wide reporting. Results of our analysis confirm the existence of substantial heterogeneity in the environmental impact of consumption across a community providing

[1] Neighbourhood is used throughout the paper to describe geographically contiguous populations of 400 to 700 households.

local decision-makers with additional information to direct policy, programming, and planning efforts towards sustainability.

The lack of examples measuring the environmental impact of consumption at the neighbourhood level largely reflect challenges around gathering the necessary data to populate models and limited local capacity to conduct analyzes (time, resources, skill sets) (Graymore et al, 2008; Scipioni et al., 2011; van Zeijl-Rozema et al., 2011). Haq and Owen (2009) suggest a potential approach for calculating neighbourhood carbon footprints for the City of York (United Kingdom). Using the Stockholm Environment Institute's Resource Energy Analysis Program (REAP), a model integrating life cycle assessment and input output data, they demonstrated that neighbourhoods with the highest average carbon footprint tended to be wealthier neighbourhoods in the city centre or in commuter areas. The City of York integrated findings with survey data measuring environmental attitudes to identify neighbourhoods that would receive targeted energy efficiency programming as part of a wider city effort to meet greenhouse-gas emission targets. The REAP model is proprietary software that was designed for the United Kingdom. While the underlying datasets have been expanded to support other jurisdictions in the European Union, it is not yet a relevant model to support North American applications.

Canada, however, is in a unique position where Statistics Canada reports Census data at a fine geographic scale. The Census does not report directly on household consumption or local energy and material data. It does report on key proxies of consumption such as median income, commuting patterns and distances, and household size. Several communities in Canada also track energy data to satisfy greenhouse gas reporting protocols. Van de Weghe and Kennedy (2007) estimated intra-urban greenhouse gas emissions for Toronto using a model integrating energy consumption data and Census data. The study demonstrated that lower density suburbs report the largest greenhouse gas emissions due primarily to private auto use. Klinsky et al. (2009) estimated the ecological footprint of seven suburban boroughs of Montreal. The results were communicated spatially using geographic information system (GIS) software in a study exploring the role of GIS tools in supporting public engagement for local sustainability planning. Here, we continue to advance efforts to understand environmental impact at the local level by reporting ecological footprint values by neighbourhood. More importantly, we make the argument that using tools such as the ecological footprint to highlight variability in environmental impact within a community makes them more relevant to local decision-makers and provides a robust basis upon which targeted education and policy implementation can occur.

6.3.1 Study Community

With a population of 170,000 people, the Town of Oakville, Ontario, is located 85 kilometres southwest of Toronto, Canada's largest city, on the shores of Lake Ontario. An affluent bedroom community of Toronto, Oakville has a median household income approximately 50% higher than the provincial average (Statistics Canada, 2006). In 2009, Oakville confirmed its commitment to environmental sustainability in the official Town Plan including reducing the community's ecological footprint, reporting on the state of the environment, and measuring progress towards a list of environmental goals (Town of Oakville, 2009). According to several environmental performance indicators tracked by the Town, efforts to become a more sustainable community have had limited success. For example, per capita residential energy use between 2004 and 2008 increased by 6% even though reducing community energy use has been a Town priority for some time (Doyle, personal communication, 2010). The disconcerting trends have prompted interest among Town officials to better understand differences in household environmental impact so they can better design and target sustainability efforts to foster footprint reductions.

6.4 Methods

6.4.1 Defining Neighbourhoods

We estimate average per capita ecological footprints for 249 neighbourhoods across the Town of Oakville. We assume that neighbourhoods coincide directly with the Statistics Canada designation dissemination area. A dissemination area (DA) is defined as a small, relatively stable geographic unit composed of one or more neighbouring dissemination blocks with a population of 400 to 700 persons. It is the smallest standard geographic area for which all Canadian census data are disseminated. Dissemination areas are a subset of census tracts which are defined by a committee of local specialists (for example, planners, health and social workers, and educators) in conjunction with Statistics Canada (Statistics Canada, 2007). Oakville has 259 dissemination areas. Due to data suppression we did not estimate a footprint value for ten dissemination areas. The advantage of reporting results at the DA level is that results can be rolled up into larger units if other neighbourhood boundaries are preferred.

6.4.2 Estimating Neighbourhood Ecological Footprints

To derive neighbourhood ecological footprint values we adopt a top down-bottom up calculation approach. Of the six components of the ecological footprint: 1) shelter, energy; 2) shelter, non energy; 3) consumer goods and services; 4) mobility; 5) food; and, 6) government, we calculate the energy footprint associated with shelter directly. For the remaining five categories, we estimate values using a top down calculation approach by adjusting the Town of Oakville's ecological footprint for the respective categories based on differences in average per capita consumption levels between each neighbourhood and the Oakville average. We calculated an Oakville average ecological footprint following the ecological footprint calculation strategy for Canadian communities proposed by Wilson and Grant (2009) (Table 6.1). That method offers a consistent and accessible calculation framework taking a top down calculation approach that draws on public data sets and complies with the Global Footprint Network Ecological Footprint Standards (Global Footprint Network, 2009). We deviate from Wilson and Grant's approach slightly by estimating the energy footprint associated with shelter using direct data from the Town of Oakville. The energy footprint associated with shelter represents approximately 10% of the average ecological footprint.

6.4.3 Calculation Approach

Shelter – energy

The energy footprint associated with shelter refers to the direct energy demands of households. We calculate this component by converting household electricity consumption and natural gas consumption into the equivalent energy land area required to sequester the associated greenhouse gas emissions using Environment Canada greenhouse gas conversion factors (2010) and the Global Footprint Network CO_2e to energy land conversion factor (Ewing, 2008). Electricity and natural gas represent over 99% of total residential energy utilized by Oakville residents (Doyle, personal communication, 2010). Oakville Hydro and Union Gas provided electricity data and natural gas data respectively by postal codes. Both organizations have exclusive utility contracts for the Town of Oakville. Employees of the Town of Oakville rolled up data to the dissemination area level before providing it to us to ensure household confidentiality. We cross referenced our neighbourhood derived results with greenhouse gas inventory data for the residential sector collected by the Town of Oakville to ensure consistency.

Shelter – non-energy

The non-energy component of the shelter footprint refers to the construction, maintenance, and other material inputs to support shelter. Neighbourhood ecological footprint values for this component were estimated by comparing average total floor area occupied per person of respective neighbourhoods with the Oakville average and scaling the non-energy portion of the shelter footprint accordingly. The non-energy portion of the average Oakville footprint per person is 0.5 hectares per capita. Total residential floor area is assumed to be a proxy for total resource inputs associated with shelter. Using residential floor area per person neglects to consider other factors that may influence resource inputs such as types of construction materials used and architectural styles. Residential floor area data aggregated at the neighbourhood level were provided by the Town of Oakville (Thompson, personal communication, 2011).

Consumer goods and services

The goods and services component represents approximately 30% of the average ecological footprint of an Oakville resident. Neighbourhood ecological footprint values per capita for the consumer goods and services category were estimated by scaling the portion of the average goods and services footprint per Oakville resident by differences in available income between the Oakville average and respective neighbourhoods. Available income refers to the total income that households have at their disposal to spend on the consumption of goods and services. For each neighbourhood, it was calculated by subtracting average expenditure on gross rent or mortgage payment, pension contributions, savings, insurance payments, charitable donations, and support payments such as alimony and child support from median after tax income. Removing income categories such as major housing payments, for example, assumes that income allocated to mortgages or rent is not available for spending on consumer goods and services. Rent and mortgage expenditure data available at the dissemination area level are from Statistics Canada (2006). Pension contributions, savings, insurance payments, charitable donations, and support payments expenditure data are from the Survey of Household Spending (Statistics Canada, 2010). Data are not, however, available for each dissemination area. For these expenditure categories, we assume similar expenditure levels by income quintile to the respective Ontario-wide averages.

Assuming that higher available income equates with higher material consumption overlooks two critical points. First, households with more available income could be buying higher priced items and not necessarily more items. Secondly, having more expenditure dollars can facilitate lifestyle decisions and purchase decisions that may have lower associated consumption impacts than alternative decisions. Living in a condominium in the downtown core in a certified sustainable building may have a lower environmental impact than living in a single dwelling home in the suburbs. Relying on available income as a proxy of consumption fails to distinguish between differences in the impact of the basket of goods and services consumed. A hundred dollars spent on goods and services by one household may or may not have the same impact as a hundred dollars spent by another household.

Mobility

We divide the mobility or transportation component of the ecological footprint into personal transportation, air travel, and other passenger transportation. We estimate the ecological footprint associated with personal transportation based on neighbourhood level commuting patterns. Commuting represents a significant portion of personal transportation use and reflects overall dependency on the automobile (Clarke, 2000; Turcotte, 2005). The commuting footprint for each neighbourhood is calculated by multiplying median commuting distance by commuting type (personal vehicle as driver, personal vehicle as passenger, motorcycle, public transit, bicycling or rollerblading, walking) by number of commuters of each type. For example, our calculation considers the number of commuters using public transit multiplied by median commute distance travelled by public transit for that neighbourhood. Total distance travelled by commuter type are converted to carbon dioxide equivalent using greenhouse coefficients by travel mode from Environment Canada's Greenhouse Gas Inventory (2010). Carbon dioxide equivalents are subsequently converted into energy land using the Global Footprint Network energy footprint coefficient (2009). Statistics Canada provided detailed commuting data as a special data run. We adjust the air travel portion of the ecological footprint and the portion of the ecological footprint associated with other passenger transportation based on respective expenditure data from the Survey of Household Spending (Statistics Canada, 2010). Other passenger transportation includes rail and recreational vehicles. For these expenditure categories we assume similar expenditure levels by income quintile to the respective Ontario-wide averages. We adjusted the average ecological footprint of an Oakville resident associated with these categories by the difference in expenditure between the respective neighbourhoods and Oakville average.

Food

No food consumption statistics or expenditure data were available at the neighbourhood level. A Canadian study by Mackenzie et al. (2008), analyzing ecological footprints by income deciles found low variability in the food footprint regardless of income category. For example, Mackenzie and colleagues (2008) found that Canadian households in the highest income decile had a food footprint 5% above the Canadian average. The spread between those in the lowest decile and those in the highest decile was 8%. To derive neighbourhood food footprints we scale the Oakville wide average food footprint based on median household income in proportion to the differences in values by income decile found by Mackenzie et al. (2008).

Government

No neighbourhood specific adjustments were made to the government services portion of the ecological footprint. The community wide value was used for all neighbourhoods based on the assumption that Oakville residents have equal access and use of municipal, provincial, and federal government services. The government portion of the Oakville ecological footprint is one hectare per capita. It was calculated following the Global Footprint Network Canadian Land Use Matrix where the land categories of the ecological footprint are allocated to footprint components using input-output tables (Global Footprint Network, 2010). Approximately 11% of the ecological footprint is attributed to the provision of government services.

6.4.4 Limitations of a Mixed Calculation Approach

We acknowledge that household and neighbourhood level assessments of environmental impact would be better if based directly on detailed local data. Collecting local data using surveys or other means, however, is generally well beyond the resource capacities of most communities. A mixed approach, may be the only feasible option given the lack of readily available detailed household consumption or local energy and material flow data.

Calculating local ecological footprints using proxy data can be problematic. Barrett et al. (2004) and Graymore et al. (2008), both argue that extrapolating national and regional data to estimate ecological footprints at finer scales produces generic results that have little local bearing. Data extrapolation is usually based on population or household characteristics. In our case, footprint data were scaled using neighbourhood level proxies of consumption to estimate several components of the ecological footprint.

We are very careful to be transparent about assumptions behind the indicators selected. An open and transparent model allows for adjustments to be made if more reliable data become available. Given our interest in the distribution of values around the Oakville mean ecological footprint we could have reported results indexed to average Oakville raw data instead. Our choice to report neighbourhood ecological footprint values in global hectares was intended to highlight the magnitude of the results and to provide perspective in relation to other ecological footprint benchmarks and studies.

6.4.5 Reporting Multiple Metrics

Calculating neighbourhood ecological footprints requires collecting a significant amount of data. The extensive background data can be used to report several other environmental impact metrics that support sustainability decision making. For example, as part of our analysis we report:

- Consumption potential based on available income (in dollars per household);

- Average household greenhouse gas emissions (in kilograms of CO_2 equivalents); and,
- Commuting related greenhouse gas emissions (in kilograms of CO_2 equivalent per commuter).

Background ecological footprint data were provided to the community for manipulation and integration into other community environmental data management systems. Our decision to report multiple measures follows Poritosh and colleagues (2009) recommendation regarding life cycle assessment that presenting multiple outputs helps users better interpret and understand the implications of results. In addition, we recognize the disparate views of reporting biophysical impacts as a single aggregate value. McManus and Haughton (2006), for example, argue that presenting the ecological footprint as a single aggregate value oversimplifies results to a point where they are no longer useful to support decision-making. Stoeglehner and Narodoslawsky (2008) commend the simplicity of the footprint as being an asset for planning and policy-making. They argue that the ecological footprint provides results which are intuitively understandable and easy to communicate supporting community decisions which have a strong public participation component or involve decision-makers with diverse backgrounds (2008). While we recognize the merit of a single aggregate value, we concur with McManus and Haughton that in some decision contexts, more data may be required to support decision-making.

6.4.6 Data Presentation

We present results to highlight the distribution of environmental impacts across the community of Oakville. We present total ecological footprint values by neighbourhood. To convey the distribution of neighbourhood ecological footprint values we report the mean footprint value and standard deviation. In addition, we report dissemination area counts by per capita ecological footprint hectare. Results were also categorized into quintiles, colour coded and displayed graphically using ArcMap 9.3 by ESRI Canada. Quintiles were deemed to be an appropriate level of aggregation with which to achieve the goal of geographically discerning variation in neighbourhood footprint values across the Town of Oakville. In addition, to showcase other metrics calculated as part of our analysis we mapped the 10% of neighbourhoods with the highest average per capita ecological footprint, the highest average household greenhouse gas emissions, the highest average commuting impact per commuter, and the highest consumption potential. Our intention was to identify high environmental impact neighbourhoods within Oakville and to explore the extent to which high impact neighbourhoods, as defined using different metrics, overlap. A benefit of presenting data in GIS format is that it is well suited for further decision-making analysis. The data behind the maps can be overlaid and manipulated based on other environmental impact related inquiries. The spatial results need to be interpreted cautiously as the size of a DA corresponds to population (400-700 people) and not geographic area. The actual size of the DA in terms of land area will vary according to population density.

6.5 Ecological Footprint Results

6.5.1 Community-Wide Ecological Footprint

The Oakville average ecological footprint is 9.0 global hectares per capita (Table 6.1).[2] In terms of total area, Oakville's ecological footprint occupies 1.5 million global hectares. This is over one hundred times the town's total land area (13,850 hectares) or more than double the size of the Greater Toronto Area (590, 365 hectares).

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[2] A global hectare is a standardized hectare to account for the fact that different land types and different land categories have different productivity or biocapacity potentials. A common unit allows for the meaningful summation of different land types and categories and also allows for meaningful comparisons of footprint results between regions and countries. Land types are adjusted, reflecting the fact that land types (for example, agriculture land) have different productivity potentials depending on the region. Productivity potential can vary both within a country and across countries. The productivity potential of the different land categories are also converted to global hectares so the different land categories can be summed into a total ecological footprint value. For example, cropland in the ecological footprint methodology is considered to be more productive than pasture land. The land category conversion factors are based on global scientific data and updated by the Global Footprint Network (Ewing et al., 2008).

The consumption of goods and services accounts for 30% of the average Oakville ecological footprint per capita. An additional 30% supports food consumption. Transportation or mobility accounts for 16%. Shelter, which includes household energy consumption as well as the materials and energy used to maintain the shelter, accounts for 14%. Government services account for 10%.

Table 6.1: Ecological footprint per capita (global hectares) by component and quintile for Oakville, ON

	Goods and services	Food	Mobility	Shelter (energy)	Shelter (non-energy)	Govt.	Total
Oakville	**2.7**	**2.6**	**1.4**	**0.7**	**0.5**	**1.0**	**9.0**
Lowest quintile	1.7	2.6	0.9	0.5	0.4	1.0	7.1
Second quintile	2.1	2.6	1.2	0.7	0.5	1.0	8.1
Third quintile	2.4	2.6	1.4	0.7	0.5	1.0	8.6
Fourth quintile	2.8	2.6	1.6	0.8	0.5	1.0	9.4
Highest quintile	3.8	2.7	1.8	1.2	0.7	1.0	11.1

6.5.2 Neighbourhood-Level Ecological Footprints

Average ecological footprints by neighbourhood range from 5.5 global hectares per capita to 15.2 global hectares per capita (Figure 6.1). The range of ecological footprint values is clustered heavily around the Oakville mean footprint of 9.0 global hectares per capita with a standard deviation of 1.5 global hectares.

The tight spread of values around the mean (Figure 6.1) is unsurprising given that Oakville is a relatively homogenous study sample as an affluent bedroom community of Toronto. The short tail to the left of the mean suggests there is a conceivable footprint floor around the five hectares per capita mark, while the longer tail to the right suggests there are no structural parameters limiting high per capita footprint values.

The categorization and mapping of neighbourhood level per capita footprint results by quintile (Figure 6.2) allows for the quick identification of neighbourhoods within Oakville that have the highest and lowest ecological footprints and conveys variability in ecological footprint values across a community.

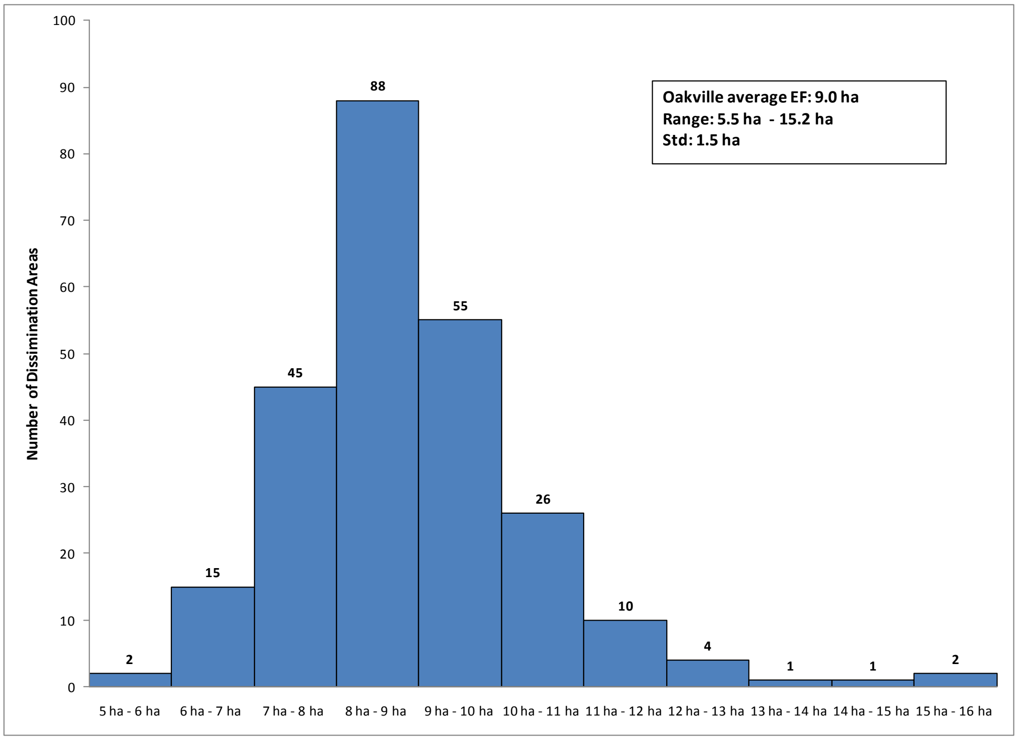

Figure 6.1: Frequency distribution of neighbourhood ecological footprint counts by global hectares per capita for Oakville, ON

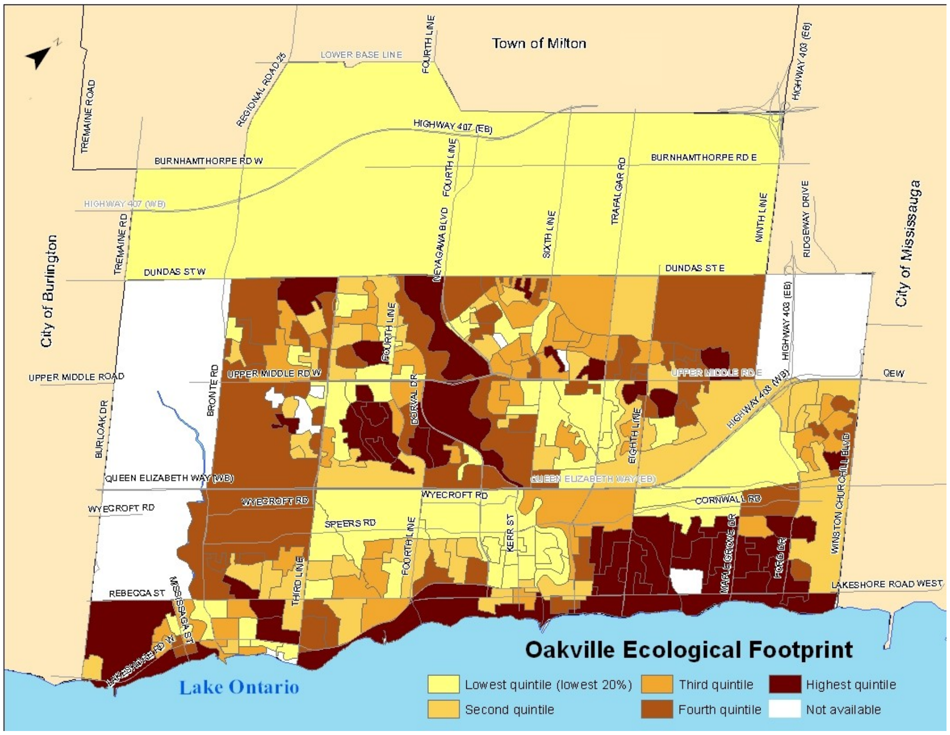

Figure 6.2: Neighbourhood ecological footprints by quintile

High ecological footprint neighbourhoods tend be located along the Town's lake shore, in what is known as old Oakville, and near an internationally recognized golf course. Low footprint neighbourhoods tend to be clustered in the urban core and around industrial and commercial zones. The spatial presentation of results reveals patterns of high and low ecological footprint neighbourhoods within a community. Interestingly, while some neighbourhoods with high per capita ecological footprints also experience high household greenhouse gas emissions, commuting-related impact and/or consumption potential, others do not. Mapping the 10% of neighbourhoods with the highest ecological footprint per capita (Figure 6.3a), highest consumption potential (Figure 6.3b), household greenhouse gas emissions (Figure 6.3c), and highest commuting impact (Figure 6.3d), illustrates the extent to which neighbourhoods rank differently depending on the environmental performance metric analyzed (Figure 6.3).

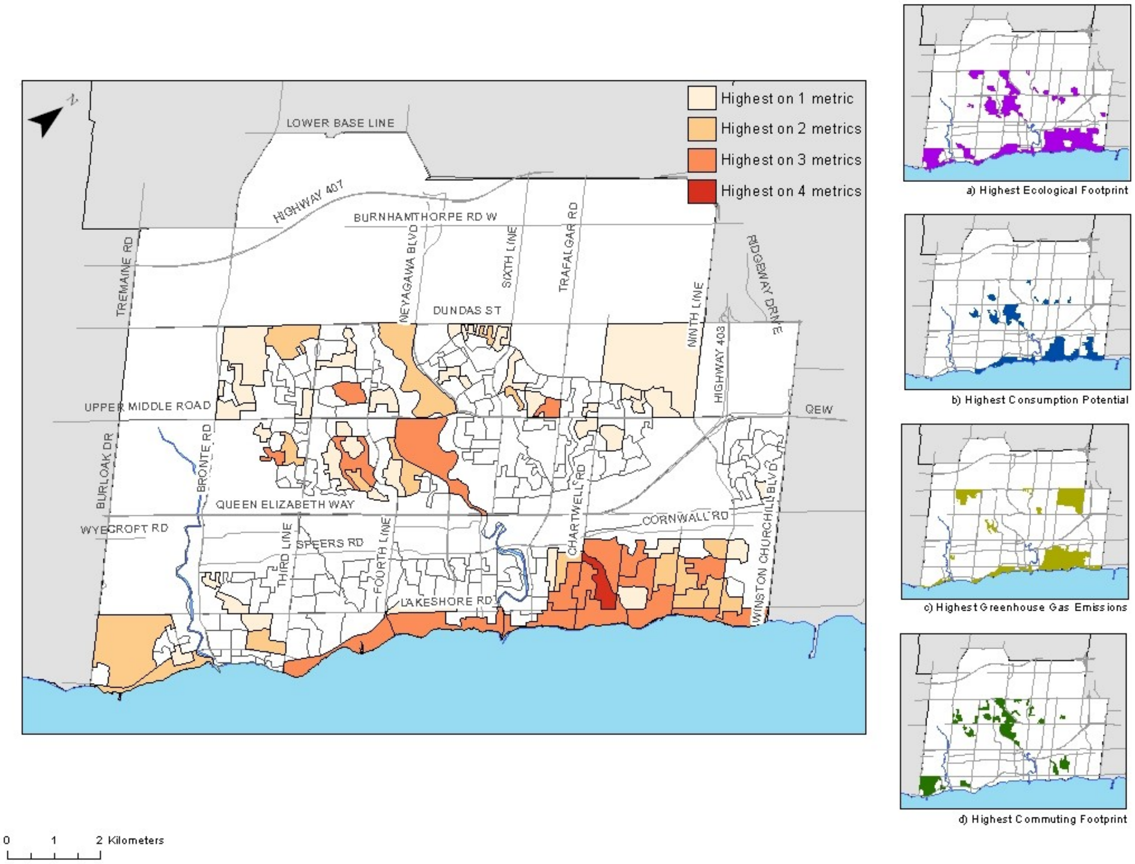

Figure 6.3: Neighbourhoods ranking highest by select footprint-related metrics (top 10%)

In addition to demonstrating how supporting data used to estimate the ecological footprint can be used to report different environmental performance-related metrics, Figure 6.2 also highlights how results can be used to differentiate neighbourhoods using spatial analysis.

6.5.3 Drivers of Variability in Ecological Footprint Values

Analyzing results by ecological footprint component offers insight into what drives variability between neighbourhoods (see Table 6.1). If we consider the 20% of neighbourhoods with the lowest ecological footprints, the difference in ecological footprint values between these neighbourhoods and the Oakville average can almost entirely be attributed to the goods and services and mobility sub-components. The combined impact of these two categories explains 80% of the difference in footprint estimates. For the 20% of neighbourhoods with the highest ecological footprints, the goods and services sub-component explains 46% of the difference between these neighbourhoods and the Oakville average ecological footprint. The shelter-energy and mobility sub-components are also important; they explain 24% and 17% respectively. If we consider that there is little difference in footprint size for the food and government services sub-components of the ecological footprint, it is unsurprising that variability in the goods and services, mobility and shelter-energy sub-components contribute most to overall ecological footprint differences.

6.6 Discussion

6.6.1 Increasing the Relevance of Ecological Footprint Rules

A comprehensive study by Benders and colleagues (2006) found that blanket sustainability programs across large jurisdictions have been largely ineffective. Measuring ecological impact by neighbourhood provides a higher level of detail when compared to community wide assessments to support policy and planning decisions. Categorizing neighbourhoods into high and low ecological footprint groups offers a strategic approach to allocate limited program funding and to prioritize program delivery. The Town of Oakville, for example, is engaging households in neighbourhoods with ecological footprints falling in the top quintile to participate in EcoAction Programming. EcoAction Programming is an Earth Day Canada program that aims to build capacity of households to be more sustainable. Similarly, the Town is directing energy efficiency education to neighbourhoods with high household energy footprints. The Environmental Policy Department is using footprint data to support budget decisions, a protocol mandated by the Town. In addition, the department is presenting neighbourhood ecological footprint results as part of the public consultation process supporting Oakville's Environmental Strategic Plan update to engage participants. The Transportation Planning Department is using the results to support the Master Transportation Plan update. Before calculating neighbourhood ecological footprint results, Town officials used the Oakville wide ecological footprint as a reporting variable in their State of the Environment Report. The results were not used to support any direct sustainability programming or policy development.

6.6.2 Income and Environmental Impact

Income is widely recognized as a determinant of household environmental impact. Numerous studies have demonstrated the role of income as a main driver of direct and indirect energy use, and the household carbon footprint (Benders et al., 2006; Druckman and Jackson, 2008; Hertwich and Peters, 2009; Moll et al., 2005; Peters, 2010; Poortinga et al., 2004; Tucker et al, 2010; Weber and Mathews, 2008b). Based on an extensive review of household environmental impact literature, Peters (2010) concludes the dominant factor influencing the carbon footprint of households is available income. The elasticity between the footprint and expenditure is typically between 0.6 and 1.0, reflecting that as households get wealthier their consumption shifts to higher value-added or more service-based goods and services (Peters, 2010). Previous research has demonstrated a high correlation between income and the ecological footprint (Cranston et al., 2010; Csutora et al., 2009; Mackenzie et al., 2008). The strength of this relationship is not surprising as the ecological footprint is a measure of consumption related impacts and, as Druckman and Jackson (2008) note income facilitates expenditure and almost every expenditure incurs use of resources.

Previous sub-national ecological footprint studies have used either median income or average income as a variable to distribute ecological footprint values among different population segments (Venetoulis, 2004; Wackernagel and Rees, 1996; Wackernagel, 1998; Wilson and Anielski, 2005; Wilson and Grant 2009). We use the concept of available income as a means to adjust the goods and services component of the ecological footprint. Available income, as quantified here, is essentially income stripped of various categories of consumption (e.g. housing payment, savings, taxes...) to better represent actual income spent on goods and services.

The apparent income-impact correlation needs to be interpreted cautiously. Most household environmental impact models (including footprint models) connect household expenditure data with input-output analysis to derive estimates of material and energy flows or use expenditure data to estimate environmental impact by extrapolating national data. These models are based on the core assumption that higher expenditure generates more energy and material consumption. While models typically account for the different environmental impacts of expenditure categories, for example expenditures on food versus expenditures on recreation, higher expenditures still equate with higher consumption and subsequently higher environmental impact. The problem, however, is that while income drives expenditures and expenditures drive resource consumption, not all expenditures have the same resource consumption impact. At some point, the impact likely begins to level off and the nature of the relationship will vary due to household preference profiles (Girod and Haan, 2010). Baiocchi et al. (2010) in a study on household greenhouse gas emissions in the United Kingdom note that the relationship between income and emissions is not perfect. The authors found that for every 10% increase in household income, there was an average of a 7% increase in emissions across households. Interestingly, however, they found that a 10% increase in income among lower and higher income household groupings were associated with emission increases of 12% and 16% respectively. Girod and Haan (2010) in a study of greenhouse emissions of Swiss households found that modelling household impact based on monetary units overestimates the impact of marginal consumption and neglects the potential of decoupling income and environmental impact by consuming better instead of more. Girod and Haan (2010) propose using household consumption models based on functional units (eg. kg of food, person kilometres) as an alternative to models based on monetary units. The challenge with Girod and Haan's proposal is that jurisdictions do not typically track functional unit data explaining why most studies default to monetary based analyses.

6.6.3 Income as a Driver of Variability in Neighbourhood Ecological Footprints

Ehrlich and Holdren's (1971) IPAT equation (impact = population * affluence * technology) emphasizes the contribution of income as a major differentiating factor in determining neighbourhood ecological footprint values. Available technology, infrastructure, and macro-economic policies are typically similar across a community highlighting the importance of affluence as a driver of variability in environmental impact at the neighbourhood level. For example, following the IPAT equation, we measure the ecological footprint at a static point in time where available technology is constant across a community and express ecological footprint results on a per capita basis negating the impact of population growth. The remaining variable driving impact is affluence, which we equate with income. Understandably, income is an important variable influencing the ecological footprint. As noted in the previous paragraph, using income as a direct proxy of energy and material consumption is problematic.

6.6.4 Lower Limits on Ecological Footprint Values

While income appears to be a major driver of variability in ecological footprint values within a community, there is likely a footprint floor for a community dictated by urban form and locked in socio-economic structures. In the case of Oakville, such a lower bound is probably around 5.0 global hectares per capita (Figure 6.1). Existing infrastructure, production systems, product availability, and government policies influence a significant portion of a community's footprint, factors that households have little direct control over.

Having a sense of what a realistic lower bound ecological footprint range is for a community provides critical information for setting footprint targets and understanding the magnitude of footprint reductions that are conceivable for a community to achieve. For example, establishing a short-term ecological footprint target for Oakville below 5 hectares per capita would be setting the community up for failure. There are several examples in Canada, however, where political leaders have set bold footprint reduction targets failing to recognize the magnitude of change required to make significant ecological footprint reductions (Wilson, 2011).

Meadowcraft (2007) argues that the ecological footprint is largely determined by the fundamental organization of our communities. He describes the influence of pre existing structures as path dependency and argues that we need to dramatically redesign communities to facilitate low environmental impact if we want to reduce the ecological footprint (Meadowcraft, 2007). Similarly, Halme and colleagues (2004) argue that local authorities, urban planners, and service providers limit a household's capacity to reduce consumption. Transportation infrastructure, for example, sets parameters within which households are able to decide how to fulfill their mobility needs. An important take away for leaders, planners and policy makers from this research is that changing urban form, infrastructure, and resource use patterns may be critical in many settings if large scale ecological footprint reductions are to be realized. As major infrastructure and planning decisions made in the past influence a city's current ecological footprint, current infrastructure and planning decisions will lock a community into consumption patterns that are difficult to overcome. The long term influence that planning decisions can have over a jurisdiction's ecological footprint highlights the importance of making sure that new development projects, major infrastructure decisions, and city planning and policy documents foster a lower footprint future (Rees, 1997; Rees, 1999).

While urban form, production systems, and government policies influence a lower limit on household ecological footprints, less understood factors which contribute to a footprint floor include societal norms and cultural values. Spangenberg and Lorek (2002) identify intrinsic factors such as cognitive capacities, psychological factors, individual interests, and philosophic or ethical norms as drivers of household consumption. Intrinsic factors, they note, are critical because these factors determine preferences. Jackson (2005) and Jackson and Papathanasopoulou (2008), who explore the significance of urban form as a driver of locked in household consumption, also stress that consumer "lock in" flows from habits, routines, social norms and expectations, and dominant cultural values.

If we accept that physical and social structural factors influence a lower footprint floor, it becomes increasingly difficult for a household to achieve footprint savings as they approach that threshold. Higher footprint households have more options to lower their footprint by shifting spending patterns and curbing consumption than lower footprint households do. From a sustainability perspective, for an Oakville resident to reduce their environmental impact below the 'footprint floor' requires major changes in infrastructure and government policy or dramatic lifestyle changes to circumvent structural barriers. Oakville is not unique within the Canadian landscape. The concept of a footprint floor well above the global sustainability ecological footprint threshold of two hectares per capita likely applies to most, if not all, communities in Canada. While footprint values above the footprint floor seem to be largely influenced by available income, factors determining a footprint floor are largely outside of a household's control.

6.6.5 Identifying Leverage Points

From a municipal perspective, what options does the Town of Oakville have? Many drivers of the ecological footprint do not fall under a community's authority (McManus and Haughton, 2006). Municipalities are restricted to a limited set of policy levers and options to help households lower their ecological footprints. Domains under municipal jurisdiction include development regulations and approvals, building codes, transportation planning decisions, community infrastructure, waste removal, and the residential tax structure. Communities need to understand what capacities they have to support footprint reductions and target their efforts accordingly. Even when considering the capacities that communities have, meaningful reductions in environmental impact requires citizen support and engagement. Of all levels of government, however, municipalities have the closest link with households; along with the citizenry, they define the cultural tone of a community.

While communities have opportunity to lower their ecological footprint, significant footprint reductions require efforts at a multitude of levels. Leverage points to reduce ecological footprints need to be considered at several levels and in different policy contexts, covering for example, individuals, households, companies, different levels of government, and even entire nations (Peters, 2010). For example, municipal governments cannot directly influence vehicle fuel efficiency standards, industrial greenhouse gas emissions, the embodied energy associated with food supply chains, the number of televisions a household has or the temperature at which they set their thermostat. This does not mean municipalities cannot support reductions in these areas, but other levels of government or household decision makers have more immediate authority to influence these sorts of changes. Municipalities, however, can purchase hybrid vehicles, lobby the federal government to implement industrial greenhouse gas targets, support low input food provision choices, and implement energy efficiency programs targeting households, efforts that in themselves may not be immediately captured in the community footprint value but contribute toward building a more sustainable society.

6.7 Conclusion

Our neighbourhood assessment of environmental impact using the ecological footprint measure demonstrates variability in energy and material consumption within a community. Given that Oakville is a relatively homogeneous bedroom community of Toronto, we expected to see relatively uniform results across neighbourhoods. The surprising degree of variability (footprint range: 5.4 to 15.2) confirms that neighbourhood level analyzes provide policy-makers and planners additional information to develop and target programs and policies to reduce environmental impact.

We used direct household energy data to estimate the shelter-energy portion of the ecological footprint. For the remaining ecological footprint categories, we used a top down approach adjusting Oakville ecological footprint results based on different proxies of consumption between neighbourhoods and the Oakville average. While the Global Footprint Network Ecological Footprint Standards (2009) support a top down approach for sub-national ecological footprint calculations, basing calculations on direct local energy and material flow data would strengthen assessments. The capacity to collect local data and the potential costs, however, make more direct sub-community calculations prohibitive at this time. Several emerging technologies, however, offer promising approaches to support local data collection. For example, all households in Oakville are now equipped with smart meters which electronically track how much and when electricity is used. Similar meters are being installed to electronically track water use (Doyle, personal communication, 2011). Natural Resources Canada is also piloting the use of onboard vehicle software to measure greenhouse gas emissions of personal vehicles (Marshall, personal communication, 2011). Handheld devices offer a unique approach to collect direct household data. A 2007 study conducted in Halifax, Canada, had individuals carry handheld devices with a GPS unit installed to track travel patterns and related activities (Saint Mary's University, 2009).

A valuable insight from our analysis is the notion that within a community there may exist a footprint floor dictated largely by structural barriers. For the community of Oakville that footprint floor appears to be more than double the global sustainable ecological footprint target of two global hectares per capita. Reducing household consumption, shifting spending patterns, and improving household energy efficiency are important. Leaders, planners and policy makers, however, need to understand that significant ecological footprint reductions require redesigning urban form, rethinking how our communities work and adjusting government policies to support opportunities for households to achieve lower ecological footprints.

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PART 3: UNDERSTANDING DRIVERS AND DISTRIBUTION OF ENVIRONMENTAL IMPACT ACROSS AN URBAN REGION

> *"They both listened silently to the water, which to them was not just water, but the voice of life, the voice of Being, the voice of perpetual Becoming."*
>
> \- Hermann Hesse

Overview:

Part 3 presents two chapters that aim to improve our understanding of environmental impact at the local level. In Chapter 7, I conduct a multivariate analysis examining the relationship between direct GHG emissions and 20 socio-economic and wellbeing variables. The analysis is a first to include several wellbeing variables in efforts to understand better the relationship between subjective wellbeing and environmental impact. Chapter 8 investigates whether where we live matters in terms of contributions to GHG emissions. Chapter 8 reports results and statistical differences in greenhouse gas emissions for Halifax Regional Municipality between communities and urban-rural zones (inner city, suburban, and inner/outer rural commuter). The study underscores the importance of understanding the spatial distribution of GHG emissions at a sub-regional scale.

Chapter 7: An Exploration of the Relationship Between Socio-economic and Wellbeing Variables and Household Greenhouse Gas Emissions

7.1 Publication Information

This manuscript has been accepted for publication in the *Journal of Industrial Ecology* pending revisions. It was coauthored by Jeffrey Wilson (lead author), Peter Tyedmers and Jamie Spinney.

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7.2 Abstract

This research reports on a multivariate analysis that examined the relationship between direct greenhouse gas (GHG) emissions and socio-economic and wellbeing variables for 1,920 respondents living in Halifax Regional Municipality (HRM), Nova Scotia, Canada using results from the Halifax Space Time Activity Research (STAR) project. The unique data set allows us to estimate direct GHG emissions with an unprecedented level of specificity based on household energy use survey data and geographic positioning system (GPS) verified personal travel data. Of the variables analyzed, household size, age, income, marital status, and community zone are all statistically significant predictors of direct GHG emissions. Birthplace, ethnicity, educational attainment, perceptions of health, life satisfaction, job satisfaction, happiness, volunteering, or community belonging did not seem to matter. In addition, we examined whether those reporting energy efficient behaviors had lower GHG emissions. No significant differences were discovered among the groups analyzed, supporting a growing body of research indicating a disconnect between environmental attitudes and behaviors and environmental impact. Among the predictor variables, those reporting to be married, young, low income, and

living in households with more people have correspondingly lower direct GHG emissions than other categories in respective groupings. Our finding that respondents with lifestyles that generate higher GHG emissions did not report to be healthier, happier or more connected to their communities suggest that individuals can experience similar degrees of wellbeing regardless of the amount of GHG emissions associated with his or her respective lifestyles.

7.3 Introduction

People naturally aspire to personal wellbeing and high quality lives. The pursuit of wellbeing manifested by decisions regarding the goods and services we consume, where we live and work, how and how far we travel, the dwellings we choose to live in, and what we do with our leisure time all have environmental costs. Studies indicate household consumption contributes between 13% and 35% of a country's total direct greenhouse gas (GHG) emissions. When indirect emissions are included, households contribute upwards of 60% to 80% of a country's emissions (Benders et al. 2006; Kok et al. 2006; Larsen and Hertwich 2010; Lenzen and Peters 2009; Moll et al. 2005; Nansai et al. 2008; Nijdam et al. 2005; Peters and Hertwich 2006; Weber and Mathews 2008).

Most studies exploring the relationship between GHG emissions and household consumption are national in scope and attribute GHG emissions to households using input-output analysis and/ or national level survey data (Benders et al. 2006; Cohen et al. 2005; Moll et al. 2005; Pachauri 2004; Reinders et al. 2003). A limitation of country level analyses is that conclusions based on an average 'national household' lack regional and sub-regional context. Consequently, results fail to offer decision makers specific enough evidence to support decisions and policy reform toward sustainability (Wilson et al. 2012). In addition, analyses typically emphasize refinements in GHG accounting techniques without probing too deeply into understanding the key drivers of GHG emissions. When drivers are considered, they consist of a standard selection of socio-economic and demographic variables collected through Census surveys or as household identifier information (Benders et al. 2006; Druckman and Jackson 2008; Weber and

Mathews 2008). Variables identified in the literature as influencing household-level environmental impact include income, household size (number of occupants), age, dwelling type, and tenure. Of these, income is frequently cited as a strong determinant of energy use or GHG emissions (Abrahamse and Steg 2011; Benders et al. 2006; Druckman and Jackson 2008; Hertwich and Peters 2009; Moll et al. 2005; Peters 2010; Poortinga et al. 2004; Tukker et al. 2010; Weber and Mathews 2008; Weisz and Steinberger 2010; Zacarias-Farah and Geyer-Allély 2003). Despite efforts to understand household-level drivers of environmental impact, analyses lack sub-regional scope and fail to consider a broad suite of potential predictor variables.

We conduct a multivariate analysis of socio-economic and wellbeing variables affecting household GHG emissions for an urban region. Our analysis estimates direct GHG emissions with an unprecedented level of specificity, using household energy use data and geographic positioning system (GPS) verified travel data. We then analyze direct GHG emissions in relation to a broad suite of socio-economic and personal wellbeing variables collected as part of detailed household entry and exit surveys. We also examine GHG emissions in relation to an eco-efficiency index to discern if there is a relationship between energy efficient behaviors and attitudes and environmental impact. In addition to providing a robust test of household-level drivers of impact and the ability to validate if drivers identified using national data apply at the local level, our results provide municipal/regional planners and policy-makers with critical information to support household GHG reduction strategies.

Our inclusion of wellbeing variables such as physical health, mental health, time stress, happiness, life satisfaction, job satisfaction, sense of community and civic engagement reflects the need to better understand the relationship between subjective wellbeing and environmental impact. Long-term sustainability requires visions of wellbeing and quality of life decoupled from high consumption lifestyles. To the best of our knowledge, to date no other multivariate analysis has tested the relationship between impacts (GHG emissions) and subjective assessments of wellbeing (Lenzen and Cummins 2011).

7.4 Methods

7.4.1 Survey Data

Our study employs data from the Halifax Space-Time Activity Research (STAR) Project. The Halifax STAR project collected time diary and questionnaire data over 373 days of data collection between April 2007 and May 2008 from 1,971 households, with a cooperation rate of 25% and overall response rate of 21% (Millward and Spinney 2011). Researchers were unsuccessful in their attempts to stratify for proportional distributions for age, gender, and geography, especially for younger age groups and rural households, but the data approximate a proportional distribution for gender, days of the week, and the four seasons (see Spinney and Millward, 2010). Survey respondents carried a GPS-device (Hewlett Packard iPAQ hw6955) for a 48-hour reporting period providing travel distance data. The "day after" the two-day reporting period, questionnaire and time diary data were collected using a computer-assisted telephone interview technique. See Millward and Spinney (2011) for a detailed description of the survey data including sampling strategy and survey methods. The STAR survey was modeled after Statistics Canada's (2006) General Social Survey, Cycle 19, which asked questions about personal characteristics, household characteristics, neighbourhood features, socio-economic data, and subjective dimensions of wellbeing. The STAR survey supplemented those questionnaire items with questions regarding electricity consumption, home heating, and energy efficiency behaviors, among others.

7.4.2 Estimating Direct Greenhouse Gas Emissions

Our analysis estimates direct GHG emissions associated with the household sector using household energy use data and geographic positioning system (GPS) verified travel data collected as part of the STAR project. Direct GHG emissions refer to Scope 1 and Scope 2 emissions as defined by the World Resources Institute. Direct GHG emissions associated with shelter and transportation were quantified for each respondent in terms of kg of CO_2 equivalents (CO_2e) for an average one-day period. Shelter-related direct GHG emissions are a function of electricity and home heating fuel consumption while transportation-related direct GHG emissions are a function of fossil fuels combusted to provide personal transportation based on actual distances traveled by mode using the geo-referenced time diary data for a random two-day sample period for each respondent.

Our analysis focused exclusively on direct GHG emissions associated with the household sector. Direct GHG emissions refer to Scope 1 and Scope 2 emissions as defined by the World Resources Institute. Related to shelter, we do not include indirect GHG emissions associated with energy production, distribution and trade, electricity and heating infrastructure, construction and maintenance, and operation of energy services. Our analysis also does not include the indirect emissions associated with the physical shelter such as construction, maintenance, and waste removal. For transportation, we do not include the indirect emissions associated with transportation energy production, distribution and trade; emissions related to the manufacture, maintenance and disposal of private vehicles; their transportation infrastructure, construction and maintenance; and, operation of the transport business. We also do not include GHG emissions associated with air travel. In Nova Scotia, air travel represents approximately 7% of total GHG emissions associated with transport (Environment Canada 2010). Of the 1,971 STAR survey participants, 51 respondents were screened out due to either incomplete travel data (n=36) or unreasonably high estimates of shelter related emissions (n=15) (see below) leaving a total of 1,920 participants for whom daily average direct shelter and transport GHG emissions were calculated.

7.4.3 Shelter - Electricity Consumption

Respondents reported their average monthly electricity expenditure. To determine the annual kilowatt hours consumed, we multiplied the monthly average by twelve and subtracted taxes, provincial rebate and the Nova Scotia Power (the provincial private electricity utility operator) base charge to derive an annual kilowatt hour charge. This charge was divided by the 2007 residential price of electricity, to determine the number of kilowatt hours consumed. Household electricity consumption was converted to CO_2 equivalents based on the GHG intensity of electricity in Nova Scotia for the year 2007 reported in the Environment Canada National Inventory Report on GHG emissions (2010). Our estimates do not include energy losses associated with the transmission of electricity from point of generation to point of consumption. Average electrical line losses for the residential sector are 10.5% (McLean, 2012).

7.4.4 Shelter - Home Heating

Respondents reported their household primary and secondary heat sources. Ninety-one percent (91%) of households use either oil (62%) or electricity (29%) as their primary heat source. Other heat sources include wood (5%), heat pump (2%) and natural gas (1%). For households using oil and natural gas, respondents reported their monthly average fuel expenditure. In the case of oil, for example, to determine annual litres consumed, we multiplied the monthly average bill by twelve, adjusted the value to reflect taxes and the provincial oil rebate, and divided by the 2007 average annual retail price of heating oil per litre for Halifax Regional Municipality (Statistics Canada 2007). We converted litres of oil to CO_2 equivalent based on the GHG intensity of heating oil for the year 2007 from the Environment Canada National Inventory Report on GHG emissions (2010). Survey respondents heating with wood reported the number of cords (where 1 cord is equivalent to ~3.62m^3 of split and stacked wood) burnt per year. We converted cords of wood to CO_2 equivalent based on the GHG intensity of burning mixed hardwood. The GHG coefficient for mixed hardwood was provided by Efficiency Nova Scotia and is used in their space heating comparison report (2011). The energy consumption for those heating with electricity and the electricity required to run a heat pump is captured in electricity consumption data. Table 1 presents CO_2 equivalent emission factors by heating source used in our analysis. As noted oil and electricity are the two primary heat sources. The GHG intensity factor for electricity is 0.74 kg CO_2e/kwh. The GHG factor for oil is 0.27 kg CO_2e/kwh. In comparison to other Canadian provinces, electricity in Nova Scotia has a relatively high associated CO_2e impact per unit of energy. Coal and refined petroleum products accounted for 68% of electricity generation in Nova Scotia in 2007.

Table 7.1: Greenhouse gas emission factors by heating source

Heating oil (kg CO_2-e/ litre)	2.91
Electricity (kg CO_2-e/kwh)	0.74
Natural gas (kg CO_2-e/m^3)	1.90
Wood (kg CO_2-e /cord)	2,653

Greenhouse gas emissions estimates for 68 households were more than two standard deviations above or below the annual mean value of 14,775 kg of CO_2e per household. We had an energy expert (McLean 2011) at a provincial energy efficiency agency (Efficiency Nova Scotia) review the legitimacy of the values. To inform the expert's opinion, the expert was provided respective household GHG estimates stratified by electricity and other heating sources, number of household members, dwelling type, square footage, primary and secondary heat sources, and age of furnace (if relevant). The expert ranked the likelihood of results on a scale from 1 to 5 with values corresponding to very possible, possible, unlikely, highly unlikely, and not possible. The expert also included a brief rationale explaining his ranking decisions. The expert deemed 15 entries as either highly unlikely or not possible. The 15 entries were removed from our analysis. Potential errors leading to unrealistic results include data entry errors, recall errors, ignorance, and subtle non-response.

7.4.5 Transportation

STAR project participants carried a GPS logger tracking their travel for two consecutive "diary days" (48-hour period). The distribution of diary day pairs is as follows: two weekdays, 61% of respondents; one weekday and one weekend day, 26% of respondents; two weekend days, 13% of respondents. In addition to carrying the GPS unit, respondents also completed a time use diary, which provided insight into such things as the purpose of trips taken, the travel mode and whether they were the driver or passenger. We tallied travel episodes in meters by mode of travel for the two-day reporting period. Mode of travel includes personal vehicle, bus, taxi, motorcycle, walk, and bicycle. Personal vehicles include car, van, sport-utility vehicle (SUV), pick-up truck, other truck, and other vehicle. Respondents used personal vehicles for 92% of the total distance travelled. We further categorized personal vehicle use into driver and passenger.

We categorized travel episodes into three groups, travel to and from work, work-related travel, and personal or non-work travel. Travel to and from work includes travel from home to work destination and any stops along the way. Work related travel includes travel associated with an individual's work such as driving a taxi, or delivering pizzas. It also includes any travel required to deliver work-related services. This would include traveling to someone's home to deliver a service or visiting an off-site location for a work-related function. Personal travel includes travel for 24 different potential trip purposes such as travel for sports and entertainment events, travel for crafts and hobbies, travel for religious services, or travel to restaurants. Across all respondents, personal travel accounted for approximately 70% of total kilometers travelled, followed by work-related travel (20%), and travel to and from work (10%). As our analysis focused on understanding the influence of socio-economic and other influences on household-level impacts, we removed all travel *during work* from our estimates of direct GHG emissions attributed to personal transportation.

For trips made using a personal vehicle, we assumed each respondent used the household-owned vehicle for which they were identified as the primary driver. Five percent of respondents reported being the primary driver of more than one vehicle. In these cases, we assumed a vehicle profile based on the average of the respective vehicles. In instances where respondents indicated 'other', 'did not know' or 'refused' (2%) in relation to their primary vehicle use, we assumed vehicle characteristics that were an average of car and light duty truck. Personal vehicle fuel type was also noted for each vehicle. Gasoline vehicles made up 95% of total privately owned vehicles in our dataset.

For each travel episode using personal vehicles, we knew whether the respondent was the driver or a passenger and, in instances when they drove, whether passengers were present or not. We did not, however, know the total number of passengers present. Over the entire dataset drivers had a passenger in the vehicle for 15.7% of trips. In these cases, we attribute half the trip-related emissions to the respondent drivers. Further partitioning of emissions was not undertaken on the assumption that the number of trips with more than one passenger was relatively small. We also expect that backseat passengers (usually children) are often 'captive passengers' going along on the trip with an adult. In cases where the respondent was the passenger, we attribute half the total emissions of the trip to the respondent. When trips were taken by taxi, we assumed the vehicle employed was a car.

For all privately owned vehicle-based trips, transport-related GHG emission estimates were made by multiplying total meters travelled by each travel mode's respective GHG emissions factors from Canada's National Inventory Report on GHG emissions (2010). The emissions factors for "light duty trucks" in the National Inventory Report were applied to all transport undertaken by van, pick-up truck, and SUV. For all modes of transport, emission factors in the National Inventory Report assume tier one emission factors for gasoline vehicles and light duty trucks, advanced control emission factors for diesel vehicles and light duty truck and non-catalytic controlled for motorcycles. To express results in CO_2 equivalent we adopted the global warming potential (GWP) conversion factors for various GHGs used by the Intergovernmental Panel on Climate Change IPCC (2007) published in its Fourth Assessment Report.

For trips made using public transit buses, we apply an average load factor for a 58-passenger bus used in the Halifax Regional Municipality as reported by O'Keefe and colleagues (2009) and GHG emission rates per passenger/kilometer of bus travel based on data from the United States Department of Transportation Bureau of Transportation Statistics (2008). No GHG emissions were attributed to active forms of transportation such as walking and biking. Table 2 summarizes the GHG conversion factors applied in our analysis.

Table 7.2: Greenhouse gas emission factors for all modes of transport employed

	Kg CO_2 equivalent per km
Car (gasoline)	0.206
Car (diesel)	0.150
Car (hybrid)	0.117
Light duty truck (gasoline)	0.294
Light duty truck (diesel)	0.281
Motorcycle	0.119
Taxi	0.206
Bus (per passenger/km)	0.018
Walking	0.000
Biking	0.000

Thirty-six respondents (36) had no travel episode data and were screened from our analysis. Reasons for having no travel episode data include: participants made no trips during the two day period (did not leave the house); there was no location information provided for the origins and/or destinations; or, the locations that were given were unable to be found when run through the network analyst tool in ArcGIS. In the STAR data, there are instances of people traveling long distances for various purposes, such as visiting relatives and friends or simply taking a pleasure drive. Thirty-five respondents had GHG emissions associated with travel events that were three standard deviations above the mean. Although outliers, we did not remove these cases. Travel behavior research confirms that "extreme" trips are realistic events. While these events appear to be atypical amongst those surveyed, a review of travel behavior studies by Kang and Scott (2010) confirm considerable variability in people's activities and travel across days of the week.

7.4.6 Statistical Analysis

Correlation analysis and multivariate modelling were used to explore the statistical relationship between direct GHG emissions (CO_2e kg person^{-1} day^{-1}) and several variables in order to gain a better understanding of the potential drivers of emissions at the household level. Influenced by an extensive review of the literature and supported by a comprehensive list of questionnaire items available in the STAR data, twenty variables were considered for inclusion in the multivariate regression model (Table 3).

Table 7.3: Variables considered for inclusion in model (arranged alphabetically)

Age	Gender
Birth place	Happiness
Civic engagement	Household size
Community belonging	Job satisfaction
Community zone	Life satisfaction
Educational attainment	Marital status
Employment status	Primary heat source
Energy efficiency engagement	Personal income
Ethnicity	State of health
Financial security	Time stress

In addition to socio-economic and well-being variables, we included primary heat source as variable recognizing the prevalence of households using oil (62%) and electricity (29%). The energy efficiency engagement index and time stress index were compiled based on related questions on the respective themes in the entry and exit surveys. The energy-efficiency engagement index, for example, considers responses to five questions. Each question is a yes/no answer question. To develop the index, a 'yes' response was assigned a 1, a 'no' response 0. Values for the five questions were summed giving respondents a score range between 0 and 5. A '5' on the scale corresponds to high energy-efficiency engagement; a score of '0' on the scale corresponds to low energy-efficiency engagement. Questions included: do you take energy efficient measures to keep warm? Do you think energy efficiency is important? Do you use energy efficient light bulbs? Have you considered solar hot water? Have you considered a heat pump?

Using SPSS 20.0 software, we explored these socio-economic and subjective wellbeing variables independently to investigate the nature (e.g. linearity) of their relationship with total daily personal direct GHG emissions and to determine their statistical suitability for inclusion in our multivariate model by performing correlation analysis. Those variables for which there was a statistically significant relationship with GHG emissions, based on the initial correlation analysis, were used in a multiple regression analysis and are described in Table 4. Multiple regression analysis, using ordinary least squares, was used to investigate the strength and direction of the statistical relationship between direct GHG emissions and the independent variables chosen for inclusion in the model.

Table 7.4: Socio-economic variables included in regression model

Variable	**Description**
Direct GHG Emissions	Direct GHG emissions are measured per person per day, with a minimum value of 2.1 kg CO_2e person^{-1} day^{-1}, a maximum of 103.5, a median of 20.8, and a mean of 24.0.
Age	Age is measured in seven 10-year categories (1=15-24, 2=25-34, 3=35-44, 4=45-54, 5=55-64, 6=65-74, 7=75+).
Community Zone	There are four zones based primarily on travel distance from "downtown"; the inner-city zone (0 to 5 km), the suburbs (5 to 10 km), the inner commuter belt (10 to 25 km), and the outer commuter belt (25 to 50 km from downtown).
Employment status	Employment status is binary with unemployed (0) and employed (1).
Financial security	Financial security measures subjective feelings about the respondents' finances and values range from 1 (very dissatisfied) to 5 (very satisfied).
Gender	Gender is binary with male (0) and female (1).
Household size	Household sizes were categorized into five groups based on number of people living at home. Groups corresponded to actual number with group 5 equaling households of five or more persons.
Marital status	Marital status is binary with not married (0) and married (1).
Personal income	Personal income is measured in six categories of before tax income (1=0-$19,999, 2=$20,000-$39,999, 3=$40,000-$59,999, 4=$60,000-$79,999, 5=$80,000-$99,999, 6=$100K and above).
Time stress	Time stress is on a five-point scale, with values ranging from 1 (not stressed) to 5 (very high stress).
Heating Source	Heating source is binary with electric (0) and oil (1).

Supplemental table A offers a more complete analysis of the statistical relationship between direct GHG emissions and the independent variables chosen for inclusion in the model using a Generalised Linear Model to illustrate the influence of each of the independent variables.

7.5 Results

7.5.1 Direct GHG Emissions

Statistical analysis of the STAR household data indicates mean total direct GHG emissions of 24 kg CO_2e person^{-1} day^{-1}. Overall, median total direct GHG emissions for HRM was 21 kg CO_2e person^{-1} day^{-1}, with a range of 101 kg, and inter-quartile range of 14 kg. On average, provision of electricity and home heating contribute 78% of direct GHG emissions with a median of 15 kg CO_2e person^{-1} day^{-1}, which is significantly higher ($p = <0.001$) than emissions associated with personal transportation, which account for the balance of direct GHG emissions with a median of 4 kg CO_2e person^{-1} day^{-1}.

Table 7.5: Summary statistics for direct GHG emissions by socio-economic groups

	N	Mean GHG Emissions*	Standard Deviation	95% Confidence Interval
Age				
15-24	56	18.0	9.9	15.3 - 20.6
25-34	113	18.6	9.6	16.8 - 20.4
35-44	363	20.7	13.0	19.4 - 22.1
45-54	593	22.5	11.6	21.6 - 23.4
55-64	455	28.0	14.7	26.6 - 29.3
65-74	230	26.8	13.5	25.0 - 28.5
75+	88	29.5	15.7	26.1 - 32.8
Missing	22			
Community Zone				
Inner City	382	22.8	12.7	21.5 - 24.1
Suburbs	1039	23.4	12.8	22.6 - 24.2
Inner commuter belt	334	26.0	14.5	24.4 - 27.5
Outer commuter belt	165	26.1	15.0	23.8 - 28.4
Missing	0			
Employment status				
Employed	1288	22.6	13.3	21.9 - 23.4
Not Employed	624	26.6	13.1	25.6 - 27.6
Missing	8			

Financial Security				
Very dissatisfied	51	23.1	13.1	19.4 - 26.8
Dissatisfied	92	22.9	12.8	20.3 - 25.6
Neutral	351	23.0	12.1	21.7 - 24.2
Satisfied	936	25.3	13.6	22.9 - 24.7
Very satisfied	470	24.9	13.9	24.1 - 26.6
Missing	20			
Gender				
Female	1033	23.0	12.4	22.3 - 23.8
Male	887	25.0	14.3	24.1 - 26.0
Missing	0			
Household size				
One	233	36.3	15.4	34.3 - 38.2
Two	746	25.6	13.1	24.7 - 26.6
Three	374	21.7	10.6	20.6 - 22.8
Four	412	18.5	10.0	17.5 - 19.5
Five	155	17.4	11.0	15.7 - 19.2
Missing	0			
Marital status				
Not married	391	28.9	16.0	27.3 - 30.5
Married	1527	22.7	12.3	22.1 - 23.3
Missing	2			
Personal income				
Under $20,000	213	21.0	12.1	19.4 - 22.7
$20,000 - $39,999	409	23.6	12.1	22.4 - 24.8
$40,000 - $59,999	410	23.4	11.6	22.3 - 24.5
$60,000 - $79,999	297	23.8	14.7	22.1 - 25.4
$80,000 - $99,999	121	27.1	15.2	24.3 - 29.8
$100,000 or more	146	27.0	16.4	24.3 - 29.7
Missing	324			
Time stress				
Not stressed	324	26.0	13.8	24.5 - 27.5
Low stress	736	24.9	14.0	23.8 - 25.9
Medium stress	525	23.3	13.4	22.2 - 24.5
High stress	229	20.9	10.1	19.5 - 22.2
Very high stress	106	21.2	12.3	18.8 - 23.5
Missing	0			
Heat Source				
Electric	564	24.5	13.5	23.4 - 25.6
Oil	1179	23.9	13.4	23.1 - 24.7
Other/Missing	177	22.4	12.5	20.6 - 24.3

* Direct GHG Emissions are measured in kg CO_2e $person^{-1}$ day^{-1}

Table 5 provides summary statistics for the ten variables passing the initial univariate screen including direct GHG emissions. Mean GHG emissions increase with age category ranging from a low in the 15-24 cohort to a high in the 75+ age cohort. With respect to community zone, mean GHG emissions are similar among inner city and suburban respondents but are greater among respondents living in the inner and outer commuter belt zones. Mean emissions were higher among not employed respondents than for employed respondents. Those responding 'satisfied' and 'very satisfied' with their financial security (74% of sample) had higher mean direct GHG emissions than those responding 'dissatisfied and 'very dissatisfied'. Females (54% of sample) had slightly lower mean GHG emissions when compared to male respondents. Mean direct GHG emissions per person decrease as the number of household members increase ranging from a high in one-person households to a low in households of five persons or more. Married respondents (80% of sample) reported lower mean direct GHG emissions than respondents who were not married. Mean direct GHG emissions are virtually the same among respondents reporting incomes in the ranges, \$20,000 - \$39,999; \$40,000 - \$59,999, and \$60,000 - \$79,999. Mean GHG emissions, however, jump noticeably between the \$60,000 - 79,999 range and the \$80,000 - \$99,999 range. Respondents with time stress ratings of 'no stress' or 'low stress' (54% of sample) reported higher GHG emission in comparison to respondents with time stress ratings of 'high' or 'very high' (17%). Finally, whether a household uses electric heat or an oil furnace as their primary heat source appears to have little impact on the mean direct GHG emissions.

7.5.2 Correlation Analysis

Based on correlation analysis, 7 variables proved to be significantly associated with direct GHG emissions for those respondents who heat with electricity (Table 6). These include: age, community zone, employment status, household size, marital status, personal income, and time stress. For those respondents who heat with oil (Table 7), financial security and gender were also significantly associated with direct GHG emissions. In terms of the larger list of variables, excluded variables of note include the household engagement in eco-efficiency index and respondent's personal assessment of their state of happiness and health.

Table 7.6: Correlation analysis of direct GHG emissions and socio-economic groups (oil heat)

	Direct GHG Emissions	Age	Community Zone	Employment Status	Financial security	Gender	Household size	Marital Status	Personal income	Time stress
Direct GHG Emissions	1	0.206	0.073	-0.117	0.065	0.074	-0.374	-0.225	0.13	-0.117
Sig. (2-tailed)		*< 0.000*	*0.012*	*< 0.000*	*0.027*	*0.011*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*
Age	0.206	1	-0.164	-0.596	0.173	0.131	-0.485	-0.013	-0.012	-0.294
Sig. (2-tailed)	*< 0.000*		*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*	*0.667*	*0.713*	*< 0.000*
Community Zone	0.073	-0.164	1	0.098	-0.046	0.004	0.159	0.107	0.013	0.061
Sig. (2-tailed)	*0.012*	*< 0.000*		*0.001*	*0.116*	*0.891*	*< 0.000*	*< 0.000*	*0.687*	*0.035*
Employment Status	-0.117	-0.596	0.098	1	-0.101	0.022	0.365	0.113	0.266	0.315
Sig. (2-tailed)	*< 0.000*	*< 0.000*	*0.001*		*0.001*	*0.452*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*
Financial security	0.065	0.173	-0.046	-0.101	1	0.103	-0.109	0.109	0.227	-0.203
Sig. (2-tailed)	*0.027*	*< 0.000*	*0.116*	*0.001*		*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*
Gender	0.074	0.131	0.004	0.022	0.103	1	-0.004	0.185	0.378	-0.096
Sig. (2-tailed)	*0.011*	*< 0.000*	*0.891*	*0.452*	*< 0.000*		*0.89*	*< 0.000*	*< 0.000*	*0.001*
Household size	-0.374	-0.485	0.159	0.365	-0.109	-0.004	1	0.391	0.086	0.237
Sig. (2-tailed)	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*	*0.89*		*< 0.000*	*0.007*	*< 0.000*
Marital Status	-0.225	-0.013	0.107	0.113	0.109	0.185	0.391	1	0.191	0.066
Sig. (2-tailed)	*< 0.000*	*0.667*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*	*< 0.000*		*< 0.000*	*0.023*
Personal income	0.13	-0.012	0.013	0.266	0.227	0.378	0.086	0.191	1	0.028
Sig. (2-tailed)	*< 0.000*	*0.713*	*0.687*	*< 0.000*	*< 0.000*	*< 0.000*	*0.007*	*< 0.000*		*0.376*
Time stress	-0.117	-0.294	0.061	0.315	-0.203	-0.096	0.237	0.066	0.028	1
Sig. (2-tailed)	*< 0.000*	*< 0.000*	*0.035*	*< 0.000*	*< 0.000*	*0.001*	*< 0.000*	*0.023*	*0.376*	

Table 7.7: Correlation analysis of direct GHG emissions and socio-economic groups (electrical heat)

	Direct GHG Emissions	Age	Community Zone	Employment Status	Financial security	Gender	Household size	Marital Status	Personal income	Time stress
Direct GHG Emissions	1	0.31	0.094	-0.191	0.049	0.059	-0.401	-0.129	0.106	-0.106
Sig. (2-tailed)		*< 0.000*	*0.026*	*< 0.000*	*0.251*	*0.159*	*< 0.000*	*0.002*	*0.021*	*0.012*
Age	0.31	1	-0.079	-0.539	0.137	0.159	-0.426	0.051	-0.012	-0.213
Sig. (2-tailed)	*< 0.000*		*0.061*	*< 0.000*	*0.001*	*< 0.000*	*< 0.000*	*0.228*	*0.797*	*< 0.000*
Community Zone	0.094	-0.079	1	0.086	-0.029	-0.043	0.13	0.158	0	0.075
Sig. (2-tailed)	*0.026*	*0.061*		*0.042*	*0.488*	*0.303*	*0.002*	*< 0.000*	*0.996*	*0.074*
Employment Status	-0.191	-0.539	0.086	1	-0.125	-0.011	0.311	0.024	0.231	0.253
Sig. (2-tailed)	*< 0.000*	*< 0.000*	*0.042*		*0.003*	*0.787*	*< 0.000*	*0.566*	*< 0.000*	*< 0.000*
Financial security	0.049	0.137	-0.029	-0.125	1	0.035	-0.067	0.099	0.233	-0.3
Sig. (2-tailed)	*0.251*	*0.001*	*0.488*	*0.003*		*0.402*	*0.115*	*0.019*	*< 0.000*	*< 0.000*
Gender	0.059	0.159	-0.043	-0.011	0.035	1	-0.022	0.144	0.335	-0.069
Sig. (2-tailed)	*0.159*	*< 0.000*	*0.303*	*0.787*	*0.402*		*0.608*	*0.001*	*< 0.000*	*0.103*
Household size	-0.401	-0.426	0.13	0.311	-0.067	-0.022	1	0.406	0.131	0.198
Sig. (2-tailed)	*< 0.000*	*< 0.000*	*0.002*	*< 0.000*	*0.115*	*0.608*		*< 0.000*	*0.004*	*< 0.000*
Marital Status	-0.129	0.051	0.158	0.024	0.099	0.144	0.406	1	0.176	0.066
Sig. (2-tailed)	*0.002*	*0.228*	*< 0.000*	*0.566*	*0.019*	*0.001*	*< 0.000*		*< 0.000*	*0.118*
Personal income	0.106	-0.012	0	0.231	0.233	0.335	0.131	0.176	1	-0.043
Sig. (2-tailed)	*0.021*	*0.797*	*0.996*	*< 0.000*	*< 0.000*	*< 0.000*	*0.004*	*< 0.000*		*0.346*
Time stress	-0.106	-0.213	0.075	0.253	-0.3	-0.069	0.198	0.066	-0.043	1
Sig. (2-tailed)	*0.012*	*< 0.000*	*0.074*	*< 0.000*	*< 0.000*	*0.103*	*< 0.000*	*0.118*	*0.346*	

Tables 6 and 7 presents the correlation coefficients and significance of the variables included in our multivariate model stratified by electric and oil heat respectively. In both of the tables, household size, age, and marital status are most strongly correlated with direct greenhouse gas emissions. Age has a stronger association with GHG emissions for electric heat, but GHG emissions for both heat sources are significantly associated with age. Community zone is significantly associated with GHG emissions with similar strength of association in households that heat primarily with oil or electricity. Personal income and time stress exhibit statistically significant associations with GHG emissions for both heat sources, the association, however, is highly significant for respondents heating with oil. Tables 6 and 7 also highlight potential issues for autocorrelation. For example, time stress is significantly correlated with all other variables for households that heat with oil, save personal income (Table 6), while personal income is significantly associated with all variables except age and community zone for those who use electric heat (Table 7).

7.5.3 Multiple Regression Analysis

Results of the fitted multivariate linear regression model using direct GHG emissions (kg CO_2e person^{-1} day^{-1}) as the dependent variable appear in Table 8, while the results from the generalised linear model can be found in Appendix A. Beta values (i.e. standardised coefficients) from the model (Table 8) indicate the strength of the statistical association and, because they are standardized, enable direct comparison of the relative strength of each independent variable. A negative Beta value indicates an inverse relationship between the two variables and as values move away from zero the strength of the relationship increases. Scores from the t-test and their significance are also presented to illustrate the significance of each variable in the fitted model. Multiple regression analysis, using ordinary least squares, indicates that five of the ten variables (age, community zone, household size, marital status, and personal income) are significantly associated with direct GHG emissions, when all other variables are accounted for. Employment status, time stress, gender, financial security, and heating source failed to significantly contribute to the explanation of direct GHG emissions while household size, income, community zone, age, and marital status all did. None of the originally considered wellbeing variables were found to be significant predictors of household greenhouse gas emissions.

Table 7.8: OLS regression of direct GHG emissions against significant socio-economic drivers

	Unstandardized Coefficients	Standard Error	Beta	t	Sig.
Constant	22.29	2.951		7.554	<0.001
Household Size	-3.564	0.346	-0.309	-10.300	<0.001
Personal Income	1.518	0.255	0.164	5.945	<0.001
Community Zones	2.598	0.391	0.159	6.647	<0.001
Age	1.364	0.338	0.135	4.039	<0.001
Marital Status	-4.398	0.892	-0.134	-4.933	<0.001
Gender	0.795	0.692	0.030	1.149	0.251
Time Stress	-0.156	0.164	-0.024	-0.952	0.341
Heat Source	-0.643	0.669	-0.023	-0.962	0.336
Employment Status	0.257	0.967	0.009	0.265	0.791
Financial Security	-0.083	0.375	-0.006	-0.222	0.825
R-square			**0.221**		

The fitted regression model was able to explain 22.1% of the total variation in direct GHG emissions using five independent variables. Relatively, the association between direct GHG emissions and household size (β = -0.309, p= <0.001) appears to be twice as strong as the other predictor variables. Among them, personal income (β = 0.164, p= <0.001), location along the urban-rural continuum (β = 0.159, p= <0.001) exceed both age (β = 0.135, p= <0.001) and marital status (β = -0.134, p= <0.001).

7.6 Discussion

7.6.1 GHG Predictor Variables

Our findings, based on an urban focus confirm number of people per household, income, community zone, age, and marital status, as predictors of direct GHG emissions. Results are consistent with studies drawing on national level data (Abrahamse and Steg 2011; Druckman and Jackson 2008; Hertwich and Peters 2009; Peters 2010; Poortinga et al. 2004; Tukker et al. 2010; Weber and Mathews 2008; Weisz and Steinberger 2010). The number of people per household had the strongest association with direct greenhouse gas emissions. While shelter-related electricity and home heating fuel consumption is greater in larger households, it does not increase in a linear fashion. As a result, an increase in the number of people per household offsets the marginally greater impact associated with electricity and home heating. Tucker and colleagues (2010) and Weber and Mathews (2008) note a strong association between household size and GHG emissions but suggest that the influence of income is greater. Both studies consider indirect GHG emissions, which may explain the increased prominence of income as a predictor variable.

Other predictor variables identified in our analysis including age, marital status and income suggest that where people are along the life continuum influences GHG emissions. Younger, lower income, and married were associated with lower greenhouse gas emissions when compared to other groupings within the same category. Conceptualizing environmental impact around life stage offers a unique perspective for planning GHG reduction strategies. For example, what are the GHG implications of changing demographic trends toward older populations in high consumption countries?

Income, while related to life stage, is a unique variable as it directly determines capacity to consume. The relationship between income and GHG emissions will be different whether indirect GHG emissions were included. Indirect GHG emissions refer to the emissions that result from earlier stages in the life cycle of the goods and services that we consume. Indirect emissions have been found to correlate closely with income (Baiocchi et al. 2010; Lenzen et al. 2008; Peters 2010; Satterthwaite 2009). The association is logical, income drives expenditures and all expenditures mobilize resources. As Wilson and colleagues (2012) note, however, there is a need to interpret the degree of correlation cautiously. Methods to estimate indirect emissions are often based on expenditure surveys which auto-correlate with income. In other words, it is not a surprise that studies have found correlations between indirect emissions and income when the former is, in part, quantified based on the latter. Direct GHG emissions are assumed to begin decoupling at higher income levels based on the assumption that people can only drive so much and have so many electronic appliances (Lenzen et al. 2008). We found similar GHG emissions among those in lower-mid income ranges with a noticeable jump in GHG emissions among those in the $80,000 to $99,999 category. Those falling in the highest income range considered in our analysis ($100,000 or more) reported similar GHG emissions to those in the $80,000 to $99,999 category suggesting the possibility of decoupling. The similarity of GHG emissions among low-middle income groups may indicate that incomes up to a certain amount are allocated to support general lifestyle needs. Once that level has been attained, income can be allocated toward buying the bigger home, travelling more, or having more electronic gadgets, thus explaining the jump at higher income levels.

Community zone, the other predictor variable identified in our analysis, is not typically included in other analyses studying drivers of GHG emissions. Vandeweghe and Kennedy (2007) estimated residential GHG emissions by census tract for the Toronto Census Metropolitan Area (Canada) based on household fuel payments and transportation survey data. They found, for the most part, that census tracts located further from the inner city had comparatively higher per capita GHG emissions, although some inner city census tracts had higher GHG emissions than suburban census tracts. As expected, we also found those living greater distance from the inner city generally had higher associated GHG emissions. Distance, however, is not the sole explanatory factor. When examined by zone, we found similar levels of GHG emissions between respondents living in the inner city and suburb zones and between those living in the inner and outer commuter belt zones. The lack of difference between those living in the inner city and those living in the suburbs questions the widely held assumption that people living in the urban core have lower environmental impact lifestyles than do their suburban counterparts (Hoornweg et al. 2011). Understanding the relationship between where we live and environmental impact presents an interesting area of future research.

Our analysis took a consumption-based approach assessing environmental impact of Halifax Regional Municipality focusing on direct greenhouse gas emissions associated with the household sector. Urban environmental sustainability can also be assessed using metabolism based and complex systems approaches. See Baynes and Wiedmann (2012) for useful descriptions of the three approaches and prominent studies using the various techniques. As noted by Baynes et al. (2011), the approaches offer different conceptualizations for understanding urban environmental impact and subsequently convey different results. Urban metabolism studies, which are defined by geo-political boundaries, for example, highlight population density and urban form as critical factors influencing environmental impact (Baynes and Wiedmann, 2012). In terms of advancing sustainability, the approaches are complementary and support different uses. Baynes and Wiedmann (2012) suggest consumption based techniques are most useful for informing policies directed at influencing household consumption and least useful for informing urban land use planning as they lack direct territorial representation.

Our results offer interesting insights but clearly, different insights might be obtained if the study was conducted in different jurisdictions. For example, larger cities with higher population densities, warmer climates, and less GHG intensive heating and electricity sources may produce different findings. In Halifax Regional Municipality, for example, heating and electricity related emissions dominate total GHG emissions. The contribution of heating and electricity to overall emissions can be explained in part by Halifax's reliance on oil and electricity as a primary heating source coupled by the fact that the greenhouse gas intensity of electricity generation in the province is high, over triple the Canadian average. Jurisdictions relying on hydro generated electricity and heating primarily with natural gas, for example, would have lower GHG emissions associated with heating and electricity consumption. In comparison to other regions in Canada, the housing stock is older and there are more single-family dwellings as a percentage of the total housing stock (NRCAN, 2010).

Given the dominance of oil and electricity as a primary source of home heating, 62% and 29% respectively, we considered the distinction between the two types of heating as a potential variable influencing GHG emissions. We found similar mean direct GHG emissions between the primary heat sources. As noted above, the carbon intensity of electricity in Nova Scotia is very high and in fact slightly above that of oil. As the province reduces the carbon intensity of the electrical grid and natural gas increasingly becomes available across the municipality, the influence of primary heat source will need to be readdressed. Exploring the implications of a shift to alterative heat sources would offer valuable insights to inform energy policy. Understanding, for example, the potential reduction in GHG emissions associated with wider use of natural case as heating fuel may increase political pressure to increase areas serviced by natural gas. The expansion of natural gas as a heating source has been slow in HRM. The current provider has been reluctant to expand services unless a guaranteed number of households per area switch to natural gas heating systems.

7.6.2 Wellbeing Variables

Our analysis pioneers including wellbeing variables in efforts to better understand the relationship between subjective wellbeing and environmental impact at an urban level. In the end, however, all wellbeing variables were dropped from our model. Regardless of differences in health, life satisfaction, job satisfaction, happiness, civic engagement, or sense of community belonging, we did not find any significant differences in GHG emissions. Respondents with lifestyles that generate higher GHG emissions did not report to be healthier, happier or more connected to their communities. The results suggest that individuals can experience similar degrees of wellbeing regardless of the amount of GHG emissions associated with lifestyle. The New Economics Foundation (2012) similarly did not find a relationship between wellbeing and environmental impact using national data.

Their analysis compared national ecological footprints and national carbon footprints, as proxies of environmental impact, with global data on well being and life expectancy. Other nation based studies, however, have noted a relationship between improved environmental quality and increases in wellbeing. Studies by Welsch (2006) and Di Tella, and MacCulloch (2008), for example, examining the relationship between air pollutants and wellbeing in European countries and OECD countries respectively found that lower levels of pollutants correspond with higher levels of wellbeing. Welsch (2006) found a significant negative relationship between nitrogen dioxide and lead and life satisfaction. Di Tella and MacCulloch (2008) found sulphur dioxide emissions have an adverse effect on reported wellbeing. In the respective studies, wellbeing is the dependent variable as opposed to environmental impact as in our case. Further, our study examines the relationship between wellbeing and greenhouse gas emissions, as a proxy of personal environmental impact. Welsch's (2006) and Di Tella, and MacCulloch's (2008) research suggest a less polluted environment corresponds to higher levels of wellbeing. Our research more broadly suggests that wellbeing is not contingent upon high environmental impact lifestyles challenging a widely held view that levels of consumption correspond to wellbeing. Understanding the relationship between wellbeing and environmental impact offers a fascinating and needed area of future research.

7.6.3 Energy Efficiency Engagement Index

The energy efficiency engagement index variable was dropped from the model. Our results support a small but emerging consensus in the literature that environmental attitudes and behaviors do not translate into lower environmental impact. Gatersleben and colleagues (2002) and Custora (2012) found that household energy use and carbon footprints respectively were weakly related to environmental attitudes with income being a better predictor in both studies. Further analysis of 'green' consumers by Custora (2012) found no association between footprint score and 'green' attitude. Baiocchi and colleagues (2010) in a study of carbon dioxide emissions in the United Kingdom using geodemographic consumer segmentation data found that carbon emissions increased with membership in environmental organizations. Environmental attitudes do not appear to influence travel patterns either. Susilo and colleagues (2012) in a study of the UK found that travel behavior did not necessarily match concern about the environment. Exploring the relationship between environmental behaviors, impact and income is an important area of future research. The potential that income trumps environmental attitudes has significant public policy implications.

7.7 Conclusion

The sustainability imperative requires that we reduce the aggregate environmental impact of our lifestyles. Identifying drivers of GHG emissions offers insight at multiple levels of decision making to support GHG reductions. Of the 20 variables considered in our analysis, number of persons per household, income, community zone, age, and marital status were identified as predictors of GHG emissions explaining 22% of the variation in findings. The variables not associated with GHG emissions are perhaps more telling, calling for additional avenues of interdisciplinary research. Two notable areas include better understanding the connection between subjective well being and environmental impact and finding opportunities to decouple income from environmental impact. Further research must also extend into barriers to personal action, and psychological drivers of the consumer lifestyle. Redefining personal aspirations independent of affluence and high consumption is essential for long-term sustainability. Our findings offer mixed views toward achieving this goal. Degree of happiness, life satisfaction, health, sense of community belonging and civic engagement were not associated with GHG emissions, suggesting that lower GHG emission lifestyles do not compromise wellbeing. Our finding, however, that eco-efficient attitudes are not associated with environmental impact highlights the complexity of leading smaller footprint lifestyles and the role that income plays in trumping environmental attitudes. Reconciling visions of wellbeing and reduced environmental impact is paramount for long-term human sustainability and a critical research agenda.

7.8 Supplemental Table, Generalised Linear Model

Parameter	B	Standard Error	95% Wald Confidence Interval (Lower – Upper)	Significance
			(38.783 - 53.497)	< 0.000
Age=1	-7.462	2.722	(-12.798 - -2.127)	0.006
Age=2	-7.567	2.217	(-11.913 - -3.221)	0.001
Age=3	-4.413	2.049	(-8.428 - -0.398)	0.031
Age=4	-3.960	2.007	(-7.895 - -0.026)	0.049
Age=5	-0.789	1.863	(-4.441 - 2.862)	0.672
Age=6	-1.719	1.864	(-5.372 - 1.934)	0.356
Age=7	0a			
Community Zone=1	-6.922	1.468	(-9.800 - -4.045)	< 0.000
Community Zone=2	-4.782	1.331	(-7.391 - -2.174)	< 0.000
Community Zone=3	-0.905	1.517	(-3.879 - 2.069)	0.551
Community Zone=4	0a			
Employment=.00	-0.205	1.141	(-2.440 - 2.031)	0.858
Employment=1.00	0a			
Financial Security=1	-1.690	1.523	(-4.675 - 1.295)	0.267
Financial Security=2	2.468	1.546	(-0.561 - 5.498)	0.110
Financial Security=3	-0.304	0.976	(-2.217 - 1.608)	0.755
Financial Security=4	-0.132	0.854	(-1.804 - 1.541)	0.877
Financial Security=5	0a			
Gender=.00	-0.818	0.658	(-2.108 - 0.472)	0.214
Gender=1.00	0a			
Household Size	-3.484	0.319	(-4.109 - -2.860)	< 0.000
Marital Status=.00	4.459	0.985	(2.529 - 6.390)	< 0.000
Marital Status=1.00	0a			
Personal Income=1	-7.045	1.620	(-10.221 - -3.870)	< 0.000
Personal Income=2	-5.819	1.442	(-8.645 - -2.993)	< 0.000
Personal Income=3	-5.612	1.409	(-8.374 - -2.850)	< 0.000
Personal Income=4	-3.594	1.526	(-6.585 - -0.603)	0.019
Personal Income=5	-0.444	1.886	(-4.140 - 3.252)	0.814
Personal Income=6	0a			
Time Stress=0	-1.354	2.550	(-6.352 - 3.643)	0.595
Time Stress=1	-1.528	2.493	(-6.414 - 3.358)	0.540
Time Stress=2	-1.004	2.530	(-5.962 - 3.954)	0.691
Time Stress=3	-0.843	2.550	(-5.840 - 4.154)	0.741
Time Stress=4	-1.578	2.560	(-6.595 - 3.440)	0.538
Time Stress=5	-2.749	2.546	(-7.740 - 2.241)	0.280
Time Stress=6	-3.384	2.645	(-8.567 - 1.799)	0.201
Time Stress=7	-2.869	2.592	(-7.948 - 2.211)	0.268
Time Stress=8	0a			
Heat Source=.00	0.699	0.690	(-0.654 - 2.051)	0.311
Heat Source=1.00	0a			

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Chapter 8: Blame the Exurbs, Not the Suburbs: Exploring the Distribution of Greenhouse Gas Emissions Within a City Region

8.1 Publication Information

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8.2 Abstract

This research investigates whether where we live matters in terms of contributions to direct greenhouse gas (GHG) emissions. Using results from the Halifax Space Time Activity Research (STAR) project, we estimate GHG emissions for 1,920 randomly selected respondents in Halifax Regional Municipality, Nova Scotia, Canada. The unique data set allows us to report direct GHG emissions with an unprecedented level of specificity at the sub-regional scale using household energy-use survey data and GPS-verified travel data. We report results and investigate statistical differences between communities and urban-rural zones (inner city, suburban, and inner/outer rural commuter). Results reveal considerable spatial variability in direct GHG emissions across the study area. Our findings indicate that individuals living in the suburbs generate similar amounts of GHG emissions (20.5 kg CO_2e person^{-1} day^{-1}) to those living in the inner city (20.2 kg CO_2e person^{-1} day^{-1}), challenging a widely held assumption that living in inner city is better for sustainability. However, individuals in more rural areas have significantly higher transport-related GHG emissions than those living in the inner city and suburbs. Our results underscore the importance of understanding the spatial distribution of GHG emissions at the sub-regional scale.

8.3 Introduction

Influencing where people live is a strategy available to municipalities for reducing household greenhouse gas (GHG) emissions. The objective has been to increase the population density of the urban core and curb urban sprawl (Parshall et al., 2010; Weisz and Steinberger, 2010). The rationale is based on a poorly tested, yet widely held, assumption that people living in the urban core have lower environmental impact lifestyles than do their suburban and rural counterparts (Hoornweg et al., 2011). The premises held that square-footage of houses are smaller in inner city zones, and that individuals are less vehicle-dependent than those living in the suburbs and rural fringes of urban regions. However, the relationship between the built environment and travel mode choice remains poorly understood (Ewing and Cervero, 2010). Furthermore, as most sustainability and land-use planning studies are urban-focused, the policy debate has focused on an urban (inner-city) versus suburban dichotomy. The suburbs are perceived as the locus of 'unsustainable' development and living, compared to the urban core (Hoornweg et al., 2011). In the North American context, however, the rapidly-developing "exurbs" of the urban fringe (Davis et al., 2007; Lamb, 1983; Nelson, 1991; Taylor, 2011) extend far into the quasi-rural commuter belt, and it is therefore also important to compare GHG emissions of exurbanites to those of urban and suburban residents.

The notion that where we live influences our environmental impact has been an important meta-theme within the "smart growth" planning movement. Understanding of the distribution of GHG emissions across urban regions is, however, poor at best. Few studies report GHG emissions or carbon footprints at the sub-regional or sub-municipality scales. Lack of data at finer spatial scales presents a significant limitation for sustainability analyses (Wilson and Grant, 2009). Lenzen and colleagues (2004) and Haq and Owen (2009) estimate both energy consumption and carbon footprints at the sub-city level by linking input-output analysis with household spending survey data. Druckman and Jackson (2008) for the United Kingdom, and Weber and Mathews (2007) for the United States, estimate GHG emissions of households by income category and other household characteristics using models that link household expenditure data to energy consumption. Neither study specifies a geographic region, but their models could support a sub-regional analysis. In absence of input-output data, Vandeweghe and Kennedy (2007) estimate residential GHG emissions by census tract for the Toronto Census Metropolitan Area (Canada) based on household fuel payments and transportation survey data. Wilson and colleagues (2012) present the most refined analysis to date, estimating the ecological footprint, including the carbon footprint, by Census Dissemination Area (400 to 700 people) for Oakville, Ontario (Canada). Ecological footprint values were derived using household energy use data, square footage data, household spending data, and commuting survey data.

The literature provides convincing evidence that environmental impacts vary substantially at the sub-city level. What is of interest, however, is to discern whether such variation is repetitive and predictable, and here the results are mixed. Wilson et al. (2012) found substantial variability in ecological footprints within what one might expect to be a relatively homogeneous group (the study community, Oakville, is a bedroom community of Toronto and considered a suburb in its entirety). In Oakville, higher footprint households tended to concentrate along the waterfront and green spaces indicative of higher-income households. Lenzen and colleagues (2004) found that energy requirements of subdivisions are higher nearer the city centre. Haq and Owen (2009) noted neighbourhoods reporting the largest carbon footprints were in either the inner city or rural areas. Vandewehge and Kennedy (2007) found census tracts located in the suburbs had the highest per capita GHG emissions. All of these efforts, however, quantify the environmental impact of households based, in part, on spending data or other indicators. To the best of our knowledge, no studies have estimated GHG emissions (or similar measurement units, i.e. carbon footprint) based directly on energy use information collected at the household level.

The present analysis complements existing sub-city environmental impact analyses by reporting direct GHG emissions using household-specific energy use data and GPS-verified personal travel data for 1,920 respondents living in Halifax Regional Municipality (HRM), Nova Scotia, Canada. The dataset allows us to report direct GHG emissions with an unprecedented level of specificity at a sub-regional level. Results are reported by community, confirming variability in direct GHG emissions across HRM. In addition, results are reported by urban-rural zones, highlighting differences in direct GHG emissions between urban, suburban, and rural commuter zones.

8.4 Data and Methods

8.4.1 Data

Our study employs data from the Halifax Space-Time Activity Research (STAR) project. When completed, the STAR project represented the world's largest deployment of global positioning system (GPS) technology for a household activity survey (Bricka, 2008). The Halifax STAR project sampled 1,971 randomly selected households in HRM, or about one household in 78, between April 2007 and May 2008 (TURP, 2008). HRM is the provincial capital of Nova Scotia and the largest municipality in Atlantic Canada, spanning almost 5,500 square kilometers, with a population of over 390,000 people in 2007 (Statistics Canada, 2012). In many ways, HRM is highly representative of Canadian mid-sized metropolitan areas, having a diverse and moderately prosperous economy with population growth of about 0.5% per year. The study area has a clearly-defined downtown area that has experienced little inner city decay, and income variations across the region are modest. Low-density exurban sprawl is particularly evident in a commuter belt within 50 km of downtown Halifax, while very few people reside in the fully rural area beyond 50 km (Millward and Spinney, 2011a).

The Halifax STAR project collected time diary and questionnaire data over 373 days of data collection, with a cooperation rate of 25% and overall response rate of 21% (Millward and Spinney, 2011b). The primary sampling frame was all residential households with listed telephone numbers. A pre-notification letter was used to make initial contact with each household, followed by a recruitment telephone call. Using computer-assisted telephone interview (CATI) software, recruited households completed a brief intake questionnaire to collect age and other information for all household members. The age information was used to randomly select the secondary sampling unit; a single household member over the age of 15 who was randomly selected as the "primary respondent", and was assigned a consecutive pair of diary days (e.g. Thursday-Friday, Friday-Saturday, etc.). The "primary respondent" carried a GPS-device (Hewlett Packard iPAQ hw6955) throughout the 48-hour reporting period. The "day after" the two-day reporting period, both questionnaire and two-day time diary data were collected using specially-designed CATI software. See TURP (2008, 2010a, and 2010b) for a detailed description of the sampling strategy, survey methods, and data user guide. The STAR survey was modeled after Statistics Canada's (2006) General Social Survey (GSS) Cycle 19, which asked questions about personal characteristics, household characteristics, neighbourhood features, socio-economic data, and subjective dimensions of well-being. The STAR survey supplemented those questionnaire items with questions regarding electricity consumption, home heating, and energy efficiency behaviours, among others.

Using ArcGIS 9.3®, the geographic coordinates (latitude and longitude) of each household's residential location within the study area were "spatially joined" with communities (based on municipal administrative boundaries) and urban-rural zones (delimited operationally on the basis of both urban form and commuting linkages to the urbanized area). Given the measurement of transportation-related GHG emissions is based primarily on average travel distance, the zones defined by Millward and Spinney (2011a) are used in this study. These zones are based on street network travel distance outwards from "downtown", and, therefore, offer a more functional approach to examine the extent to which GHG emissions vary within the study region. The "inner city" of HRM is defined as the area within a 5 km travel distance of Halifax and Dartmouth's "downtown" areas. The "suburbs" are adjacent developed areas with city water and sewer services, mostly lying between 5 and 10 km from downtown. The "inner commuter belt (ICB)" comprises unserviced communities lying between 10 to 25 km travel distance to downtown while the "outer commuter belt (OCB)" lies between 25 to 50 km to downtown. Millward and Spinney (2011a) define a fifth zone, "remote rural", which encompasses areas beyond 50 km travel distance to downtown, or lacking road access, but due to a lack of sample, we do not report GHG emissions for this zone (Figure 8.1).

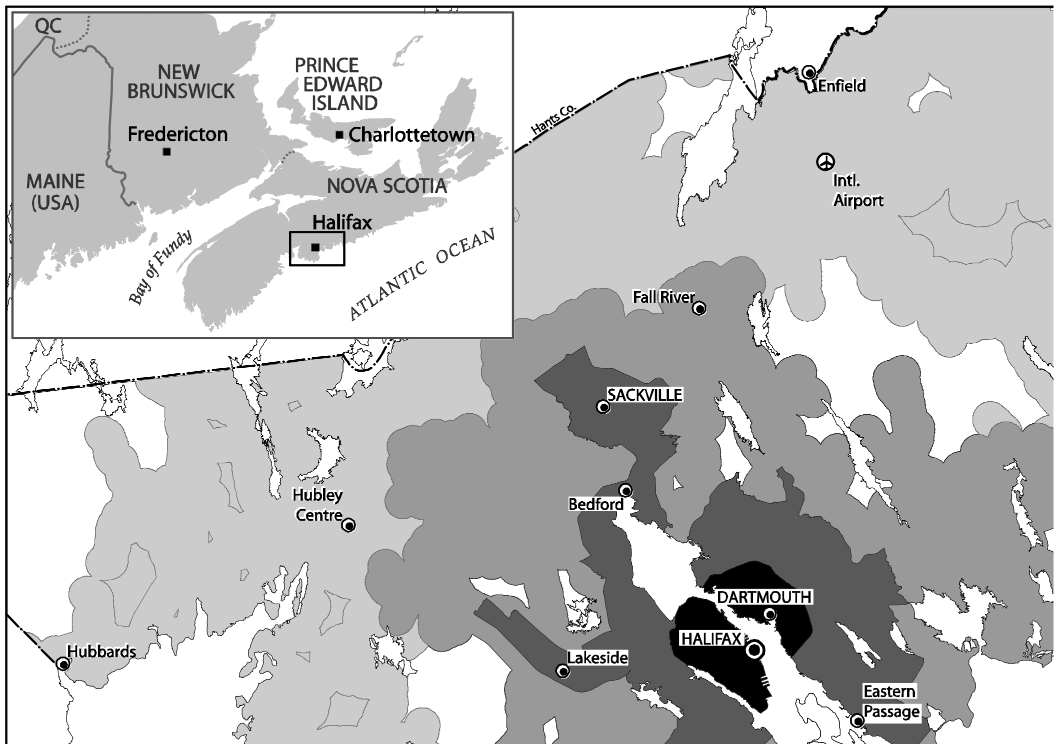

Figure 8.1: Urban-Rural Zones in the Halifax Regional Municipality

The STAR sample approximates a proportional distribution for gender, days of the week, and the months of the year. Spinney and Millward (2011b) provides a detailed description of the sample. A few of the more pertinent survey items to the study are summarized and presented in Table 10 using measures of central tendency and dispersion that are stratified by urban-rural zone. The survey items that are summarized in Table 8.1 are respondents' ages; persons per household; housing size (heated square footage); heated square footage per person; number of household vehicles, "car-dependency" that is measured using daily mean travel distance (km) in a private vehicle or taxi; and the prevalence of light duty vans and trucks (bigger vehicles) as a percentage of vehicle fleet of respondents.

Table 8.1: Summary statistics for select survey items by urban-rural zone

	Inner City	Suburbs	Inner Commuter Belt	Outer Commuter Belt
	mean (S.D.)	*mean (S.D.)*	*mean (S.D.)*	*mean (S.D.)*
Age of respondents	55.3 (14.2)	52.6 (13.9)	48.7 (11.0)	48.9 (12.1)
Persons per household	2.5 (1.1)	2.7 (1.1)	3.1 (1.2)	2.9 (1.2)
Heated square feet	1,983 (1,128)	1,979 (820)	2,237 (915)	2,177 (786)
Heated square feet per person	920 (626)	848 (505)	836 (514)	857 (443)
Number of household vehicles	1.4 (0.7)	1.7 (0.7)	2.1 (1.0)	2.0 (0.7)
Car-dependency (km/day)	25.8 (38.5)	32.6 (42.7)	49.3 (50.6)	53.2 (52.1)
Percent light duty vans and trucks	21%	31%	34%	38%

If we consider adjacent successive urban-rural zones as an ordinal classification of increasing distance from downtown, both household size and the number of persons per household increases as we move away from the inner city toward the inner and outer commuter belts. The heated square footage of households per person, however, decreases with increasing distance from the inner city. The mean heated square feet of housing increases with increasing distance from the inner city, but when the size of the home is adjusted for the size of the family, there appears to be an inverse relationship between heated square feet per person and increasing distance from the inner city. Both dwelling size and family size increase with distance from the inner city. There is a steady decline in the age profile with increasing distance from the inner city; average age decreases from a high of 55 years old in the inner city to 48 years old in the inner and outer commuter belts. More pronounced is the pattern of increasing car dependency (km travelled per day) with increasing distance from the inner city, and decreasing opportunities for alternate travel modes. About one-quarter of inner city respondents are car dependent. That value rises to about one-third in the suburbs, to almost one-half in the inner commuter belt, and exceeds one-half in the outer commuter belt. Not surprising then, the average number of vehicles per household also increases with increasing distance from the inner city, where the average is 1.4 vehicles per household and that number reaches its maximum in the inner commuter belt at 2.1 vehicles per household. In the same vein, there is a modest, but steady, increase in the prevalence of light-duty vehicles with increasing distance from the inner city to the rural countryside. Overall, however, we found little distinction in prevalence of light-duty vehicles between the inner and outer commuter belt zones.

8.4.2 Methods

We estimated shelter-related and transportation-related direct GHG emissions per respondent per day in kilograms carbon dioxide equivalents (kg CO_2e person^{-1} day^{-1}). Shelter-related GHG emission estimates are based on respondent-recall of electricity bills, home heating bills, and amount of wood burned. Respondents were asked to consult bills, although we are not able to verify which houses did and did not. We reported shelter-related emissions per respondent by dividing the total household emissions by the number of household members. We confirmed results with an expert from the provincial energy efficiency agency, which resulted in 15 cases being screened out of our analysis, because of unreasonably high shelter-related energy use. For transportation-related direct GHG emissions, we estimated personal transportation based on distances travelled, using household activity survey data. Trip distances were computed using the shortest path between GPS-verified origins and destinations. Thirty-six respondents had incomplete travel data and were screened out of our analysis. Also, we do not include air travel in our estimation of GHG emissions, because many of the out-of-country coordinates were not captured in the STAR survey. In Nova Scotia, air travel (domestic aviation) represents approximately 7% of total transport-related GHG emissions (Environment Canada, 2010). For a more detailed description of specific methods and assumptions used to estimate direct GHG emissions, see Wilson et al. (under review).

Each respondent's residential location was used to summarize the spatial distribution of direct GHG emissions across the 62 communities within the study area. Communities, in this context, do not reflect a specific population size. Rather, they represent longstanding geographic areas with meaningful similarities in both perceptions and interpretations of both place and context. Median direct GHG emissions were calculated for communities with three or more sampled households and mapped using

ArcGIS 10® to illustrate the spatial distribution of emissions. In addition, shelter-related, transportation-related, and total direct GHG emissions were assigned to urban-rural zones. Statistical Package for the Social Sciences (SPSS, v.15.0) software was used to measure the median and inter-quartile range (IQR) for GHG emissions by urban-rural zones. Medians were chosen in preference to means, because each of the three emission variables exhibit skewed distributions, which was confirmed by the Kolmogorov-Smirnov Z test. We tested two-tailed significance of differences among adjacent successive urban-rural zones using a Mann-Whitney U test. The U-test is a non-parametric difference-of-ranks test that tests whether two independent samples of observations are drawn from the same distribution, and it is employed in preference to the two-sample t-test because the distributions of the variables under scrutiny are skewed.

8.5 Results

In total, direct GHG emissions were estimated for 1,920 respondents. We omitted 51 of the 1,971 households due to incomplete or problematic data specific to this research. Statistical analysis of the STAR household data indicates the median total direct GHG emissions for HRM is 21 kg CO_2e person^{-1} day^{-1}, with a range of 101 kg, and inter-quartile range of 14 kg. On average, provision of electricity and home heating contribute 78% of direct GHG emissions with a mean of 14.6 kg CO_2e person^{-1} day^{-1}, which is significantly higher ($p = <0.001$) than emissions associated with personal transportation, which account for the balance of direct GHG emissions and average 4.1 kg CO_2e person^{-1} day^{-1}.

8.5.1 Greenhouse Gas Emissions by Community

A total of 44 communities have three or more households with estimates of direct GHG emissions (Figure 8.2). We found variability in direct GHG emissions across the HRM, with median values ranging from 13.9 to 35.8, and a median value across all communities of 21 kilograms of CO_2e person^{-1} day^{-1}. In general, median GHG emissions per person tend to increase as we move toward the perimeter of the HRM, although that pattern does not appear consistent. For example, the communities of Stillwater and Waverley both have very high emissions (>30 kg CO_2e person^{-1} day^{-1}) while the community of Westphal has very low median emissions (<18kg CO_2e person^{-1} day^{-1}) (Figure 8.2). The overall pattern is that communities in the commuter belt tend to have higher values than the inner-city or suburbs, and it is noteworthy that all communities with median direct emission values over 24 kg CO_2e person^{-1} day^{-1} lie in the exurban commuter area. Relatively high per capita daily emission values (>24) are noted for wealthier commuter-belt communities, such as Waverley, Stillwater, and Oakfield, while relatively lower emission values (<20) in the commuter belt tend to occur in medium or low-medium income areas, such as Enfield, Lawrencetown, and Williamswood. These community-level patterns echo results produced previously by Wilson and colleagues (2012) for the town of Oakville, Ontario. However, some community-level median emission values are based on few observations, which prevents significance testing. To allow such testing, we turn to an examination of zonal differences along the urban-rural continuum.

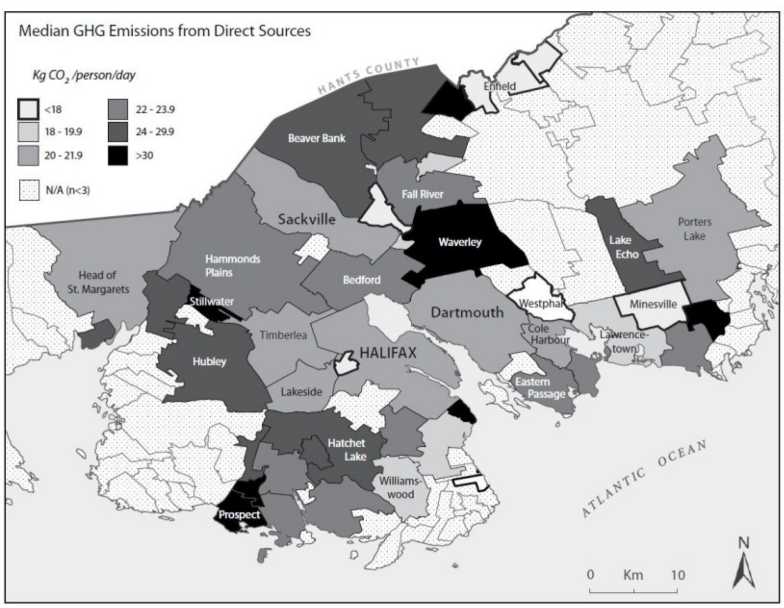

Figure 8.2: Spatial distribution of median GHG estimates by communities

8.5.2 Greenhouse Gas Emissions by Rural-Urban Zone

To further elucidate the spatial distribution of shelter-related, transportation-related, and total direct GHG emissions across the study area, we present median and IQR emissions per person per day by urban-rural zone in Table 8.2. In addition, in Table 8.2, we report Mann-Whitney U test results of differences between adjacent successive urban-rural zones.

Table 8.2: Median direct greenhouse gas emissions, kg CO_2e person^{-1} day^{-1}

Zone	N	Total Direct GHG	Transportation-Related GHG	Shelter-Related GHG
		median (IQR)	*median (IQR)*	*median (IQR)*
Overall	1,920	20.8 (14.1)	4.1 (7.0)	14.6 (10.0)
Inner city	382	20.2 (12.6)	2.2 (3.9)	16.2 (10.5)
Significance		*0.382*	*<0.001*	*0.026*
Suburbs	1,039	20.5 (13.5)	4.0 (6.1)	14.6 (10.0)
Significance		*0.003*	*<0.001*	*0.023*
Inner commuter belt	334	22.9 (15.4)	6.2 (9.1)	13.6 (9.5)
Significance		*0.944*	*0.513*	*0.818*
Outer commuter belt	165	22.0 (19.6)	7.1 (11.4)	13.4 (8.2)

Results in Table 8.2 indicate that total emissions increase away from the urban core; median daily direct GHG emissions for respondents living in the inner and outer commuter belt zones are approximately 11% higher than for respondents living in the inner city zone. Interestingly, there is little difference (<2%) in median direct GHG emissions between the inner city zone and the suburban zone, and any difference appears purely due to chance. For total direct GHG emissions, the only significant difference ($p = 0.003$) between adjacent successive zones is for the suburbs and ICB, the suburbs being lower. Basically, the results in Table 8.2 illustrate that as we move away from the inner city zone, there is a decrease in shelter-related emissions and an increase in transportation-related emissions.

Results in Table 8.2 indicate that the overall median transportation-related GHG emissions are 4.1 kg CO_2e person^{-1} day^{-1}. Our sample indicates that private vehicle use contributes about ninety-nine percent (99%) of direct GHG emission associated with personal transportation. Transportation-related GHG emissions increase from the inner city (2.2 kg CO_2e person^{-1} day^{-1}) to the suburbs (4.0 kg CO_2e person^{-1} day^{-1}), and another increase to the inner commuter belt (6.2 kg CO_2e person^{-1} day^{-1}). Despite another increase (0.9 kg CO_2e person^{-1} day^{-1}) from the inner to the outer commuter belt, values for these two zones are not significantly different (Table 8.2). Any increase in transportation-related emissions moving outwards from the inner city appears to be offset by significantly reduced shelter-related emissions. Results in Table 8.2 indicate that significantly lower shelter-related emissions were reported in the suburbs (14.6 kg CO_2e person^{-1} day^{-1}) compared to the inner city (16.2 kg CO_2e person^{-1} day^{-1}), and significantly lower again in the inner commuter belt (13.6 kg CO_2e person^{-1} day^{-1}). Just like travel-related emissions, shelter-related emissions are not statistically different between respondents living in inner and outer commuter belts.

8.6 Discussion

The present analysis complements previous sub-city environmental impact analyses by reporting direct GHG emissions using household-specific energy use data and GPS-verified travel data for 1,920 respondents living in HRM. Our unique dataset allows us to report direct GHG emissions with an unprecedented level of specificity at a sub-regional level. The data demonstrate a wide range in direct GHG emissions by respondent, reflective of the sampling strategy and consistent with trip diary data. Data collection occurred over a one-year period. Depending on the time of year, households could have very different home energy use demands. Further, with respect to personal transportation, large ranges in travel distances are consistent with trip diary surveys (Kang and Scott, 2010; Millward and Spinney, 2011). While individual travel events may appear to be atypical for the person undertaking them (e.g., the case of visiting a distant relative), taken in aggregate they accurately represent variability in GHG emissions based on people's activities, which take place at disparate locations. Not all trip purposes are repetitive (as is the case for travel to and from work). Rather, there exists considerable variability in activities and travel across days of the week (Kang and Scott, 2010).

Our analysis is not without its limitations. For example, it reports GHG emissions per respondent, because the travel data are based on a personal, wearable, GPS collection device. Conceptualizations of urban regions in terms of households may be a more functional unit. It would also help mitigate issues of attribution of transportation- related GHG emissions if other household members are present in the vehicle. For example, what is the appropriate approach to split the associated GHG emissions if the trip purpose is taking a child to sports practice or if a child is present on a shopping trip? A potential research direction could be to incorporate out-of-home activity diary data from other household members, which could be used to identify if and why other household members are present in the vehicle. Using trip purpose by travel segment could provide a useful mechanism for mitigating attribution issues.

Although studies suggest that indirect emissions contribute 40-65% of total GHG emissions associated with typical Western lifestyles (Benders et al., 2006; Kok et al., 2006; Moll et al., 2005; Nansai et al., 2008; Weber and Mathews, 2007), our analysis does not include indirect GHG emissions. Indirect GHG emissions refer to the emissions that result from earlier stages in the life cycle of the goods and services that we consume and correlate strongly with income (Benders et al., 2006; Druckman and Jackson, 2008; Lenzen et al., 2004; Weber and Mathews, 2007). We considered including indirect emissions by allocating regional input-output data to households by income category, using household expenditure survey data. We decided not to, however, to avoid estimating sub-city results by extrapolating from national-level data. Further, HRM does not appear to have a high concentration of higher-income households in one specific zone. In the sample, all zones reported a median personal income range category of $40,000-$59,999. Future research will look at including indirect GHG emissions.

Our results illustrate the patterning of GHG emissions within a city region at an exceptional spatial scale. These fine-resolution results offer planners and policy makers a unique perspective regarding variation in GHG emissions at a sub-city level. The community-level mapping, for example, revealed considerable variation in emissions at the local level, consistent with other sub-city analyses of environmental impacts (Haq and Owen, 2009; Vandeweghe and Kennedy, 2007; Wilson et al., 2012) and challenges the notion of a simple gradient in increasing emissions moving outward from the urban core. Although peripheral communities tend to show higher median total emissions, these medians also seem to relate to income levels, with wealthier communities showing higher values (Wilson et al., under review). Small sample sizes in many rural communities, however, prevented the use of significance testing.

Shelter-related emissions per person decrease as we move away from the inner city toward the outer commuter belt. While houses become bigger as we move outward from the inner city, to the commuter-belt zones, the STAR sample indicates there are more people per household (Table 8.1). When we factor in number of persons per household, therefore, the square footage per person is very similar in all zones except the inner city, where it is noticeably larger. This may reflect the "empty-nest" characteristic of many inner-city households, compounded by under-sampling of renters, such as students and those with lower incomes.

We feel reporting results by urban-rural zone to be a more useful approach for analyzing direct GHG emissions at the sub-city level than reporting by community. The zones organize the municipality based on urban form and distance to the regional centre, which are both widely perceived factors influencing energy consumption patterns. Categorizing HRM by zone can support policy and planning decisions that cater to different challenges and opportunities faced by each zone to reduce GHG emissions. For example, public transportation either does not service or poorly services the inner and outer commuter zones, limiting travel options. Also, the housing stock in the urban core is typically older than in the suburban and commuter zones, presenting different challenges to reducing shelter-related GHG emissions. The lack of significant differences between inner and out commuter belt zones in direct, transportation-related, and shelter-related emissions, plus select survey items, suggesting remarkable similarities among these two regional groups of respondents.

Our results for HRM show little difference in per capita direct GHG emissions between the inner city and the serviced suburbs, challenging a widely-held assumption that inner city living is inherently more sustainable than suburban living. The notion that inner city living is 'greener' is partially based on the assumptions that people in the inner city drive less and live in smaller households (square footage). We did find that those living in the suburbs had higher median transportation related GHG emissions. Respondents in the suburbs, on average, traveled more per day as the driver or passenger of a private vehicle (Table 8.1) and drove bigger vehicles (SUV, van, pick-up) as a percentage of the total vehicle fleet. Median transportation related GHG emissions of respondents living in the inner city are 2.2 kg CO_2e person^{-1} day^{-1} compared to 4.0 kg CO_2e person^{-1} day^{-1} for respondents living in the surburbs. Our dataset, however, does not support the assumption of bigger homes and higher shelter related GHG emissions in the suburbs in comparison to the inner city. Average heated square footage of households was remarkably similar. When we consider square footage per household member, it was less in the suburbs compared to the inner city. Median shelter related GHG emissions of respondents living in the inner city are 16.1 kg CO_2e person^{-1} day^{-1} compared to 14.6 kg CO_2e person^{-1} day^{-1} for respondents living in the surburbs. Lower GHG emissions associated with electricity use and home heating in the suburbs offset the higher transportation related emissions.

In HRM, heating and electricity related emissions dominate direct GHG emissions accounting for 78% of total GHG emissions. The substantial contribution of heating and electricity to overall emissions can be explained in part by Halifax's reliance on oil and electricity as a primary heating source coupled by the fact that the greenhouse gas intensity of electricity generation in the province is high, over triple the Canadian average. Additionally, in comparison to other regions in Canada, the housing stock is older and there are more single-family dwellings as a percentage of the total housing stock (77% vs. 58%) (NRCAN, 2010). In comparison, jurisdictions relying on hydro generated electricity and heating primarily with natural gas would have lower associated household GHG emissions. If GHG emissions associated with shelter were lower, the difference in transportation related emissions would be more pronounced in terms of influencing overall emissions. As the province of Nova Scotia moves toward lower-carbon electricity, shelter-related emissions should decline placing greater emphasis on transportation related emissions.

The dynamics driving transportation related GHG emissions are not straight-forward. For example, the notion of a single travel flow from suburbs to inner city for work and shopping in the HRM context is a misnomer. In the case of HRM, several major industrial-business hubs are located in the suburbs. Ninety-six percent of new office space between 2008 and 2012 was built in the suburbs indicating a decreasing need to enter the downtown core for work (HRM Alliance, 2012). Further, those living in the urban core may travel to suburbs to access services and shopping creating a reverse travel requirement. Assuming that inner city living is inherently more sustainable than suburban living masks that both zones present sustainability challenges that require attention. In the case of HRM, the inner city is dealing with a number of critical issues: high carbon electricity and heating sources, inefficient homes, low population density, large housing size per person, low public transit use, and an infrastructure that favours travel to suburban zones for shopping, services, and potentially even employment.

Efforts to reduce greenhouse gas emissions must also include the exurbanites. Those living in commuter belt zones and on the rural fringe have substantially higher greenhouse gas emissions, and particularly higher transport-related direct GHG, which often get overlooked in a policy debate that has focused on an urban (inner-city) versus suburban dichotomy. The exurbs of the commuter belt are characterized by low-density, large-lot development, lacking city water and sewer services. Though commuter-dependent, they have the appearance of rurality, and provide sought-after environments for family living (Millward, 2006). However, lack of public transport, and limited options for active transport, lead to greater car-dependency and longer trip distances per person, such that transport-related direct emissions are three times those in the inner city, and 1.5

times those in the suburbs. Interestingly, median transport-related GHG estimates for the inner and outer commuter zones are virtually the same, which is likely due to transport-minimizing accommodations made by outer commuter belt residents, reported by Spinney and Millward (2011a). In particular, respondents in the outer commuter belt make significantly fewer car trips than those in the inner commuter belt, though average trip duration is slightly longer.

8.7 Conclusion

Policy and planning efforts to reduce the carbon footprints of our cities requires an understanding of the variability in household environmental impacts within a community (Wilson et al., 2012). The STAR data provide a more nuanced perspective on direct GHG emissions than previously available, particularly because they reflect accurate and precise location information. Analysis of these data by urban-rural zones offers a functional categorization around which to frame GHG reduction strategies. Findings challenge the assumption that development patterns outside of the urban core, notably in the serviced suburbs, are inherently less sustainable and/or detrimental to sustainability. The assumptions that those living in the suburbs have bigger homes, bigger cars, and drive more are only partially correct. We did find bigger cars and higher transportation-related GHG emissions, but lower shelter-related emissions appear to offset them. Shelter-related emissions account for 70% of total GHG emissions, on average. We do, however, see a noticeable increase in direct GHG emissions for the inner and outer commuter belts, raising concerns regarding exurban sprawl. The increase can be almost entirely attributed to transportation-related emissions.

While where we live may be important, other factors influence direct GHG emissions. The results of the current study confirm considerable variability in household GHG emissions across HRM, but also significant predictability between urban-rural zones. Other factors such as income, household size, and respondent age may be better predictors of GHG emissions (Wilson et al., under review), and are inter-related with zonal variations. Research supporting a more nuanced understanding of household GHG emissions within cities, targeting the different challenges and opportunities of different zones, will benefit efforts to achieve GHG reductions.

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Chapter 9: Conclusion and Future Research

9.1 Introduction

My dissertation argues in support of an economic view that ecological limits ultimately constrain throughput economic activity. Specifically, my work explored approaches to account for and better understand drivers and distribution of environmental impact at the urban and sub-regional scale. Accounting for energy and material throughput and associated impacts at the local level are critical to improve decision-making toward sustainability. My work bridges a number of key disciplines oriented around ecological sustainability, including ecological economics, industrial ecology, urban planning, and population health.

The preceding chapters each presented important elements of my research agenda. Part 1, Chapters 2 through 4, focused on rethinking measurement from an ecological economics perspective. Part 2 applied biophysical based approaches to account for environmental impact at a local level. Chapter 5 tested a calculation strategy to estimate municipal ecological footprint values within Canada. Chapter 6 measured environmental impact at a neighbourhood level, extending our ability to account for environmental impact at finer scales of resolution. Part 3 explored drivers of environmental impact using Halifax Regional Municipality as a case study. Chapter 7 examined the relationship between direct GHG emissions and socio-economic and wellbeing variables using a multivariate model. Chapter 8 explored whether where we live matters in terms of GHG emissions. My research is part of a wider effort to encourage decision making that supports sustainable human systems by applying biophysically based sustainability assessments at the sub-regional level. I suggest new approaches to account for environmental impact at fine scales and analyze drivers and the spatial distribution of environmental impact at an urban level.

9.2 Future Research

Extending biophysically-based sustainability assessments to regional and sub-regional applications is not without its challenges. As highlighted in the discussion sections of each chapter in my dissertation, several critical areas of future research and inquiry would improve how we account for and understand environmental impact at the local level. I highlight four thematic areas that continue to challenge my thinking including calculation challenges, the role of income as a driver of environmental impact, understanding the environment-wellbeing connection, and understanding the relationship between environmental attitudes and impact. Further, to improve our understanding of environmental impact at the local level I see a need to expand urban and intra-urban biophysically-based sustainability assessments. The list offers an interesting research agenda forward and a wide array of exciting opportunities to occupy future students and me for years to come. The order does not reflect preference or priority.

9.2.1 Calculation Challenges

Data limitations prove a major challenge for sub-regional assessments (see Chapters 2, 3, 5, 6). Data availability, data collection costs, and compatibility between datasets are persistent issues limiting measurement at finer spatial resolutions. The absence of data and lack of straight-forward calculation approaches restrict local biophysical assessments. Responding to this challenge, my thesis suggested an approach forward to estimate municipal ecological footprints (see Chapter 5). In doing so, I also raised a more profound question; given calculation challenges at the local level, are these exercises relevant as decision support tools for sustainability. In Chapter 6, I demonstrate that finer scale analyses produces results that can support policy and planning decisions. I found reporting environmental impact by neighbourhood demonstrates wide variability in impact across a community. Differences between neighbourhoods were largely influenced by income. The confounding factor, however, is that I used a modified version of income to estimate the ecological impacts associated with household consumption of goods and services. My approach reflects similar strategies adopted at a sub-national level. As described in Chapters 5 and 6, techniques used to estimate sub-regional ecological footprints typically estimate indirect impacts associated with the consumption of goods and services using input-output analysis, expenditure surveys, income data or some combination of the three. Using income as a proxy of consumption is logical in the absence of actual energy and material flow data because the two variables are correlated. National level studies indicate income as a strong driver of greenhouse gas emissions, energy use and ecological footprint (Baiocchi et al., 2010; Benders et al., 2006; Borucke et al., 2013; Drunkman and Jackson, 2008; Weber and Mathews, 2008). Relying on monetary-based variables, however, to estimate biophysical indicators is problematic (Chapter 4). Future research efforts ought to explore options independent of monetary variables to ensure accurate attribution of environmental impacts to households. Concern around limitations associated with using expenditure surveys and income to estimate indirect impacts caused me to focus exclusively on direct GHG emissions in the analyses exploring drivers of GHG emissions (Chapters 7 and 8). Two important areas of future research to address these concerns include improving data collection techniques to build robust local data sets and improving methods to account for indirect impacts associated with household consumption.

Future research: improve data collection techniques

Finding better means to collect robust local data is critical toward advancing sustainability measurement at fine scales. The STAR project collected detailed travel data using handheld personal digital assistant (PDA) devices loaded with global positioning system software. PDA devices could be used to collect other personal data of value in assessing impacts of lifestyle. Using applications to collect consumption and energy use data would reduce reliance on extrapolated data and in the case of the STAR dataset replace lengthy entry and exit surveys, which may have been a barrier to participation. Preloading survey tools on the PDA would improve response accuracy and detail, and reduce data transfer time from paper to database program and associated error. The applications could also be designed to ask questions reflective of previous responses or have questions triggered by geographic positioning data. The comprehensive information would change how we approach local sustainability analyses and reduce dependency on extrapolated data and top-down calculation models. Collecting spatial data alongside throughput data offers a new lens to understand flows of energy and material through the urban landscape.

Future research: improve methods to account for embodied impacts of consumption

Improving methods to account for embodied impacts of goods and services would substantially improve the use of biophysical metrics at sub-national scales. I see an opportunity to compile provincial embodied impact models drawing on recent advances in life cycle assessment databases and input-out models. Nijdam and colleagues (2005) assigned an impact variable to expenditure categories reported in the Dutch Expenditure Survey using such an approach. Their method, however, follows financial flows as opposed to material flows. Moran et al., (2009) have attempted to assign embodied ecological footprint values to international trade flows. Their research uses the United Nations COMTRADE database. As a result, they are limited to higher-level categories of traded goods. A provincial model would need to be refined to link impact to consumption categories used in household surveys. Attributing impact based on material and energy demands as opposed to expenditure or income data would dramatically improve throughput analyses at the sub-national scale.

9.2.2 The Role of Income as a Driver of Environmental Impact

The role of income as a driver of environmental impact is an important issue I continue to struggle with. In my calculation approach to estimate municipal and neighbourhood environmental impacts I used a modified version of income as a proxy of consumption to assign the embodied footprint associated with the consumption of goods and services. For Oakville (Chapter 6), the footprint of goods and services represents 30% of the total ecological footprint. In terms of explaining variation within a community, the category is significant because the portion of the footprint attributed to government (10%) was held constant across the community and there was very little variation in the food footprint (30%). The role of income raises several important questions in regards to my research. 1) Is income an appropriate proxy of environmental impact? 2) If and at what point does income decouple from consumption? Future research into these questions may offer insights to address challenging public policy issues. Do public policy goals directed at increasing household income undermine sustainability? If income is a significant driver of environmental impact, should governments cap income and if so, what is an appropriate income cap?

Future research: improve understanding of income and environmental impact relationship

Theoretical concepts such as IPAT (impact = population * affluence * technology) have long stipulated that affluence is a key factor driving environmental impact (Ehrlich and Holden, 1971). The relationship between affluence and environmental impact has been widely articulated at the global level comparing impacts between high-income countries and low-income countries. On a per capita basis, richer nations contribute more significantly to global environmental impacts than poorer nations (Wilson et al., 2007). The breakdown of global environmental impact will vary depending on what impacts are evaluated, how environmental impacts are allocated between regions and groups and other methodological assumptions (table 9.2). The message, however, is unmistakable; on a per capita basis, high income countries are disproportionately responsible for a larger share of global environmental impact than middle and low income countries.

Table 9.2: Environmental impact of high income countries

	Richest 20% (high income countries)	**Remaining 80%** (middle and low income countries)
GHG emissions (UNEP, 2005)	65%	35%
GHG emissions (Dodman, 2009)	46%	54%
Ecological footprint (GFN, 2010)	45%	55%

The United States and Canada together account for 19.4 per cent of global greenhouse gas emissions. Canadians on average produce almost 24 tonnes of CO_2-e per capita per year. In comparison, Bangladeshis, who live in one of the poorest countries in the world, produce on average less than 0.5 tonnes of CO_2-e per capita per year (Dodman, 2009). Correlation analysis between national ecological footprint values and Gross Domestic Product (GDP) Purchase Price Parity reveals that substantial variation in ecological footprint between countries corresponds with variations in the GDP (Wilson et al., 2007).

The general premise that greater wealth, as measured by income or GDP per capita, results in greater environmental impact at a global level is more or less accepted. Although less documented, rising incomes are consistently identified as a main driver of household energy use and/ or GHG emissions at a sub-national level as well (Baiocchi et al., 2010; Benders et al., 2006; Drunkman and Jackson, 2008; Weber and Mathews, 2008). As discussed in Chapter 7, the strength of the relationship depends on whether direct and indirect energy/ GHG emissions are included and what calculation approach was used. Results from the multivariate analysis (Chapter 7) identified income as one of five statistically significant determinants of direct greenhouse gas emissions. The association between income and greenhouse gas emissions, however, was half that of household size and on par with location along the urban-rural continuum. In addition, the relationship between direct greenhouse gas emissions and income range is not linear. Compared to respondents earning less than $20,000 a year, respondents earning $100,000 or more generated 7.0 kg CO_2e more per day. Mean direct GHG emissions, however, were virtually the same among respondents reporting incomes in the ranges, $20,000 -

$39,999; $40,000 - $59,999, and $60,000 - $79,999. Mean GHG emissions jumped noticeably between the $60,000 – 79,999 range and the $80,000 - $99,999 range.

The relationship between income and environmental impact is not straightforward at the household level. Redefining personal aspirations independent of affluence and high consumption, however, is essential for long-term sustainability. In efforts to decipher the relationship between income, consumption and ecological footprint, the critical messages should not be lost. Advances in economic prosperity that result in increased consumption of goods and services cannot occur without the increased appropriation of natural capital.

Future research: explore if and at what level income decouples from consumption and environmental impact

For me as a researcher and practitioner, the relationship between income and environmental impact of lifestyles raises interesting questions for local studies. In the absence of biophysically based datasets and detailed local datasets generally, is income an appropriate proxy of impact and what role, if any, should income be used in local modeling exercises? The time and financial cost of a detailed analysis can be substantial, especially for jurisdictions that may operate on small budgets. The right approach will depend and must consider purpose of exercise, scope of study, intended use, time, budget, and access to expertise.

In Chapter 6, I suggested that urban form and structural factors likely define an ecological footprint floor for a community, whereas income most likely sets a footprint ceiling. The conclusion reflects an assertion that income determines household capacity to consume goods and services. As long as households spend income on goods and services, income will be accompanied by environmental impact. At some point, however, the amount consumed should begin decoupling from income based on the assumption that people can only drive so much and have so many electronic appliances (Lenzen et al., 2008). As consumption levels off so should environmental impact. As noted in Chapter 7, signs of decoupling may only occur at very high income levels. Based on the STAR data I found no evidence of decoupling by income category. The highest income strata analysed in the STAR dataset were those individuals who reported earning $100,000 and above. Peters (2010) found that household carbon emissions begin to level off at higher incomes as consumption shifts to higher value added or more service based goods. Lenzen and colleagues (2008) note a leveling off for direct energy consumption at higher income levels but less so for indirect energy consumption. Baiocchi and colleagues (2010), however, found among U.K. households, that increases in income among higher income households corresponds to larger increases in CO_2 emissions than for increases in income among lower income households. If and at what level income and consumption decouple is not clear.

9.2.3 Improve Understanding of the Environment, Wellbeing Relationship

Reconciling visions of wellbeing and reduced environmental impact is paramount for long-term human sustainability. In Chapter 7, I found that self-reported health and wellbeing indicators did not correlate with direct transport and shelter-related greenhouse gas emissions. Respondents with lifestyles that generate higher GHG emissions did not report being healthier, happier or more connected to their communities. This suggests we can change lifestyles in ways that reduce GHG emissions without undermining wellbeing. Understanding the connections between health, wellbeing, consumption and environmental impact is a critical area of future research. Deciphering, for example, the extent to which increased income and consumption are required conditions of wellbeing will influence how understand and operationalize sustainability. Understanding the environment wellbeing relationship supports a growing effort to define societal progress based on quality of life as opposed to economic growth.

9.2.4 Improve Understanding of Relationship Between Environmental Attitudes and Impact

Also in Chapter 7, I found that attitudes supportive of energy efficient behaviours (or lack of them) did not correlate with direct GHG emissions. This finding is consistent with a small but emerging consensus in the literature that environmental attitudes and behaviours do not translate into lower environmental impact (Baiocchi et al., 2010;

Csutora, 2012; Gatersleben et al., 2002; Susilo et al., 2012). Efforts to reduce environmental impact through the adoption of energy efficient technologies and behaviours may be offset by increased consumption driven by higher incomes. A disconnect between environmental attitudes and environmental impact has profound implications on policies and programs designed to promote lower footprint lifestyles. Further research is required to confirm the relationship and to understand the dynamics underlying the relationship.

9.2.5 Expand Biophysical Based Assessments at Urban Level for Canadian Cities

Thinking of a city in terms of energy and material flows and subsequent environmental impacts presents a unique perspective for understanding sustainability opportunities. My results from the STAR data analysis (Chapters 7 and 8) indicated that in Halifax Regional Municipality suburbanite and inner city resident direct GHG emissions were effectively equal challenging an assumption that inner city living is necessarily more sustainable than living in other zones. It could be, and clearly living in the urban core offers opportunities to make decisions around how you live and how you move that can reduce environmental impacts. The results, however, highlight the importance of supporting sustainable living options regardless of if that is in the suburbs or in the urban core. Improving an understanding of energy and material flows for urban populations is essential to inform sustainability focused decision-making. Building a repertoire of studies using a comparable methodology would provide a deeper understanding of drivers and spatial distribution of environmental impact at the urban level across Canada.

9.2.6 Support Efforts to Advance Standardized Calculation Approaches or Best Practices for Ecological Economic Measures

The lack of a standard calculation methodology is often cited as a limitation preventing wider adoption of sustainability metrics at the local level (Lawn, 2003; Posner and Costanza, 2011; Wilson and Grant, 2009). The ecological footprint calculation suggested in Chapter 5 provides a consistent and easily adoptable approach for Canadian municipalities. Several other Canadian efforts are working to promote standardized frameworks that can be adopted at the local level. The Canadian Index of Wellbeing (CIW), for example, has worked with the City of Guelph to adapt the national index of wellbeing framework to reflect data that is either available or can be collected at the local level (Smale personal communication, 2012). The intent is that a standardized approach will allow for comparisons between jurisdictions and relate back to the Canadian CIW. Anielski promotes a standardized set of indicators that can populated with Statistics Canada data allowing for wider adoption of the Genuine Wealth Model. As communities collect more data, the model can be expanded and refined as needed (Anieski personal communication, 2012). Efforts to standardize calculation frameworks are important to encourage adoption of sustainability metrics for local applications. Integrating the academic community, practitioners, and appropriate government agencies in the conversation is critical.

9.3 Final Remarks

In a time of sustainability challenges and urgency, it is imperative that we account for the impacts of human activity and use that information toward positive change. My research agenda challenges how we think about the relationship between human economic activity and the natural world. A focus on sustainability measurement brings new insights toward understanding material and energy throughput at refined scales, essential for designing sustainable human systems. The economic worldview that we collectively subscribe to sets an agenda that defines how we organize and manage the human endeavor. Our economic ideas drive policies, priorities, and investment decisions, which have ramifications at a multitude of scales from the global to local. It influences how countries, regions and people interact. It influences how we perceive the natural world. At a fundamental level, it influences how we choose to live our lives.

The defining point that distinguishes ecological economics from the conventional economic worldview can be simplified in a question: do you believe that there are ecological limits to throughput economic growth? The answer matters. Accepting the idea of a sustainable scale to economic activity relative to biocapacity changes how we think about allocation of resources and their distribution. Additionally, it changes how we think about the ends of economics. My body of research is part of a larger effort to rethink the current economic story; a story, as told, may not end well. My commitment to support the spread of new ideas and principles of economic decision-making is part of a revolution in how we think about economics and ultimately in how we structure and organize our societies. This research has been a wonderful exploration to help advance a new economic vision based on the premise that the scale of throughput economic activity is limited by ecological constraints.

9.4 Afterword

I recently mentioned to my grandma that I am nearing the end of my PhD. I have been deliberate to not commit an end date and only speak in vague terms about progress. Not offering definitive timelines is a protective strategy students in their late thirties adopt. My long life as a student was perceived by my grandmother as avoiding the necessity of growing up. Having a family and working as a consultant on the side did not seem to count. I was still a university student. My grandma was raised in a time when university was a luxury. Physical hard work defined an individual's worth. Sitting at a computer for long periods constitutes idleness. Her view of students is one of gathering in pubs to exchange socialist ideas. On occasion, they riot in the streets and break some windows, especially if they live in Quebec. 'Did you learn anything' she asked. The tone was antagonistic but full of love. I replied, 'not a thing'. I was pleasantly tormenting her, as grandsons (even adult grandsons) are obliged to do. She was proud of me and I knew it. What did I learn though? Was it enough to justify a doctoral degree? Did it justify all the years, tuition fees, and foregone income by not working fulltime? Suddenly, what I learnt seemed small and insignificant. After a brief bout of anxiety, I changed my focus from the small 'learnt' pile to the enormous pile next to it of unknowns, curiosities, and problems that desperately need solutions. The important pile, I thought. The one I am most comfortable digging in. Yes, I am nearing the end of a very long university career as a student. I have more wrinkles on my face than when I started. My hair is cut shorter and has some grey in it. I am probably less radical. I typically wear collared shirts and no longer own a pair of cords. I gave up playing my two Bob Dylan songs at open mic night long ago. I increasingly opt for light beer and read the Economist. What has not changed, however, is my hunger to learn. I am not sure if it is getting older or a heightened sense of ecological uncertainty but my drive seems more urgent. The limitation is not lack of things to study or what to study, it is time. My excitement and energy remains strong, whether I am a student or not, younger or older. It is, however, time to close this chapter; with closure comes new adventures. I am comfortable giving up the 10% Tuesday student discount at the grocery store. It was starting to feel awkward anyway. For that matter, the seniors discount is only 20 years away.

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