ABSTRACT

METZGER, ALEXANDER E. Using Urban Metabolism to Measure the Sustainability of Urban Ecosystems. (Under the direction of Dr. Melissa McHale).

Urban ecosystems will play an important future role in balancing the needs of our populations with the environmental and ecological health of our planet. Current demands of urban populations negatively affect natural resource supplies and essential services provided by global ecosystems. Urban sustainability is becoming increasingly urgent in rectifying this destructive relationship as urban populations continue to expand.

Urban metabolism analyses are designed to explore urban sustainability by analyzing flows of water, food, energy, and material resources utilized by human-dominated ecosystems. The utility of this method, however, has been historically limited by low temporal resolution of data, inconsistent definition of urban boundaries, and a lack of quantifiable sustainability metrics.

Our goal was to design an urban metabolism study that addressed these three limitations. We then assessed the utility of our method in quantifying the sustainability of urban ecosystems. Conducting our study across four, five-year increments allowed us to increase the temporal resolution of the analysis beyond the typical metabolism study's single time increment. To create consistency in the spatial scale of our analysis and capture more of the urban ecosystems' important metabolic processes, we defined urban boundaries at the county-level rather than city-level, leading to a comparative study of Durham, Orange, and Wake Counties in North Carolina.

Creating sustainability metrics involved consulting the literature on sustainability of urban resource flows to identify four guiding themes: 1) use of clean, renewable resources, 2) closing resource loops, 3) dense, mixed-use development, and 4) localization of resource supplies. These themes guided our creation of ten quantifiable sustainability metrics that we scored on a scale of 0 to 1 and summed to determine each county's overall sustainability

score. Durham County earned the highest score of 1.47 out of a possible 10 due to its balanced water cycle and reductions in water consumption, gaseous pollutant outputs, and solid waste production. Our quantitative metrics also allowed to us link specific metabolic changes within the study area to changes in the overall sustainability of the three counties.

By increasing temporal resolution, setting appropriate urban ecosystem boundaries, and incorporating quantifiable sustainability metrics we increased the utility of this urban metabolism study and provided a more thorough basis for comparison among urban ecosystems. If adopted, our method has the potential to make urban metabolism a more powerful and widely utilized tool for sustainability efforts.

Using Urban Metabolism to Measure the Sustainability of Urban Ecosystems

by
Alexander E. Metzger

A thesis submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the degree of
Master of Science

Natural Resources

Raleigh, North Carolina

2013

APPROVED BY:

Dr. Toddi Steelman

Dr. George Hess

Dr. Melissa McHale
Chair of Advisory Committee

BIOGRAPHY

Alexander Metzger worked toward Master's degrees in Natural Resources Analysis and Assessment at North Carolina State University and Forest Ecology at the University of Helsinki through the Atlantis Transatlantic Master's Program. This program presented him with a great opportunity to become part of a global community involved in enhancing humans' relationship with the resources we depend on. This opportunity to connect with a brilliant, dedicated group of urban ecologists and a diverse, international group of forestry students and professionals was invaluable to his education and research.

Alexander will continue exploring human beings' connection to the resources that sustain them. He hopes to use his knowledge and experience to cultivate sustainability and vitality in urban ecosystems and in the actions of people worldwide.

ACKNOWLEDGEMENTS

There are many people I wish to thank for their dedication and support of my work. I would like to thank Dr. Melissa McHale, who did her best to stay in contact no matter where in the world we both found ourselves, Dr. George Hess, Dr. Toddi Steelman, and Dr. Eero Nikinmaa, who were patient and accommodating and shared their fantastic insights, and all of my graduate peers, who kept me engaged and motivated to think differently. I also wish to extend thanks to my many contacts at local, state, and federal agencies that helped me obtain data and guided me in using it correctly. Without their willingness to help, my research would not have been possible.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF APPENDICES

APPENDIX E

CHAPTER 1. Literature Review on Urban Metabolism and Sustainability of Urban Ecosystems

Section 1. Introduction

While over 50% of our current global population resides in urban areas, the United Nations predicts a rapid increase in global urbanization, bringing this figure to about 70% by 2050 (United Nations Department of Economic and Social Affairs 2008). Urban ecosystems in which these populations reside are known to have large impacts on earth's ecology and environmental health. These systems consume more resources than earth's ecosystems can collectively continue to produce, resulting in a majority of anthropogenic greenhouse gas emissions, land cover change, and alteration of global biogeochemical cycles (Grimm et al. 2000 and 2008, Vitousek et al. 1997, Botsford et al. 1997, Noble and Dirzo 1997). To mitigate these issues and move toward the ideal of sustainable societies, we must focus our efforts on changing the resource consumption patterns of our global urban ecosystems.

Perhaps the most widely referenced definition for sustainable development is '...development that meets all the needs of the present without compromising the ability of future generations to meet their own needs' (World Commission on Environment and Development 1987: 43). Phillis et al. (2010: 16) summarized that many well-known definitions of sustainability generally include the following elements:

a. Renewable resources should be harvested at a rate no greater than the natural regeneration rate.

b. Nonrenewable resources should be used at a rate no greater than the rate of creation of substitutes for them.
c. Waste emissions should not exceed the relevant assimilative capacities of ecosystems.

Thus, many urban ecologists have begun to address sustainability by studying flows of resources and energy through urban ecosystems. Urban metabolism, which Kennedy et al. (2007: 44) defined as “the sum total of the technical and socioeconomic processes which occur in cities, resulting in growth, production of energy, and elimination of waste,” is one promising method of analysis originating from this approach. Urban metabolism quantifies flows of energy and material resources into an urban ecosystem and wastes expelled from the system, providing insight into the internal “metabolic” processes which drive these flows. By understanding how urban metabolism drives resource consumption, global sustainability initiatives could be built around intelligently altering them.

The global community looks upon sustainability as an increasingly urgent goal (Holden et al. 2008) and urban metabolism has the potential to enhance sustainable development efforts with a detailed knowledge of urban resource flows. I reviewed the literature on sustainability of urban resource flows and present here a discussion of how urban metabolism can better address the main themes found in the urban sustainability literature.

Section 2. Sustainable Urban Resource Flows

2.1 Energy and Fuel

2.1.1 Urban Energy Systems

Electricity and fuel are consumed in practically every major activity required to fulfill the needs of urban residents. In a simplified view of this resource flow, energy is imported into the urban ecosystem as electricity and fuel and consumed in transportation, heating or cooling of buildings, industrial activities, and other processes. Fuels are most often used in combustion processes, which release many harmful pollutant emissions and greenhouse gasses. The transportation sector, for example, is almost exclusively powered by liquid petroleum and produces more greenhouse gasses than residential, commercial, or industrial activities (United States Energy Information Administration 2011).

While the use of electricity does not directly generate pollutants, our country's electricity grid depends mainly on the combustion of fossil fuels, producing even more greenhouse gasses than vehicle transportation (United States Energy Information Administration 2011). Our heavy reliance on non-renewable fossil fuels, such as oil and coal, threatens to destabilize our societies as we approach the limits of their capacity and contribute to the threat of potentially catastrophic changes in global climate (de Almeida and Silva. 2011, Nel and Cooper. 2009, Parry et al. 2007, Verbruggen and Al Marchohi. 2010). Much of the literature on sustainability of fuel and energy discusses alternative and renewable fuel and energy sources, reducing consumption, and increasing efficiency.

2.1.2 Renewable energy sources

The National Renewable Energy Laboratory (2012) defined renewable energy sources as those that rely on resources that are infinitely replenished, such as solar, wind, hydropower, tidal, biomass, and geothermal. Although nuclear is commonly discussed along with renewable energy sources, its reliance on a limited supply of uranium fuel excludes it as a renewable energy source. The literature on renewable energy sources is extensive and detailed. Rather than delving into a discussion on the benefits and issues with each source individually, I will focus on the best methods to integrate renewable energy systems into an urban environment.

The majority of renewable energy systems today are created by large government or corporate entities with no particular ties to the location of production (Bagliani et al. 2010). For example, the European Union Renewables Directive considered a plan to import electricity from large solar thermal plants that would be built in the North African deserts (Battaglini et al. 2009). Although large projects such as these may cause a great overall increase in the global production of renewable energy, Bagliani et al. (2010) noted the advantages of localized, cooperative renewable energy solutions. Input by local stakeholders would provide motivation to avoid social acceptance issues and local environmental impacts. Economic benefits may also follow, as local entrepreneurs, investors, and others have the opportunity to be involved in creating this infrastructure.

2.1.3 Localized energy production

Localized energy production may increase reliability and distribution efficiency of energy systems through custom-tailoring systems to utilize local resources and address peak demand characteristics (Pepermans et al. 2005). The US national electricity distribution grid experiences normal annual losses of 7% (United States Energy Information Administration 2011). Distributing energy production to multiple locations in the same grid can increase efficiency by 10-15%, while addressing voltage and peak demand issues (Dondi et al. 2002). Localized systems that utilize biomass and other combustibles have the option to incorporate combined heat and power systems, which capture the thermal energy created in electricity production and distribute it for space and water heating. These combined systems are known to be highly efficient (Obernberger 1998) and are already being successfully utilized in many large urban complexes and in some cities such as Helsinki, Finland.

2.1.4 Alternative fuels and transportation

Creating sustainable fuel and energy systems also necessitates a focus on our methods of transportation. Some of the more promising renewable transportation fuels include biofuels, such as biodiesel and ethanol, hydrogen and other gasses for use in fuel cells, or electricity. These sources have various benefits and issues, but all have the potential to utilize renewable energy sources.

One important barrier to large-scale adoption of renewable fuels is the relatively high cost of the fuels, vehicles that can utilize them, and physical infrastructure compared to the already well-established petroleum-based transportation scheme (MacLean and Lave 2003). Replacing or retrofitting our current petroleum infrastructure with renewable fuels will require significant time and monetary investment and may involve taking intermediate steps. The infrastructure could begin to incorporate cleaner fossil fuels, such as natural gas, and then transition to renewable sources over time (Muradov and Veziroglu 2005). Soloman et al. (2007) discussed the growth of ethanol use in response to the 1988 Alternative Motor Fuel Act, Corporate Average Fuel Economy (CAFE) standards and the 1992 Energy Policy Act. Their account suggests that some progress toward incorporation of renewable fuels has been accomplished through government incentives and regulatory actions coupled with a strong public desire to address social and environmental issues related to petroleum-based transportation.

If renewable alternative fuels are to become a major component of our transportation infrastructure, however, care must also be taken to create a system in which the production of these alternative fuels is not dependent on fossil energy. Corn ethanol, for example, has been shown to yield only 25% more energy than is used in its production (Hill et al. 2006).

Many other suggestions for increasing sustainability of transportation revolve around decreasing vehicle travel altogether. There are high embodied energy costs in the construction and maintenance of vehicles and infrastructure, as well as the production of fuels (MacLean and Lave 2003, Woodcock et al. 2007). Negative effects to human health have also been seen, including exposure to harmful pollutants, automobile accidents, and physical inactivity encouraged by the automation of travel. From a social justice perspective, the “car culture” creates increased social segregation and unequal access to certain amenities (Woodcock et al. 2007). Public transport and active transport (walking and bicycling) are favorable alternatives which help to avoid these problems by decreasing automobile travel (Woodcock et al. 2007). Reorganizing the form of urban ecosystems to make vehicle travel less necessary is essential to these alternatives.

2.1.5 Dense, mixed-use development

Urban form can play a large role in dictating energy use and the efficiency of transportation. Urban sprawl and low-density development have been shown to increase per capita consumption of energy and fuel (Gueneralp and Seto 2012), as have fragmentation and irregularity of urban land use patters (Chen et al. 2011). High urban density may be able to mitigate some of this inefficiency by decreasing the length of travel for urban residents (Chen et al. 2011, Woodcock et al. 2007). Other research indicates that much of the benefit of urban density stems from vehicle choice, for example making residents less likely to drive large pickup trucks and sport utility vehicles that are less fuel efficient. A comprehensive analysis of the benefits of density on fuel and energy efficiency concluded that benefits accrue mostly by influencing the number of vehicles per household, speed of vehicle travel, type and size of vehicles, and availability of public and alternate transit (Liu and Shen 2011). In addition, density increases the efficiency of public mass transit systems (Newman and Kenworthy 1989).

Transportation efficiency may also depend largely on locating residences, schools, stores, businesses, grocery stores, and other amenities in close proximity to one another in a mixed-use development scheme. This strategy decreases reliance on automobiles by making goods, services, and activities more accessible by bicycle and pedestrian transportation. Active, alternative modes of transportation are also associated with a range of social, psychological, and health benefits related to community-oriented urban design (Jabareen 2006, Leyden 2003, Woodcock et al. 2007).

Mixed-use development can help create ecological, social, and economic functionality throughout the urban ecosystem (Lovell and Johnston 2009). Urban green spaces, such as parks, green roofs, and street trees, are examples of incorporating ecological components into the urban landscape. Although green spaces such as parks may slightly decrease overall density, they can reduce energy consumption of heating and cooling through shading and microclimate effects while reducing psychological stress and contributing to a sense of well-being (Lafortezza et al. 2009, Simpson and McPherson 1998).

2.1.6 Summary of sustainable energy systems

Renewable energy sources are a major focus of the literature on sustainability. Localization of these energy sources has many benefits in the way of efficiency and socioeconomic impacts. Incorporating density and mixed-use development into the urban structure can greatly improve the efficiency of energy and transportation systems by making combined heat and power systems possible and by encouraging bicycle, pedestrian, and mass transport.

2.1.7 Urban energy systems in metabolism studies

The need to address energy consumption and transportation is consistently expressed in the urban metabolism literature, as they collectively drive the majority of fuel consumption and gaseous pollutant emissions in all urbanized areas. Baccini's (1997) and Newman's (1999) criteria for sustainable energy systems both emphasized reduction in energy and fuel consumption and pollutant emissions through decreasing vehicle transportation, increasing energy efficiency, and changing the energy infrastructure to rely solely on renewable energy sources. Per capita electricity and fuel consumption has increased through time in most urban areas, and renewable energy has either been absent or shown to represent a nearly negligible percentage of total demand (i.e. Kennedy 2007, Newman 1999, Sahely et al. 2003, Warren-Rhodes and Koenig 2001).

Metabolism studies have linked high population densities to lower vehicle energy consumption and well-developed public transportation and pedestrian networks (Kennedy 2007, Newcombe et al. 1978, Warren-Rhodes and Koenig 2001). Hong Kong's transportation-related energy consumption, for example, has been much lower than less-dense urban areas such as the Greater Toronto Area and Los Angeles County (Ngo and Pataki 2008, Sahely et al. 2003, Warren-Rhodes and Koenig 2001). These studies and others in Newman and Kenworthy's (1991) research on a set of 32 cities across the globe appear to show that higher urban density results in lower per capita transportation-related energy consumption. This finding is also supported by Barles (2009) more recent comparison of

Paris and its encompassing Île-de-France region, which showed lower overall per capita fossil fuel consumption and gaseous pollutant emissions in the dense urban core of Paris than in the sprawling Île-de-France region.

Though urban metabolism studies have been able to demonstrate a link between urban form, transportation, and energy consumption, some important topics related to sustainable urban energy systems remain untouched. Renewable energy and the beneficial effects of green spaces are two such topics that have gained wide visibility in the literature, but are so far missing from most urban metabolism assessments.

2.2 Water

2.2.1 Urban Water Systems

Water is one of the most basic of human needs, essential in nearly all aspects of life. Within urban ecosystems water typically follows a linear flow pattern, entering the urban ecosystem through ground water reservoirs, surface flow, precipitation, or facilitated transport, such as truck or pipeline delivery. Some of this water is used by people in various domestic, commercial or industrial processes. This water is then expelled directly or through wastewater treatment plants into ground and surface water reservoirs and leaves the system through surface runoff or evapotranspiration. While urban water systems have historically focused primarily on meeting demands for clean, usable water, current sustainability-centered designs also include considerations for water self-sufficiency, water quality, and environmental impacts (Lim et al. 2010, Rygaard et al. 2011).

2.2.2 Water self-sufficiency

Water self sufficiency is considered a key component of sustainability partly due to the large economic and environmental costs of importing water. Desalinization and rainwater collection is practiced in many countries throughout the world to fulfill a significant portion of domestic and drinking water requirements. As local water supplies worldwide are already being utilized beyond their limits, especially in arid environments, reclamation and reuse of water is becoming increasingly important (Rygaard et al. 2011). A sophisticated model designed by Lim et al. (2010) to optimize urban water systems determined that using reclaimed water from wastewater treatment to fulfill demands, diluted with surface water to reach quality standards, contributed significantly to self-sufficiency. Various well-developed, integrated processes exist for reclaiming water during treatment (Listowski et al. 2011), which could reduce pressure on local ground and surface water sources or the need for imports. Riparian and other ecological systems may also benefit from the reduction (Lim et al. 2010) or replacement (Rygaard et al. 2011) of water withdrawals from surface sources.

2.2.3 Minimizing environmental impacts of water systems

Urban water systems can have large environmental impacts through consumption of energy in water treatment and distribution. Researchers have created models of the urban water cycle to identify the best means of dealing with these issues. Fagan et al.’s (2010) model suggested that smaller individual grey water recycling systems collectively produce more greenhouse gas emissions than large, centralized grey water systems. They also found that individual hot water systems account for the vast majority of residential and commercial energy consumption. These findings suggest that because dense urban areas allow for larger water distribution networks and incorporation of combined heat and power systems, they may be the key to reducing the energy consumption of urban water systems.

Another model by Icke et al. (1999) suggested that investments toward separating storm runoff from sanitary sewer systems can have a large impact on controlling nutrient leakage and pollution in an urban water system. Reclamation of the nutrients present in wastewater could also serve as a resource for fertilizing agricultural land in developed and developing counties, although risks of infectious disease transmission must be mitigated (Winker et al. 2009).

2.2.4 Summary of sustainable urban water systems

The literature makes some clear suggestions for designing a more sustainable urban water system. Designing wastewater systems to reclaim water and nutrients creates a more closed, circular flow of resources which contributes to self-sufficiency. Increased efficiency of water systems can be accomplished through utilizing urban density to create larger, distributed water networks and incorporate combined heat and power systems.

2.2.5 Urban water systems in metabolism studies

The sustainability of water resources has consistently remained a focus of urban metabolism literature due to the prevalence of issues with urban ecosystems meeting their current and future demands. Hong Kong and Los Angeles County have historically depended on imported water to fulfill their demand (Ngo and Pataki 2008, Warren-Rhodes and Koenig 2001). Although the Swiss Lowlands were able to satisfy their current demands locally in 1990, Baccini (1997) predicted a depletion of groundwater resources over time. Baccini (1997) and Newman (1999) stated that a sustainable urban ecosystem should fulfill its demand with only local water resources, thus emphasizing the importance of the local water cycle.

In urban ecosystems such as Hong Kong, which is surrounded by seawater, and Los Angeles, which does not consistently receive adequate precipitation to cover its water demands, local water sources may never be able to completely fulfill demands. Decker et al. (2000), Newman (1999), and others discuss rooftop catchments, reclaimed wastewater and other methods for improving the use of existing water resources on both the input and output phases of the water cycle. Ngo and Pataki (2008) were successful in quantifying reclaimed wastewater, concluding that it represented only a small percentage contribution to consumption and water supplies.

Environmental pollution by wastewater output has also been an important topic in the urban metabolism literature. Wastewater treatment infrastructure varies greatly among cities and generally increases with wealth and development (Decker et al. 2000). In many cities, wastewater was diverted directly into waterways, causing extensive risk to human and ecosystem health. Although Hong Kong, like other cities, has increased their treatment of raw sewage between 1971 and 1997, much is still expelled directly into its main harbor and other marine waterways (Newcombe et al. 1978, Warren-Rhodes and Koenig 2001). The reclamation of nutrients for use as agricultural fertilizer is a topic not adequately addressed by urban metabolism literature, nor is the possible benefits of urban green spaces or water storage techniques.

2.3 Food

2.3.1 Urban Food Systems

In a typical urban ecosystem in the United States, food is imported from large, distant agricultural operations, processing plants, and warehouses. Human consumption produces both solid waste, in the form of packaging and food waste, and wastewater, in the form of human excreta. Solid waste is landfilled, incinerated, or recycled, and wastewater is treated to remove pollutants and expelled from the system. Creating a sustainable urban food system will require a restructuring of food supply and agriculture to reduce environmental impacts and continuing to meet the needs of the population.

2.3.2 Local agriculture

Localization of agriculture is a common, though debated, suggestion for increasing the sustainability of our food systems. One typical argument for localizing agriculture is that it reduces the distance, and thus the energy input, required to transport food from production areas to consumers (Cowell and Parkinson 2003). Other researchers question this argument, showing that transportation-related energy input may be similar in local food economies due to lower economies of scale. Local food systems may rely on many smaller vehicles as opposed to fewer larger vehicles, for example, and require consumers to travel to multiple locations for goods (Mariola 2008). Pirog et al. (2001) found that the most efficient type of distribution systems incorporate larger-capacity vehicles and mass distribution networks within a regional (statewide) food market.

Alternatively, Mariola (2008) points out that urban and community gardens may diminish the importance of these transportation and economic networks altogether, as they produce food for consumption in the most direct way possible. A material flow analysis of food production in Switzerland showed that of the total energy consumption in food systems, 27% is due to mechanization, fertilizer and pesticide application, greenhouse operations, and other agricultural processes, 15% to retailing, 12% to processing, and 8% to transportation (Kytzia et al. 2004). Urban and community gardens can almost completely avoid the latter three categories, and significantly reduce mechanization, assuming these gardens rely mainly on manual labor. In addition, the cultivation of vacant lots in destitute areas may increase property values, encourage reinvestment and revitalization, and boost visibility and community interest, perpetuating the process in other unstable areas of the urban ecosystem (Metcalf and Widener 2011).

Despite the great potential of urban gardening in vacant lots, this strategy alone is unlikely to provide enough agricultural area to produce a significant amount of food. Rooftop gardens are another method for converting largely underutilized urban surfaces into vegetated, food-producing areas. Vegetated roofs aid in controlling the rate of storm water runoff through absorption and slower release (Teemusk and Mander 2011) and have been found to significantly reduce the energy costs of cooling buildings (Kumar and Kaushik 2005, Wong et al. 2003). Another proposed method, peri-urban agriculture, utilizes land at the periphery of urban development. As many as 800 million people worldwide are estimated to be involved in peri-urban agriculture, and this method has been found to be economically viable (Midmore and Jansen 2003).

These methods of incorporating agriculture into an urban setting also present a number of challenges. The quality of urban soil and its contamination of lead and other harmful substances has long been a concern (Jim 1998, Mielke et al. 1983, Rodrigues et al. 2009). These hazards can be mitigated by several soil remediation methods or restricting human and vehicle traffic (Millward et al. 2011, White and Claxton 2004).

The values of land and economic potential for development, as discussed in cultivation of vacant lots, may also be a limiting factor. Rising land values that accompany the expansion of urban ecosystem may make commercial peri-urban agriculture economically uncompetitive with agriculture in rural areas (Midmore and Jansen 2003). Combining urban community agriculture with regional agricultural systems may prove to be the most realistic way to create an efficient food system of the appropriate capacity (Mariola 2008). Care must be taken, however, to incorporate these elements in a way that does not drastically decrease the density of the urban ecosystem and negatively impact density-dependent sustainability projects.

2.3.3 Elements of organic agriculture

Increased use of fossil fuel-reliant agrochemicals (mainly fertilizers and pesticides) and intensified agricultural practices from "the Green Revolution" of the 1960s have greatly increased our ability to meet food demands. Despite these benefits, intensive agricultural methods have negatively affected soil and environmental quality, increased insect predation, and created disease resistance to formerly-effective crop treatments, signaling the need for alternative production methods (Matson et al. 1997, Pimentel et al. 2005).

Organic agricultural practices that incorporate manure and natural fertilizers, cover crops, and crop rotation have been shown to reduce agrochemical inputs, soil erosion, and water consumption and improve soil quality and biodiversity (Fliessbach et al. 2007, Pimentel et al. 2005). Although organic agricultural methods remain a controversial topic (Rigby and Caceres 2001), we will attempt to address the most fundamental aspect of the debate: replacement of chemical fertilizers and pesticides.

Soil nutrients in organic agriculture are typically supplied by livestock animal manure and nitrogen-fixing legume cover crops. These nutrient sources do not rely directly on fossil fuel inputs, and can be produced at the farm. While cover crops are very feasible in an urban setting, animal manure may not be available. Instead, nutrients recovered from local wastewater and solid waste could conceivably replace animal fertilizer (Matson et al. 1997). Composted organic solid waste is already used to various degrees worldwide as a fertilizer and soil conditioner (Hargreaves et al. 2008). One study in the US found that 21% of an analyzed solid waste stream consisted of organic waste, illustrating the further potential for utilization of this resource (Zeng et al. 2005). Fertilizers derived from wastewater, as previously mentioned, could also cover a significant portion of agricultural nutrient demand in countries around the world (Winker et al. 2009).

There are, however, some problems with solid waste and wastewater fertilizers. Many compost operations, especially ones using sewage sludge as a feedstock for decomposition, contain heavy metals and salts in potentially harmful concentrations. Hargreaves et al. (2008) point out that heavy metal contamination could be mitigated by proper monitoring, separation of solid waste before collection, and avoiding the use of sewage sludge with high metal concentrations. Excluding industrial sources of wastewater can help to avoid this problem, as domestic sources are found to have low metal concentrations. An extensive study of solid waste compost and sewage sludge determined that proper sorting processes minimized the risks posed by heavy metal contamination (Smith 2009).

Organic agriculture uses a wide variety of techniques to replace chemical pesticides, fungicides, and other destructive agrochemicals, such as hand weeding to avoid herbicide use, rotation of multiple crops to control soil insects and disease, use of oils and soaps instead of chemical insecticides, and pest-resistant cover crops. Other more advanced techniques, such as monitoring and balancing of beneficial insect activity, selection of resistant cultivars, and advanced soil management techniques, rely on more intensive ecological knowledge (Zinati 2002). Encouragingly, the vast majority of these techniques can be applied very easily to agriculture in urban settings, and do not require extensive resource inputs.

2.3.4 Summary of sustainable urban food systems

The literature on urban food systems suggests that at least a portion of total food demand could be fulfilled locally through urban and rooftop gardens and peri-urban agriculture. Regional-scale agriculture is a more realistic way to fulfill the bulk of urban food demand, and would increase energy efficiency and shift away from the current national and global food networks. Organic agricultural methods have the potential to reduce petrochemical inputs by utilizing locally available resources and nutrients from solid waste and wastewater. All of these solutions emphasize the importance of closing resource loops and relying on local, renewable resources.

2.3.5 Urban food systems in metabolism studies

One of the most common discussions in urban metabolism literature related to sustainability of food systems is the contribution of local agriculture to sustainability. Metabolism researchers tend to agree with the literature's suggestion that local food systems cut down on transportation energy and contribute to self-sufficiency (Baccini 1997, Ngo and Pataki 2008). Baccini (1997) found that the Swiss Lowland region's government-controlled agricultural production provided more than half of the region's food in 1990. Newcombe et al. (1971) estimated that Hong Kong produced a quarter of total food consumed locally, although Warren-Rhodes and Koenig's (2001) 1997 update showed that this ratio had dropped. Despite the prevailing trends of decreasing local food production and increased reliance on national and global food networks, Los Angeles County's local vegetable consumption increased from 2% to 3% of its total vegetable consumption, perhaps signaling a shift in this trend (Kennedy et al. 2007, Ngo and Pataki 2008).

Another lesser-discussed topic is the contribution of urban nutrient cycles to food systems. Some researchers were able to quantify the fertilizers, animal feed, and other resources imported for agricultural purposes. Researchers who have constructed balances for nutrients such as nitrogen and phosphorus indicated low levels of nutrient recycling (Barles 2009, Kennedy et al. 2007, Newcombe 1978, Warren-Rhodes and Koenig 2001). They also discussed the importance of analyzing detailed nutrient cycles connected to organic solid waste and wastewater in efforts to reduce nutrient pollution and increase fertility of local agricultural land (Kennedy 2007).

2.4 Construction materials

2.4.1 Urban Materials Systems

The flow of construction materials through the urban ecosystem is a much slower process than other resource flows discussed. Materials such as brick, wood, and concrete are imported and stored in the form of buildings, roads, bridges and other urban infrastructure. These materials degrade over long periods of time, resulting in waste and the need for replacement. Construction of the built environment is one of the largest consumers of environmental resources, and also one of the largest polluters (Ding 2008).

The impact that construction has on the sustainability of an urban ecosystem is largely tied to the sustainability of materials used. The vast majority of construction materials are non-renewable, and although stocks of most basic material resources (sand, gravel, iron ore, minerals used in cement and concrete, and others) are plentiful and not in serious danger of depletion (Horvath 2004), the *Institut der Deutschen Wirtschaft* (Cologne Institute for Economic Research) estimates the remaining supplies of several important metals will last less than 50 years (Lehmann 2011).

2.4.2 Production and embodied energy

While limited resource stocks present one possible issue, the indirect impacts of their life-cycle are perhaps a much more important concern (Horvath 2004). The first important phase to consider in the life-cycle of construction materials is their creation. The extraction of raw materials and transformation into useable components consume energy and natural resources, and result in environmental impacts and production of wastes. The energy used in producing a particular construction material and preparing it for use is known as its embodied energy. Brick, for example, requires the mining of clay and other minerals and firing to extremely high temperatures (Harris 1999). Cement, one of the most prevalent construction materials today, also uses a high-temperature kiln process. The cement industry is said to consume 2% of global primary energy and produce 5% of anthropogenic CO_2 emissions (Worrell et al. 2001).

Production and use of construction materials is estimated to represent well over half of a building's total embodied energy (Dimoudi and Tompa 2008). Material-specific values and their implications can be difficult generalize, as illustrated by the debate over construction lumber. While some calculations suggest that imported softwood timber can have a much higher embodied energy than concrete and cement (Calkins 2009, Harris 1999), many studies have shown that replacing concrete and steel structural elements with timber yields large net decreases in fossil fuel consumption and greenhouse gas production. This can be attributed to the fact that the total energy used to produce construction timber factors in solar energy that trees utilize for growth and energy required for manufacture, which can be derived from wood wastes (Joseph and Tretsiakova-McNally 2010). Overall, it is important to consider the source of energy used in producing construction materials.

2.5.3 Degradation and material recycling

Degradation and demolition of the built structure is another important phase in the life cycle of construction materials. Construction and demolition waste accounts for a very large percentage of total waste disposed of at landfills (Sagoe-Crentsil et al. 2001). Reuse and recycling of construction and demolition waste makes for a much more sustainable system by reducing inputs of new materials and outputs of waste (Yuan et al. 2011). Although many of the recycling processes are very energy consumptive, some allow for direct reuse or simple mechanical crushing or chipping into aggregates. Waste concrete can be crushed into aggregate for the production of new, non-structural concrete with only a small reduction in quality (Sagoe-Crentsil et al. 2001). Asphalt, brick, ferrous and non-ferrous metals, paper products, plastics, masonry, and timber all have properties suitable for recycling into new products. Construction timber can be directly re-used in whole timber form, cut into aggregate for use in furniture and kitchen utensils, gasified and used in energy production, or chipped and formed into wood-based paneling (Tam and Tam 2006).

2.5.4 Summary of sustainable construction material systems

Construction may represent one of largest challenges in moving toward the idealized notion of sustainability. The literature makes it is unclear whether construction material flows in urban ecosystems can be made sustainable, but the answer seems to lie in use of less energy-intensive materials, producing and utilizing these materials with renewable energy sources, reusing or recycling materials to the greatest extent, and urban design that reduces material consumption and turnover. The literature illustrates that a life cycle approach to construction material use can make the built environment more sustainable by closing resource loops, consuming less energy and fossil fuels, reducing greenhouse production and other environmental impacts, and reducing stress on raw material sources.

2.5.5 Urban construction materials in metabolism studies

The enormous impact that material consumption has on the sustainability of urban ecosystems is well discussed in the urban metabolism literature. Baccini (1997) stated that sustainability demands that only local materials can be used, in which case recycling and reuse would be imperative. Newman (1999) also focused on reducing per capita consumption. To address the physical impacts of material consumption, Decker (2000) pointed out the issue with embodied energy in producing construction materials, and discussed the closing of material resource loops to reduce throughflow.

The quantification of materials by urban metabolism studies has made it possible to discuss utilization of local resources and recycling and reuse. Baccini's (1997) study of the Swiss Lowlands, and Barles (2009) study of the Île-de-France region both included measures of extraction of local materials and discussion of their contribution to total material resource demands. The Swiss Lowlands region was found to be nearly self-sufficient in 1990 and while the Île-de-France region was far from self-sufficiency, it was deemed theoretically able to produce over half of its material demands. Baccini (1997) concluded that reducing material demands would be essential to perpetuating the Swiss Lowlands high degree of material self-sufficiency. Barles (2009) also emphasized the need for dematerialization in order to reduce the Île-de-France's need for extraction of local resources (Baccini 1997, Barles 2009).

Recycling has been proven to be a much more achievable means of increased self-sufficiency in urban ecosystems. The impact that rapid growth has had on material imports has been described in metabolism studies of Hong Kong. Although Hong Kong's programs grew to recycle about one third of its metropolitan solid waste in 1997, the authors still emphasized the large contribution of its "throwaway culture" to the massive material throughflow (Newcome 1978, Warren-Rhodes Koenig 2001). Some researchers (i.e. Ngo and Pataki 2008 and Sahely 2003) have shown growth in urban recycling programs. Kenney's (2007) comparative analysis of urban metabolism studies, however, showed that while recycling programs tend to decrease mainly residential solid waste, solid waste production has continued to trend upward in other sectors.

Section 3. Conclusion

Although the sustainability literature on each major resource flow in the metabolism of urban ecosystems tells a unique story, it is clear that all are closely connected by a set of common themes. We identified four of these themes.

Renewable resources are absolutely essential for long-term sustainability. As we have discussed, there are limits on many of the energy and materials resources our societies currently depend on, and we are increasingly subjected to the negative environmental, economic, and social impacts that these resources and their wastes can have. The literature contains proposals that renewable resources can be incorporated into the urban ecosystem through alternative electricity production and fuels, reclaiming nutrients from solid waste and wastewater, and using wood and other renewable construction alternatives. The need for clean, renewable energy sources pervades the discussion, as the use of energy factors into nearly all aspects of urban metabolism.

Dense, mixed-use development has been shown to contribute a great deal to the efficiency of transportation and water systems. The positive effects of density will require re-designing our urban ecosystems to incorporate elements such as pedestrian, bicycle, and public transportation and combined heat and energy systems. Adding mixed-use components and urban green spaces, in a way that does not contradict the benefits of urban density may play a large role in efficiently fulfilling the needs of residents.

Closing resource loops can reduce the demands of urban metabolism on diminishing global resource stocks and the burden of waste absorption on the environment. The classic, linear "resources in, waste out" model of resource flows must be shifted to a circular process in which wastes are utilized as primary resources. Recycling and reuse of construction and demolition waste materials, organic and non-organic solid waste components, and wastewater are all methods for accomplishing this. These processes can be expanded by reengineering our urban ecosystems to use only construction materials that have high potential for reuse at the end of their lifecycle, organizing our agricultural systems to rely on recycled nutrients, and increasing the use of recycled water.

Localization is integral to many of the sustainability-enhancing changes proposed in the literature. Localization of food and energy systems may contribute to efficiency, self-sufficiency, and socioeconomic stability. Decreasing an urban ecosystem's dependence on national or global food, fuel, and energy markets will serve to further insulate it from potentially unpredictable and destabilizing changes. Localization also puts residents and local organizations in close contact with the sources and consequences of their resource consumption, encouraging them to fully consider their impacts and incentivizing environmentally- and socioeconomically-friendly solutions.

We have used the literature to identify widely discussed solutions for urban sustainability and explored the insight that urban metabolism researchers demonstrate in these topics. We were able to distill these topics in the sustainability literature into four main themes relating to urban metabolic resource flows. Our intent is that this review should provide researchers, decision makers, and practitioners in the fields of urban metabolism and sustainability with a clearer view of how to address urban sustainability issues in a direct and practical manner.

References Chapter 1

Bagliani, M., E. Dansero, and M. Puttilli. 2010. Territory and energy sustainability: the challenge of renewable energy sources. *Journal of Environmental Planning and Management* 53(4): 457-472.

Battaglini, A., J. Lilliestam, A. Haas, and A. Patt. 2009. Development of SuperSmart Grids for a more efficient utilisation of electricity from renewable sources. *Journal of Cleaner Production* 17(10): 911-918.

Botsford, L. W., J. C. Castilla, and C. H. Peterson. 1997. The management of fisheries and marine ecosystems. *Science* 277(5325): 509-515.

Brundtland, G. H. 1987. Our common future: A report of the World Commission on Environment and Development.

Calkins, M. 2009. *Materials for sustainable sites*: *a complete guide to the evaluation*, *selection*, *and use of sustainable construction materials*. John Wiley & Sons: Hoboken, NJ, USA.

Chen, Y., X. Li, Y. Zheng, Y. Guan, and X. Liu. 2011. Estimating the relationship between urban forms and energy consumption: A case study in the Pearl River Delta, 2005-2008. *Landscape and Urban Planning* 102(1): 33-42.

Cowell, S. and S. Parkinson. 2003. Localisation of UK food production: an analysis using land area and energy as indicators. *Agriculture Ecosystems & Environment* 94(2: 221-236.

de Almeida, P. and P. D. Silva. 2011. Timing and future consequences of the peak of oil production. *Futures* 43(10): 1044-1055.

Dimoudi, A. and C. Tompa. 2008. Energy and environmental indicators related to construction of office buildings. *Resources Conservation and Recycling* 53(1-2): 86-95.

Ding, G. K. C. 2008. Sustainable construction - The role of environmental assessment tools. *Journal of environmental management* 86(3): 451-464.

Dondi, P., D. Bayoumi, C. Haederli, D. Julian, and M. Suter. 2002. Network integration of distributed power generation. *Journal of Power Sources* 106(1-2): 1-9.

Fagan, J. E., M. A. Reuter, and K. J. Langford. 2010. Dynamic performance metrics to assess sustainability and cost effectiveness of integrated urban water systems. *Resources Conservation and Recycling* 54(10): 719-736.

Fliessbach, A., H. Oberholzer, L. Gunst, and P. Maeder. 2007. Soil organic matter and biological soil quality indicators after 21 years of organic and conventional farming. *Agriculture Ecosystems & Environment* 118(1-4): 273-284.

Grimm, N. B., J. M. Grove, S. T. A. Pickett, and C. L. Redman. 2000. Integrated approaches to long-term studies of urban ecological systems. *Bioscience* 50(7): 571-584.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global change and the ecology of cities. *Science* 319(5864): 756-760.

Gueneralp, B. and K. C. Seto. 2012. Can gains in efficiency offset the resource demands and CO(2) emissions from constructing and operating the built environment? *Applied Geography* 32(1): 40-50.

Hargreaves, J. C., M. S. Adl, and P. R. Warman. 2008. A review of the use of composted municipal solid waste in agriculture. *Agriculture Ecosystems & Environment* 123(1-3): 1-14.

Harris, D. 1999. A quantitative approach to the assessment of the environmental impact of building materials. *Building and Environment* 34(6): 751-758.

Hill, J., E. Nelson, D. Tilman, S. Polasky, and D. Tiffany. **2006.** Environmental, economic, and energetic costs and benefits of biodiesel and ethanol biofuels. *Proceedings of the National Academy of Sciences of the United States of America* 103(30: 11206-11210.

Holden, M., M. Roseland, K. Ferguson, and A. Perl. 2008. Seeking urban sustainability on the world stage. *Habitat International* 32(3): 305-317.

Horvath, A. 2004. Construction materials and the environment. *Annual Review of Environment and Resources* 29181-204.

Icke, J., R. van den Boomen, and R. Aalderink. 1999. A cost-sustainability analysis of urban water management. *Water Science and Technology* 39(5): 211-218.

Jabareen, Y. R. 2006. Sustainable urban forms - Their typologies, models, and concepts. *Journal of Planning Education and Research* 26(1): 38-52.

Jim, C. 1998. Urban soil characteristics and limitations for landscape planting in Hong Kong RID A-2289-2010. *Landscape and Urban Planning* 40(4): 235-249.

Joseph, P. and S. Tretsiakova-McNally. 2010. Sustainable Non-Metallic Building Materials. *Sustainability* 2(2): 400-427.

Kennedy, C., J. Cuddihy, and J. Engel-Yan. 2007. The changing metabolism of cities. *Journal of Industrial Ecology* 11(2): 43-59.

Kumar, R. and S. Kaushik. 2005. Performance evaluation of green roof and shading for thermal protection of buildings. *Building and Environment* 40(11): 1505-1511.

Kytzia, S., M. Faist, and P. Baccini. 2004. Economically extended - MFA: a material flow approach for a better understanding of food production chain. *Journal of Cleaner Production* 12(8-10): 877-889.

Lafortezza, R., G. Carrus, G. Sanesi, and C. Davies. 2009. Benefits and well-being perceived by people visiting green spaces in periods of heat stress. *Urban Forestry & Urban Greening* 8(2): 97-108.

Lehmann, S. 2011. Resource recovery and materials flow in the city: zero waste and sustainable consumption as paradigm in urban development. *Journal of Green Building* 6(3): 88-105.

Leyden, K. M. 2003. Social capital and the built environment: The importance of walkable neighborhoods. *American Journal of Public Health* 93(9): 1546-1551.

Lim, S., S. Suh, J. Kim, and H. S. Park. 2010. Urban water infrastructure optimization to reduce environmental impacts and costs. *Journal of environmental management* 91(3): 630-637.

Listowski, A., H. H. Ngo, W. S. Guo, and S. Vigneswaran. 2011. A novel integrated assessment methodology of urban water reuse. *Water Science and Technology* 64(8): 1642-1651.

Liu, C. and Q. Shen. 2011. An empirical analysis of the influence of urban form on household travel and energy consumption. *Computers Environment and Urban Systems* 35(5): 347-357.

Lovell, S. T. and D. M. Johnston. 2009. Designing Landscapes for Performance Based on Emerging Principles in Landscape Ecology. *Ecology and Society* 14(1): 44.

MacLean, H. and L. Lave. 2003. Evaluating automobile fuel/propulsion system technologies. *Progress in Energy and Combustion Science* 29(1): 1-69.

Mariola, M. J. 2008. The local industrial complex? Questioning the link between local foods and energy use. *Agriculture and Human Values* 25(2): 193-196.

Matson, P., W. Parton, A. Power, and M. Swift. 1997. Agricultural intensification and ecosystem properties. *Science* 277(5325): 504-509.

Metcalf, S. S. and M. J. Widener. 2011. Growing Buffalo's capacity for local food: A systems framework for sustainable agriculture. *Applied Geography* 31(4): 1242-1251.

Midmore, D. J. and H. G. P. Jansen. 2003. Supplying vegetables to Asian cities: is there a case for peri-urban production? *Food Policy* 28(1): 13-27.

Mielke, H., J. Anderson, K. Berry, P. Mielke, R. Chaney, and M. Leech. 1983. Lead Concentrations in Inner-City Soils as a Factor in the Child Lead Problem. *American Journal of Public Health* 73(12): 1366-1369.

Millward, A. A., K. Paudel, and S. E. Briggs. 2011. Naturalization as a strategy for improving soil physical characteristics in a forested urban park. *Urban Ecosystems* 14(2): 261-278.

Muradov, N. and T. Veziroglu. 2005. From hydrocarbon to hydrogen-carbon to hydrogen economy. *International Journal of Hydrogen Energy* 30(3): 225-237.

National Renewable Energy Laboratory. 2012. *Renewable Energy Basics*. http://www.nrel.gov/learning/re_basics.html. Accessed 14 June 2012.

Nel, W. P. and C. J. Cooper. 2009. Implications of fossil fuel constraints on economic growth and global Warming. *Energy Policy* 37(1): 166-180.

Newman, P. and J. Kenworthy. 1989. Gasoline Consumption and Cities - a Comparison of United-States Cities with a Global Survey. *Journal of the American Planning Association* 55(1): 24-37.

Noble, I. R. and R. Dirzo. 1997. Forests as human-dominated ecosystems. *Science* 277(5325): 522-525.

Obernberger, I. 1998. Decentralized biomass combustion: State of the art and future development. *Biomass & Bioenergy* 14(1): 33-56.

Parry, M. L., C. O.F, and P. J.P. 2007. *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK: Cambridge University Press.

Pepermans, G., J. Driesen, D. Haeseldonckx, R. Belmans, and W. D'haeseleer. 2005. Distributed generation: definition, benefits and issues RID A-5523-2008. *Energy Policy* 33(6): 787-798.

Phillis, Y. A., V. S. Kouikoglou, and V. Manousiouthakis. 2010. A Review of Sustainability Assessment Models as System of Systems. *Ieee Systems Journal* 4(1): 15-25.

Pimentel, D., P. Hepperly, J. Hanson, D. Douds, and R. Seidel. 2005. Environmental, energetic, and economic comparisons of organic and conventional farming systems. *Bioscience* 55(7): 573-582.

Pirog, R., T. van Pelt, K. Enshayan, and E. Cook. 2001. Food, fuel and freeways: an iowa perspective on how far food travels, fuel usage, and greenhouse gas emissions. Ames, IA: Leopold Center for Sustainable Agriculture. www.leopold.iastate.edu/pubs-and-papers/2001-06-food-fuel-freeways. Accessed 14 December 2011.

Rigby, D. and D. Caceres. 2001. Organic farming and the sustainability of agricultural systems RID A-9606-2009. *Agricultural Systems* 68(1): 21-40.

Rodrigues, S., G. Urquhart, I. Hossack, M. E. Pereira, A. C. Duarte, C. Davidson, A. Hursthouse, P. Tucker, and D. Roberston. 2009. The influence of anthropogenic and natural geochemical factors on urban soil quality variability: a comparison between Glasgow, UK and Aveiro, Portugal. *Environmental Chemistry Letters* 7(2): 141-148.

Rygaard, M., P. J. Binning, and H. Albrechtsen. 2011. Increasing urban water self-sufficiency New era, new challenges. *Journal of environmental management* 92(1): 185-194.

Sagoe-Crentsil, K., T. Brown, and A. Taylor. 2001. Performance of concrete made with commercially produced coarse recycled concrete aggregate. *Cement and Concrete Research* 31(5): 707-712.

Simpson, J. R. and E. G. McPherson. 1998. Simulation of tree shade impacts on residential energy use for space conditioning in Sacramento. *Atmospheric Environment* 32(1): 69-74.

Smith, S. R. 2009. A critical review of the bioavailability and impacts of heavy metals in municipal solid waste composts compared to sewage sludge. *Environment international* 35(1): 142-156.

Solomon, B. D., J. R. Barnes, and K. E. Halvorsen. 2007. Grain and cellulosic ethanol: History, economics, and energy policy. *Biomass & Bioenergy* 31(6): 416-425.

Tam, V. and C. Tam. 2006. A review on the viable technology for construction waste recycling. *Resources Conservation and Recycling* 47(3): 209-221.

Teemusk, A. and U. Mander. 2011. The Influence of Green Roofs on Runoff Water Quality: A Case Study from Estonia. *Water Resources Management* 25(14): 3699-3713.

United Nations Department of Economic and Social Affairs. 2008. *World Urbanization Prospects: The 2007 Revision*. ESA/P/WP/205. New York, NY: United Nations.

United States Energy Information Administration. *Annual Energy Outlook 2011 with Projections to 2035*. DOE/EIA-0383(2011): United States Energy Information Administration.

Verbruggen, A. and M. Al Marchohi. 2010. Views on peak oil and its relation to climate change policy. *Energy Policy* 38(10): 5572-5581.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of Earth's ecosystems. *Science* 277(5325): 494-499.

White, P. and L. Claxton. 2004. Mutagens in contaminated soil: a review. *Mutation Research-Reviews in Mutation Research* 567(2-3): 227-345.

Winker, M., B. Vinneras, A. Muskolus, U. Arnold, and J. Clemens. 2009. Fertiliser products from new sanitation systems: Their potential values and risks. *Bioresource technology* 100(18): 4090-4096.

Wong, N., D. Cheong, H. Yan, J. Soh, C. Ong, and A. Sia. 2003. The effects of rooftop garden on energy consumption of a commercial building in Singapore. *Energy and Buildings* 35(4): 353-364.

Woodcock, J., D. Banister, P. Edwards, A. M. Prentice, and I. Roberts. 2007. Energy and health 3 - Energy and transport. *Lancet* 370(9592): 1078-1088.

Worrell, E., L. Price, N. Martin, C. Hendriks, and L. Meida. 2001. Carbon dioxide emissions from the global cement industry. *Annual Review of Energy and the Environment* 26303-329.

Yuan, F., L. Shen, and Q. Li. 2011. Emergy analysis of the recycling options for construction and demolition waste. *Waste management (New York, N.Y.)* 31(12): 2503-2511.

Zeng, Y., K. Trauth, R. Peyton, and S. Banerji. 2005. Characterization of solid waste disposed at Columbia Sanitary Landfill in Missouri. *Waste Management & Research* 23(1): 62-71.

Zinati, G. 2002. Transition from conventional to organic farming systems: I. Challenges, recommendations, and guidelines for pest management. *Horttechnology* 12(4): 606-610.

CHAPTER 2. Urban Metabolism and Metrics of Sustainability: Measuring the Progress of Urban Ecosystems

1. Introduction

Anthropogenic consumption of natural resources has caused global changes in land cover, ecological function, and biogeochemical cycles (Botsford et al. 1997, Grimm et al. 2000, Noble and Dirzo 1997, Vitousek et al. 1997). Urban populations currently account for over half of the world's population, and are responsible for a majority of resource consumption and waste production (Grimm et al. 2008, Rees and Wackernagel 1996). As urban ecosystems expand to an expected 70% by 2050, mitigating the growing burden that global populations have on ecosystems and the environment will require a focus on restructuring urban ecosystems (United Nations Department of Economic and Social Affairs Population Division 2008). Urban planning and resource management will be essential in fulfilling the needs of our populations while decreasing their contribution to issues such as global climate change, resource shortages, and ecosystem degradation (Agudelo-Vera et al. 2011, International Panel for Climate Change 2007, Lehmann 2011, Rygaard et al. 2011). Thus, urban ecosystems are increasingly seen as both the key and the biggest challenge to achieving sustainable societies (Rees and Wackernagel 1996).

Energy and material flows within an urban ecosystem provide great insight into sustainability, as they are directly linked to the environmental impacts of resource procurement and consumption (Graedel and Allenby 2003). In typical energy and material flow models, resources are imported into an ecosystem, utilized for growth and maintenance through a variety of anthropogenic processes, and partially transformed into wastes; a process known as urban metabolism (Wolman 1965). A typical urban metabolism analysis quantifies water, food, fuel, electricity, and material resources entering the system and solid waste, waste water, and gaseous pollutant emissions produced (Baccini 1997, Barles 2009, Decker et al. 2000, Kennedy et al. 2007, Newcombe et al. 1978, Newman 1999, Ngo and Pataki 2008, Sahely et al. 2003, Warren-Rhodes and Koenig 2001, Wolman 1965). These data can be used to describe metabolic changes of single urban ecosystem over time, known as a time-series study, or to compare many urban ecosystems at a single point in time, also called a cross-sectional analysis (Kennedy et al. 2011).

By quantifying resource inputs and waste outputs at the urban boundary, urban metabolism analyses allow researchers to describe their underlying anthropogenic drivers. This knowledge can make urban metabolism a valuable tool in assessing and enhancing sustainability incorporating an understanding of resource limits, environmental burdens, and societal impacts of metabolic processes (Kennedy et al. 2011, Maclaren 1996, Wolman 1965). Despite this framework's potential for use in sustainable development, it is only beginning to be utilized in urban design and planning applications (Kennedy et al. 2011).

The limited utilization of urban metabolism analyses in urban planning, ecosystem studies, and sustainability efforts may stem from persistent issues with temporal resolution, definition of the urban boundary, and lack of clearly-defined sustainability metrics. Low temporal resolution has been an issue in most urban metabolism studies to date. Resource flows within an urban ecosystem operate on widely varying time scales. Water, for example, can be imported, utilized, and expelled from the system in a matter of days, while building materials such as timber are incorporated into its structure and slowly degrade over a lifespan of 75 years of more (Winistorfer et al. 2005). Despite these varying timescales, many metabolism studies base their findings on data from only two points in time, often 10 to 30 years apart (see: Newman 1999, Ngo and Pataki 2008, and Sahely et al. 2003) and include few intermediate data points within the study period (see: Huang and Chen 2009). With such low temporal resolution, these studies may fail to see the immediate effects of processes operating on shorter time scales, such as a new water-efficiency technology or pollution regulation.

Components of the urban ecosystem also operate at a range of spatial scales, influencing researchers' and planners' definition of the urban boundary. Processes connected to natural landscape features, such as river basins or vegetation zones, imply a very different spatial boundary than processes associated with socioeconomic features, such as highway networks and development patterns (Campbell 1996). Although city, county, and regional level analyses have often been grouped together in cross-sectional comparisons (e.g. Ngo and Pataki 2008, Sahely et al. 2003), it has been demonstrated that these various spatial scales are not comparable. Barles' (2009) study of Paris and its region confirmed that per capita consumption of resources varies dramatically between a city-level and regional-level analysis. A regional approach captures important metabolic processes, such as local extraction of minerals and other resources that would be found slightly outside the boundaries of a city-level analysis. Thus, if the goal of a study is to completely describe the metabolic activities of an urban ecosystem, a county or regional approach may be optimal, especially as the footprint of modern urban ecosystems tends to extend far beyond their metropolitan core in both physical structure and influence over resources.

Researchers of urban metabolism have often struggled to translate consumption of natural resources and environmental impacts into a meaningful assessment of sustainability. This issue could be resolved by creating a universally applicable system of quantitative metrics with which to assess and compare an urban ecosystem's sustainability. In chapter one, we used the literature on urban sustainability to identify four main themes to guide the creation of these sustainability metrics: 1) use of renewable resources, 2) dense, mixed-use development, 3) localization of resource chains and 4) closing of resource loops. Renewable resources, such as solar or wind electricity production, can reduce demand for fossil fuels, and by extension decrease combustion-related gaseous pollutant outputs (de Almeida and Silva 2011, Nel and Cooper 2009). Dense, mixed-use development can have the same effect on gaseous emission by shortening the travel distance between consumers and resources or services. Localizing resource chains can reduce transportation by bringing raw material resources and manufacturers closer together, also decreasing resource throughflow (Fagan et al. 2010, Jabareen 2006, Newman and Kenworthy 1989, Pirog et al. 2001). Closing resource loops further reduces the need for resource inputs and waste outputs through recycling or re-using waste products as primary resources in subsequent metabolic processes (Mathews and Tan 2011).

Further social, environmental, and financial benefits of these actions also make them good bases for sustainability metrics. Renewable resource alternatives and dense, mixed-use development reduce dependence on declining supplies of non-renewable resources, and can mitigate the socially, environmentally, and financially destabilizing effects of resource shortages (de Almeida and Silva 2011, Graedel and Allenby 2003, International Panel for Climate Change 2007, Nel and Cooper 2009, Parry et al. 2007, Verbruggen and Al Marchohi 2010). Urban density and mixed-use development can also have positive health and psychological effects by allowing for increased pedestrian and bicycle travel and incorporation of green spaces (Leyden 2003, Woodcock et al. 2007). Additional financial and environmental benefits are also associated with localizing resource chains and closing resource loops. Both themes signal investment into an urban ecosystem's local resource infrastructures and strengthening and diversifying of the local economy (Bagliani et al. 2010). These processes increase self-sufficiency of urban ecosystems along with accountability for undesirable byproducts and visibility of negative externalities (Lim et al. 2010, Rygaard et al. 2011, Winker et al. 2009, Yuan et al. 2011).

Our goal was to rectify the spatial and temporal issues associated with urban metabolism analyses and incorporate quantifiable sustainability metrics. First, we increased temporal resolution and defined urban boundaries at a county-level scale. We then created a system of quantifiable metrics by which the sustainability of urban ecosystems could be compared. Finally, we discussed the strengths and weaknesses of our approach and its possible impact on the field of urban sustainability.

2. Methods

2.0 Experimental Design

2.0.1 The Study Area

The Triangle is a collection of eight counties within the Piedmont geological region of North Carolina, USA. The Triangle region as a whole has been rapidly growing, with a population increase of 30% between 2000 and 2010, compared to the US national growth rate of 10% during that period (United States Census Bureau 2012). Durham, Orange, and Wake Counties, the core of the Triangle region, make for an interesting case study. These counties show a great degree of interconnectedness through organizations such as the Triangle J Council of Governments, which facilitate collaboration in economics, city planning, and resource conservation, their shared public transportation system, and academic collaborations of major universities housed in each county.

Despite their strong connections, the triangle counties can be differentiated by demographic and socio-economic characteristics. Wake County is the most highly populated county with more than 750,000 people in 2005, the end year of our study. Orange County is the smallest with only 122,000 and has a population density roughly one third of Durham and Wake. Daily commuting patterns suggest that Wake and Durham Counties are much larger employment centers, as they gain thousands of workers in daily commuting from other counties, while Orange loses over one thousand (United States Census Bureau 2012).

2.0.2 Sustainability Metrics

We conducted an urban metabolism analysis of Durham, Orange, and Wake Counties between 1985 and 2005, analyzing transportation and urban structure and flows of water, food, fuel and electricity, gaseous pollutant emissions, and solid waste. For all categories we created quantitative sustainability metrics that fell into one of two classifications: status metrics, which indicated sustainability at a single point in time, or trend metrics, which referred to the change in sustainability within the study period. We measured each urban ecosystem's level of sustainability for individual metrics on a scale of 0 to 1. A score of 0 represented the lowest level of sustainability and 1 the highest based on the urban ecosystems fulfillment of quantitative sustainability goals we defined for each metric. To compare the overall sustainability among urban ecosystems, we calculated a total "sustainability score" by summing the individual metrics (Table 1, Table 2).

2.1 Urban Structure and Transportation

We collected data for population, population density, worker travel between counties, and travel time, by means of transportation, from the United States Census Bureau (2012). Total county population data were available for the entire study period, and complete major metropolitan area population data were only available for 2000 and 2005. Data on worker travel between counties were available in 1990 and 2000 and commuter means of transportation were available for 2000 and 2005.

Our sustainability metrics in this category focused on alternative transportation, including methods such as bus, train, and other public transport, walking, and biking. Our status metric for 2005 was the percentage of total travel time in each county that commuters used alternative transportation. The sustainability goal for this metric was for all commuting in the county to rely on alternative transportation methods. Thus, if 100% of commuting time in a county was by alternative transportation, the county earned a sustainability score of 1.00. Due to a lack of data in previous years, we were not able to create a trend metric based on the change in this percentage throughout the study period.

2.2 Water

We modified a basic water balance formula from Ngo and Pataki (2008) by including surface water inflow, as we found it represented a large portion of water entering the counties:

$$\text{Available water balance } (G + \Delta S) = \text{Inputs } (P + F + I) - \text{Outputs } (E + R + W) \tag{1}$$

where G is groundwater, ΔS is change in storage, P is precipitation, F is surface water input, I is anthropogenic withdrawals, E is evapotranspiration, R is runoff, and W is wastewater (Figure 1).

We calculated water consumption using the United States Geological Survey's county-level water use data (United States Geological Survey 2012b). County-level precipitation data from the North Carolina State Climate Office (2012) were used to calculate the total volume of precipitation in each county. We calculated surface water inflow and runoff using annual average flow data from the United States Geological Survey's water monitoring stations located along major surface water bodies (United States Geological Survey 2012a). Flow data were not available for all years at some flow stations, but the major flows into and out of each county were taken into account. Flow data contained some inaccuracy due to the distance of flow monitoring stations from actual county boundaries, resulting in increased or decreased flow levels (Figure 2). This inaccuracy is both unavoidable and difficult to quantify.

Average annual evapotranspiration data were available through the State Climate Office of North Carolina (2012). Due to incomplete evapotranspiration data for Durham County, we calculated an average evapotranspiration rate for each year using annual data from climate monitoring stations in each county. We multiplied this rate over the land area of each county to reach a total value for each year (Appendix B: Table 3). The use of potential evapotranspiration, rather than actual evapotranspiration is a possible source of inaccuracy in our water balance. Potential evapotranspiration represents water lost to the atmosphere if a sufficient source of water were consistently available from land surface and vegetation, and tends to be higher than actual evapotranspiration due to limited actual water availability, microclimate effects, and other factors. In our study region, for example, high humidity in the summer months is one factor that would limit actual evapotranspiration, causing it to be much lower than potential.

Annual wastewater flow was calculated using the United States Environmental Protection Agency's National Pollutant Discharge Elimination System data for point source discharge, provided by the North Carolina Department of Environment and Natural Resources Division of Water Quality (2011)). Data for groundwater (G) and change in storage (ΔS) were not available, but were collectively used as the "balance" of the equation in order to indicate recharge or depletion of water resources.

Water importation data were not available, possibly resulting in an underestimate of total water inputs. Because it was unclear whether any county regularly imported water from outside the county, we did not include this flow in our water balance equation.

We created both a status metric and trend metric for the urban water system. Our status metric was based on the water balance equation. The sustainability goal for this metric was for each county to water inputs that were equal to or greater than water outputs, as this indicated sufficient ability of the ecosystem to retain water for anthropogenic consumption, natural processes, and recharge of ground and surface water reservoirs. We used the following formula to calculate an average water balance for each county throughout the study period:

$$1/5 \sum_{i=1985}^{2005} \text{Total water input (i) / Total water output (i)} \qquad (2)$$

Our rationale for this method was that averaging the data points should decrease the likelihood of an uncharacteristic year skewing the results. A water balance percentage of 100% indicated the highest degree sustainability for this metric.

A trend metric for this category was based on change in per capita water consumption in each county. The sustainability goal was to decrease per capita water consumption. We defined the sustainability goal as a 100% decrease in water consumption over the 20-year study. Although a full 100% decrease to 0 m^3/cap is not logically obtainable, it served as an absolute lower limit to consumption, allowing for the calculation of a percent decrease.

2.3 Food

Per capita food inputs, in total weight of food, were estimated using food availability data produced by the United States Department of Agriculture's Economic Research Service (2012), which served as a proxy for actual food consumption. The food availability calculations included spoilage, which is necessary as it represents a portion of food inputs for any urban ecosystem.

Data on food produced locally for direct consumption and total value of agricultural sales in thousands of dollars were available in five-year increments between 1992 through 2007 through the United States Department of Agriculture's National Agricultural Statistics Service's Census of Agriculture reports. These data refer to consumable foods that are produced in a county and sold in roadside stands, farmers markets, and "pick-your-own" sites. Because the location of consumption was not specified and individual criteria for "local" may differ, we assumed that these sales venues generally included food produced in close proximity. Some farmers markets in the study area required vendors from within state boundaries, but others restricted vendor participation to as low as a 50 mile radius. Therefore, our data on local food for each county refers to food produced within the study area and sold within the state of North Carolina.

We used data on total county-level food sales from the North Carolina Department of Revenue's State Sales and Use Tax Statistics reports for additional food system analyses.

Our status metric for the food category quantified the value of locally produced food sales as a percentage of total food sales in the state. Achieving 100% local food sales within a county was our high sustainability benchmark. While fully achieving this goal may be currently unobtainable for most urbanized systems, it can serve as an example of diversifying the local economy, reducing resource throughflow, increasing self-sufficiency, and greater visibility and accountability for food production practices. Our trend metric was based on the percent increase of local food sales over time, with a goal of 100% increase over the study period.

2.4 Fuel and Electricity

We collected annual energy consumption data for residential, commercial, industrial, and transportation sectors in each county as BTU's from the United States Energy Information Administration and converted them into Joules of energy. Downscaling these sector-level data from the state- to county-level relied on methods previously used by Ngo and Pataki (2008). We downscaled residential and transportation consumption using the ratio of county populations to North Carolina state population, and industrial and commercial sector consumption using a ratio of the number of people employed in these sectors in each county to the entire state's employment in these sectors (Appendix A: table 2). Our use of employment ratios to downscale the industrial and commercial sector consumption was a means of quantifying each county's relative contribution to the entire states industrial and commercial energy consumption. Since county employment data were only available for 1990 and 2000, we estimated employment levels in 1985, 1995, and 2005 based on the rate of change of employment in each sector between 1990 and 2000.

Data on fuel importation for energy production were not included in our dataset. While we were able to locate an estimate of average annual energy production for Wake County's nuclear power plant through the United States Energy Information Administration, data were not available for University of North Carolina at Chapel Hill's cogeneration facility in Orange County; the only other power plant in our study area.

Our two sustainability metrics in this category focused on renewable fuels and overall energy consumption. We created a status metric for renewable fuels based on biomass and fuel ethanol consumption. Our sustainability goal was the status metric was for 100% of fuels consumed to be renewable. This status metric focused only on fuels and not imported electricity because local renewable energy data was not available. Our trend metric focused on change in per capita energy consumption, with a goal of 100% decrease.

2.5 Emissions

State-level CO_2 emissions data for 1985 to 2005 were available through the United States Environmental Protection Agency (2012b), and were downscaled using the same methods as our energy consumption estimates. We downscaled sector-specific emissions using county-to state-level population ratios for residential and transportation sectors and employment ratios for commercial and industrial sectors.

We compared our CO_2 emissions estimates to Purdue University's Vulcan Project database, a dataset incorporating high-resolution data such as local and regional air quality monitoring, United States Census, and road travel data in spatial scales of less than 100 km^2 (Gurney et al. 2009). This database's single temporal data point in 2002 made it unfeasible to use as a source for our analysis, but the comparison allowed us to assess the validity our estimation methods.

County-level pollutant emissions data were available through United States Environmental Protection Agency's AirData database for years 1990, 2000, and 2008 (United States Environmental Protection Agency 2012a). We estimated pollutant emissions for 1985 and 1995 using the rate of change between 1990 and 2000, and emissions for 2005 using the rate of change between 2000 and 2008.

Our trend metric for sustainability in this category was change in total per capita pollutant emissions, for which our sustainability goal was 100% reduction.

2.6 Solid waste

Solid waste production data were available in North Carolina Department of Environment and Natural Resources county solid waste reports (North Carolina Department of Environment and Natural Resources Division of Waste Mangement 2012). Construction and demolition (C&D) waste data and detailed data on solid waste movement among counties were only available for 2000 and 2005. We used the following solid waste balance equation to check the completeness of solid waste data:

$$\text{Waste Landfilled} = \text{Total Waste Produced} + \text{Waste Imported} - \text{Waste Exported} \tag{3}$$

Only Orange and Wake County’s 2005 levels of solid waste recycling were available from county solid waste management plans. We estimated Durham County’s 2005 recycling levels based on the change between its 2004 and 2007 levels (North Carolina Department of Environment and Natural Resources Division of Waste Mangement 2012).

We created a status and trend metric for solid waste based on percent of solid waste recycled and change in total solid waste output. Our status metric goal was 100% recycling of solid waste produced in the county. We quantified the change in per capita solid waste production to score the trend metric, with a 100% decrease as the sustainability goal.

3. Results

3.1 Urban Structure and Transportation

We found Orange County to be unique in urban structure, having the lowest county population density, 118 people/km^2, but the highest metropolitan density at 989 people/km^2 in 2005. Durham and Wake counties were similar in county population density, with 322 and 351 people/km^2, respectively.

Orange County also had the largest percentage of alternative transportation commuting time in 2005 at 9.3%, earning the highest score for this sustainability metric. Durham County was close behind in alternative transportation commuting time, with 8.7% (Table 2). Additional data on commuting patterns showed that Durham County gained over 53,000 workers in daily commuting, and Wake over 21,000. Orange County experienced a net loss of 1,711 workers daily (Figure 5).

3.2 Water

Our water balance calculations showed notable variations among counties (Appendix B: Table 1). Durham experienced a positive balance of water resources in all years except 2005, while Orange and Wake showed a water balance deficit for all years (Figure 7). Durham had an average water balance greater than 1.0 for the study period, earning it a score of 1 for the water balance metric. Orange had a small deficit in average water balance and Wake a slightly larger deficit, earning 0.91 and 0.84 points, respectively (Table 2).

There was also a visible distinction between counties in the trend metric dealing with water consumption. Orange County had the lowest per capita water consumption throughout the study period and Durham the highest. Wake County added 45 million m^3 in 1990 due to power generation, accounting for a spike of over 100 m^3 per capita and contributing to its 3% increase in per capita water consumption (Appendix B: Table 2). Durham County showed the largest decrease of 7.9% in per capita water consumption, earning 0.079 points. Wake increased in consumption, earning no points for this category (Table 2).

3.3 Food

Our estimate of per capita food consumption increased by 8.4% during the twenty-year study period, from 0.69 tons in 1985 to 0.74 in 2005. The direct sales of locally harvested crops increased in all counties from 1992 to 2007, with the greatest increase in Orange County (Figure 8). Less than 0.1% of total food sales in each county were from local sources (Figure 9), resulting in a negligible contribution of points to both the status and trend metrics (Table 2). Of the total value of agricultural production within each county in 2007, only 2.3% was sold locally in Orange County, and 1.1% in Wake.

3.4 Fuel and Electricity

Trends in fuel consumption suggested a shift in all counties from coal to other sources of energy. Per capita coal consumption decreased in all counties by over 40% and per capita retail electricity consumption increased in all counties. Durham and Wake Counties increased in per capita natural gas consumption by 4.1% and 26.1%, respectively (Appendix C: Table 1,2,3).

We found the lowest per capita consumption of petroleum fuel in Orange County, at 111 GJ, and it was the only county to decrease in per capita consumption (Appendix C: Tables 1, 2, 3). Wake County had the largest proportion of renewable fuels, with 5.7% of their total energy from ethanol and biomass consumption, earning the county 0.057 points (Table 2). Durham followed closely with 5.1% of its fuel consumption from renewable sources (Figure 10).

Orange County decreased by 2.7% in per capita energy consumption (Figure 11), a score of 0.027, while Durham and Wake Counties both increased (Table 2). Wake County's nuclear power plant produced an estimated 27 thousand terajoules of energy annually (United States Energy Information Administration 2011), the equivalent of 60% of Wake's total 2005 electricity demand.

3.5 Emissions

Per capita CO_2 emissions decreased by 7% in Orange County, while Wake increased by 3.2% (Figure 12). Per capita emissions of other pollutants were very similar among Durham, Orange, and Wake Counties, and decreased universally with a few exceptions. Orange County maintained higher per capita of NH_3 emissions than Durham and Wake and began the study period with much higher per capita SO_2 emissions, but decreased to similar levels of SO_2 by the end of the study period (Figure 13) (Appendix D: Table 1,2). Total gaseous pollutant emissions decreased by 10.6% in Orange County, contributing 0.106 points to its overall sustainability score (Table 2).

We found the Vulcan Project's county-level CO_2 estimates for 2002 to be higher than our downscaled state-level estimate, especially in Durham County (Figure 14). Per capita transportation emissions in this dataset were the lowest in Orange County, with 7.3 metric tons, while Durham and Wake both produced 7.6 (Appendix D: Table 3).

3.6 Solid waste

Solid waste recycling estimates showed that Orange and Wake Counties recycled 13.6% and 9.8% of their solid waste, respectively, while Durham recycled only 1.4% (Figure 15). Thus, Orange county earned the most points for the solid waste recycling metric, with 0.14 (Table 2).

The solid waste equation was balanced for all counties except Durham in 2000, for which 396 metric tons of waste was unaccounted. Per capita production of solid waste decreased in all counties throughout the study period, but most dramatically in Durham County with a 16% reduction (Figure 16). Thus, all counties made progress in reducing per capita solid waste production, with Durham County earning the most points in this metric (Table 2). Construction and demolition waste differed greatly among the counties. Orange County decreased in per capita construction and demolition waste from 2000 to 2005, while Durham and Wake increased (Figure 17).

4. Discussion

4.1 Urban Structure and Transportation

Our analysis of transportation characteristics and urban structure led to discoveries about their effects on metabolism and sustainability of the counties. Orange County had the highest percentage of alternative transportation, and the highest metropolitan density, which agrees with the popular hypothesis that a positive correlation exists between urban density and alternative transportation (Chen et al. 2011, Jabareen 2006, Liu and Shen 2011). Thus, Orange County's planning and urban development policies that led to higher population density appear to have positively affected the county's sustainability by encouraging alternative transportation.

Counties with denser metropolitan areas and greater use of alternative transportation have the potential to decrease their metabolism of transportation-related fuels and production of gaseous pollutant emissions. In fact, the Vulcan dataset showed that Orange County, the densest metropolitan area with the greatest use of public transportation, had the lowest transportation-related emissions of CO_2 in 2002. Unfortunately, the other consumption and emissions data that were available could not be used to analyze the relationship between density and transportation emissions. This was because transportation fuel consumption and pollutant outputs data were only available at the state scale, requiring us to downscale by population. This method created the assumption of equal per capita consumption across the region. Although, vehicle miles traveled would have been an ideal dataset by which to downscale, it was not available at the county level. For future metabolism analyses to be useful in a comparative framework, county or city level data will be necessary. Despite this data availability issue, our sustainability metrics suggested that Orange County was effective in increasing its sustainability through greater use of alternative transportation.

4.2 Water

The water balance analyses illustrated the importance of placing metabolic resource demands into context with resource supplies when assessing sustainability. Although Durham had the highest per capita water consumption, it showed an overall surplus of water resources. This result suggested that Durham County's demands for water were not depleting its existing supply at the current rate of metabolism. Thus, our sustainability metrics were able to identify a sustainable water system in Durham County. Orange and Wake Counties were found to deplete their water resources, suggesting that their level of metabolism was not sustainable.

These findings could be partially explained by the position of each county in the Neuse watershed. Orange County is located at the headwaters of the Neuse River watershed, and the majority of its surface water runs into Durham County through the Neuse River and other smaller water bodies (Figure 2). Thus, Durham County's available water supplies were regularly supplemented by surface water runoff from Orange, which was forced to rely solely on precipitation.

This case highlights the importance of a larger-scale approach toward sustainable water systems. Attempts to increase the water resources of each county should consider effects on neighboring counties further down the watershed. For example, Orange County may benefit from capturing more precipitation inputs, but in doing so could decrease surface water flow to Durham and Wake Counties. Thus, sustainably managing water resources depends on a watershed approach. Managers must also consider the effects of global climate change, ecological sensitivity to hydrological changes, and water quality throughout the watershed (Baron et al. 2002).

The increased temporal resolution of our analysis made it possible to discover some of the metabolic effects of establishing Wake County's nuclear power plant. Wake's sudden increase in water consumption between 1985 and 1990 coincided with its power plant's initiation in 1986. Nuclear power plants consume more water for cooling purposes than all other major electricity production technologies, and non-thermal renewable technologies such as wind and photovoltaic consume the least (Macknick et al. 2011).

While the entire region benefited from electricity produced in this power plant, its high levels of water consumption only affected Wake County's already strained water resources. This example of a shared resource flow creating an unequal burden of resource demands is a case for the importance of comparing urban ecosystems on a regional basis. Regional comparisons can aid in understanding the larger scale effects of sustainability initiatives taking place in a single urban ecosystem. For example, replacing Wake County's nuclear plant with non-thermal renewable electricity production could lessen the burden on its water resources, but may put strain on the electricity grid to meet the regions demands without other counties investing in their own energy infrastructure.

4.3 Food

A strong local food infrastructure may decrease an urban ecosystem's fuel consumption and gaseous emissions related to transportation of food products (Cowell and Parkinson 2003, Mariola 2008). Furthermore, local food economy would decrease metabolism by reducing food imports and solid waste production. Local food represented too small a percentage of total food sales in all counties to cause any noticeable effect on metabolism. This result suggests that local food infrastructure should be a major concern for future sustainability efforts in this region.

Because the majority of food produced in each county was being exported, it stands to reason that some of this food may be diverted into local markets to increase the local food economy. However, completely meeting the urban ecosystems' food demands would require expanding and diversifying agricultural production. Orange County's lower population density and level of urbanization may give it more potential to expand agriculture to peri-urban and rural areas. Durham and Wake Counties, however, are more populated and sprawling, and may need to explore urban farming and community gardening to further increase their local food production.

4.4 Electricity and Fuel

Orange County's success in decreasing its per capita energy consumption makes it a valuable case study for other urban ecosystems in efforts to control their metabolism of fuel and electricity. The linkage between Orange County's metropolitan density, alternative transportation, and transportation emissions demonstrates how urban structure and transportation can influence the metabolism of transportation fuels. Increasing alternative transportation through investment in public transport systems and bicycle and pedestrian corridors may reduce per capita demand for transportation fuels within a matter of years. Metropolitan density, on the other hand, can only be incorporated through long-term urban planning. Despite Orange County's successes in transportation and emissions, our metrics indicate that all counties have much more progress to make in meeting sustainability goals.

Along with this necessary decrease in the metabolism of transportation-related resources, we also need to consider energy sources. Only a small fraction of the energy consumed in each county was from renewable fuels, illustrating the unsustainable nature of these resource flows. Due to data availability, our study did not include renewable energy sources such as biodiesel fuel, biogas from solid waste landfills, and electricity from private, non-utility solar panels and windmills. Future analyses should attempt to account for these additional sources in order to create a sustainability metric that extends beyond fuels to all sources of renewable energy.

4.5 Emissions

The nearly universal decreases we discovered in per capita pollutant emissions suggest that considerable metabolic changes took place throughout the study period. One possibility is that enhanced air pollution regulations, such as the 1990 amendments to the Clean Air Act dealing with vehicle and fuel emissions, may have helped to control pollutant emissions. Another explanation may lie in the decreases of per capita coal consumption and apparent shift to natural gas and retail electricity. Natural gas is a much cleaner energy source than coal, which produces large amounts of CO_2 and nitrous and sulfur oxides. Increased use of retail electricity, by relocating pollutant emissions to where energy is produced, would decrease emissions on a local scale but would have very little effect on the global environmental burden. Therefore, emissions reductions experienced by switching to retail electricity are not a true benefit for sustainability in global terms.

The lack of agreement between our estimates of CO_2 emissions and the Vulcan Project's could suggest inaccuracy in our emissions dataset or the downscale methods we employed. Since the Vulcan Project data were collected at a higher resolution and from a more diverse set of data sources, it is possible that our dataset fails to account for all actual CO_2 sources. Our consistently lower estimates would support this theory. Errors associated with our methods of downscale from state to county level could easily have the same effect. Although our downscaled methods have been used in previous studies, we did not find any other attempts to validate these methods through comparisons to higher-resolution datasets. Despite the comparability issues between emissions datasets, the Vulcan Project data was able to support a connection between alternative transportation and CO_2 emissions, as previously discussed.

4.6 Solid Waste

Solid waste management divisions in all counties appear to be having success in reducing the outputs of solid waste and increasing recycling rates. This trend represents a decrease in material throughflow and based on our metrics, an increase in sustainability of the three counties. The increased recycling rates probably account for some reduction in solid waste output, but the lack of a noticeable quantitative relationship between these two measurements suggests that other unknown metabolic factors have a greater influence.

Although we did not see a correlation between per capita solid waste production and construction and demolition waste, these data are useful in deducing trends in overall flows of construction materials. The increase of construction and demolition waste outputs seen in Durham and Wake indicate increased construction activities and larger inflows of construction materials by association. Although these inferences give us some information regarding material flows, actual data on material inputs would allow us to further discuss metabolism related to the urban structure. These data could also serve as the basis for a metric of material consumption and help us analyze the energy content of the built structure of an urban ecosystem.

4.6 Benefits of Experimental Design

Analyzing the changes in metabolism over four, five-year increments enriched our study by providing a higher resolution view of metabolic changes. Our analysis of the water cycle, for example, illustrated that variation in weather from year to year had a significant impact on the water cycle of each county. Higher temporal resolution also allowed us to pinpoint the link between Wake's increase in water consumption and the initiation of its nuclear power plant. Since our water balance sustainability metric relied on an average balance throughout the study period, the greater number of data points available theoretically reduced the impact of any one year with abnormal weather on this average.

Our decision to define the boundary of the urban ecosystem at the county level was beneficial because it allowed us to include important metabolic processes, increase data availability, and improve the overall sustainability assessment. Important components of the urban ecosystem, such as extraction points for ground and surface water sources, were located almost exclusively outside the major metropolitan boundaries, making the metropolitan areas dependent upon the rural surroundings to fulfill their water demands (Figure 18). Such situations are nearly universal in developed countries, as cities are estimated to require a productive land area of up to hundreds of times their own size to fulfill their basic demands for water, food, fuel, and other resources (Rees and Wackernagel 1996, Folke et al. 1997). Therefore, any method of measuring urban sustainability would be biased away from sustainability and self-sufficiency in a city-level analysis.

Additionally, a city-scale urban metabolism analysis in the United States would likely be subject to extreme data limitations. Our study and others (see: Pataki et 2008, Sahely et al. 2003, Wolman 1965) show that scale-appropriate data availability is frequently an issue, making downscaling from state and national databases or up-scaling from average per capita values an unfortunate necessity. With city-level datasets even less common, researchers would be forced to either first establish a system of collecting data over time, or rely almost solely on heavily downscaled data.

4.7 Applying Sustainability Metrics

The utility of our metrics lies in what they reveal about the sustainability of urban metabolic flows, their repeatability, and real-world application. Our metrics made it possible to quantify the contribution of each metabolic component to the urban ecosystem's overall sustainability. For example, we were able to determine that Durham County was ahead of Orange and Wake in use of water balance, reduction of solid waste, and reduction of water consumption, and that Orange County was the strongest in alternative transportation, solid waste recycling, and reducing gaseous pollutant emissions. This type of information could allow governments and decision-makers to prioritize sustainability initiatives based on the specific needs of their urban ecosystem, and to allocate resources to solutions that are proven to have a measurable impact.

For these methods to be easily repeatable and yield comparable analyses, the sustainability metrics must be built upon broadly available data. The state and national datasets we utilized are widely available in the United States, and similar data for European countries have been obtained through the "Eurostat" method used by Barles et al. (2009) and others. The downscale techniques we used for some datasets also rely on widely available data, and have been deemed acceptable by other researchers.

One real-world application of the sustainability metrics is the possibility of creating standardized national and international rankings, which may stimulate competitive interest in making urban areas more sustainable. In our analysis, Durham County received the highest sustainability score of 1.47, while Orange was very close behind with 1.45 and Wake trailed behind with 1.08. With the highest possible score being a 10, however, it is clear that all three counties could use many of our suggestions for increasing the sustainability of each individual category to improve their sustainability score.

4.8 Comparison to Previous Studies

Past urban metabolism studies have focused primarily on per capita values in comparative cross-sectional analyses or in time-series studies of a single urban ecosystem. Our approach has achieved a greater depth of analysis by combining per capita values and other sustainability indicators into a comparative study that includes both cross-sectional and time-series elements. Therefore, we were able to add our study area to the existing body of per capita comparisons and create a new set of comparisons based on our systems of sustainability metrics.

The population and density of the counties in our study area were much lower than other urban ecosystems compiled by Newman et al. (2007) and Ngo and Pataki (2008). While most per capita output values for our study area fell somewhere between those of other urban ecosystems, the input categories stood out as a cause for concern. Wake County had the highest per capita water consumption, followed by Durham, Los Angeles County, and Orange County, respectively. Durham, Orange, and Wake Counties also consumed more energy per capita than all other urban ecosystems, with Los Angeles County close behind. These observations seem to evidence the inverse relationship between population density and the efficiency of resource flows.

To demonstrate the value of a new set of urban ecosystem comparisons based on sustainability metrics, we applied the metrics to another previously studied urban ecosystem and compared it to the counties in our study area. Hong Kong was the most acceptable for this comparison, as studies by Newcombe et al. (1978) and Warren-Rhodes and Koenig (2001) provided more compatible data than other urban metabolism studies available (Table 4). Despite a lack of data on alternative transportation, water balance, and renewable energy, we were able to determine that Hong Kong has made more progress than Durham, Orange, and Wake Counties in reducing gaseous pollutant emissions, recycling solid waste, and producing local food (Figure 19).

Hong Kong's large decreases in gaseous pollutant emissions could be connected to its extremely high urban density compared to the counties in our study area. This high density may translate to less personal vehicle travel and lower per capita fuel consumption, as demonstrated by our study and others (see: Kennedy et al. 2007). Warren-Rhodes and Koenig (2001) point out the efficiency of this dense urban form in terms of per capita energy demands, which were only 71 GJ/cap in 1997; less than one quarter the demand in Durham, Orange, or Wake Counties.

Although local food production decreased between 1971 and 1997, Hong Kong continued to fulfill much more of their food demand with local production than the counties in our study area. This observation is difficult to explain, as the researchers of Hong Kong do not explicitly discuss local food in their analysis, but the answer may lie in cultural differences.

5. Conclusion

We conducted an urban metabolism analysis using methods that addressed both the temporal and spatial shortcomings present in previous analyses. These improvements allowed us to identify more detailed metabolic trends and engage in a deeper discussion of their local and regional impacts. We also augmented our urban metabolism analysis by creating a system of sustainability metrics that are an appropriate diagnostic tool for assessing progress toward sustainability. Finally, our comparison of metabolic flows between urban ecosystems has added to the body of urban metabolism literature and presented insights into urban sustainability. Our intent is for this research to provide inspiration for planners, researchers, and decision-makers to use urban metabolism as a tool for enhancing the sustainability of our global urban ecosystems.

Figures and Tables

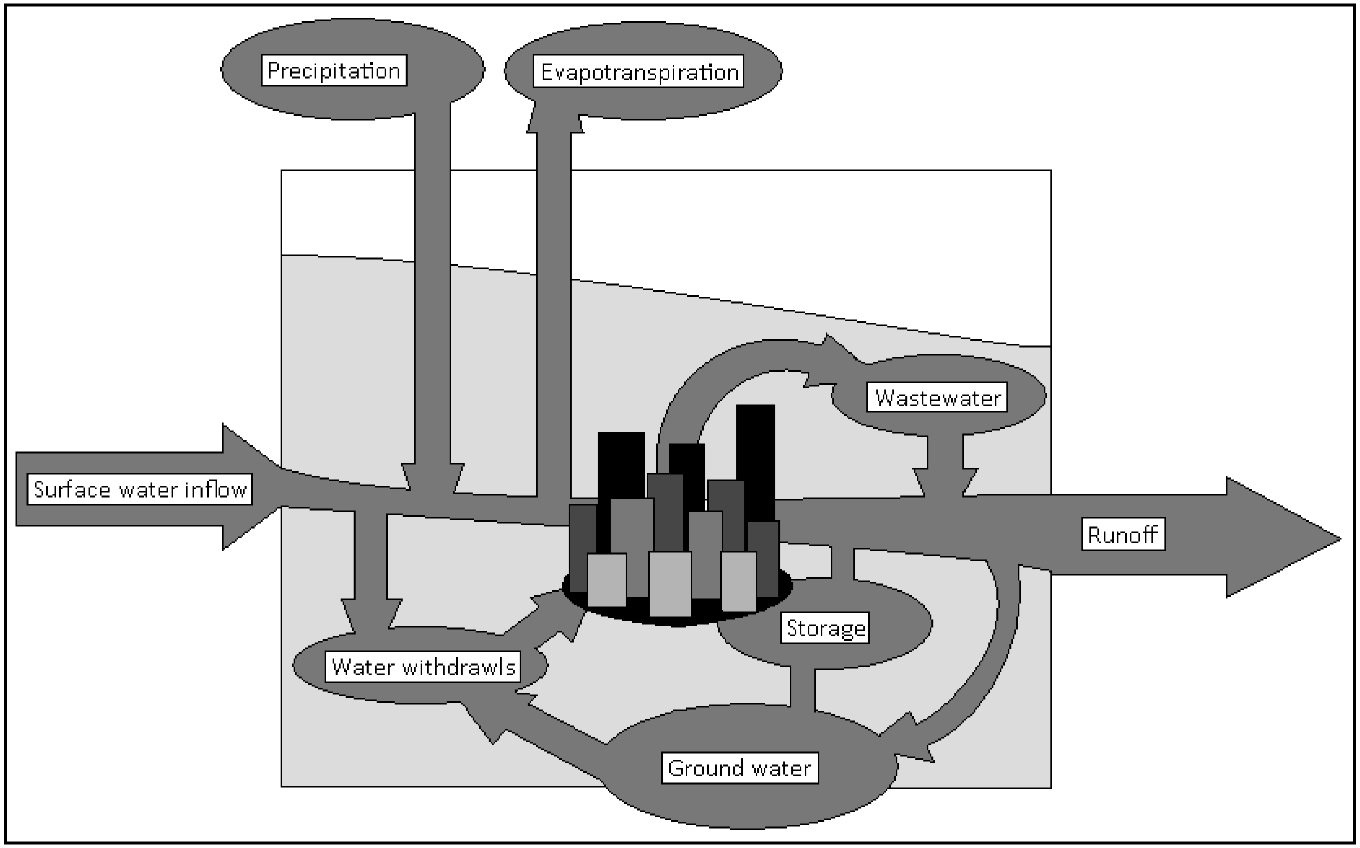


Figure 1. Flow diagram of the components included in the water balance

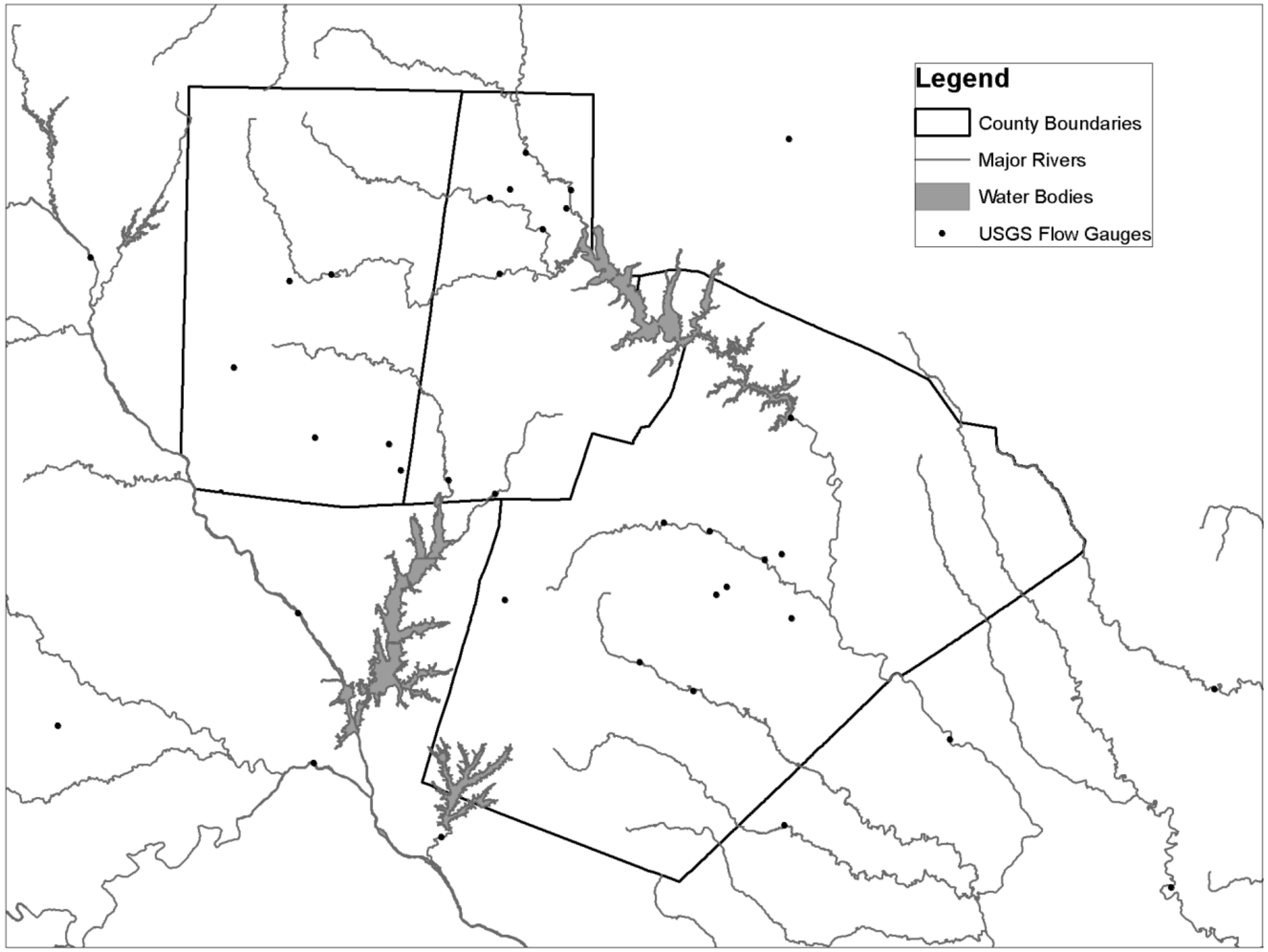


Figure 2. Location of USGS surface water flow monitoring stations relative to county boundaries

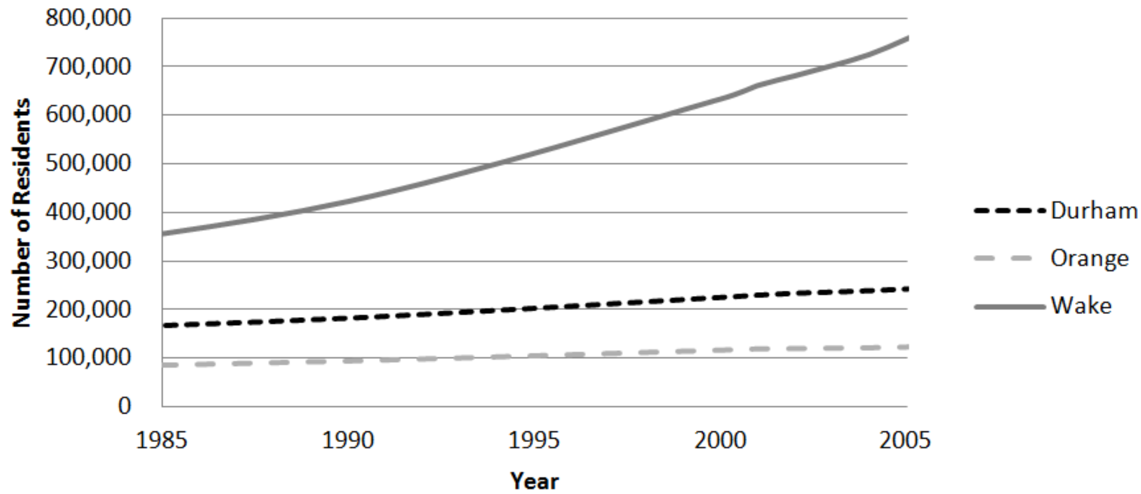
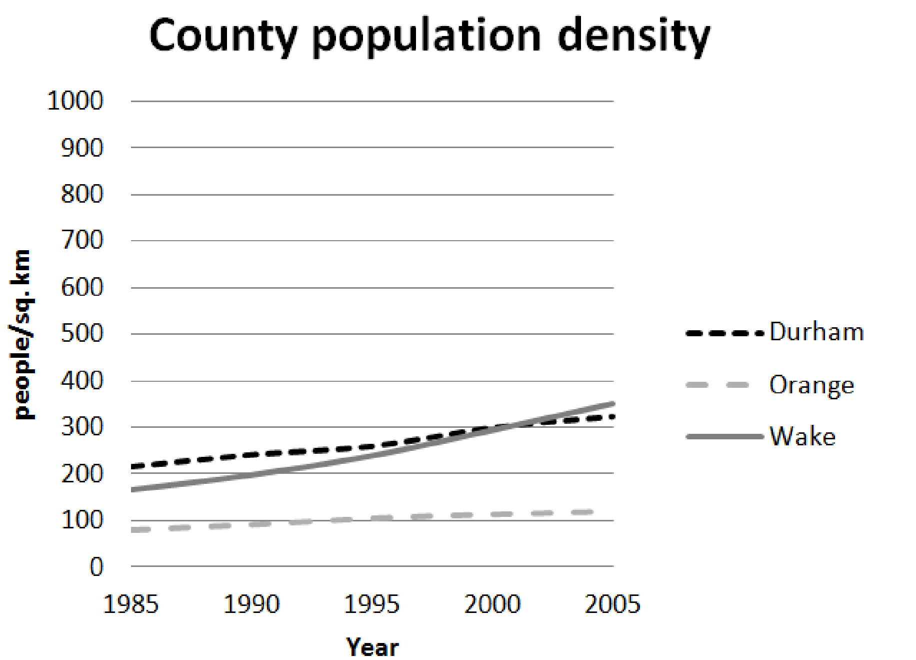


Figure 3. Change in county population



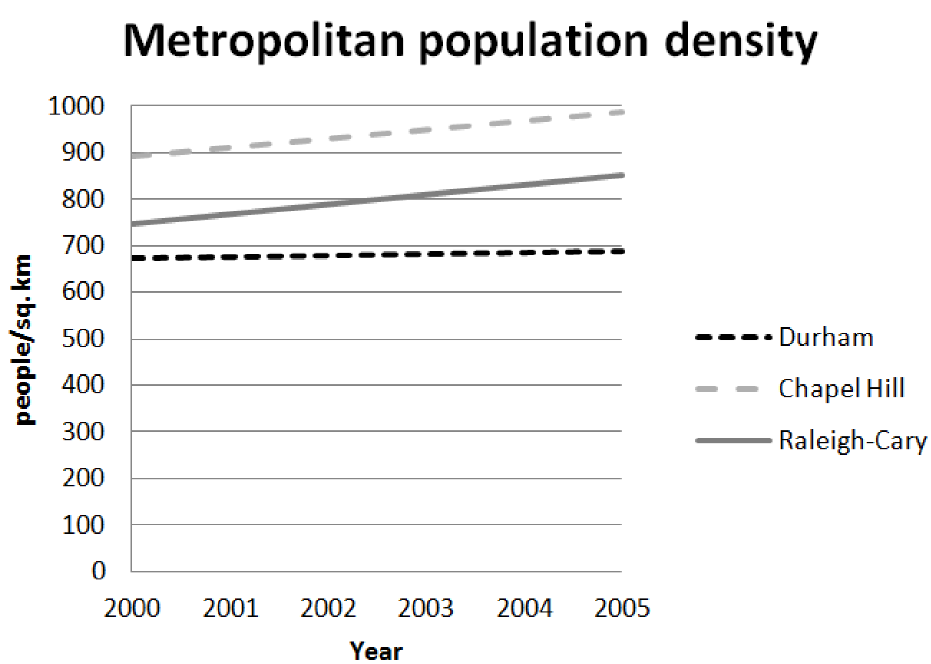


Figure 4. County and major metropolitan area population density

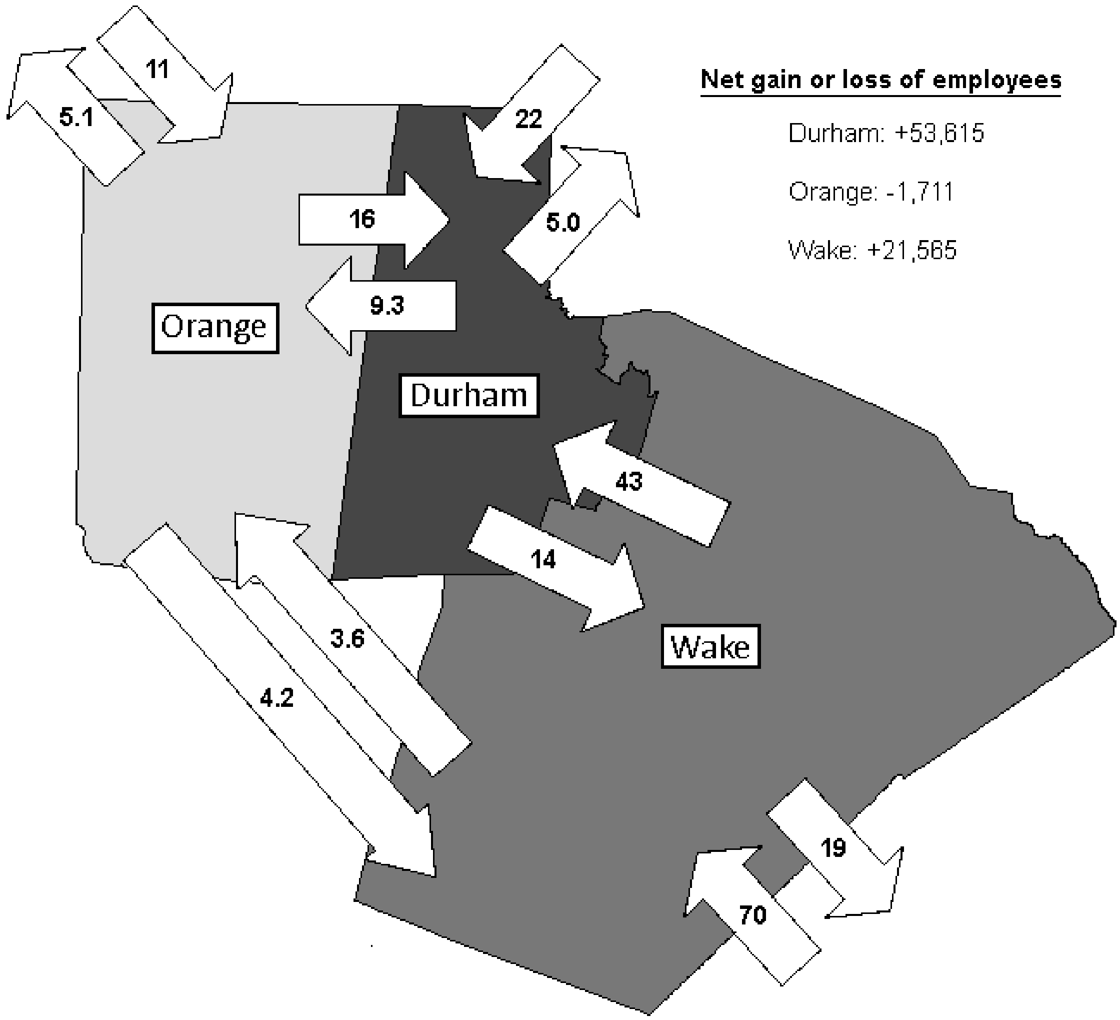


Figure 5. Flow of workers among triangle counties and other locations (thousand workers)

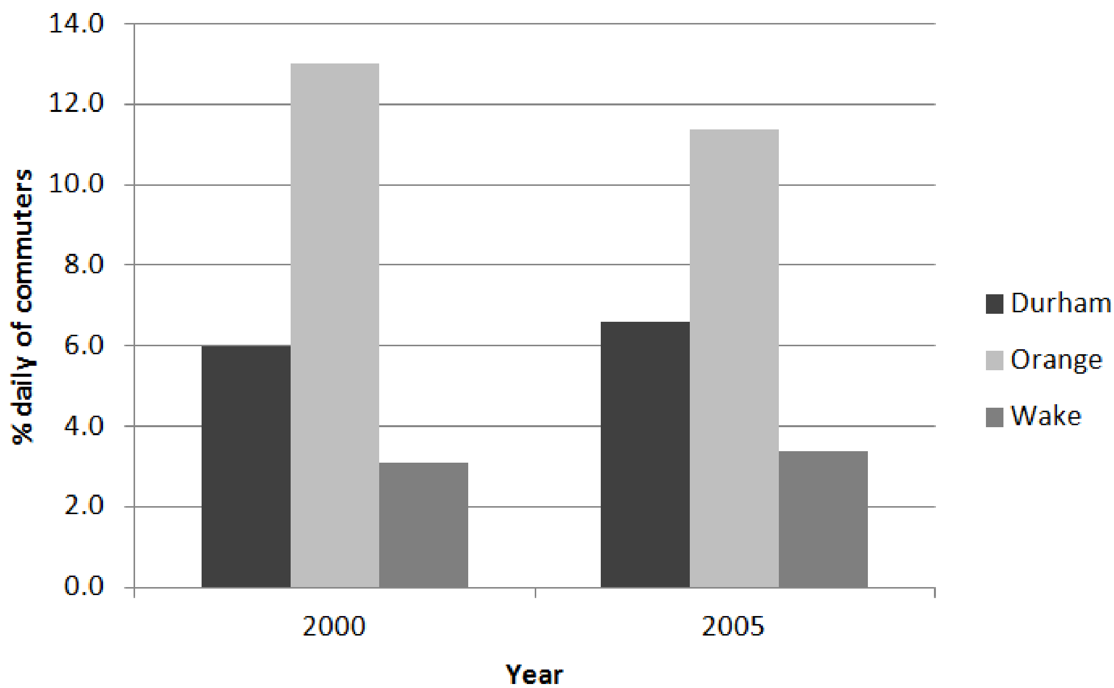


Figure 6. Percentage of daily commuters using alternative transportation in 2000 and 2005

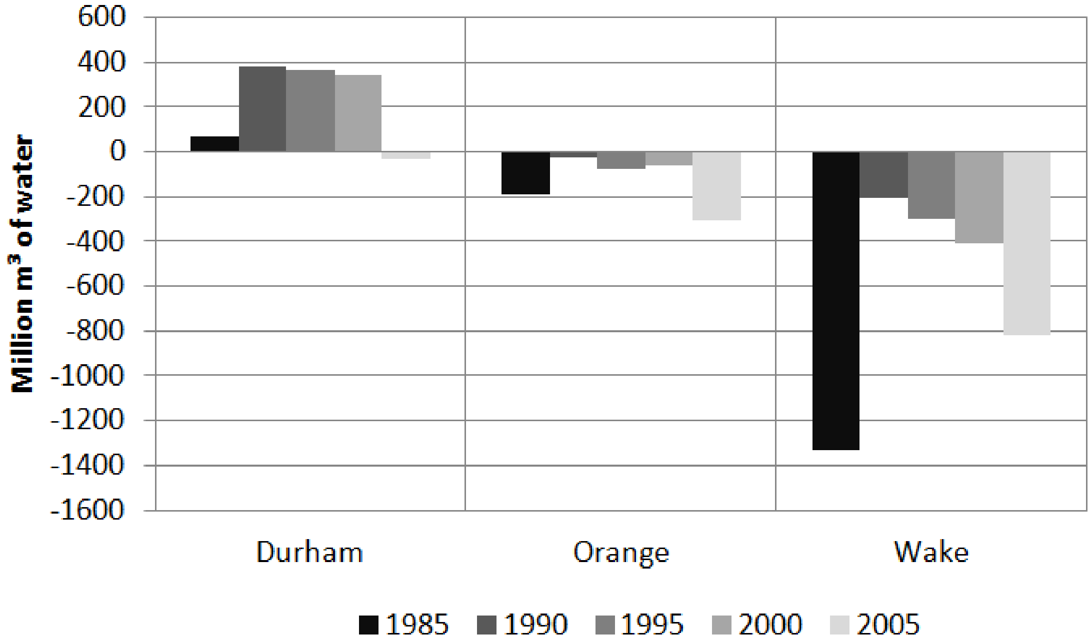


Figure 7. Results of water balance equation for each year, showing net surplus as a positive value and deficit as a negative value (Total inputs – Total Outputs)

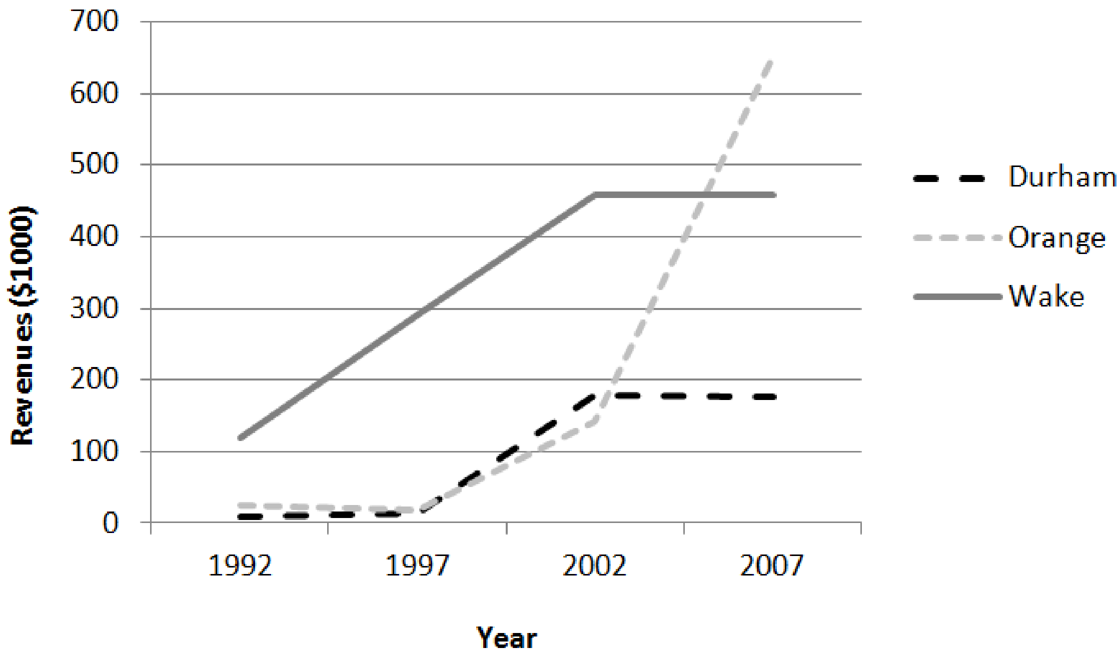


Figure 8. Total revenues from local food in each county

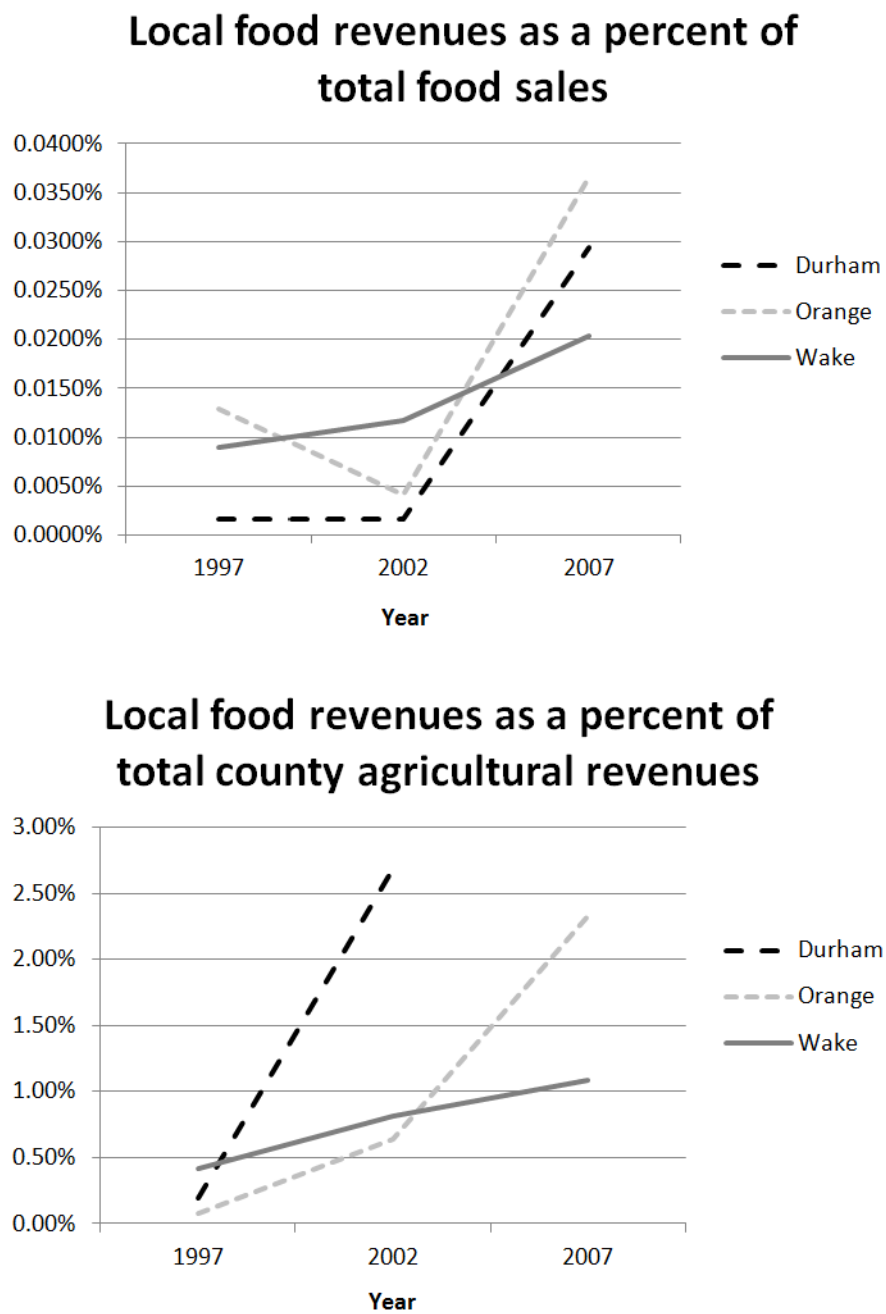


Figure 9. Local food revenues as a percentage of total food sales and total agricultural revenues in each county

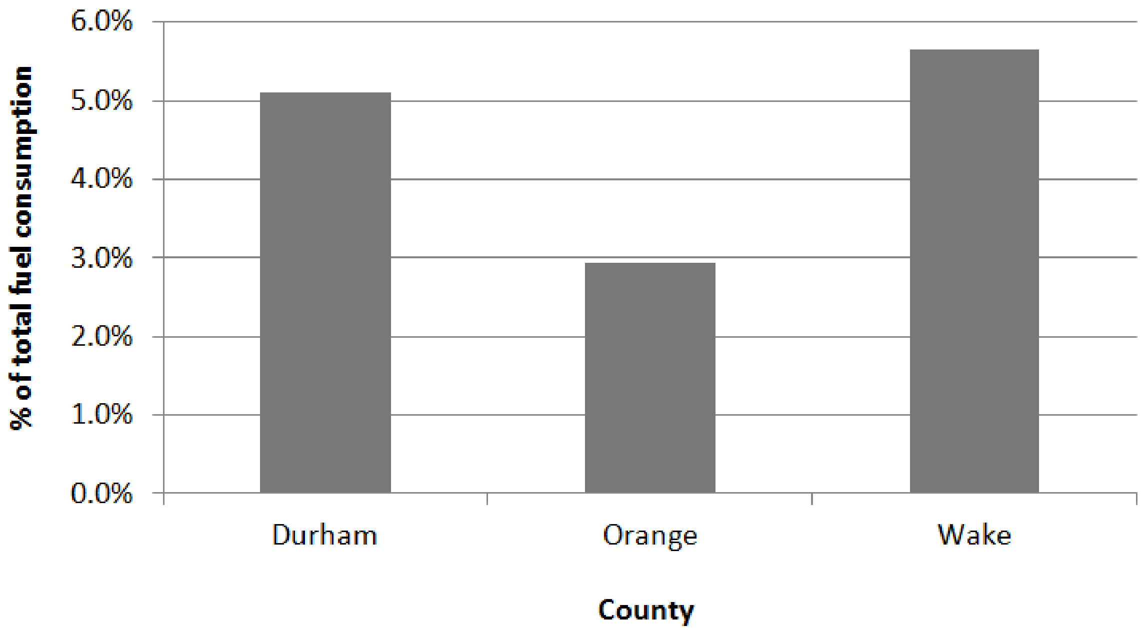


Figure 10. Biomass and ethanol energy consumption as a percent of total fuel consumption in each county in 2005

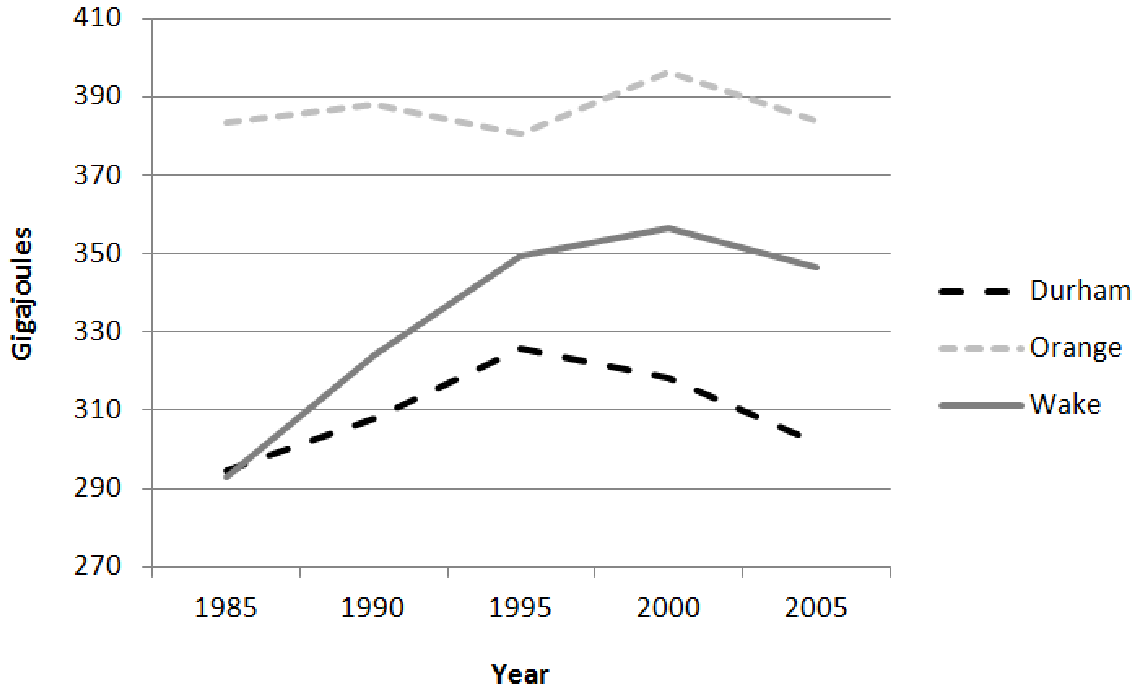


Figure 11. Change in total per capita energy consumption

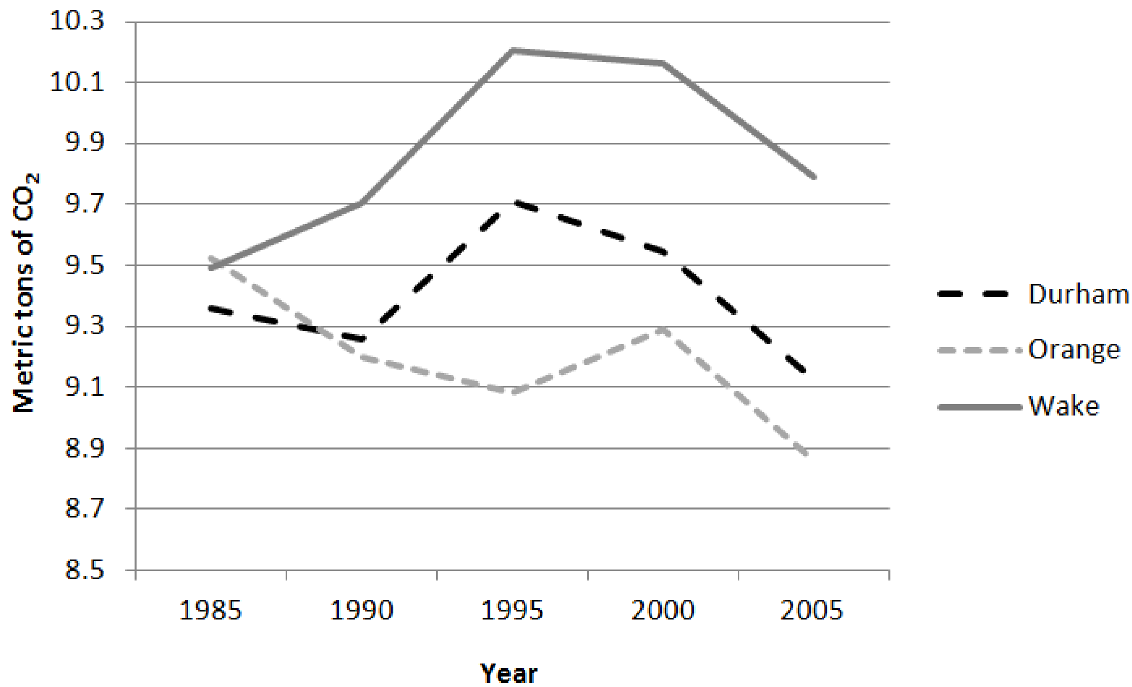


Figure 12. Change in per capita CO_2 emissions

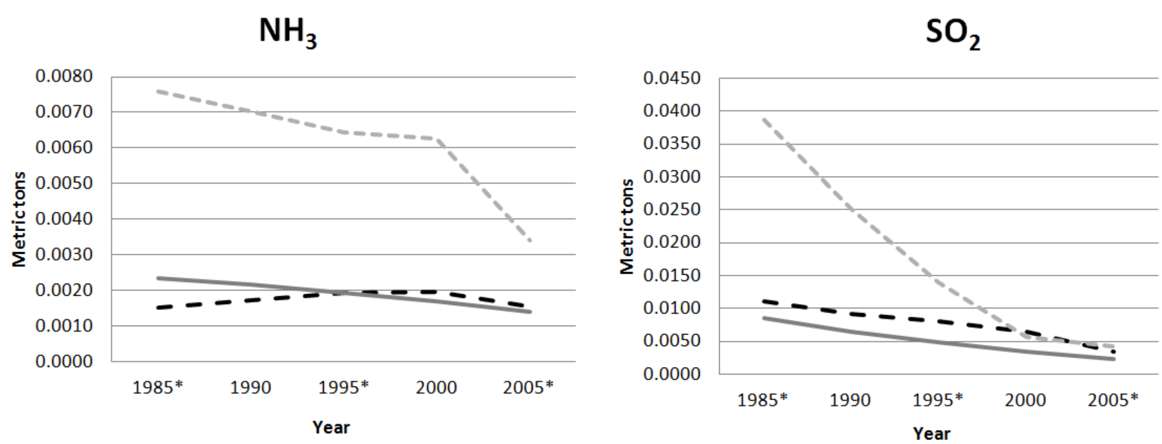


Figure 13. Change in per capita NH_3 and SO_2 emissions

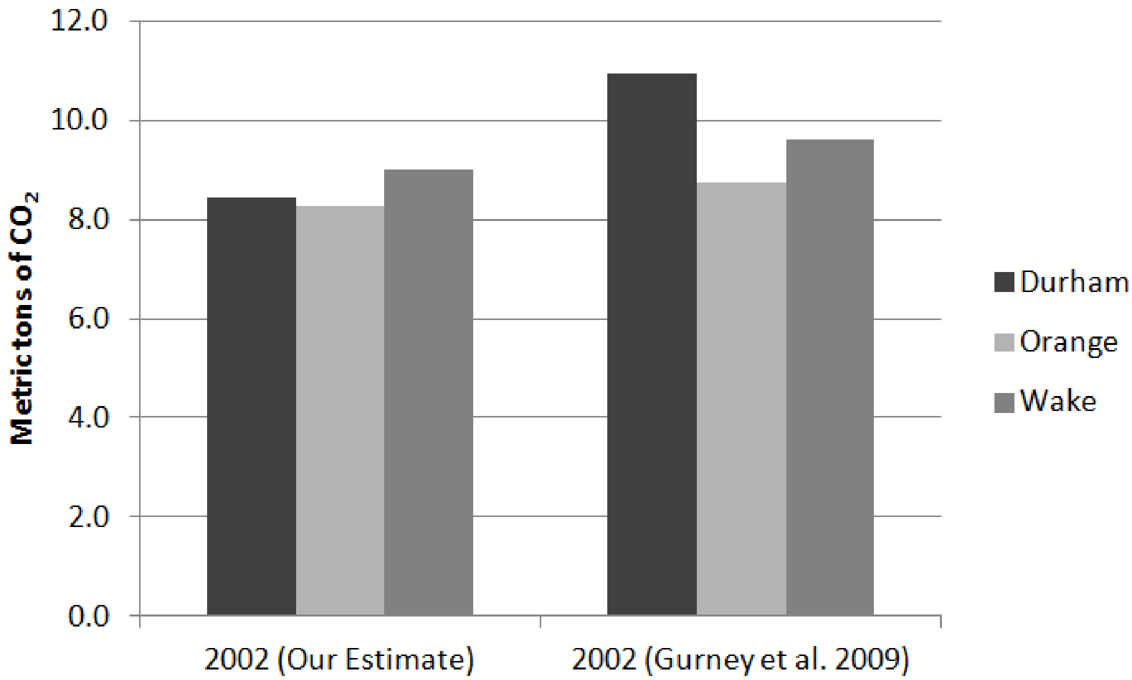


Figure 14. Comparison between data obtained from the Vulcan Project county-level CO2 database (2002) and data downscaled from United States Environmental Protection Agency's state-level CO2 database

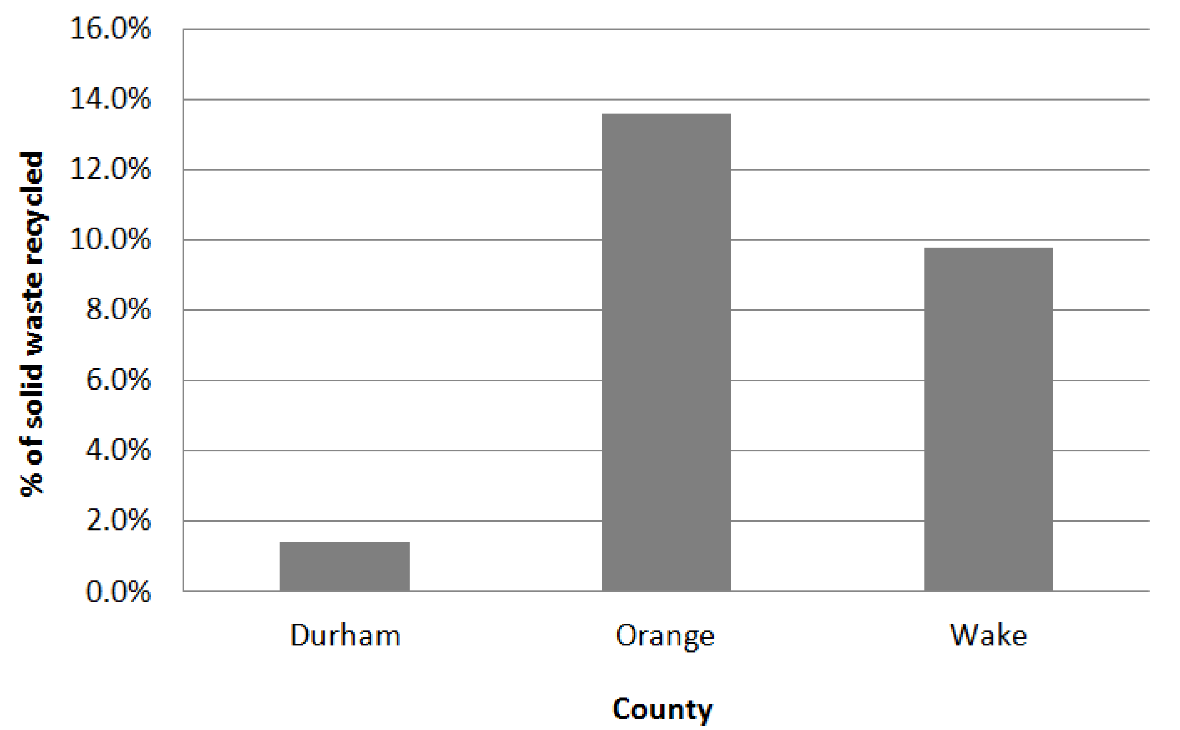


Figure 15. Percent of solid waste recycled in 2005

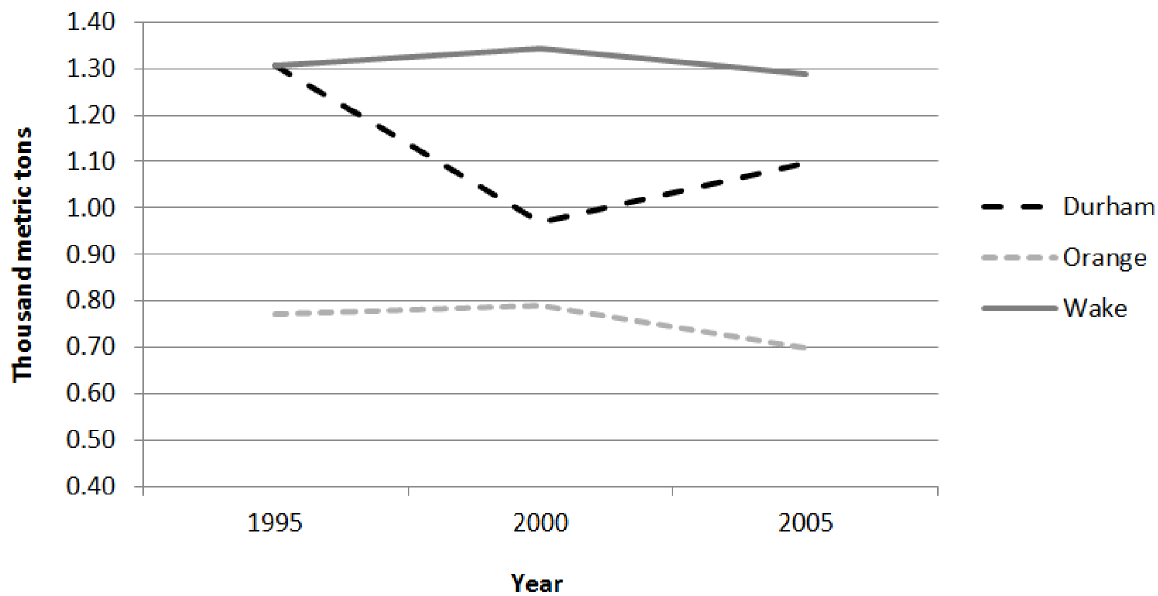


Figure 16. Change in per capita solid waste production

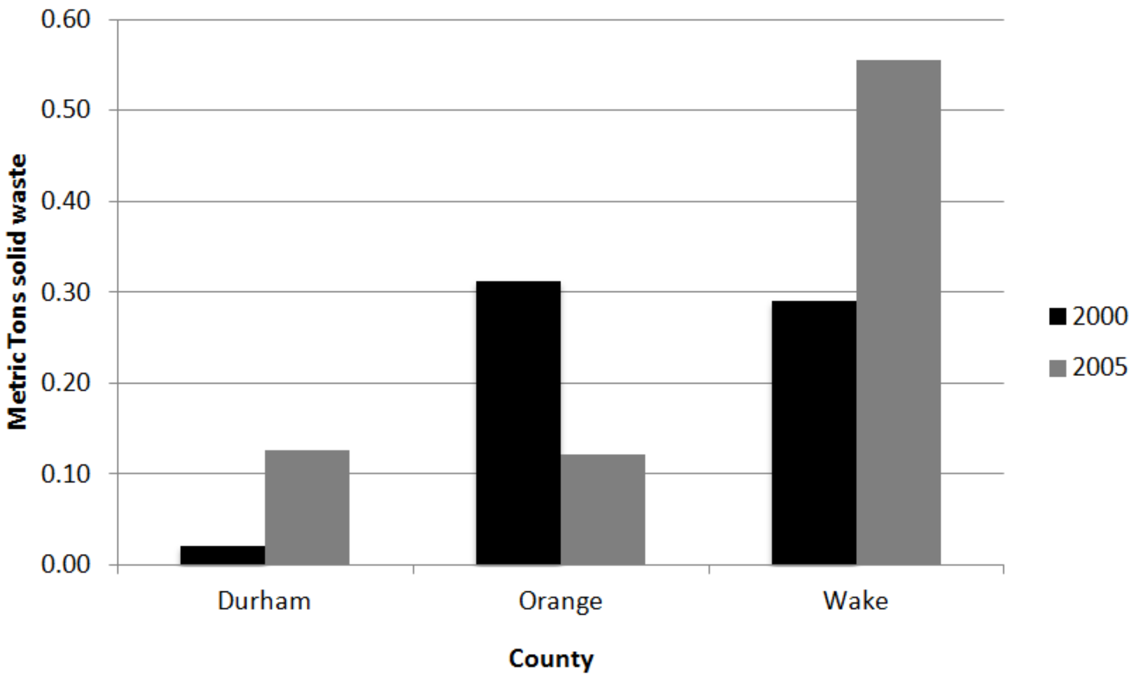


Figure 17. Change in per capita construction and demolition waste (2000 to 2005)

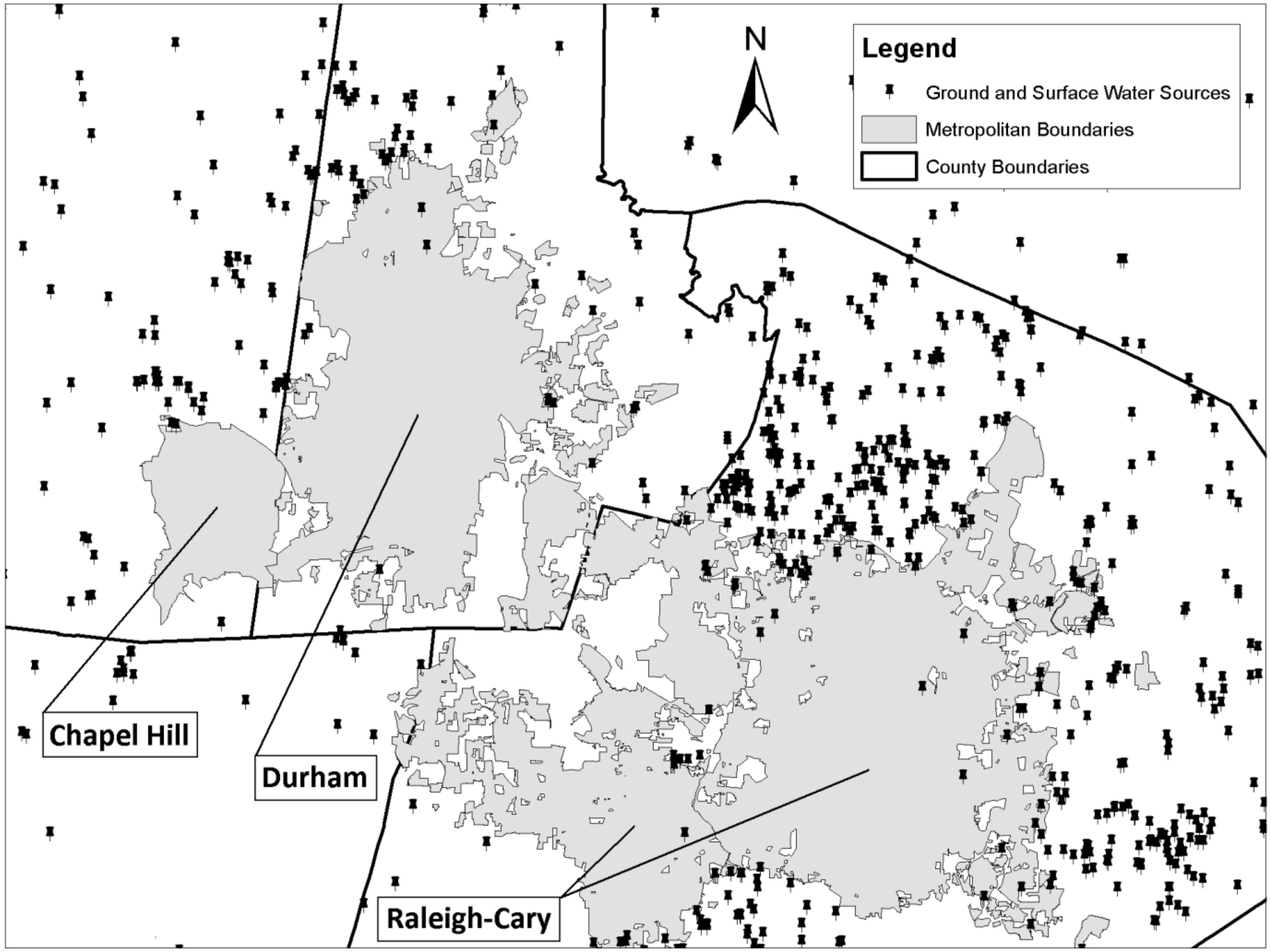


Figure 18. Ground and surface water sources relative to metropolitan areas

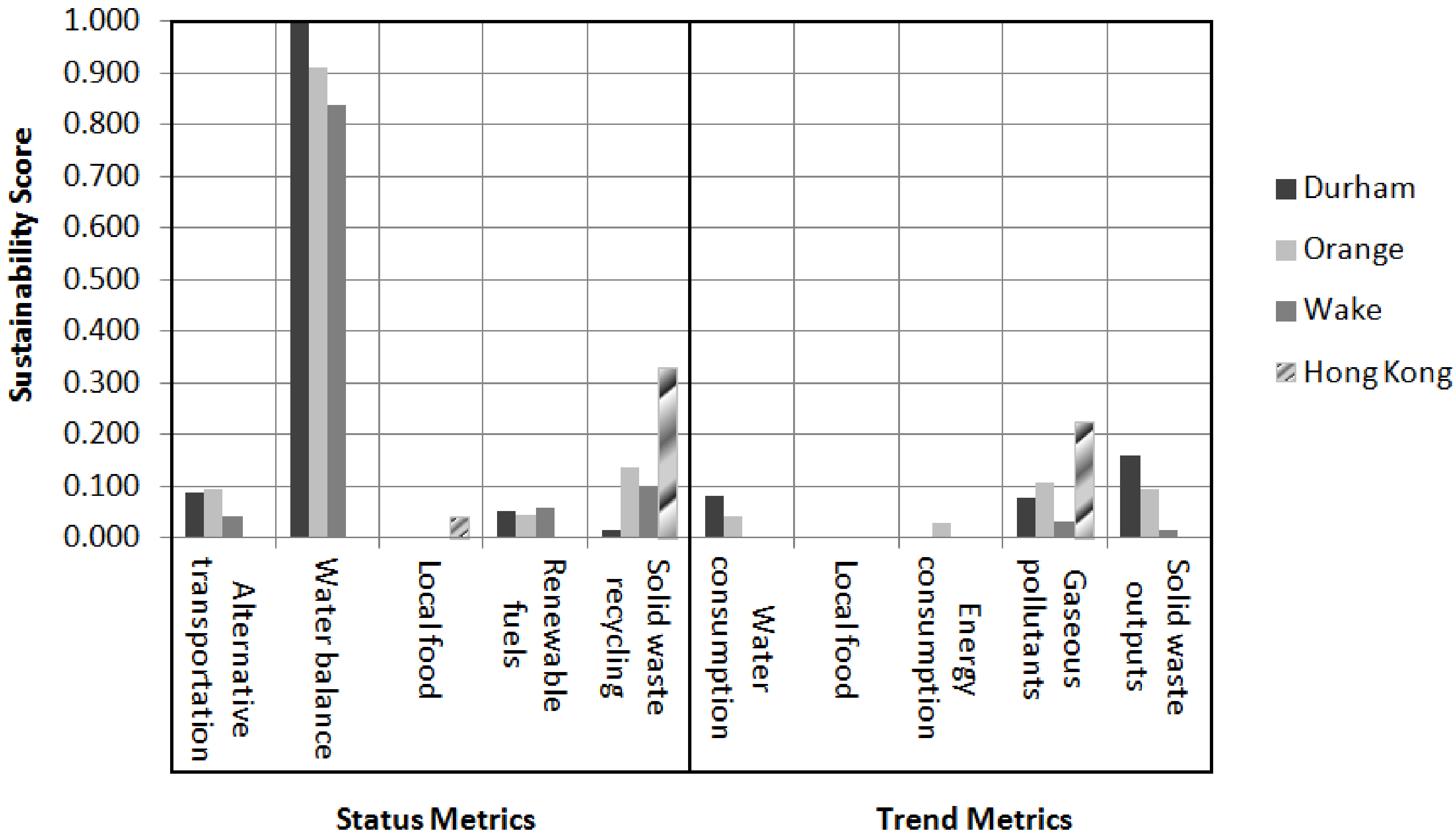


Figure 19. Comparison of sustainability metrics between Durham, Orange, and Wake counties and Hong Kong

Table 1. List of sustainability metrics used, goals for each metric, and equations used in quantifying metrics

Status Metrics	**Goal**	**Equation**
Alternative transportation	100% of commuters utilize walking, bicycling, public transport, and other alternatives	(Commuting time using alternative transportation) / (Commuting time using personal vehicles)
Water balance	Total water inputs into the county are equal to or greater than water outputs	$1/5 \sum_{i=1985}^{2005}$ [(Total water input) - (Total water output)]
Local food	100% of food consumed is from local sources	(Local food sold locally) / (total food consumption)
Renewable fuels	100% of fuels consumed are renewable fuels	(energy from renewable energy sources) / (total energy consumption)
Solid waste recycling	100% of solid waste is recycled	(solid waste recycled) / (solid waste landfilled or exported)
Trend Metrics		
Water consumption	Per capita water consumption decreases by 100%	[(Per capita consumption at beginning of period) - (Per capita consumption at end of period)] / (Per capita consumption at beginning of period)
Local food	Increase to 100% of food from local sources	(% local food at beginning of period) / (% local food at end of period)
Energy consumption	Per capita energy consumption decreases by 100%	[(Per capita consumption at beginning of period) - (Per capita consumption at end of period)] / (Per capita consumption at beginning of period)
Gaseous pollutant emissions	Per capita gaseous pollutant emissions decrease by 100%	[(Per capita emissions at beginning of study period) - (Per capita emissions at end of study period)] / (Per capita emissions at beginning of study period)
Solid waste	Per capita solid waste outputs decrease by 100%	(Per capita output at beginning of study period) / (Per capita output at beginning of study period)

Table 2. Sustainability scorecard for Durham, Orange, and Wake Counties showing status and trend sustainability metrics. All metrics range from 0 (low sustainability) to 1 (high sustainability).

Status Measurements (% of total, 2005)	**Durham**	**Orange**	**Wake**
Alternative transportation	0.087	0.093	0.040
Water balance	1.000	0.910	0.839
Local food*	0.000	0.000	0.000
Renewable fuels	0.051	0.046	0.057
Solid waste recycling	0.014	0.136	0.098
Trend Measurements (1985-2005)			
Water consumption (% reduction)	0.079	0.040	0.000
Local food (% increase)	0.000	0.000	0.000
Energy consumption (% reduction)	0.000	0.027	0.000
Gaseous pollutant emissions (% reduction)	0.079	0.106	0.032
Solid waste outputs (% reduction)	0.160	0.094	0.014
	1.47	**1.45**	**1.08**

* Data from 2007

Table 3. Comparison of per capita resource and waste balances between this study and others compiled by Kennedy et al. (2007) and Ngo and Pataki (2008).

Urban Ecosystem	Year	Population (×1,000)	Population Density (cap/km²)	Inputs: Food (tons/year)	Inputs: Water (tons/year)	Inputs: Total energy (GJ/year)	Outputs: CO2 (tons/year)	Outputs: Solid waste (tons/year)	Outputs: Wastewater (tons)
Sidney	1990	3,657	2,000	0.22	180	114	9.1	0.77	128
Vienna	1990s	1,500	3,710	–	147	–	–	3	144
Hong Kong	1997	6,617	58,000	0.68	138	71	4.8	2.11	102
Greater Toronto Area	1999	5,071	2,920	0.85	183	–	14	–	157
London	2000	7,000	6,730	0.34	117	–	8.6	1.68	–
Cape Town	2000	3,000	3,900	–	109	42	–	0.68	67
Los Angeles	1990	8,863	843	0.84	273	268	13.2	–	79
County, CA	2000	9,159	905	0.91	258	249	13	0.91*	98
Durham	1990	182	242	0.70	272	308	9.3	–	159
County, NC	2000	225	299	0.76	281	318	9.5	0.97	139
Orange	1990	94	91	0.70	221	388	9.2	–	138
County, NC	2000	116	112	0.76	236	396	9.3	0.79	115
Wake	1990	423	196	0.70	351	324	9.7	–	139
County, NC	2000	633	294	0.76	304	356	10.2	1.34	138

* Data from 1999

Table 4. Sustainability assessment of Hong Kong between 1971 and 1997 using data from Newcombe (1978) and Warren-Rhodes and Koenig (2001).

Status Measurements (1997)	**Progress Metric**	**Point Scale**	**Hong Kong Metric**
Alternative transportation	% of total	0 to 1	Missing
Water balance	% of total	0 to 1	Missing
Local food*	% of total	0 to 1	0.040
Renewable fuels	% of total	0 to 1	Missing
Solid waste recycling	% of total	0 to 1	0.33
Trend Measurements (1971-1997)			
Water consumption	% reduction	0 to 1	0
Local food	% increase	0 to 1	0
Energy consumption	% reduction	0 to 1	0
Gaseous pollutant emissions	% reduction	0 to 1	0.23
Solid waste outputs	% reduction	0 to 1	0
	Total Score	**0 to 10**	**0.60**

* Data from 2007

References Chapter 2

Agudelo-Vera, C. M., A. R. Mels, K. J. Keesman, and H. H. M. Rijnaarts. 2011. Resource management as a key factor for sustainable urban planning. *Journal of environmental management* 92(10): 2295-2303.

Baccini, P. 1997. A city's metabolism: Towards the sustainable development of urban systems. *The Journal of urban technology* 4(2): 27.

Barles, S. 2009. Urban Metabolism of Paris and Its Region. *Journal of Industrial Ecology* 13(6): 898-913.

Baron, J. S., N. L Poff, P L. Angermeier, C. N. Dahm, P. H. Gleick, N. G. Hairston, Jr. R. B. Jackson, C. A. Johnston, B. D. Richter, A. D. Steinman. 2002. Meeting ecological and societal needs for freshwater. *Ecological Applications* 12(5): 1247-1260.

Botsford, L. W., J. C. Castilla, and C. H. Peterson. 1997. The management of fisheries and marine ecosystems. *Science* 277(5325): 509-515.

Campbell, S. 1996. Green cities, growing cities, just cities? Urban planning and the contradictions of sustainable development. *Journal of the American Planning Association* 62(3): 296-312.

Chen, Y., X. Li, Y. Zheng, Y. Guan, and X. Liu. 2011. Estimating the relationship between urban forms and energy consumption: A case study in the Pearl River Delta, 2005-2008. *Landscape and Urban Planning* 102(1): 33-42.

Cowell, S. and S. Parkinson. 2003. Localisation of UK food production: an analysis using land area and energy as indicators. *Agriculture Ecosystems & Environment* 94(2): 221-236.

de Almeida, P. and P. D. Silva. 2011. Timing and future consequences of the peak of oil production. *Futures* 43(10): 1044-1055.

Decker, E., S. Elliott, F. Smith, D. Blake, and F. Rowland. 2000. Energy and material flow through the urban ecosystem. *Annual Review of Energy and the Environment* 25685-740.

Fagan, J. E., M. A. Reuter, and K. J. Langford. 2010. Dynamic performance metrics to assess sustainability and cost effectiveness of integrated urban water systems. *Resources Conservation and Recycling* 54(10): 719-736.

Folke, C., A. Jansson, J. Larsson, and R. Costanza. 1997. Ecosystem appropriation by cities. *Ambio* 26(3): 167-172.

Food and Agriculture Organization of the United Nations. 2012. *Food Balance Sheets*. faostat.fao.org/site/354/default.aspx. Accessed October 2011.

Graedel, T. E. and B. R. Allenby. 2003. *Industrial Ecology*. 2nd edition. Upper Saddle River, Jew Jersey: Peason Education, Inc.

Grimm, N. B., J. M. Grove, S. T. A. Pickett, and C. L. Redman. 2000. Integrated approaches to long-term studies of urban ecological systems. *Bioscience* 50(7): 571-584.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global change and the ecology of cities. *Science* 319(5864): 756-760.

Gurney, K. R., D. L. Mendoza, Y. Zhou, M. L. Fischer, C. C. Miller, S. Geethakumar, and Du Can,Stephane De La Rue. 2009. High Resolution Fossil Fuel Combustion CO(2) Emission Fluxes for the United States. *Environmental science & technology* 43(14): 5535-5541.

Huang, S. and C. Chen. 2009. Urbanization and Socioeconomic Metabolism in Taipei. *Journal of Industrial Ecology* 13(1): 75-93.

International Panel for Climate Change. 2007. Summary for Policymakers. In *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* , edited by M. L. Parry et al. Cambridge, UK: Cambridge University Press.

Jabareen, Y. R. 2006. Sustainable urban forms - Their typologies, models, and concepts. *Journal of Planning Education and Research* 26(1): 38-52.

Kennedy, C., S. Pincetl, and P. Bunje. 2011. The study of urban metabolism and its applications to urban planning and design. *Environmental Pollution* 159(8-9): 1965-1973.

Kennedy, C., J. Cuddihy, and J. Engel-Yan. 2007. The changing metabolism of cities. *Journal of Industrial Ecology* 11(2): 43-59.

Lehmann, S. 2011. Resource recovery and materials flow in the city: zero waste and sustainable consumption as paradigm in urban development. *Journal of Green Building* 6(3): 88-105.

Liu, C. and Q. Shen. 2011. An empirical analysis of the influence of urban form on household travel and energy consumption. *Computers Environment and Urban Systems* 35(5): 347-357.

Macknick, J., R. Newmark, G. Heath, and K.C. Hallett. 2011. *A Review of Operational Water Consumption and Withdrawal Factors for Electricity Generating Technologies*. NREL/TP-6A20-50900. Golden, Colorado: United States Department of Energy National Renewable Energy Laboratory.

Maclaren, V. 1996. *Developing Indicators of Urban Sustainability: A Focus on the Canadian Experience*. Toronto, ON: Intergovernmental Committee on Urban and Regional Research, ICURR Press.

Mathews, J. A. and H. Tan. 2011. Progress Toward a Circular Economy in China The Drivers (and Inhibitors) of Eco-industrial Initiative. *Journal of Industrial Ecology* 15(3): 435-457.

Metzger, A. 2013. Literature review on urban metabolism and sustainable urban ecosystems. Unpublished Master's thesis, North Carolina State University, Raleigh.

Nel, W. P. and C. J. Cooper. 2009. Implications of fossil fuel constraints on economic growth and global Warming. *Energy Policy* 37(1): 166-180.

Newcombe, K., J. Kalma, and A. Aston. 1978. Metabolism of a City - Case of Hong-Kong. *Ambio* 7(1): 3-15.

Newman, P. 1999. Sustainability and cities: extending the metabolism model. *Landscape and Urban Planning* 44(4): 219-226.

Newman, P. and J. Kenworthy. 1989. Gasoline Consumption and Cities - a Comparison of United-States Cities with a Global Survey. *Journal of the American Planning Association* 55(1): 24-37.

Ngo, N. S. and D. E. Pataki. 2008. The energy and mass balance of Los Angeles County. *Urban Ecosystems* 11(2): 121-139.

Noble, I. R. and R. Dirzo. 1997. Forests as human-dominated ecosystems. *Science* 277(5325): 522-525.

North Carolina Department of Environment and Natural Resources Division of Waste Mangement. 2012. *Solid Waste Management Plans*. portal.ncdenr.org/web/wm/sw/managementplans. Accessed February 2010.

North Carolina Department of Environment and Natural Resources Division of Water Quality. 2011. Unpublished wastewater monitoring data.

North Carolina Department of Revenue. 2012. *State Sales and Use Tax Statistics*. www.dornc.com/publications/fiscalyearsales.html. Accessed March 2011.

North Carolina State Climate Office. 2012. *NC Climate Retrieval and Observations Network of the Southeast Database*. www.nc-climate.ncsu.edu/cronos. Accessed March 2011.

Obernberger, I. 1998. Decentralized biomass combustion: State of the art and future development. *Biomass & Bioenergy* 14(1): 33-56.

Parry, M. L., C. O.F, and P. J.P. 2007. *Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, UK: Cambridge University Press.

Phillis, Y. A., V. S. Kouikoglou, and V. Manousiouthakis. 2010. A Review of Sustainability Assessment Models as System of Systems. *Ieee Systems Journal* 4(1): 15-25.

Pirog, R., T. van Pelt, K. Enshayan, and E. Cook. 2001. Food, fuel and freeways: an iowa perspective on how far food travels, fuel usage, and greenhouse gas emissions. Ames, IA: Leopold Center for Sustainable Agriculture. www.leopold.iastate.edu/pubs-and-papers/2001-06-food-fuel-freeways. Accessed 14 December 2011.

Rees, W. and M. Wackernagel. 1996. Urban ecological footprints: Why cities cannot be sustainable - And why they are a key to sustainability. *Environmental Impact Assessment Review* 16(4-6): 223-248.

Rygaard, M., P. J. Binning, and H. Albrechtsen. 2011. Increasing urban water self-sufficiency New era, new challenges. *Journal of environmental management* 92(1): 185-194.

Sahely, H., S. Dudding, and C. Kennedy. 2003. Estimating the urban metabolism of Canadian cities: Greater Toronto Area case study. *Canadian Journal of Civil Engineering* 30(2): 468-483.

United Nations Department of Economic and Social Affairs Population Division. 2008. *World Urbanization Prospects: The 2007 Revision*. New York: United Nations.

United States Census Bureau. 2012. www.census.gov. Accessed August 2009.

United States Department of Agriculture Economic Research Service. 2012. *Food Availability (Per Capita) Data System*. www.ers.usda.gov. Accessed September 2010.

United States Department of Agriculture National Agricultural Statistics Service. 2012. *Census of Agriculture.* www.agcensus.usda.gov. Accessed November 2009.

United States Energy Information Administration. 2011. *North Carolina Nuclear Profile*. www.eia.gov/cneaf/nuclear/state_profiles/north_carolina/nc.html. Accessed July 2009.

United States Environmental Protection Agency. 2012a. *AirData website.* www.epa.gov/airdata/ Accessed January 2010.

United States Environmental Protection Agency. 2012b. *Greenhouse Gas Emissions*. www.epa.gov/climatechange/emissions/index.html. Accessed September 2010.

United States Geological Survey. 2012a. *Water Resources of North Carolina*. nc.water.usgs.gov. Accessed March 2011.

United States Geological Survey. 2012b. *Water Use in the United States*. water.usgs.gov/watuse. Accessed March 2011.

Verbruggen, A. and M. Al Marchohi. 2010. Views on peak oil and its relation to climate change policy. *Energy Policy* 38(10): 5572-5581.

Vitousek, P. M., H. A. Mooney, J. Lubchenco, and J. M. Melillo. 1997. Human domination of Earth's ecosystems. *Science* 277(5325): 494-499.

Warren-Rhodes, K. and A. Koenig. 2001. Escalating trends in the urban metabolism of Hong Kong: 1971-1997. *Ambio* 30(7): 429-438.

Winistorfer, P., Z. Chen, B. Lippke, and N. Stevens. 2005. Energy consumption and greenhouse gas emissions related to the use, maintenance, and disposal of a residential structure. *Wood and Fiber Science* 37128-139.

Wolman, A. 1965. Metabolism of cities. *Scientific American* 213(3): 179.

Woodcock, J., D. Banister, P. Edwards, A. M. Prentice, and I. Roberts. 2007. Energy and health 3 - Energy and transport. *Lancet* 370(9592): 1078-1088.

World Commission on Environment and Development. 1987. *Our Common Future*. Oxford: Oxford University Press.

Zeng, Y., K. Trauth, R. Peyton, and S. Banerji. 2005. Characterization of solid waste disposed at Columbia Sanitary Landfill in Missouri. *Waste Management & Research* 23(1): 62-71.

APPENDICES

APPENDIX A

Table 1. Population change between 1985 and 2005 (number of people)

	Durham	Orange	Wake
1985	162,700	82,500	354,100
1990	181,840	93,850	423,370
1995	195,260	107,650	513,640
2000	224,619	116,017	633,461
2005	242,121	122,466	756,873
% growth	49%	48%	114%

Table 2. Ratios of county to state population and employment used for downscale (Source: United States Census, 2010)

	Year	Durham	Orange	Wake
Population	1985	2.6%	1.3%	5.8%
	1990	2.7%	1.4%	6.4%
	1995	2.7%	1.4%	7.2%
	2000	2.8%	1.4%	7.8%
	2005	2.8%	1.4%	8.7%
Industrial Sector Employment	1985*	1.6%	0.82%	3.8%
	1990	1.6%	0.81%	4.6%
	1995*	1.7%	0.80%	5.4%
	2000	1.8%	0.79%	6.2%
	2005*	1.9%	0.77%	7.3%
Commercial Sector Employment	1985*	4.3%	2.3%	8.1%
	1990	3.9%	2.1%	9.1%
	1995*	3.5%	1.8%	10.1%
	2000	3.0%	1.6%	11.1%
	2005*	2.7%	1.4%	12.4%

* Estimated based on data from 1990 and 2000

APPENDIX B

Table 1. Detailed water balance between 1985 and 2005 (Million m³ unless otherwise noted)

	Inputs				Outputs				
	Precipitation	**Anthropogenic Withdrawals**	**Consumption per capita (m³)**	**Surface water inflow**	**Evapotranspiration**	**Wastewater**	**Wastewater per capita (m³)**	**Runoff**	**Balance**
Durham									
1985	1,063	44	269	217	1,116	28	170	119	61
1990	841	49	272	441	680	29	159	255	367
1995	955	51	261	353	921	28	145	48	361
2000	755	63	281	391	644	31	139	129	404
2005	815	66	273	199	937	32	131	165	-53
% change	-	51%	2%	-	-	15%	-23%	-	
Orange									
1985	1,407	11	204	0	1,538	13	159	65	-197
1990	1,179	21	221	0	936	13	138	276	-26
1995	1,400	26	237	0	1,269	13	120	217	-73
2000	1,117	28	236	0	887	14	115	224	20
2005	1,137	24	200	0	1,291	12	100	187	-329
% change	-	107%	-2%	-	-	-10%	-37%	-	
Wake									
1985	2,117	85	240	401	3,199	48	136	697	-1,342
1990	2,267	148	351	846	1,948	59	139	1,479	-225
1995	2,880	125	243	473	2,640	78	151	1,064	-305
2000	2,308	191	304	876	1,846	87	138	1,673	-232
2005	2,222	200	267	437	2,686	105	140	933	-866
% change	-	136%	11%	-	-	118%	3%	-	

Table 2. Change in per capita water consumption by use type (million m^3)

	Public Supply	Domestic Self Supplied	Industrial Self Supplied	Irrigation Self Supplied	Livestock Self Supplied	Mining	Power Generation	Commercial Self Supplied	Total
Durham									
1985	234	31	0.85	2.2	0.68	0	0	0	269
1990	244	14	0.076	13	0.61	0	0	0.38	272
1995	188	26	28.3	19	0.5	0	0	0.35	261
2000	236	25	0	20	0.25	0	0	0	281
2005	233	20	0	17	0.34	2.3	0	0	273
% change	0%	-36%	-100%	684%	-50%	-	-	-	2%
Orange									
1985	156	20	0.17	13	6.9	6.5	0	0	204
1990	186	10	0	12	6.2	5.9	0	0.29	221
1995	169	22	0	39	6.2	0	0	0.39	237
2000	187	23	0	23	3.5	0	0	0	236
2005	162	17	0	18	2.5	1.5	0	0	200
% change	3%	-18%	-100%	31%	-64%	-77%	-	-	-2%
Wake									
1985	179	28	11	16	1.5	5.2	0	0.078	240
1990	193	5.3	10	35	0.88	0.1	106	0.065	351
1995	147	25	21	27	1.37	2.7	17	0.67	243
2000	183	14	0	31	0.26	0	76	0	304
2005	164	14	0	17	0.77	2.5	69	0	267
% change	-8%	-49%	-100%	6%	-49%	-51%	-	-100%	11%

Table 3. Data sources used for average evapotranspiration data (million m^3) (Source: North Carolina State Climate Office, 2010)

	RDU Airport - Morrisville, Wake, NC	Lake Wheeler Rd Lab - Raleigh, Wake, NC	Reedy Creek Field Laboratory - Raleigh, Wake, NC	Williams Airport - Chapel Hill, Orange, NC	Duke Forest - Durham, Durham, NC	Avg.
1985	53.0	63.9	-	-	-	58.5
1990	53.5	17.7	-	-	-	35.6
1995	48.2	missing	-	-	-	48.2
2000	47.6	incomplete	19.8	43.6	incomplete	33.7
2005	51.3	45.9	51.9	45.2	47.2	49.1

APPENDIX C

Table 1. Durham County fuel and electricity consumption between 1985 and 2005 (terajoules)

		Coal	Nat. gas	Petroleum	Biomass	Fuel ethanol	Retail Electricity	Electrial system loss	Total
Residential	**1985**	30	819	1,776	791	0	2,533	5,835	11,784
	1990	23	1,044	1,325	338	0	3,272	7,564	13,567
	1995	20	1,475	1,544	512	0	3,897	8,850	16,298
	2000	8.8	1,933	1,511	428	0	4,658	10,595	19,134
	2005	8.8	1,952	1,289	389	0	5,441	11,901	20,981
Commercial	**1985**	172	1,174	1,256	32	0	2,965	6,829	12,429
	1990	131	1,322	999	53	0	3,565	8,240	14,310
	1995	179	1,410	833	88	0	3,875	8,802	15,186
	2000	87	1,426	883	77	0	4,280	9,736	16,489
	2005	100	1,415	837	60	0	4,318	9,443	16,173
Industrial	**1985**	921	1,275	1,940	1,067	0	1,476	3,401	10,079
	1990	1,285	1,534	2,232	1,428	0	1,841	4,256	12,577
	1995	1,111	1,989	2,875	1,531	0	2,096	4,759	14,361
	2000	879	2,066	3,389	1,517	0	2,200	5,002	15,052
	2005	726	1,770	3,307	1,292	0	2,020	4,418	13,534
Transportation	**1985**	0	136	13,488	0	22	0	0	13,645
	1990	0	188	15,349	0	0	0	0	15,537
	1995	0	182	17,226	0	3	0	0	17,412
	2000	0	217	20,419	0	97	0	0	20,733
	2005	0	133	22,022	0	62	0	0	22,216
Total	**1985**	1,123	3,403	18,459	1,890	22	6,974	16,065	47,937
	1990	1,439	4,088	19,905	1,820	0	8,678	20,060	55,990
	1995	1,310	5,056	22,478	2,131	2.9	9,868	22,411	63,257
	2000	974	5,642	26,201	2,022	97	11,138	25,333	71,408
	2005	835	5,271	27,454	1,742	62	11,778	25,763	72,905
	change	-26%	55%	49%	-8%	180%	69%	60%	52%
Per Capita (GJ)	**1985**	6.9	21	113	12	0.14	43	99	295
	1990	7.9	22	109	10	0	48	110	308
	1995	6.7	26	115	11	0.015	51	115	324
	2000	4.3	25	117	9.0	0.43	50	113	318
	2005	3.4	22	113	7.2	0.26	49	106	301
	change	-50%	4.1%	0%	-38%	88%	13%	7.8%	2.2%

Table 2. Orange County fuel and electricity consumption between 1985 and 2005 (terajoules)

		Coal	Nat. Gas	Petroleum	Biomass	Fuel Ethanol	Retail Electricity	Electrial system loss	Total
Residential	**1985**	16	417	905	403	0	1,291	2,975	6,007
	1990	12	538	682	174	0	1685	3,896	6,988
	1995	11	779	815	270	0	2,058	4,673	8,606
	2000	4.5	999	780	221	0	2,406	5,474	9,885
	2005	4.5	988	652	197	0	2,752	6,021	10,614
Commercial	**1985**	92	629	673	17	0	1,588	3,657	6,656
	1990	70	706	533	28	0	1,904	4,400	7,642
	1995	95	750	443	47	0	2,062	4,684	8,081
	2000	46	755	468	41	0	2,267	5,157	8,735
	2005	53	747	442	32	0	2,279	4,985	8,537
Industrial	**1985**	486	673	1,025	564	0	779	1,796	5,324
	1990	638	762	1,109	709	0	914	2,113	6,245
	1995	520	930	1,344	716	0	980	2,226	6,716
	2000	388	911	1,495	669	0	970	2,206	6,640
	2005	302	736	1,374	537	0	839	1,836	5,623
Transportation	**1985**	0	69	6,876	0	11	0	0	6,956
	1990	0	97	7,906	0	0	0	0	8,002
	1995	0	96	9,096	0	2	0	0	9,194
	2000	0	112	10,549	0	50	0	0	10,711
	2005	0	67	11,141	0	31	0	0	11,239
Total	**1985**	594	1,789	9,478	984	11	3,659	8428	24,943
	1990	720	2,102	10,230	912	0	4,503	10,410	28,877
	1995	625	2,555	11,699	1,033	1.5	5,100	11,582	32,596
	2000	438	2,778	13,292	931	50	5,644	12,837	35,970
	2005	359	2,537	13,608	766	31	5,871	12,841	36,014
	change	-40%	42%	44%	-22%	178%	60%	52%	44%
Per Capita (GJ)	**1985**	7.2	22	115	12	0.14	44	102	302
	1990	7.7	22	109	10	0	48	111	308
	1995	5.8	24	109	10	0.014	47	108	303
	2000	3.8	24	115	8.0	0.43	49	111	310
	2005	2.9	21	111	6.3	0.26	48	105	294
	change	-59%	-4.4%	-3%	-48%	87%	8%	3%	-2.7%

Table 3. Wake County fuel and electricity consumption data between 1985 and 2005 (terajoules)

		Coal	Nat. Gas	Petroleum	Biomass	Fuel Ethanol	Retail Electricity	Electrial system loss	Total
Residential	**1985**	67	1,799	3,903	1,739	0	5,568	12,826	25,902
	1990	54	2,448	3,106	793	0	7,670	17,734	31,826
	1995	53	3,879	4,061	1,346	0	10,252	23,281	42,904
	2000	25	5,452	4,260	1,208	0	13,137	29,880	53,986
	2005	28	6,101	4,028	1,217	0	17,005	37,198	65,622
Commercial	**1985**	326	2,224	2,378	60	0	5,615	12,930	23,525
	1990	309	3,115	2,353	125	0	8,401	19,415	33,727
	1995	524	4,132	2,440	257	0	11,356	25,795	44,527
	2000	318	5,222	3,235	282	0	15,679	35,663	60,411
	2005	457	6,449	3,812	274	0	19,672	43,025	73,714
Industrial	**1985**	2,221	3,076	4,681	2,575	0	3,561	8,206	24,317
	1990	3,605	4,301	6,261	4,006	0	5,163	11,936	35,277
	1995	3,513	6,290	9,090	4,842	0	6,627	15,050	46,370
	2000	3,067	7,211	11,828	5,293	0	7,677	17,456	53,156
	2005	2,856	6,967	13,012	5,086	0	7,950	17,386	53,822
Transportation	**1985**	0	298	29,646	0	49	0	0	29,993
	1990	0	441	35,983	0	0	0	0	36,424
	1995	0	479	45,315	0	7.6	0	0	45,794
	2000	0	612	57,585	0	273	0	0	58,197
	2005	0	415	68,829	0	194	0	0	69244
Total	**1985**	2,615	7,397	40,608	4,374	49	14,744	33,963	103,736
	1990	3,968	10,306	47,703	4,925	0	21,233	49,085	137,254
	1995	4,091	14,780	60,907	6,445	7.6	28,236	64,126	179,594
	2000	3,409	18,497	76,908	6,783	273	36,493	83,000	225,751
	2005	3,341	19,931	89,681	6,576	194	44,626	97,609	262,401
	% change	28%	169%	121%	50%	298%	203%	187%	153%
Per Capita (GJ)	**1985**	7.4	21	115	12	0.14	42	96	293
	1990	9.4	24	113	12	0.00	50	116	324
	1995	8.0	29	119	13	0.015	55	125	350
	2000	5.4	29	121	11	0.43	58	131	356
	2005	4.4	26	118	8.7	0.26	59	129	347
	change	-40%	26%	3%	-30%	86%	42%	34%	18%

APPENDIX D

Table 1. Total gaseous pollutant outputs between 1985 and 2005 (metric tons unless otherwise stated)

		CO_2 (1000 Metric Tons)	CO	NH3	N_2O	Particles < 10 µm	Particles < 2.5 µm	SO_2	VOC's
Durham	**1985***	1,523	117,166	247	8,841	5,482	1,670	1,796	18,671
	1990	1,683	102,000	311	9,051	5,535	1,716	1,679	16,092
	1995*	1,896	86,834	376	9,261	5,588	1,762	1,563	13,514
	2000	2,144	71,668	441	9,472	5,642	1,808	1,446	10,935
	2005*	2,209	63,028	376	8,232	5,837	1,320	831	9,861
	% change	45%	-46%	52%	-7%	6%	-21%	-54%	-47%
Orange	**1985***	786	56,406	627	3,920	3,098	1,057	3,198	7,803
	1990	863	54,154	659	4,588	3,246	1,122	2,353	7,255
	1995*	977	51,902	692	5,257	3,395	1,186	1,508	6,707
	2000	1,078	49,650	725	5,926	3,543	1,251	663	6,159
	2005*	1,085	41,396	416	5,397	3,822	1,219	514	5,690
	% change	38%	-27%	-34%	38%	23%	15%	-84%	-27%
Wake	**1985***	3,361	254,231	835	19,244	22,382	6,055	3,045	38,743
	1990	4,108	238,821	915	20,980	20,783	5,864	2,769	34,912
	1995*	5,242	223,410	996	22,716	19,184	5,672	2,494	31,080
	2000	6,437	208,000	1,076	24,452	17,585	5,481	2,218	27,248
	2005*	7,411	177,306	1,067	21,489	20,667	4,580	1,731	24,689
	% change	121%	-30%	28%	12%	-8%	-24%	-43%	-36%

*Estimate based on 1990, 2000, and 2008 data

Table 2. Per capita gaseous pollutant outputs between 1985 to 2005 (metric tons)

		CO_2	CO	NH3	N_2O	Particles < 10 µm	Particles < 2.5 µm	SO_2	VOC's
Durham	**1985**	9.4	0.72	0.0015	0.054	0.034	0.0103	0.0110	0.115
	1990	9.3	0.56	0.0017	0.050	0.030	0.0094	0.0092	0.088
	1995	9.7	0.44	0.0019	0.047	0.029	0.0090	0.0080	0.069
	2000	9.5	0.32	0.0020	0.042	0.025	0.0080	0.0064	0.049
	2005	9.1	0.26	0.0016	0.034	0.024	0.0054	0.0034	0.041
	% change	-3%	-64%	2%	-37%	-28%	-47%	-69%	-65%
Orange	**1985**	9.5	0.68	0.0076	0.048	0.038	0.0128	0.0388	0.095
	1990	9.2	0.58	0.0070	0.049	0.035	0.0120	0.0251	0.077
	1995	9.1	0.48	0.0064	0.049	0.032	0.0110	0.0140	0.062
	2000	9.3	0.43	0.0062	0.051	0.031	0.0108	0.0057	0.053
	2005	8.9	0.34	0.0034	0.044	0.031	0.0100	0.0042	0.046
	% change	-7%	-51%	-55%	-7%	-17%	-22%	-89%	-51%
Wake	**1985**	9.5	0.72	0.0024	0.054	0.063	0.0171	0.0086	0.109
	1990	9.7	0.56	0.0022	0.050	0.049	0.0138	0.0065	0.082
	1995	10.2	0.43	0.0019	0.044	0.037	0.0110	0.0049	0.061
	2000	10.2	0.33	0.0017	0.039	0.028	0.0087	0.0035	0.043
	2005	9.8	0.23	0.0014	0.028	0.027	0.0061	0.0023	0.033
	% change	3%	-67%	-40%	-48%	-57%	-65%	-73%	-70%

Table 3. Vulcan Project per capita CO_2 emissions estimates for 2002 (metric tons). Source: Gurney et al. 2009

	Commercial	Industrial	Residential	Electricity Production	Transportation	Total
Durham	0.98	1.3	0.93	0.10	7.6	10.9
Orange	0.49	0.065	0.91	0	7.3	8.7
Wake	0.73	0.34	0.91	0.0017	7.6	9.6

APPENDIX E

Table 1. Solid waste production, construction and demolition waste, and recycling (Thousand metric tons unless otherwise stated)

	Durham	Orange	Wake
Total Waste Produced			
1995	252	82.0	678
2000	217	93.8	841
2005	269	98.5	1,078
% change (15-year)	7%	20%	59%
Per Capita (metric tons)			
1995	1.31	0.77	1.31
2000	0.97	0.79	1.34
2005	1.10	0.70	1.29
% change (15-year)	-16%	-9%	-1%
Construction and Demolition Waste			
2000	4.65	36.9	182
2005	30.4	14.8	419
% change (10-year)	554%	-60%	130%
Per Capita (Metric tons)			
2000	0.02	0.31	0.29
2005	0.13	0.12	0.56
% change (10-year)	503%	-61%	91%
Waste recycled			
2005	3.8	13	106
Per capita (metric tons)			
2005	0.016	0.11	0.14