

ABSTRACT

Title of Document: SPATIAL AND SEASONAL DISTRIBUTION OF CARBON DIOXIDE EMISSIONS FROM FOSSIL-FUEL COMBUSTION; GLOBAL, REGIONAL, AND NATIONAL POTENTIAL FOR SUSTAINABLE BIOENERGY FROM RESIDUE BIOMASS AND MUNICIPAL SOLID WASTE.

Jay Sterling Gregg, Doctor of Philosophy, 2009

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Combustion of fossil fuels releases carbon dioxide (CO₂) into the atmosphere, and has led to an increase in the atmospheric concentration of CO₂. CO₂ is a greenhouse gas, and the increase in concentration leads to an increase in global temperatures and global climatic change. Fossil-fuel consumption, along with cement production, is responsible for 80% of anthropogenic carbon emissions and consumption of fossil fuels continues to increase. Despite its importance to the global climate and the global carbon cycle, data for fossil fuel CO₂ emissions are traditionally maintained only on national levels and annual time steps. A method is developed to improve the spatiotemporal resolution to the leading energy consuming countries of the world. The method uses energy consumption datasets as well as other ancillary datasets to apportion national annual emissions totals into sub-national

and monthly emissions datasets by fuel type. Emissions patterns are highly variable both temporally and spatially by fuel type, and detailed information on the distribution of emissions improves our understanding of the global carbon cycle and leads to better understanding of the spatial and seasonal distribution of the drivers of global change.

In the endeavor to develop alternatives to fossil fuels, advanced biomass energy has garnered much attention because of its renewable nature and its potential to approach carbon-neutrality. As co-products, agricultural and forestry residues as well as municipal solid waste (MSW) are potential low-cost and sustainable biomass feedstocks for energy production. The role of residue biomass within the future global energy portfolio is projected and quantified under the context of environmental and economic sustainability. The potential for residue biomass is projected for the next century under a reference (business-as-usual) scenario and a scenario that includes a hypothetical climate policy that limits carbon emissions. While residue biomass alone cannot replace fossil fuels, a substantial amount of energy potentially could come from this resource, particularly in a global economic market under a climate policy that caps CO₂ emissions from fossil fuels.

SPATIAL AND SEASONAL DISTRIBUTION OF CARBON DIOXIDE
EMISSIONS FROM FOSSIL-FUEL COMBUSTION;
GLOBAL, REGIONAL, AND NATIONAL POTENTIAL FOR SUSTAINABLE
BIOENERGY FROM RESIDUE BIOMASS AND MUNICIPAL SOLID WASTE.

By

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Dissertation submitted to the Faculty of the Graduate School of the
University of Maryland, College Park, in partial fulfillment
of the requirements for the degree of
Doctor of Philosophy
2009

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Preface

This dissertation consists of six manuscripts, presented here in two parts. The first three manuscripts concern the emissions of carbon dioxide (CO₂) from the combustion of fossil fuels. The second three manuscripts focus on the potential for bioenergy from agricultural and forestry residue biomass and municipal solid waste (MSW).

The purpose of the three manuscripts in the first section is to develop and apply a methodology for improving the temporal and spatial resolution of CO₂ emissions estimates. The first manuscript uses the United States (US) to develop the technique of apportioning national annual emissions statistics to states by month based on (often incomplete) fuel consumption and other ancillary datasets. The second manuscript applies the technique to China. At the time of publication, China appeared to have recently passed the US in annual emissions from fossil-fuel consumption. The third manuscript expands upon the first, applying the technique to better understand emissions patterns across North America.

The second set of manuscripts shifts the focus to one of the alternatives to fossil fuels: bioenergy. Specifically, this suite of three manuscripts concentrates on the potential for bioenergy from residue biomass. The fourth manuscripts analyses the potential for this fuel using an integrated assessment modeling framework, and projects the global and regional utilization of agricultural and forestry residue biomass with and without the presence of a global climate policy that limits

anthropogenic CO₂ emissions. The fifth manuscript looks more at the effect of residue harvest on erosion, future crop yields, and carbon balance at the field level for a selection of US crops and locations. Finally, the sixth manuscript contains an analysis of the biomass in the MSW stream, and projects the waste-to-energy potential from discarded biomass for the next century.

I do not intend to suggest, by the organization of this document, that residue biomass can serve as a complete replacement for fossil fuels. Given the size of the resource, residue biomass could only replace a fraction of current fossil-fuel consumption, much less when considering projected future growth in energy consumption. It is often said that there is no silver bullet to the energy-environment-economy trilemma, and the future energy portfolio will have a suite of complementary resources to compete with fossil fuel. Residue biomass is no panacea.

Fossil fuel CO₂ emissions research presented here supports development and improvement of the fossil fuel emissions database maintained by the Carbon Dioxide Information Analysis Center (CDIAC) at Oak Ridge National Laboratory. This database, one of the most complete and well-documented of its kind, is used by social and physical scientists worldwide. In addition, much of the high resolution spatiotemporal data on emissions presented in this dissertation are currently being integrated into the Carbon Tracker model under development at the National Oceanic and Atmospheric Administration (NOAA). It is hoped that the high resolution data within the Carbon Tracker model will improve our understanding of regional carbon fluxes from both anthropogenic and natural sources. Analysis of the fossil carbon

emissions patterns for North America also meet goals of the North American Carbon Program (NACP).

The residue biomass research presented here serves to aid in the development of the O^bJECTS MiniCAM at the Joint Global Change Research Institute by improving the detail and capabilities of the bioenergy, agriculture, and land use forecasts of that model. This model is used by the US Department of Energy to build scenarios for the US Climate Change Science Program (CCSP) and also contributes to analysis in the Intergovernmental Panel on Climate Change (IPCC) reports. Modeling of residue biomass feedstock potential supplements work currently being pursued by the Environmental Sciences Division at Oak Ridge National Laboratory. In addition, determination of sustainable residue removal rates is one of the goals for the sustainability thrust of the Great Lakes Bioenergy Research Center (GLBRC), a long-term US Department of Energy research program designed to inform energy policy makers on how to most sustainably implement a national bioenergy program.

Foreword

Part I.

Bring me my Bow of burning gold:

Bring me my Arrows of desire:

Bring me my Spear: O clouds unfold!

Bring me my Chariot of Fire.

-William Blake (1757-1827)

(from 'Jerusalem' in *Milton*, 1804-1808)

Part II.

WHAT things for dream there are when spectre-like,

Moving among tall haycocks lightly piled,

I enter alone upon the stubble field,

From which the laborers' voices late have died,

And in the antiphony of afterglow

And rising full moon, sit me down

Upon the full moon's side of the first haycock

And lose myself amid so many alike.

-Robert Frost (1874-1963)

(from 'Waiting Afield at Dusk' in *A Boy's Will*, 1915)

Dedication

To those who offered me unwavering encouragement, friendship, and support.

Thank you.

Acknowledgements

I have had the opportunity to collaborate with a number of leading scientists and have had the chance to work in some of the top research labs in the world. None of this would have been possible without the generosity and support of many people, institutions, and programs.

The work on fossil fuel CO₂ emissions began at the Department of Space Studies at the University of North Dakota, under Robert J. Andres. Since then, work on China emissions was done at the Institute for Geographic Studies and Natural Resources Research (IGSNRR) in Beijing, China, under my host, Shaoqiang Wang. I was given the opportunity to conduct research in China through the East Asia and Pacific Summer Institute (EAPSI) run by the National Science Foundation (NSF) and cooperating institutions in the host countries, in this case, the Chinese Ministry of Science and Technology (MOST). The CO₂ Fossil Fuel Emissions Effort (CO₂FFEE) team, led by Kevin Gurney at Purdue University has been key in supporting my work on high resolution emissions, fostering collaboration and focusing research efforts, as well as offering travel support to attend meetings. Collaboration with the National Oceanic and Atmospheric Administration (NOAA) Carbon Tracker modeling team has been especially beneficial in improving format issues and refining the data products. In particular, I wish to extend my gratitude to Pieter Tans, Gabrielle Petron, Joceyln Turnbull, and John Miller for inviting me to present the most recent findings and to discuss future direction for this work. My research on fossil-fuel emissions

continued through the summer of 2008 where I served as an Advanced Short Term Research Opportunity (ASTRO) intern under Gregg Marland, TJ Blasing, and Tom Boden at the Environmental Sciences Division of Oak Ridge National Laboratory (ORNL).

A special thanks goes to the Joint Global Change Research Institute (JGCRI), which has been my home base for research since the Fall of 2006. It has been an exciting privilege to be a part of an institution that delivers timely, policy relevant research in the face of one of humanity's biggest challenges. I would like to extend a thanks to all of the staff there for including me as part of the team.

In particular, I extend my gratitude to Steve Smith, who was always available to discuss research ideas and helped shape my thinking when confronted with enormous and complex issues. He introduced me to the world of integrated assessment modeling, as well as research in the field of bioenergy.

R. César Izaurrealde of JGCRI also helped focus my ideas and fueled my interest in bioenergy research by including me in field research on Tennessee switchgrass plots (with ORNL) and flying me to the Kellogg Biological Station near Battle Creek, Michigan for meetings of the sustainability thrust to the Great Lakes Bioenergy Research Center (GLBRC). Like Steve Smith, César Izaurrealde has always been available to meet and discuss ideas and is one of the most inspirational scientists I have had the pleasure to meet. I also must thank both Steve Smith and César Izaurrealde for funding me as a graduate research assistant for the last three years.

Ning Zeng of the Atmospheric and Oceanic Sciences Department has also been an inspiration and always willing to discuss ideas and include me on his research projects. He invited me to co-author a paper that analyzed the challenges China faces in confronting climate change, a paper that found its way into the journal *Science*. I have had the valuable experience of being closely involved with his Gemstone honor students, Team Carbon Sinks, and served as a mentor to the team. He has invited me on numerous fact finding field trips, and fostered an appreciation for meeting people on the ground in the real world to check that the model results made sense.

Six of the eight chapters of my dissertation have been submitted to academic journals for publication. As of the date of submission, two have been published (Chapters 2 and 3), two are in press (Chapters 4 and 7), and two have been accepted (Chapters 5 and 6). Five of these manuscripts have multiple authors, and below, I document the work I have done on these manuscripts versus that of the other authors.

Chapter 2: *A method for estimating the temporal and spatial patterns of carbon dioxide emissions from national fossil-fuel consumption*, was co-authored by Robert Andres of Oak Ridge National Laboratory. For this chapter, I developed the methodology, I collected all of the data, did all of the analyses, created all of the figures, and composed the entire manuscript. Robert Andres had the original concept of the apportioning methodology and had previously applied it to a short pilot study. I adapted it to reduce uncertainty and improve accuracy, as well as modified it to handle missing data.

Chapter 3: *China: the emissions pattern of the world leader in CO2 emissions from fossil fuel consumption and cement production*, was co-authored Robert Andres and Gregg Marland of Oak Ridge National Laboratory. For this chapter, I developed the methodology, I collected all of the Chinese data from the National Bureau of Statistics, and did the all of the analyses with regard to the emissions patterns in China. I also created all of the figures and composed the entire manuscript. Robert Andres had the original concept of the apportioning methodology (a similar methodology was employed in Chapter 2) and assisted with the uncertainty analysis by analyzing the magnitude of revisions in past entries in the CDIAC database. The other part of the uncertainty analysis was done by me, by looking at past revisions within the National Bureau of Statistics, as well as the agreement between annual totals and provincial sums within that source. Gregg Marland was in charge of maintaining the CDIAC database which contains national annual emissions totals, and he projected the emissions totals to the present day using BP data (there is a approximately a 3-year lag in fuel consumption data from the UN that is typically used to create the CDIAC emissions totals).

Chapter 4: *The temporal and spatial distribution of carbon dioxide emissions from fossil-fuel use in North America*, was co-authored by London Losey of the University of North Dakota, and Robert Andres, T. J. Blasing, and Gregg Marland of Oak Ridge National Laboratory. For this chapter, I developed the methodology, collected all of the data, completed all of the analyses, created all of the figures, and composed the entire manuscript. London Losey had created preliminary estimates for

emissions from Canada and Mexico and had identified national data sources that contained fuel consumption statistics. I then updated, refined, and otherwise corrected these estimates. Robert Andres had the original concept of the apportioning approach (a similar approach was used in Chapters 2 and 3). T.J. Blasing had previously published emissions patterns for the US (state-by-state annual, and monthly national) using a different (bottom-up) methodology. I adapted the original concept of Andres to include elements of Blasing's approach to further reduce uncertainty and error in the emissions estimates. Gregg Marland provided the emissions coefficients to convert BP fuel consumption statistics to CO₂ emissions estimates.

Chapter 5: *Global and regional potential for bioenergy from agricultural and forestry residue biomass*, was co-authored by Steve Smith of the Joint Global Change Research Institute. For this chapter, I developed the parameterization for estimating residue biomass, I collected all of the data, completed all of the analyses, created all of the figures, and composed the entire manuscript. Steve Smith was involved with getting the parameterization into the MiniCAM integrated assessment model and testing its implementation. Steve Smith also collaborated on discussions on how best to parameterize residue biomass.

Chapter 6: *Effect of crop residue harvest on long-term crop yield, soil erosion, and nutrient balance: trade-offs for a sustainable bioenergy feedstock*, was co-authored by R. César Izaurralde of the Joint Global Change Research Institute. For this chapter, I developed the parameterization for estimating residue biomass, I collected all of the data, completed all of the analyses, created all of the figures, and

composed the entire manuscript. César Izaurrealde collaborated on discussions concerning the factorial study design.

Chapter 7: *National and regional generation of biomass residue biomass and the future potential for waste-to-energy implementation*, was solely authored by myself.

The remainder of the dissertation, including Chapter 1 and Chapter 8, represent my work entirely. Any further or more detailed questions concerning the allocation of work duties between the various authors may be directed to me.

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Chapter 1: Theoretical framework

Sustainability and fossil fuels

Sustainability is the common thread that ties together the two major themes contained in this dissertation. As we continue grow our global economy, we require ever increasing amounts of fossil fuels (coal, petroleum, and natural gas) that are becoming more difficult to find and more expensive to produce. Currently, nearly nine tenths of the 500 EJ (exajoule or 10^{18} J) of primary energy consumed on the planet come from burning fossil fuels (BP 2008; EIA 2007d).

The supply of a non-renewable resource such as fossil fuel can only last so long. Yet, demand has continued to increase throughout the last century as populations and standards of living have both increased. According to the theory put forward by M. King Hubbert (1956), production rates follow a bell shaped curve over time, and when half of the reserves are consumed, the peak rate of production is reached. After this point, production rates continue to decline as the resource is depleted and more difficult and costly to extract. In 1956, Hubbert predicted peak United States (US) petroleum production would occur in 1965 and peak world production would occur in 2000 (Hubbert 1956).

In the US, peak petroleum occurred in 1970 (coterminous 48 states), after which America became dependent on imports. Today, approximately two-thirds of the petroleum the US consumes is imported, and over half of these imports are from OPEC (Organization of Petroleum Exporting Countries) members [calculated from

(EIA 2007a)]. According to industry forecasts, as production declines in non-OPEC decline, between 2009 and 2015, the market will once again be dominated by the handful of OPEC countries in the Middle East that still have the capacity to increase production (Campbell 2000; Kerr 2005).

The end of cheap petroleum is a matter of concern for the global economy, particularly the transportation, petrochemical, and agricultural industries. Globally, conventional oil discoveries peaked in 1962. According the US Energy Information Administration (EIA) (2007a), global oil consumption exceeded production in 2006. However, accurate determination of the point at which peak production occurs can only effectively be made in hindsight; prediction requires making assumptions about the total amount of petroleum there is left to find as well as assumptions about technology increases, and economic and policy landscapes. Therefore, predictions tend to be controversial. Predictions range from the present (Deffeyes 2005) to decades later (EIA 2008a). At the 2007 annual meeting of the Association of Petroleum Geologists (AAPG) scenarios were described that put the peak at 2020, 2030, or 2040 (Kerr 2007). Peak production of natural gas and coal are expected to occur after that of petroleum.

To extract heat energy from hydrocarbon fuels, the hydrogen and carbon must be oxidized, producing water (H_2O) and carbon dioxide (CO_2) as byproducts. Combustion of these fuels (along with cement production) releases nearly 8.5 Gt C yr^{-1} of CO_2 into the atmosphere (Marland, Boden, and Andres 2009). The rate of emissions has increased rapidly in the last three decades; since 1751, over half of the

cumulative 340 Gt C emissions have occurred between 1980-2007 (Marland et al. 2007a; Marland, Boden, and Andres 2009) (Figure 1).

As a result, the mean global atmospheric concentration of CO₂ is higher today than it has have been in the last 650,000 years and perhaps the last 20 million years (IPCC 2007). Since 1850, eleven of the twelve warmest years on record occurred between 1995 and 2006 (IPCC 2007). This warming has led to a rising of mean global sea levels of 3.1 mm per year since 1993 (IPCC 2007).

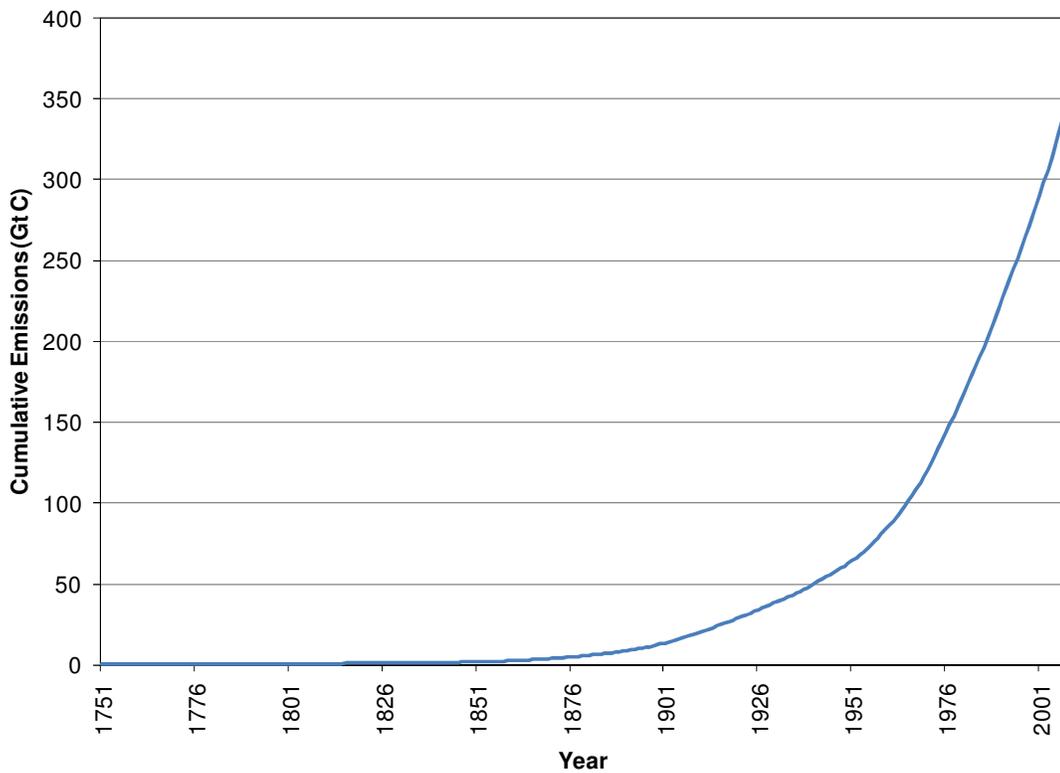


Figure 1. Cumulative emissions from global consumption of fossil fuels and cement manufacture since 1751 (Marland, Boden, and Andres 2007).

The global economy and the global ecosystem

The global economy exists within the global ecosystem because it depends on the input of raw materials and energy. The economic engine transforms these into economic services, and then outputs waste materials, pollution, and low-grade heat energy back into the ecosystem (Figure 2a). The economic services (transportation, shelter, etc.) augment the ecosystem services (clean water, oxygen, etc.) we receive from nature.

As the economy grows, it demands more resources and more energy from the global ecosystem, and as a result, creates more waste which the global ecosystem must absorb. A larger the economy has a greater rate of throughput of matter and energy, and converts these raw materials into waste at a faster rate. In a so-called "full world" (Daly and Farley 2004), we trade ecosystem services for environmental services (Figure 2b). The global ecosystem is finite, with a fixed flux of solar energy. However, global economic growth is exponential. At some point, there will be limits to the rate at which raw materials and energy resources can be consumed and converted to economic services.

Growth in the global economy has required the dramatic expansion of energy consumption, particularly fossil fuels, leading to the accumulation of the Greenhouse Gases (GHGs) in the atmosphere; the waste product of fossil-fuel combustion. Ultimately, throughput cannot continue to increase forever in a finite world. At some point the current system will no longer be sustainable. With the threat of climate change upon us and with the tightening supplies of fossil energy resources, we find

that our current system cannot be sustained indefinitely. Under these circumstances we are faced with the challenge of making changes to the system.

Systemic changes will not be immediate, but one of the first steps toward sustainability would be more prudent management of waste and by-products, such as residue biomass and Municipal Solid Waste (MSW). By using these resources we can improve the efficiency of the global economic engine: maintaining the same level of welfare without increasing material and energy throughput, thereby reducing some of the demands on the global ecosystem (Figure 2c).

Decisions we make now about energy have the potential to dramatically change the world over the next century. If we accept that the current status quo is unsustainable in terms of our reliance on an exhaustible energy supply and the limited capacity of the global biosphere to absorb the waste products of our economic system, then the motivation for making changes in the energy system should be based within the context of sustainability. Allenby (1999) suggests that we must either commit to substantial changes to our current economy, technology, and culture, or face an disruption in those systems along with an unmanaged population decline (Table 1).

The dimension of time is not considered explicitly in the response choices given by Allenby (1999) in Table 1. However, it is implied that if we act too quickly and erratically, we risk the first option, but if we move too slowly, we risk the fourth option. Therefore, one challenge to developing and implement sustainable energy options will be to have them evolve from our current energy infrastructure, and out of

our current economic and cultural systems. Otherwise, we risk disruption of these systems. A revolution, by definition, cannot be sustained.

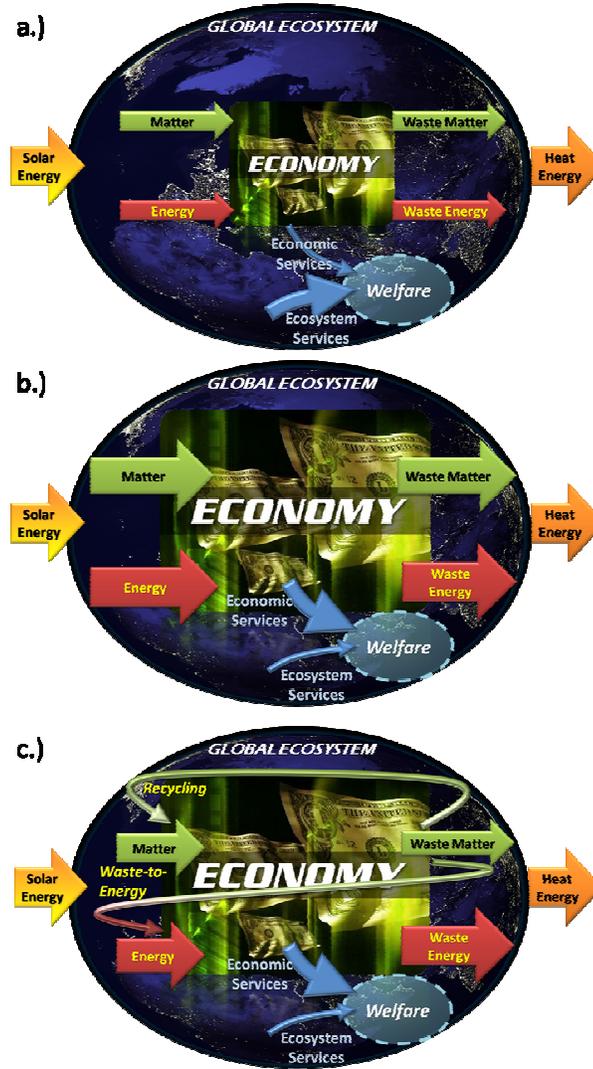


Figure 2. The global economic engine in within the global economy. As the economy grows, throughput increases and some ecosystem services are exchanged for economic services. Better management of waste streams offer a way to reduce throughput. Based on Daly and Farley (2004).

Table 1. Philosophical approaches and outcome (Allenby 1999).

Response Strategy	Effect of Technology	Implications
Radical Ecology	Return to low technology	Unmanaged population crash: economic, technological, and cultural disruption
Deep Ecology	Appropriate technology, low technology where possible	Lower population, substantial adjustments to economic, technological, and cultural status quo
Industrial Ecology	Reliance on technological evolution within environmental constraints- no basis for low technology unless environmentally preferable	Moderately higher population, substantial adjustments to economic, technological, and cultural status quo
Continuation of Status Quo	Ad hoc adoption of specific mandates, little effect on overall trend	Unmanaged population crash: economic, technological, and cultural disruption

Defining sustainability

In June of 1972, the United Nations Environment Programme (UNEP) was formed during the Conference on the Human Environment (UNCHE) in Stockholm, Sweden. It was the first meeting of the United Nations (UN) to specifically address the environment- recognizing that the “environment is a major issue which affects the well-being of peoples and economic development throughout the world” (UNCHE, 1972). While energy was not a specific focus of the meeting, two concluding principles are directly applicable to the current situation regarding fossil fuel use:

Principle 5 states that “non-renewable resources of the earth must be employed in such a way as to guard against the danger of the future exhaustion and to ensure the benefits from such employment are shared by all mankind” and Principle 6 states that “the discharge of... substances... in such quantities or concentrations as to exceed the capacity of the environment to render them harmless, must be halted to ensure serious or irreversible damage is not inflicted upon ecosystems” (UNCHE, 1972). Though “sustainability” was never mentioned at UNCHE, these two principles of nonrenewable resources and finite environmental sinks would set the framework for thinking in terms of sustainable resource utilization, and as such, these two principles would form the core of most definitions of environmental sustainability.

Sustainable development was defined in 1987 by the World Commission on Environment and Development (later renamed to the Brundtland Commission) as “development that meets the needs of the present without compromising the ability of future generations to meet their own needs” (WCED, 1987). While this definition captures the spirit of the UNCHE guidelines, it leaves many issues vague, such as what constitutes a present need and what time frame should be used in considering future generations.

Since then, definitions of *sustainability* have proliferated, and forming a consensus about what *sustainable* specifically means has proven elusive. Hundreds of definitions have been proposed since the concept was introduced in the 1980s (Durbin 1997). Some lament that the term is in danger of becoming either “pornography” in its relativism (“we will know it when we see it”), or a society’s grand desiderata, “a

landfill dump for everyone's environmental and social wish lists" (Goodland and Daly 1996).

Consensus has been elusive for three main reasons. First, assessing whether a given practice is sustainable or unsustainable (the distinction is typically assumed to be binary) is an amalgamation of objective measurements and subjective human values (Thompson 1997). The set of criteria that define sustainability have their foundation in the form of human ideals. Thus, the term "sustainability" applies to both a potential physical state and an underlying ethical substrate; a sustainable practice is a physical manifestation of a normative ethic. In other words, there is a difference between sustainability in theory (confronting the normative issues of what is desirable and why) and sustainability in practice.

Second, there are different dimensions to sustainability that are not always commensurate. "Sustainability" can refer to social sustainability, economic sustainability, or environmental sustainability. Meeting the needs of one dimension can often be at the detriment of another. Moreover, the definition can also refer to a steady state or dynamic process within these various dimensions.

Third, the definition of sustainability is dependent on the relevant spatiotemporal scale of application. Scale can refer to cartographic scale (relational scale in representation on maps), analysis scale (the size of units which phenomena are measured), or phenomenon scale (the scale at which geographic structures exist) (Montello 2001). Not only are phenomenon scale-dependent and defined by the analysis scale (Montello 2001), the operational scale is a social construction that

defines *a priori* the extent to which analysis of a phenomenon is applicable (Marston 2000). Within this conceptual framework, the choice of scale produces the geographic structure of space (Lefebvre 1991), and thus defines the methodology and the metrics employed: the smallest non-divisible unit, the breakpoints between different levels of organization, and the broadest maximum extent of relevance. These structures can be manipulated in such a way to define environmentally sustainable management schemes (Frimpong et al. 2005). Therefore, the interpretation and any generalizations for a given study are only applicable and can only be understood within the scale framework used in the study.

The 3E trilemma

The three simultaneous goals of growing the economy, procuring of energy resources, and preserving the environment interact with each other in complex ways, and this interaction is known as the 3E trilemma (Figure 3).

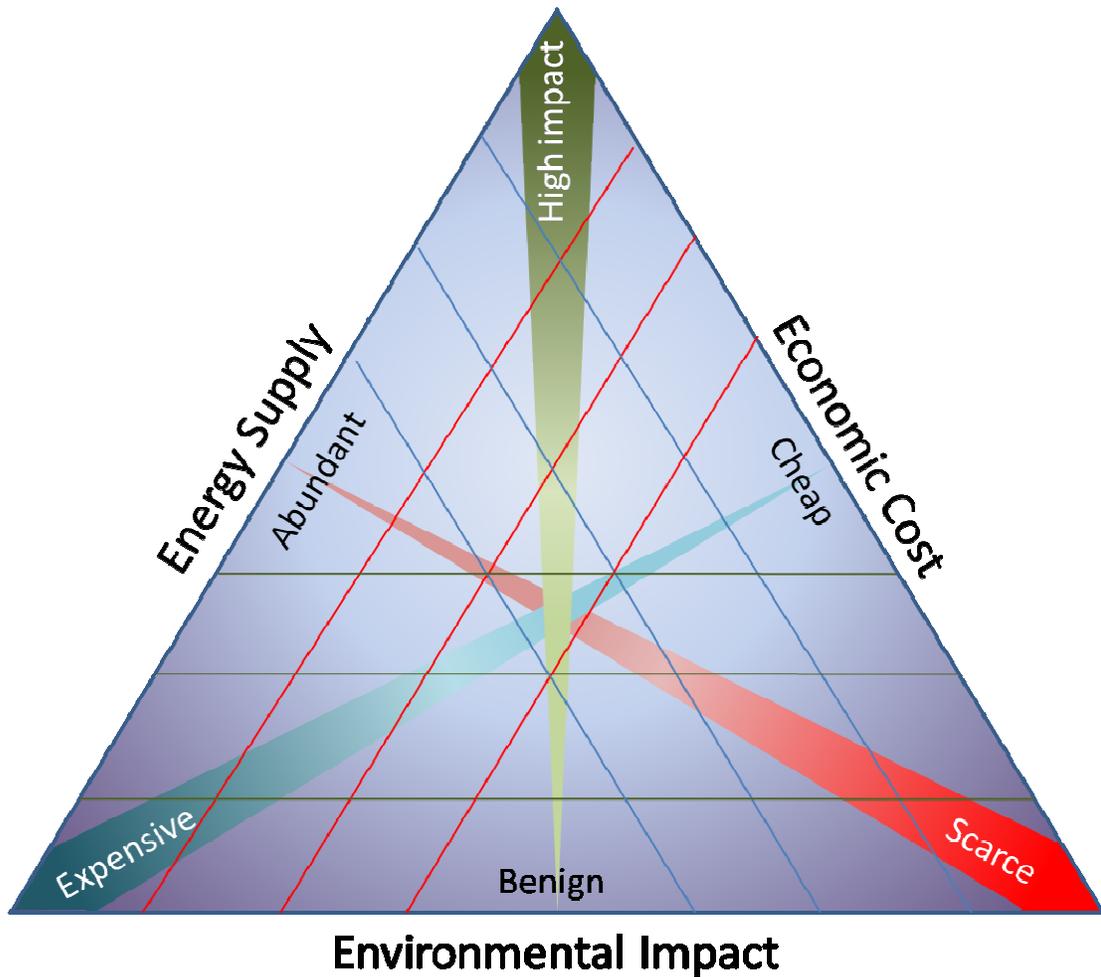


Figure 3. Diagram of the energy-environment-economy trilemma. Starting at the center, as a point moves in this space, it can improve the condition of two of the three parameters at the expense of the third.

Simplistically, when applied to fossil fuels, petroleum is relatively abundant and currently inexpensive given its energy density. Coal is cheaper and more abundant than petroleum, but also has more environmental impact (EIA 2007b).

Natural gas is cleaner burning than petroleum (less carbon per energy), but it is also more expensive (EIA 2007b).

The 3E trilemma summarizes the challenge of renewable energy. In the face of the climate crisis we look to develop more environmentally benign sources of fuel, but they are proving to be more scarce and more expensive to develop relative to fossil fuels. This trilemma is also applicable to bioenergy specifically: palm oil from Indonesia and Malaysia is becoming an inexpensive and prolific feedstock for biodiesel, but it is grown on cleared tropical rainforests and dried peat resulting in a huge environmental impact (Fargione et al. 2008). Corn ethanol is more benign than palm oil biodiesel, though production of corn does lead to nitrate runoff and eutrophication of the aquatic systems (Hill et al. 2006). However, on a energy per hectare basis, it is less abundant, and, on a cost per energy basis, it is more expensive than palm oil biodiesel (FAO 2008). Future hopes are placed in cellulosic ethanol, as it would dramatically increase the supply of ethanol, but the process is more expensive than grain distillation (FAO 2008) .

This trilemma space is dynamic, however. Technological improvements can increase supply, reduce impact, and reduce cost, but rarely all at the same time. Economies of scale reduce cost and increase supply, but often at the expense of environmental impact. For example, improvements in technology that come with an economy of scale are projected to reduce the cost of cellulosic ethanol, but to further increase supply will require increasing amounts of biomass feedstocks. To provide this, farmers and forest managers will have to remove greater amounts of agricultural

and forestry residue, or will have to expand cropping systems into land not currently in production, increasing the environmental impact.

The trilemma is also dynamic with respect to exhaustible resources; as petroleum supplies decrease, the fuel will become more scarce and more expensive. This will make the alternatives more attractive, depending on which dimension of the trilemma takes precedent. For example, coal is abundant and relatively inexpensive. On the other hand, biofuels are more expensive than coal. Biofuels could, however, could become cost competitive with petroleum and are potentially more environmentally benign, though it is doubtful that enough bioenergy could be produced to replace petroleum completely.

The challenge to policy makers is to create incentives to balance the three dimensions, and the optimal scales of production. Given an energy source, there is an optimal scale of production at which each of these factors can be met sustainably. However, without appropriate valuation of the environment (such as regulation or a cap-and-trade carbon policy that prices emissions), these scales of production will be inconsistent with each other. Ecological scale is related to the carrying capacity for a phenomenon. The optimal ecological scale is the maximum extent to which an ecosystem can support the activity without diminishing its capacity to support the activity in the future. It is the scale of an activity to which the natural capital provided by the ecosystem is not depleted and that the ecological sink can absorb waste at the rate at which it is produced (Goodland and Daly 1996).

Economic scale describes the relationship of outputs to increasing inputs. In a situation where inputs are increased by a factor m , if outputs increase by more than m , there are increasing returns to scale, if outputs increase by exactly m , there is a constant return to scale, and if outputs increase by less than m , there is decreasing return to scale. Finally, if output *decreases* with increasing input, there is a diseconomy of scale. The optimal economic scale is where marginal cost of increasing the factors of production equal marginal benefit of increased output. The corresponding geographic scale of the optimal economic scale will be equal if and only if the ecological factors of production (Ricardian land, soil nutrients, energy inputs, pollution sinks, etc.) are appropriately valued. If overvalued, then the optimal ecological scale will exceed that of the optimal economic scale, and if undervalued, the optimal economic scale will exceed the optimal ecological scale.

While research and development to are required to increase supply and lower costs, at the same time, regulation or pricing mechanisms are needed to balance the environmental dimension. While there is an optimal scale to balance these factors, there no scale of production that can maximize all the three factors simultaneously. The implications of this are that the consumption of fossil fuels, biofuels, or any other energy source leaves us choosing between factors based on the scale of production; trading off between abundance, cost, and environmental impacts. Sustainability is a necessary criterion for analyzing solutions to the 3E trilemma, and will force us to chose options that lie away from the edges of the triangle in Figure 3.

Sustainability and bioenergy from residue biomass

The concept of sustainability serves as the theoretical framework and the questions here are framed in terms of environmental and economic sustainability, in that utilization of residue biomass is done only up to the point where it both remains economically viable and environmentally benign in the long run. As awareness and political consensus builds regarding climate change, and as energy markets tighten as increases in demand reduce spare capacity in fossil energy production, it is clear we are on the cusp of substantial changes in the global energy portfolio. Much of the current race to biofuels is a result of these factors. While corn-based ethanol has gained much momentum in the US, its potential for expansion is limited and we will soon be making decisions and investing in other bioenergy sources. Depending on how these sources are produced, the bioenergy endeavor can either assuage or exacerbate energy supply shortages and environmental externalities.

Though not easily defined or objectively quantified, the concept of sustainability has become an integral part of bioenergy analysis. Environmentally, bioenergy is seen as a preferable alternative to fossil fuels in the context of GHG emissions, but only if it is produced in such a way as to minimize the negative impacts from land use change and aggressive land management. Economically, bioenergy has begun to demonstrate its profitability, leading to an influx of wealth to rural communities and a means of socially sustaining an agrarian lifestyle (Kojima and Johnson 2005), but only insofar as systems of production remain profitable and competitive with other energy options.

Pursuing biofuels as a sustainable alternative to fossil energy requires scientific research, such as that presented here, to direct policy makers and private industry so that this energy source can be brought to end users in the most efficient and environmentally benign way. This has prompted many scientists and professional organizations to examine biofuels within the context of sustainability. The International Energy Agency (IEA) is currently exploring a method for sustainability certification in biofuel production (van Dam et al. 2006). In the US, the Ecological Society of America adopted a formal position of biofuel sustainability that stresses systems thinking, conservation of ecosystem services, and scale alignment (ESA 2008).

Given that global population and standards of living are likely to continue increasing for the foreseeable future, ever more resources are needed to meet the growing global demand for energy. Therefore, the research presented here adopts the concept of sustainability under the context of expanding the utilization of byproducts and waste so that we may improve the standard of living for an expanding population, but doing so such that it minimizes possible environmental externalities. Increasing the use of by-products and waste is an essential component to sustainably meeting some part of the increasing global energy demand and more efficiently deriving added utility from goods we produce, thereby reducing throughput in the global economic engine.

Chapter 2: A method for estimating the temporal and spatial patterns of carbon dioxide emissions from national fossil-fuel consumption

Preface

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Published in:

Tellus (2008) 60B, 1-10

Abstract

A proportional methodology is presented for estimating fossil-fuel consumption and concomitant anthropogenic carbon dioxide (CO₂) emissions. This methodology employs data from representative sectors of the fossil-fuel market to determine the temporal (monthly) and spatial (provincial/state) patterns of fuel consumption. These patterns of fuel consumption are then converted to patterns of CO₂ emissions. The purpose is to provide a procedure for determining anthropogenic emissions from countries where a full accounting of emissions is impracticable due to limited data availability. To demonstrate the effectiveness of the proportional

methodology, it is applied to data from the United States (U.S.) and the results are compared to those from an independent methodology that employs a thorough accounting of all fuel sectors. Although there are some discrepancies between the two sets of CO₂ emissions estimates, overall, the approaches yield similar results. Thus, the proportional methodology developed here represents a viable method for estimating anthropogenic CO₂ emissions for other countries with limited data availability.

Introduction

Global warming from greenhouse gases such as CO₂ has been an issue of increasing scientific concern since Keeling (1960) published the Mauna Loa, Hawai'i atmospheric CO₂ concentration time series. Over the last four decades, further tabulation and modeling has been done to monitor and predict atmospheric CO₂ concentrations and the resultant climactic effects. Today, atmospheric CO₂ levels are higher than they have been in the last 420,000 years and perhaps the last 20 million years (IPCC, 2001). Moreover, the rate of increase in the atmospheric concentration over the last century is the greatest it has been in at least 20,000 years (IPCC, 2001). Of the anthropogenic activities that are related to increased atmospheric CO₂ concentrations, fossil-fuel combustion is the most substantial contributor, representing 80% of all anthropogenic CO₂ emissions (Marland, Andres, and Boden 1994).

Currently, many researchers are conducting a multitude of detailed observations of carbon (C) flux from natural processes, land cover change, and atmospheric concentrations of greenhouse gases on increasingly finer temporal and spatial scales. These observations feed a number of models to project past and present potential climates using different assumptions about natural and anthropogenic processes. Yet, fossil-fuel-based emissions are still only compiled at an annual time step for most countries; comparatively little work has been done to determine the seasonal distribution and sub-country spatial distribution of those emissions.

For example, the Carbon Dioxide Information Analysis Center (CDIAC) in the Earth Sciences Division at Oak Ridge National Laboratory (ORNL) maintains an extensive database on annual CO₂ emissions from each country (http://cdiac.esd.ornl.gov/trends/emis/tre_coun.htm) (Marland, Boden, and Andres 2005). These data were derived from apparent fuel consumption, summing domestic fuel production and imports, and subtracting exports, bunker fuels, changes in stock and production of non-fuel products (Marland et al. 1989). To ascertain the anthropogenic C emissions from fuel consumption, the C content of each specific fuel was determined via chemistry and empirical observation (Marland, Andres, and Boden 1994; Marland and Rotty 1984). Similarly, the United Nations Framework Convention on Climate Change (UNFCCC) (2005a) maintains an annual database of national greenhouse gas emissions based on self-reports from participating countries, based on reports by the Intergovernmental Panel on Climate Change (IPCC) methodological documents (IPCC, 1996). In addition, the Emission Database for

Global Atmospheric Research (EDGAR) (Olivier 2002) employs sectoral energy consumption data from the International Energy Agency (IEA) and other consumption and production data to derive annual anthropogenic greenhouse emissions into the atmosphere for each country and on a global grid. A comparison of the national annual emissions estimates contained in these databases can be found in the State of the Carbon Cycle Report (SOCCR) (Marland et al. 2006).

The lack of fossil-fuel-based emissions estimates on a seasonal and sub-national resolution are due to a lack of detailed fuel consumption data. Nevertheless, estimates of the seasonal distribution of fossil-fuel-based emissions are essential for enhancing models of potential future climates.

To date, the most spatially and temporally detailed account of fossil-fuel-based CO₂ emissions stem from the work of Blasing et al. (2004a; 2004b; 2005a; 2005b) at CDIAC. In these studies, a full accounting approach (utilizing complete statistics on consumption of all fossil fuel end-products by all market sectors) was used to produce monthly CO₂ emissions estimates for the U.S. (Blasing, Broniak, and Marland 2004b; 2005a), and annual CO₂ emissions estimates for each state and the District of Columbia (Blasing, Broniak, and Marland 2004a; 2005b). These are notable achievements, given that the U.S. is responsible for over a fifth of global emissions in 2003 [calculated from (Marland, Boden, and Andres 2005)]. That year, 20 countries were responsible for 75% of the global fossil-fuel-based CO₂ emissions [calculated from (Marland, Boden, and Andres 2005)]. However, among the top 20 fossil-fuel consuming countries in the world, not all countries maintain data detailed enough to

yield monthly fossil-fuel-based CO₂ emissions estimates from a full accounting approach. Thus, the Blasing et al. (2004a; 2004b; 2005a; 2005b) studies represent the limit in spatiotemporal resolution for CO₂ emissions estimates from a full accounting approach, which cannot feasibly be applied to all countries.

To improve the temporal resolution to monthly time intervals for countries besides the U.S., other methods of estimating CO₂ emissions from fossil-fuel consumption are necessary. A technique developed by Rotty (1987b) explores other variables that are potential indicators of fossil-fuel use (e.g., electricity production and climatological comparisons to similar countries). These can be used as estimators for fossil-fuel use, and hence CO₂ emissions. Varying proxies, depending on the country, are used to estimate emission levels (Rotty 1987a), but because different proxies are employed for each country, these methodologies lack universal applicability. In addition, population distribution has been used as a proxy to estimate the spatial distribution of emissions on a 1° x 1° latitude-longitude grid (Andres et al. 1996; Brenkert 2003). While this improves the estimates for the spatial distribution of emissions, these studies are only on an annual time scale and there is some evidence that population distribution is not perfect as a proxy for the distribution of emissions (Blasing, Broniak, and Marland 2005b).

Therefore, the goal is to devise a methodology that can be universally applied to determine monthly fossil-fuel consumption for all countries, while at the same time, producing results that are consistent with the annual data based on the U.N. statistics as prepared by CDIAC. Below, a proportional methodology is illustrated

and assessed using U.S. data. The U.S. is an ideal case for which to evaluate this methodology, not only because it is the world leader in fossil-fuel-based CO₂ emissions, but because of the detailed fuel consumption data that is maintained by the Energy Information Administration (EIA) of the U.S. Department of Energy. The U.S. data availability allows a direct comparison of the proportional method proposed here against the full accounting-based approach used by Blasing et al. (2005a; 2005b). Moreover, the proportional methodology is able to produce estimates on a monthly state-to-state level, enhancing the spatiotemporal resolution of emissions estimates from the Blasing et al. (2004a; 2004b; 2005a; 2005b) studies. Most importantly, the proportional methodology provides a means to estimate monthly CO₂ emissions for countries that do not have the same data collection commitments as the U.S.

This paper represents the publication of the methodology that produced the monthly national CO₂ emissions data that led to the results published in Bakwin et al. (1998), Lee et al. (2001), and Losey et al. (2006). Other publications that use the results of this methodology are in various stages of manuscript preparation by the authors of this manuscript and by other carbon cycle researchers.

Methodology

The underlying principle of the proportional methodology is to determine a set of market sectors whose combined fuel consumption closely approximates the

monthly distribution of total fuel use for each state (or other sub-national geographic unit). Data for fuel consumption in these sectors is used to parse the annual emissions data for a given country into sub-annual and sub-geopolitical units. Representative data is collected for each fuel type, solid (coal), liquid (petroleum-based fuel products) and gas (natural gas), and are assumed to accurately represent the seasonal and spatial distribution of the entire fossil-fuel market if they comprise a substantial portion of the entire market.

Government agencies and industry-kept statistics serve as the source for direct and indirect indicators of fuel consumption. The data employed are in physical (mass) units rather than monetary units to avoid complications from commodity pricing. For the U.S., all data are obtained from the EIA, which keeps monthly fuel consumption data for each state.

Natural gas data are obtained from the EIA (1984-2004) publication, *Natural Gas Monthly*. The EIA maintains monthly records of deliveries to all consumers by state, which occur after changes in stock (injections less withdrawals into storage) in the production chain. It is assumed that changes in stock at the consumer side of the market are small compared to actual consumption. These data also do not take into account “lease and plant fuel” and “pipeline and distribution use” (gas used in the production/processing and delivery/storage stages), which account for approximately 5% and 3% of total gas consumption, respectively [calculated from (EIA, 1984-2004)]. Data for gas consumed in these processes are not maintained at the monthly

per state level. Therefore, the proxy coverage accounts for roughly 92% of all natural gas consumed in the U.S. as seen in Table 2.

Table 2. Sectoral distribution of U.S. CO₂ emissions from natural gas (1997-2004) [calculated from (EIA, 1984-2004)].

Consumer	Percent of CO ₂ emissions	Cumulative Percent
Industrial	34.5%	34.5%
Electric Power	22.2%	56.7%
Residential	21.5%	78.2%
Commercial	13.8%	92.0%
Vehicle Fuel	0.1%	92.1%

Liquid fossil fuels present a more daunting challenge as crude oil is converted into many products, both fuel and non-fuel. From the EIA (1983-2004) publication, *Petroleum Marketing Monthly*, this method uses average daily sales data of four main types of fuel: motor gasoline (all grades), distillate fuel oils including kerosene, kerosene-type jet fuel, and propane. These four products were chosen because together they best predicted the spatiotemporal distribution of total liquid consumption when analyzed in a multiple regression-based model selection. In addition, they represent the most complete records of all individual petroleum products and the four top products of petroleum refining, together comprising about 86% [calculated from (EIA, 1983-2004)] of the U.S. petroleum market (Table 2). The daily mean sales values were multiplied by the calendar days per month to obtain

monthly sales. Again, this assumes consumer changes in stock are insignificant to consumption, which may be more problematic for products such as propane.

Table 3. Sectoral distribution of U.S. CO₂ emissions from liquid fuels (1983-2002) [calculated from (EIA, 1983-2004)].

Fuel Type	Percent of CO ₂ emissions	Cumulative Percent
Motor Gasoline	47.8%	47.8%
Distillate Fuels & Kerosene	23.5%	71.3%
Kerosene Type Jet Fuel	10.0%	81.3%
Propane (LPG)	4.2%	85.5%

Electrical utility coal consumption is the proxy chosen to represent CO₂ emissions from solid fossil fuels, as 87.3% of the coal consumed in the U.S. for the years 1984-1999 [calculated from (EIA, 1999)] was by this sector. These data are maintained by the EIA (1980-2003) in their publication, *Electric Power Monthly*.

For a given state, month, and fuel type (i.e., gas, liquid and solid), the reported consumption is divided by the sum of all states' consumption for that fuel type for the entire year. The resulting quotient, state monthly consumption over national annual consumption, represents the proportion of national fuel consumed by a given state in a given month. This proportion is then multiplied by the national annual CO₂ emissions estimate by CDIAC (Marland, Boden, and Andres 2005) based on statistics from the U.N. for each specific fuel type. For a given state and given month, this gives an estimate of the proportion of CO₂ emissions that occurred from combustion

of a particular fuel. For example, in December 2000, California consumed 213,789 MMcf ($6.05 \times 10^9 \text{ m}^3$) of natural gas, which is about 1% of the 20,772,594 MMcf ($5.88 \times 10^{11} \text{ m}^3$) of natural gas consumed in all states for the entire year (EIA, 1984-2004). Given that annual CO₂ emissions from natural gas consumption in the U.S. were 355 Tg C (Marland, Boden, and Andres 2005), the relative proportion of consumption implies that about 3.65 Tg C were emitted in December in California.

This procedure is carried out for all months and all fuel streams to create three separate time series for CO₂ emissions, one for each fuel type. Summing these across the different fuel types gives a fourth time series, the total monthly CO₂ emissions. For a given period a , time step i , m sub-geopolitical units j , and fuel type k , the total emissions (C) are expressed as follows:

$$(C)_{i,j} = \sum_k \frac{x_{i,j,k} \cdot T_{a,k}}{\sum_{i=1}^n \sum_{j=1}^m x_{i,j,k}}$$

where x_i is the sales or consumption value for time step i , and T is the total CO₂ emissions estimate. The application of this methodology to the U.S. produces monthly estimates for CO₂ emissions by state, so the time step i is set to one month, the period length n is assumed to equal 12 months (one year) and m is equal to 51 for the U.S. (50 states plus the District of Columbia). Because the monthly state proportions of consumption sum to unity, emissions estimates are mutually consistent with the CDIAC annual emissions dataset, T , for the whole U.S. The application to other countries may require different values for i , n and m , based on the temporal and

spatial resolution of the fuel consumption proportional data. This paper uses the elemental mass of carbon in CO₂ emissions estimates; the molecular mass can be obtained by multiplying by the factor 44/12.

Gaps in the proportional data, resulting from missing reports and in the case of the U.S., non-disclosure policies of the EIA, are filled using an interpolation strategy. When a gap is encountered, this strategy first computes the discrepancy between the reported total country consumption and the sum of the consumption from the individual states. This discrepancy is then used to fill the gap. In the case of multiple gaps within a single month, the discrepancy is apportioned according to the ratio of the means of the corresponding month from previous complete years. This procedure thus retains “true zeros” in states that do not consume a particular fuel. For example, California, Connecticut (for most years), the District of Columbia, Hawai’i, Idaho, Maine, Rhode Island, and Vermont consume no coal in the utilities sector, so these “true zeros” are retained. Table 4 displays the percentage of zeros encountered in each dataset, and the percentage of zeros filled by this procedure and the mean discrepancy per month per state from the national total.

Table 4. Zeros filled in proportional datasets. Discrepancy represents the difference between the reported national total and the state sum, and thus the amount filled by the gap filling procedure.

Fuel Type	Consumption = 0 (%)	Filled (%)	Mean discrepancy (per state/ per month)
Natural Gas	0.72	0.72	272.25 MMcf ($7.71 \times 10^6 \text{ m}^3$)
Liquids Gasoline	0.01	0.01	1370 US gal (5186 l)
Distillates	0.19	0.19	38 US gal (143.7 l)
Jet Fuel	6.57	4.61	302 US gal (1144 l)
Propane	7.08	5.12	129 US gal (490 l)
Solids	16.09	2.70	140 Short Tons (127,000 kg)

Results

Figure 4 shows the seasonal pattern of emissions for each fuel type for the entire U.S. from 1980-2002 produced by the proportional method described above. Because of the varying inception dates for the beginning of data collection by the EIA, the total emissions are computed only for the years 1984-2002, representing the overlap of available proportional datasets. Table 5 summarizes the CO₂ emissions from fossil-fuel consumption in the U.S. for these years. This table includes the percent of emissions from each fuel type, the long term trends (the slope from least squares regression), descriptions of the seasonal and spatial distributions, and notes the states with the greatest seasonal flux for each fuel (ranked by the quotient of the standard deviation over the mean).

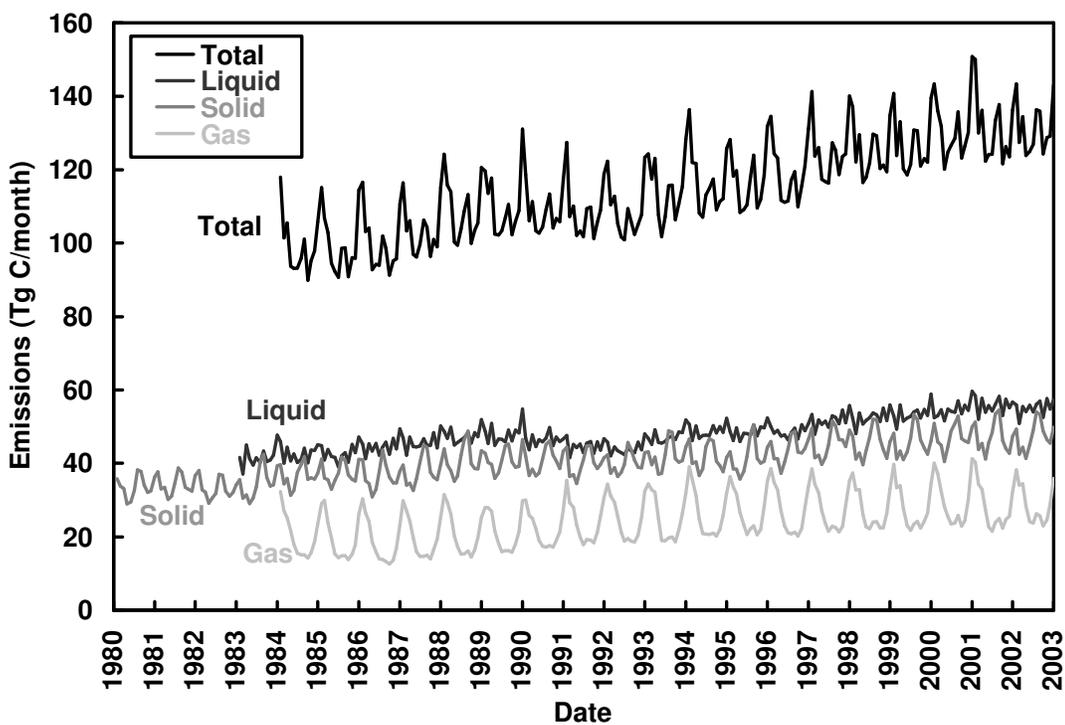


Figure 4. Monthly U.S. CO₂ emissions estimates from fossil-fuel consumption produced by the proportional method. Tick corresponds to January.

Table 5. Summary of U.S. CO₂ emissions from fossil-fuel consumption (for the period 1984-2002).

Fuel Type	Percent of U.S. Fossil Fuel CO ₂ Emissions	Long-Term Trend	Seasonal Trend	Spatial Distribution: Areas of Highest Emissions	Maximum Spatial-Seasonal Flux (100% * std. dev/mean)
Gas	21%	Increasing 7.4 Tg/yr	Peaks in winter	Great Lakes region, TX, LA and CA	Northern states (SD, IL, OH, MN, WI, ND, MI) (52-48)
Liquids	42%	Increasing 8.3 Tg/yr	Relatively uniform; follow length of calendar months	Strongly correlated to population	Northeast (RI, NH, MA, CT, VT) (18-15)
Solids	37%	Increasing 7.8 Tg/yr	Large peak in summer, smaller peak in winter	Bluegrass area, mountain states, WY, ND, and WV	Northwest (OR, WA) (43, 30)
Total	100%	Increasing 23 Tg/yr	Large peak in winter, smaller peak in summer	Densely populated areas (Atlantic Sea Board, CA), and industrial areas (TX, LA, Great Lakes region)	Northeast (RI, CT, NH, MA, VT, NY) (19-16)

Natural gas is the most distinctively seasonal, with higher winter use. This suggests that natural gas is primarily used for heating in the U.S. There is also a small summer peak that is becoming more prominent in recent years due to the increased

use of natural gas for electricity generation to meet the electrical demand created by air conditioning. In contrast, liquid fuels, which are predominately consumed by the transportation sector, are relatively constant throughout the year. Coal use has two peaks, a winter peak due to electric heating and a summer peak due to electricity generation for air conditioning.

These trends are also apparent in Figure 5, showing the mean January and July emissions of each fuel type for all states over the years 1984-2002. In general, northern states have a more drastic increase in natural gas use in the winter than southern states. In contrast, the southern states have a higher demand for coal-fired electricity in the summer than northern states. Emissions from combustion of liquid fuels are relatively constant all year and the spatial distribution is highly correlated to population, with coefficient of determination equal to 0.92 for all years in the dataset. Of all fuel types, the per capita use of liquid fuels is the most consistent state to state (Figure 6). In contrast, population versus annual natural gas carbon emissions is 0.78, and for coal, this correlation drops to 0.37. The correlation between state population and total annual fossil-fuel-based emissions is 0.70. However, there are some notable outliers for per capita emissions in all fuel types. Texas and Louisiana have large petrochemical industries, so consume a disproportionately large amount of natural gas and petroleum. Alaska, with a low population and large reserves, also has disproportionately high per capita petroleum products and natural gas use. Because electricity is more readily transported than coal, electricity generation is done closer to the areas of coal production rather than population centers. States such as

Wyoming, North Dakota, and West Virginia, which are net exporters of coal-based electricity, have per capita emissions rates an order of magnitude higher than the other states.

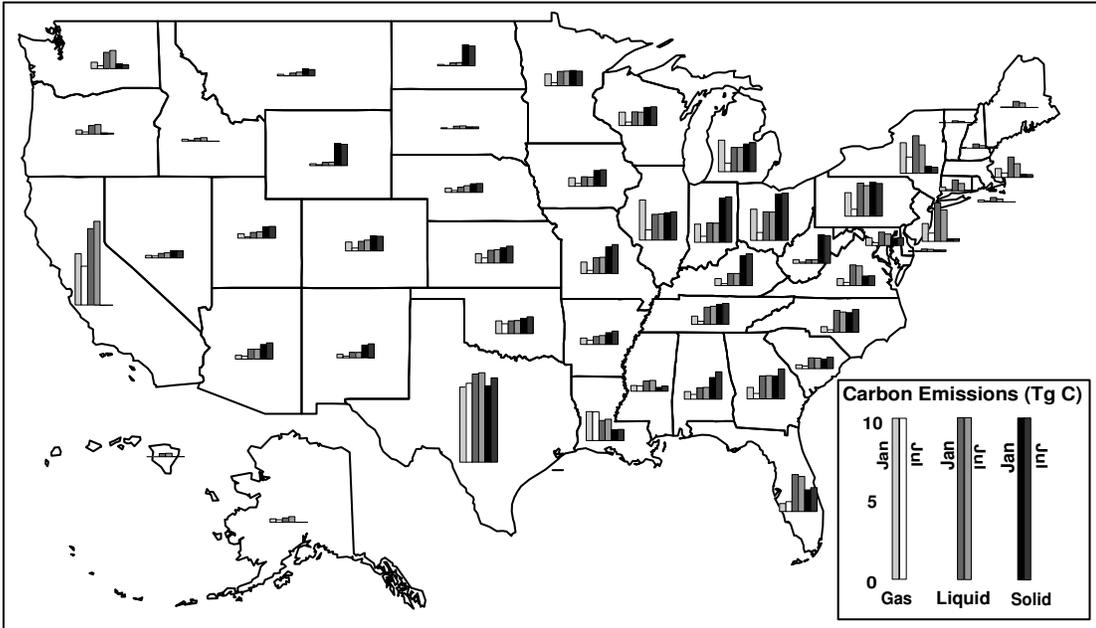


Figure 5. Winter-Summer distribution of CO₂ emissions from fossil-fuel consumption produced by the proportional method. For display purposes, DC and MD data are combined.

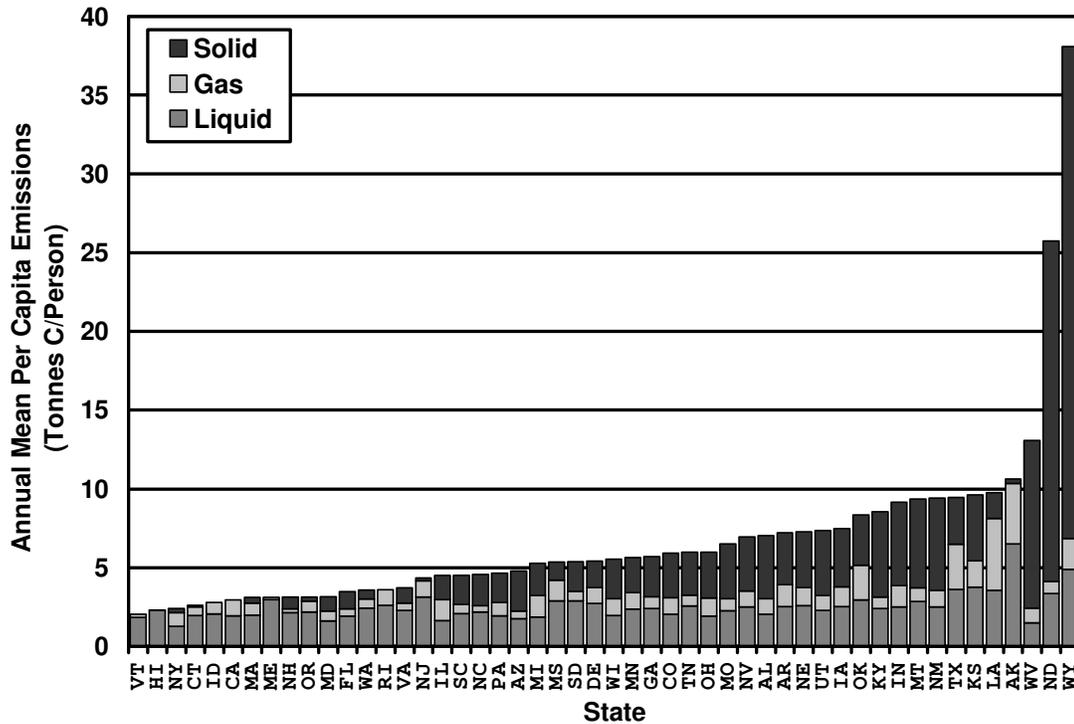


Figure 6. Mean annual CO₂ emissions per capita, by state, produced by the proportional method (1984-2002). DC and MD data are combined. Population data from U.S. Census (2001).

Discussion

To assess the ability of the proportional method to produce reliable CO₂ emissions estimates, the results for the U.S. are compared to the results produced by independent studies. Blasing et al. (2004a; 2004b; 2005a; 2005b) produced emissions estimates by conducting a data-intensive accounting of all market sectors, the fuel quality (heat content) and C content of all fuel combusted, and the combustion

coefficients (to account for incomplete combustion and soot production) for every state in the U.S. on an annual time scale (Blasing, Broniak, and Marland 2005b). This full accounting method was also applied to the entire U.S. on the monthly time scale (Blasing, Broniak, and Marland 2005a). Thus, the comparative analyses between the results from the proportional methodology and the results from the accounting methodology are performed on the temporal and spatial components separately, because the Blasing et al. approach (2005a; 2005b) produced only annual emissions estimates on a state-to-state level (Blasing, Broniak, and Marland 2004a); monthly emissions estimates were only able to be done for the entire U.S. (Blasing, Broniak, and Marland 2004b).

The two methods (proportional and accounting) produce datasets that show very similar seasonal patterns of CO₂ emissions for each fuel type. The differences in the estimates, normalized by the mean total emissions for each given month, are presented in Figure 7. In this figure, a positive result indicates the proportional method gave a higher total than the accounting method. Interannual shifts (e.g., the shift that occurs in the gas time series in Figure 7 between 1990 and 1991) are a result of the differences between the CDIAC annual emissions time series (Marland, Boden, and Andres 2005) based on UN statistics for the U.S. (which the proportional method employs as the total emissions to be parsed) and the 12-month sums of the estimates produced by Blasing et al. (2004b). Blasing et al. (2004b) gives an error estimate of 3-4% for each fuel, and by a sum of squares argument, the total error would be 5-7%. The discrepancies between the seasonal estimates of the two methods, proportional

and accounting, are within this range; the percentage differences are predominantly less than three percent, only once reaching as high as five percent (liquid fuels, December, 1991). Moreover, the mean absolute differences between the results from the two approaches result in a range from about 1 Tg C for each fuel type and about 2 Tg C for total emissions. This gives percentage differences (relative to the mean of the total monthly emissions estimates of both approaches) of 1% or less for all fuel types and less than 2% for the total emissions. These statistics are summarized in Table 6.

Table 6. Mean absolute difference between proportional method and Blasing et al. (2004b) for U.S. CO₂ emissions, 1984-2002 (Temporal Component).

Fuel Type	Absolute Difference (Tg C)	Percentage Relative to Total U.S. Emissions	Mean Monthly Emissions, Both Methods (Tg C)	Percentage Relative to Mean Monthly Emissions
Gas	0.92	0.83	24.66	3.73
Liquids	1.17	1.02	48.67	2.40
Solids	0.66	0.58	42.04	1.57
Total	1.70	1.50	115.36	1.47

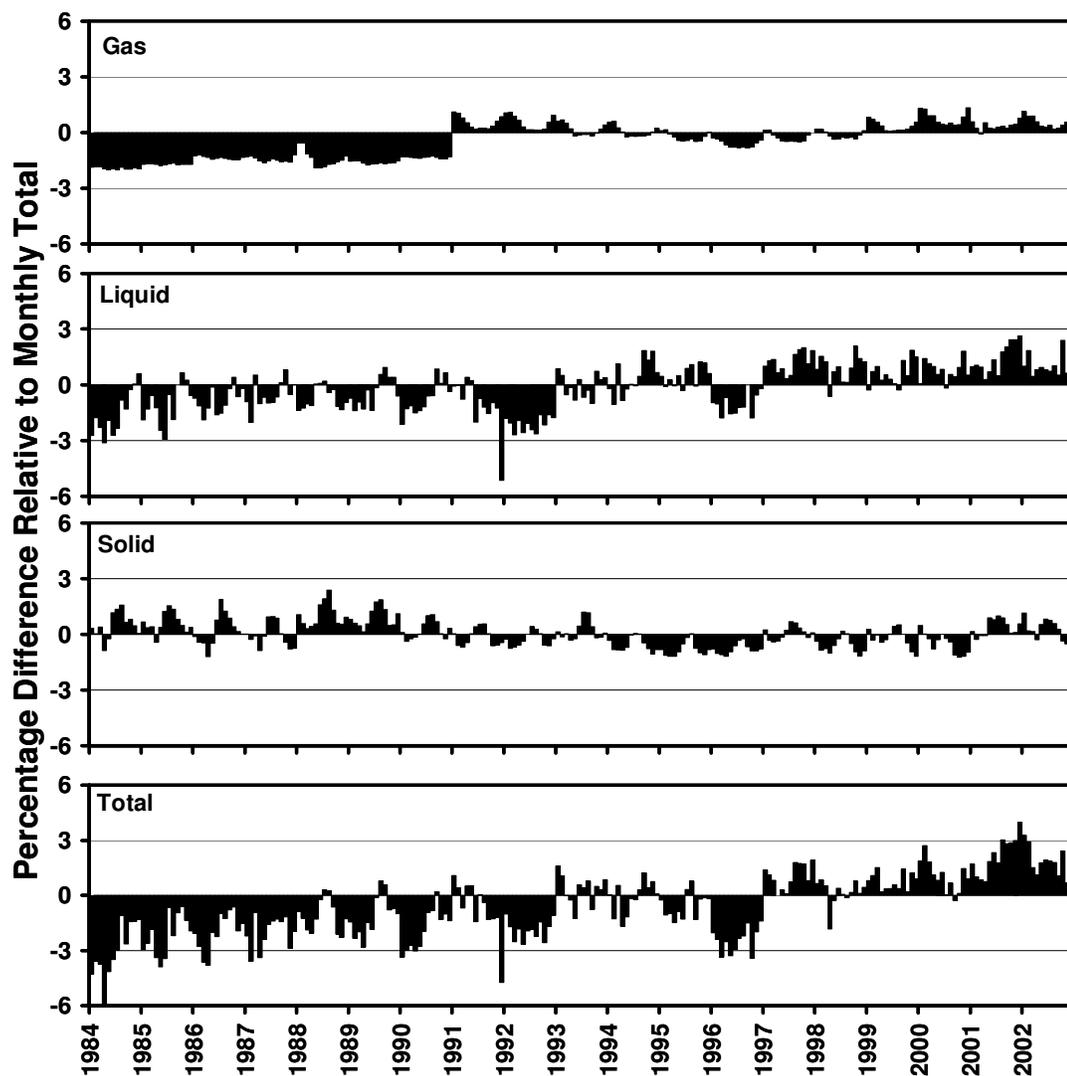


Figure 7. Percentage differences in U.S. monthly emissions estimates between the proportional method and Blasing et al. (2004b) divided by the mean of the total emissions from the two approaches. Tick corresponds to January. Interannual shifts in discrepancies are due to differences between the CDIAC annual database (Marland, Boden, and Andres 2005) and the Blasing et al. (2004b) estimates.

Because no independent monthly data are available at a state-level spatial resolution, to assess the accuracy of the spatial distributions, the annual state totals for each fuel type from the proportional method are compared to the corresponding annual estimates from the Blasing et al. (2005a; 2005b) method. Spatially, a few states represent a large fraction of the U.S. CO₂ emissions, thus the greatest absolute discrepancies between the proportional and accounting methodologies tend to occur for those states.

The top ten differences for the total CO₂ emissions per state are given in Table 7. In this table, a positive value indicates an emissions estimate from the proportional methodology that is higher than the corresponding estimate in the accounting methodology; a negative value indicates a lower emissions estimate from the proportional methodology. Pennsylvania is at the top of the list, having the largest absolute difference as well as a high percentage difference (relative to the mean of the estimates from both datasets). The absolute and percent differences do not rank uniformly, however. For example, Texas and California both have high absolute differences but relatively small percentage differences. On the other hand, Alaska has a very high percentage difference but a low absolute difference. In general, the absolute difference is sensitive to the states with high emissions levels whereas the percentage difference is sensitive to the states with low emissions levels. Moreover, the absolute differences are highly interdependent due to the proportioning from annual state sum employed in the proportional methodology. Therefore, an

underestimate in one state necessarily leads to an overestimate in at least one other state and vice versa.

Table 7. Greatest mean absolute differences between proportional method and Blasing et al. (2004a) dataset for total individual state CO₂ emissions, 1984-2001 (Spatial Component).

State	Difference (Tg C)	Percent Difference	Percentage Relative to Total U.S. Emissions	Mean Annual Total Emissions, Both Methods (Tg C)
Pennsylvania	-14.64	-22.84	-1.06	64.10
Texas	13.94	8.39	1.01	166.24
New York	-10.81	-21.96	-0.78	49.24
Louisiana	-7.78	-16.68	-0.56	46.63
Florida	-5.54	-10.96	-0.40	50.52
Kansas	5.38	24.90	0.39	21.61
West Virginia	-4.75	-18.10	-0.34	26.25
California	-4.17	-4.54	-0.30	91.78
Missouri	3.72	11.63	0.27	32.03
Alaska	-3.71	-46.39	-0.27	7.99

To further elucidate these relative measures, weighted difference scores are employed. The difference scores in Figure 8 represent the percentage difference relative to the given state's proportion of the total consumption. This is algebraically equivalent to the percentage difference relative to the total annual emissions for the entire U.S. In Figure 8, the difference scores for total emissions are less than 2% for all states, indicating a good overall agreement with the Blasing et al. (2004a) dataset.

The largest individual fuel discrepancies occur in coal consumption emissions estimates for Pennsylvania and Texas.

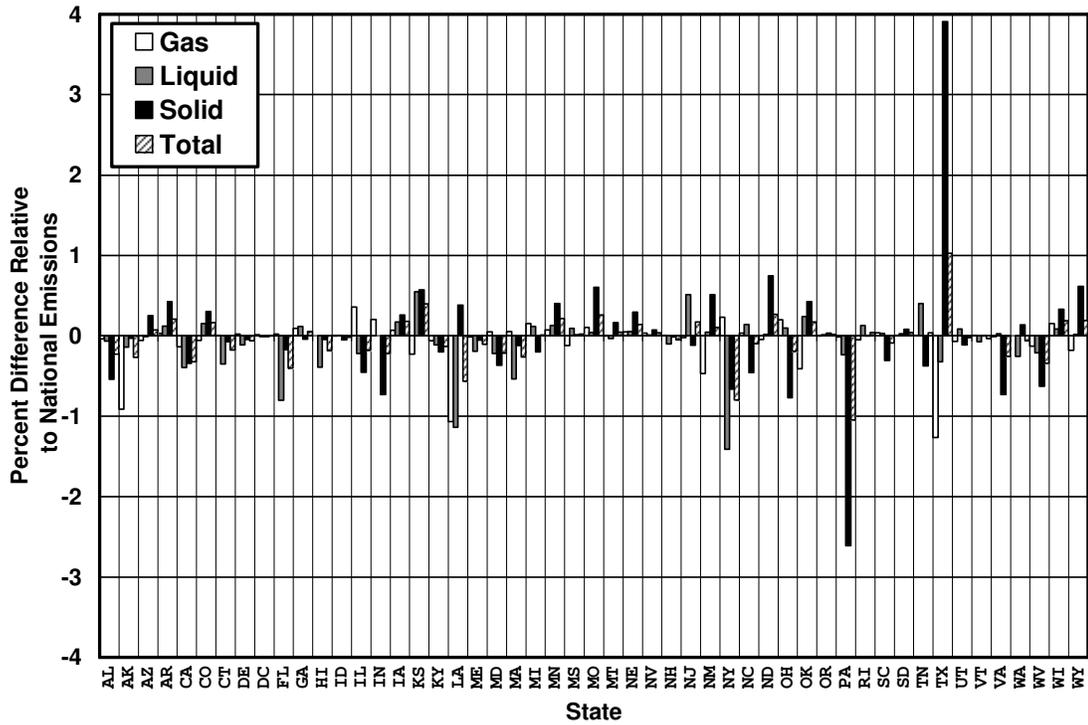


Figure 8. 1984-2001 mean percentage difference between the proportional method estimates and Blasing et al. (2004a) relative to each state’s proportion of national emissions (percentage difference to total national emissions) per fuel type.

In Pennsylvania, there is a large amount of coal that is consumed in non-utility uses. For the years 1984-1999, the coal used in electric utilities accounts for roughly 73.2% of the total coal use in the state [calculated from (EIA, 1999)], whereas the

total percentage of coal used in electric utilities for the U.S. was 87.3% for those years [calculated from (EIA, 1999)]. This causes an underestimation of Pennsylvania emissions estimates by the proportional method presented here because the proportional variable is assumed constant for all states. A low electrical utility use in this state creates an underestimation, because in this case, the method assumes only a 12.7% (= 100% - 87.3%) non-utility coal use, when this number is actually 26.8% (= 100% - 73.2%) for Pennsylvania.

Overestimation of emission levels for Texas in comparison to the Blasing et al. (2004a) dataset is due to three causes. First, the underestimation of states such as Pennsylvania causes an overestimation in other states to produce a national total that is consistent with the U.N. national data. Due to the use of relative proportions, this effect is more pronounced in the states with higher emissions, such as Texas. Second, there is a high proportion of electric utility coal use (about 94.9% in 1984-1999) in Texas [calculated from (EIA, 1999)], compared to a national average of 87.3% [calculated from (EIA, 1999)]. The reasoning is similar to the Pennsylvania case discussed above, except in the opposite direction. Third, the coal consumed in Texas has a heat content 31% lower than the national average (14.71 Mbtu short ton⁻¹ or 17.1 MJ kg⁻¹ compared to a U.S. average of 21.57 Mbtu short ton⁻¹ or 25.1 MJ kg⁻¹ for 1984-1999 [calculated from (EIA, 1984-1999)]. Because the Blasing et al. (2005b) method uses inputs in units of heat content and specifically accounts for this lower heat content, this creates a lower estimate of CO₂ emissions relative to the proportional methodology, which does not incorporate this state specific information.

Differences in C content can also impact the estimates, but in this case, the C content of the coal in Texas is essentially equivalent to the national average (212 lbs CO₂ Mbtu⁻¹ or 24.9 g C MJ⁻¹ versus a national average of 209 lbs CO₂ Mbtu⁻¹ or 24.5 g C MJ⁻¹ for the years 1984-1999; only about 1.5% higher than the national average) [calculated from (EIA, 1999)] (English units presented above are industry standards).

When the distribution is heavily skewed, the use of relative proportions produces a phenomenon where as the point estimates increase, the percentage error associated with those points also tends to increase [a.k.a. heteroscedastic (Neter 1996)]. For example, Pennsylvania and Texas are the two leading coal consumers in the U.S. and these states have the largest discrepancies for coal combustion data. Totaling over several types of fuel and larger areas helps alleviate this problem. The national total consumption value from the CDIAC database has an error rate of only 3-4% (Marland et al. 1989), therefore, at higher levels of spatial aggregation the error rates will diminish. In other words, though the proportions may be slightly incorrect with respect to each other, all fossil-fuel consumption in the country will be accounted for when the proportions are multiplied by the national total. Finally, judicious choice of proportional variables will lead to satisfactory results. In this case, the use of electrical utility coal consumption as the proportional variable resulted in a maximum of 4% error in the spatial distribution of emissions (relative to national emissions from coal consumption) when compared with an independent method. This is an acceptable level of error for a methodology that can be applied to other countries with less robust data collection commitments than the U.S.

Conclusions

The proportional method presented here produces similar distributions of CO₂ emissions, both temporarily and spatially, when compared to the results from the Blasing et al. (2005a; 2005b) method. The proportional method tends to perform best when the distribution it attempts to estimate is relatively uniform. When the distribution is skewed, which can be the case with distributions across states/provinces, the proportional method is apt to produce larger errors in the emissions estimates for the geopolitical regions with higher emissions. For this reason, the uncertainty increases in point estimates (e.g., the emissions from a specific fuel for a given state during a given month) when the distribution is greatly skewed. This is usually less problematic in the temporal component than the spatial component as the seasonal distribution of fuel use tends to be less skewed than the spatial distribution. The close agreement in the temporal component between the results produced by the proportional methodology and the Blasing et al. (2004b) dataset suggests that the proportional methodology can be applied confidently to other countries to produce accurate time series of seasonal CO₂ emissions from fossil-fuel consumption.

The proportional method presented here can be susceptible to data errors and missing records. Because the effectiveness of any method is dependent on the quality of the data inputs, incorrect source data values will diminish the capacity of any

method to estimate emissions. The proportional method uses comparatively little data and hence erroneous data points will have a more substantial impact on the overall estimates. In addition, problems associated with improper scaling or systematic bias in the input datasets will be exacerbated in the CO₂ emissions estimates that this method produces. On the other hand, the proportional method minimizes some of the effect of inaccurate data points upon aggregation by apportioning the error over all point estimates.

Many aspects of the proportional method make it an attractive alternative to the data-intensive, bottom-up accounting methods. The proportional method is computationally simple. The principle of utilizing relative proportions is applicable on many different temporal and spatial scales, allowing estimates at very detailed or broad temporal and spatial resolutions (although the uncertainty in the estimates increases at finer spatial and temporal resolutions). The ultimate benefits of the proportional method are its simplicity, wide applicability, capacity to compensate for errors, and its ability to estimate the temporal and spatial distribution of emissions from comparatively very little data. With the scarcity of available records and data for some countries, this method is particularly attractive and perhaps the only currently available option to produce monthly estimates for the CO₂ emissions in those countries. It provides a means to discern the pattern of emissions where a lack of data renders other methods inapplicable.

Acknowledgements

Gregg Marland provided helpful reviews and numerous discussion about the material presented here. T.J. Blasing and Christine Broniak graciously provided their data and early drafts of their publications (now published and referenced in this paper) to facilitate comparative analysis between our respective results. This work was funded by U.S. Department of Energy Grant DE-FG02-03ER46030.

Chapter 3: China: the emissions pattern of the world leader in CO₂ emissions from fossil fuel consumption and cement production

Preface

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Published in:

Geophysical Research Letters (2008), 35, L08806, doi:10.1029/2007GL032887

Abstract

Release of carbon dioxide (CO₂) from fossil fuel combustion and cement manufacture is the primary anthropogenic driver of climate change. Our best estimate is that China became the largest national source of CO₂ emissions during 2006. Previously, the United States (US) had occupied that position. However, the annual emission rate in the US has remained relatively stable between 2001-2006 while the emission rate in China has more than doubled, apparently eclipsing that of the US in late 2006. Here we present the seasonal and spatial pattern of CO₂ emissions in

China, as well as the sectoral breakdown of emissions. Though our best point estimate places China in the lead position in terms of CO₂ emissions, we qualify this statement in a discussion of the uncertainty in the underlying data (3-5% for the US; 15-20% for China). Finally, we comment briefly on the implications of China's new position with respect to international agreements to mitigate climate change.

Introduction

Fossil fuel combustion and cement manufacture are the principal anthropogenic sources of the greenhouse gas carbon dioxide (CO₂), and hence the principal concern in efforts to address anthropogenic climate change. The United Nations Framework Convention on Climate Change (UNFCCC) and its subsequent Kyoto Protocol were adopted in 1992 and 1997, respectively, as a beginning effort to limit the atmospheric increase in greenhouse gases. The Carbon Dioxide Information Analysis Center (CDIAC) database (Marland, Boden, and Andres 2007) shows global emissions from fossil fuels and cement have grown from 6.2 Pg C in 1990, the base year for commitments under the Kyoto Protocol, to 7.2 Pg C in 2001 and 8.4 Pg C in 2006. Rapid growth over the last five years has been dominated by economic growth in developing countries (see, for example, Raupach et al., (2007)), with 54% of the global increase in CO₂ emissions over the period 2001-2006 coming from China alone.

Historically, the United States (US) has long been the world's largest emitter of CO₂ from fossil fuel combustion and cement production (Marland, Boden, and Andres 2007). However, a recent press release from The Netherlands Environmental Assessment Agency (2007), based on preliminary analysis of data from the International Energy Agency (IEA), BP, and the US Geological Survey (USGS), suggested that emissions from China had surpassed those from the United States for the first time in 2006. Our analysis concurs with the general observation of the Netherlands Environmental Assessment Agency, and the purpose of this letter is to provide detailed monthly and province-level data on Chinese fossil fuel emissions and to comment on the uncertainties associated with emissions estimates for China. This provides context and perspective on the role of China in global CO₂ emissions from fossil fuel.

Materials and Methods

Data on yearly CO₂ emissions from China and the US are taken from CDIAC (Marland, Boden, and Andres 2007) and are based on energy data from the United Nations (UN) Statistics Office (2006) and cement production data from the USGS (Kelly and Matos 2007). Data from the UN extend through 2004 and from the USGS through 2005. Energy data from BP (2007) extend to 2006 and the fractional increases of time series data from 2004 to 2005 and 2006 have been used to extrapolate the CDIAC emissions estimates for two years. To estimate total emissions

for 2005 and 2006, we assume that the fractional change in cement production for each country was the same from 2005 to 2006 as from 2004 to 2005. Applying this extrapolation approach to historical years (back casting), we find that using the BP statistics for a one-year extrapolation of the UN data leads to emissions estimates that vary (on average) about 1% from the UN values, while using the BP data for a 2-year extrapolation leads to estimates that vary (on average) about 2% from the UN values. Estimates include emissions from the flaring of gas at oil and gas fields and processing facilities (less than 1% of total emissions) and we assume that these values have remained unchanged since the end of the UN time series in 2004.

Consistent sets of monthly data were produced by estimating the fraction of total fuel use that occurred during each month and using these fractions to allocate the annual emissions data from CDIAC [see Gregg and Andres (2008)]. For the US, the Energy Information Administration (1983-2007; 1984-2007; 1981-2007) maintains state-by-state monthly data on US coal consumption in the electric utility sector and monthly data on sales of petroleum products (gasoline, aviation fuel, kerosene type jet fuel, distillate fuel, diesel, and fuel oil) and deliveries of natural gas. Monthly data on cement manufacture and natural gas flaring are from Blasing and Hand (2007). Emissions from cement production include only the CO₂ resulting from the calcination process; emissions from fossil fuels used by the cement industry are included with fossil fuel CO₂ emissions. For China, All China Marketing Research (ACMR) (2007) provides monthly data on thermal electricity generation, steel production, coke production, and value of industrial outputs, which represent 48%,

7%, 7%, and 27% of coal consumption, respectively. These time series were weighted to estimate the monthly emissions from coal consumption (the remaining proportion of coal we assume to be uniformly distributed residential use). This source also provides quarterly data on petroleum product sales (gasoline, kerosene, diesel, and fuel oil) and monthly statistics on travel volume, which we used to estimate monthly consumption of liquid fossil fuels. In addition, ACMR maintains monthly data on natural gas output, and monthly data on cement production. These data were used to subdivide the annual emissions into a monthly time series. Finally, the China National Bureau of Statistics (2006) includes annual consumption data for coal, petroleum products (gasoline, kerosene, diesel, and fuel oil), natural gas, and cement production by province. We used these data to determine the spatial distribution of emissions in China by allocating the national annual total. These analyses do not include the special administrative regions of Hong Kong or Macao, and they also exclude Taiwan, because data are kept separately for these regions.

Results

CO₂ emissions from China increased nearly 80% from 2000 to 2006. Emissions for 2003 and 2004 saw rates of increase of 17% and 18% respectively. This outpaced the phenomenal 10% annual growth in real gross domestic product (GDP) (World Bank 2007), increasing China's carbon intensity (emissions per unit of real GDP). The rate of increase in emissions slowed to 10% and 8% for 2005 and

2006, but even at 10% annual growth, emissions would double again in less than nine years. The recent rate of growth in emissions from China has defied projections made five years prior. The IEA (2006b) had originally projected that China would become the world leader in emissions in 2030, then the following year adjusted that estimate to 2009. Cyranoski (2007) updated the estimate to late 2007. Our best estimates suggest that for the full year of 2006 emissions from the United States were possibly still larger than emissions from China, but the difference was very small and clearly within the uncertainty bounds (Figure 9). However, when considering estimates of monthly CO₂ emissions, our best estimate is that China reached US levels of emissions for the first time in November 2005, with both countries emitting 132 Tg C month⁻¹, and then eventually passed the US in September of 2006, emitting at a rate 142 Tg C month⁻¹ (Figure 10), subject again to the uncertainties in the underlying data on energy consumption, as discussed below. Therefore, our best estimate is that the crossing between the United States and China occurred late 2006.

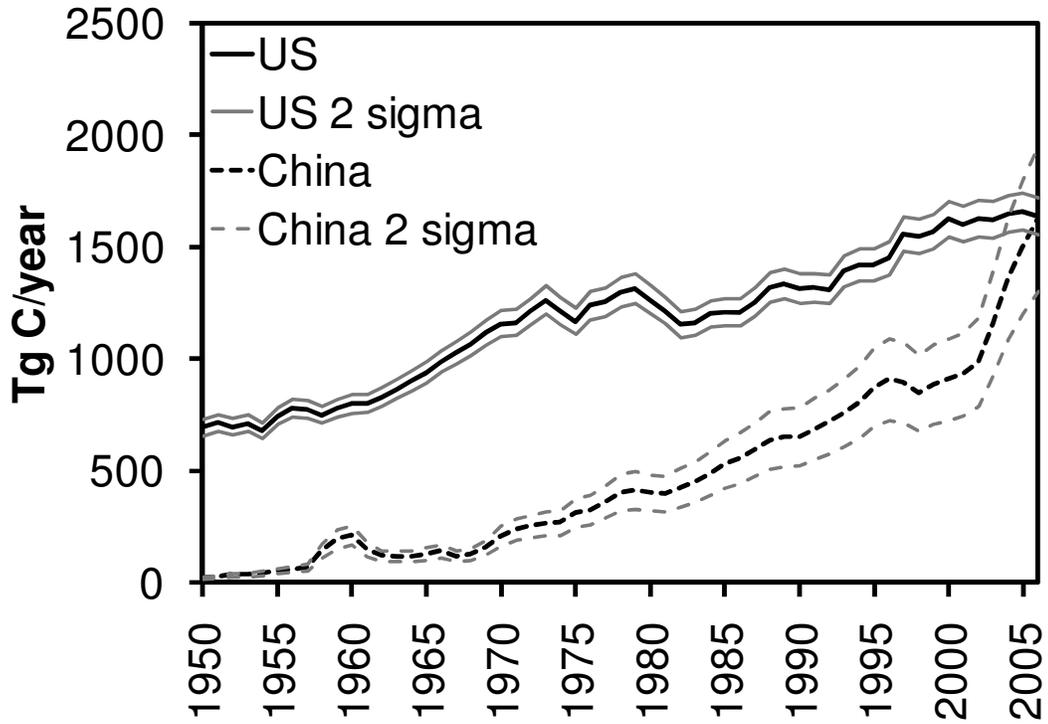


Figure 9. Historic annual emissions from fossil fuel combustion and cement production for the US and China 1950-2006. Two sigma uncertainties are represented by the adjacent gray lines. After a period of dramatic growth in China, the annual emissions for the two countries are approximately equal. Data sources as described in the text.

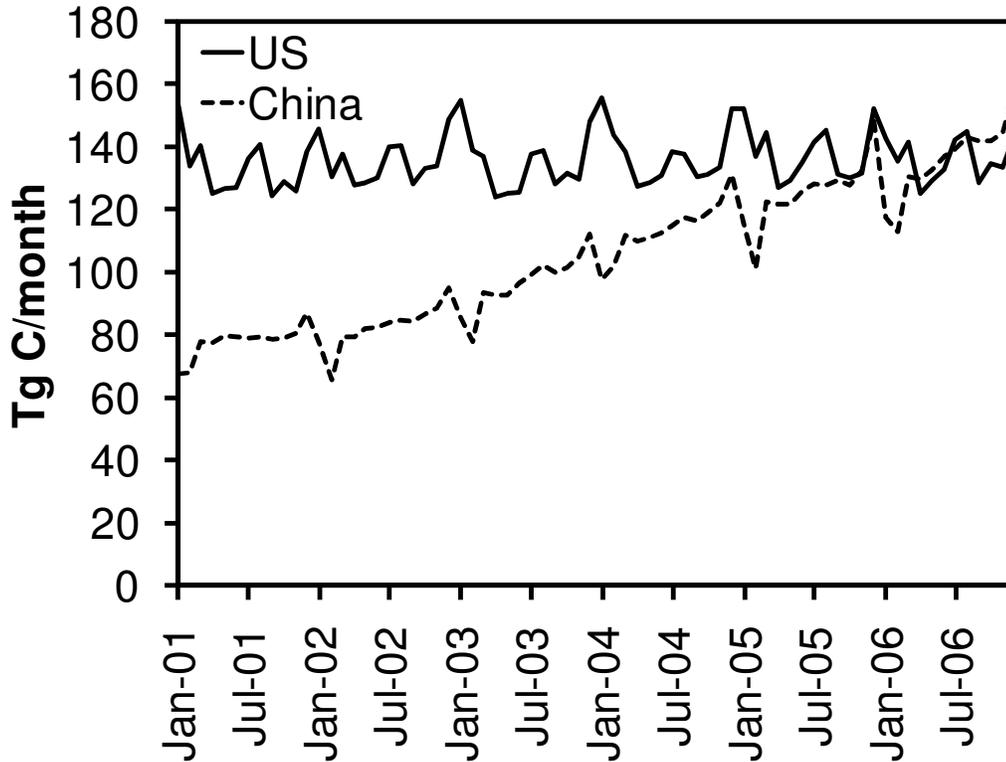


Figure 10. Monthly emissions from fossil fuel combustion and cement production for the US and China. Emissions in China begin to exceed those of the US in late 2006. Data sources as described in the text.

Not only has economic growth been very high over the last few years in China, but growth has been powered largely with coal, the primary fossil energy source with the highest emissions of CO₂ per unit of useful energy. CO₂ emissions from China in 2004 were derived from coal (72%), petroleum (17%), cement (10%), and natural gas (1%) (Figure 11) (Marland, Boden, and Andres 2007). Coal combustion is used for electric power generation and for other industrial processes.

Since 2000, China has been the world's leading producer of coal, crude steel, and cement; and China is second in electricity production (behind the US) (China National Bureau of Statistics 2000-2007). Economic sectors that use petroleum have also been rapidly expanding since 2000. For example, between 2000 and 2006 transport volume doubled (China National Bureau of Statistics 2000-2007). China is also the world leader in fertilizer production, a process that uses substantial amounts of coal and natural gas as inputs (China National Bureau of Statistics 2000-2007). But a significant portion of growth in energy consumption and CO₂ emissions has been driven by the globalization of the world economy and China is a major exporter of energy-intensive goods that are consumed elsewhere. *Shui and Harris* (2006) have estimated, for example, that between 7% and 14% of current CO₂ emissions from China are a result of producing goods that will be consumed in the US.

Estimates of monthly emissions in China show a consistent peak in December with a precipitous drop in January. There is also a slight peak in late summer in emissions from coal combustion (Figure 11). This pattern is consistent throughout the industrial production statistics from China and is also reflected in the quarterly GDP reports (ACMR, 2007). Spatially, CO₂ emissions from China are concentrated in the provinces around Beijing and along the east coast of China (Figure 12). Liquid fuel consumption is concentrated on the east coast of China, particularly in Guangdong province, which includes the special economic zone of Shenzhen and has been the beneficiary of international investment and manufacturing. Shanxi province, to the west of Beijing, is the leading consumer of coal, primarily for the production of coke,

and has the highest per capita emissions in the country. Per capita emissions from China as a whole have risen to 1.1 Mg C person⁻¹, but in Shanxi province, a leading coal producer, this rate is 3.3 Mg C person⁻¹, equivalent to per capita emissions rates in Western Europe. At 1.1 Mg C person⁻¹, the per capita emissions in China are close to the world average.

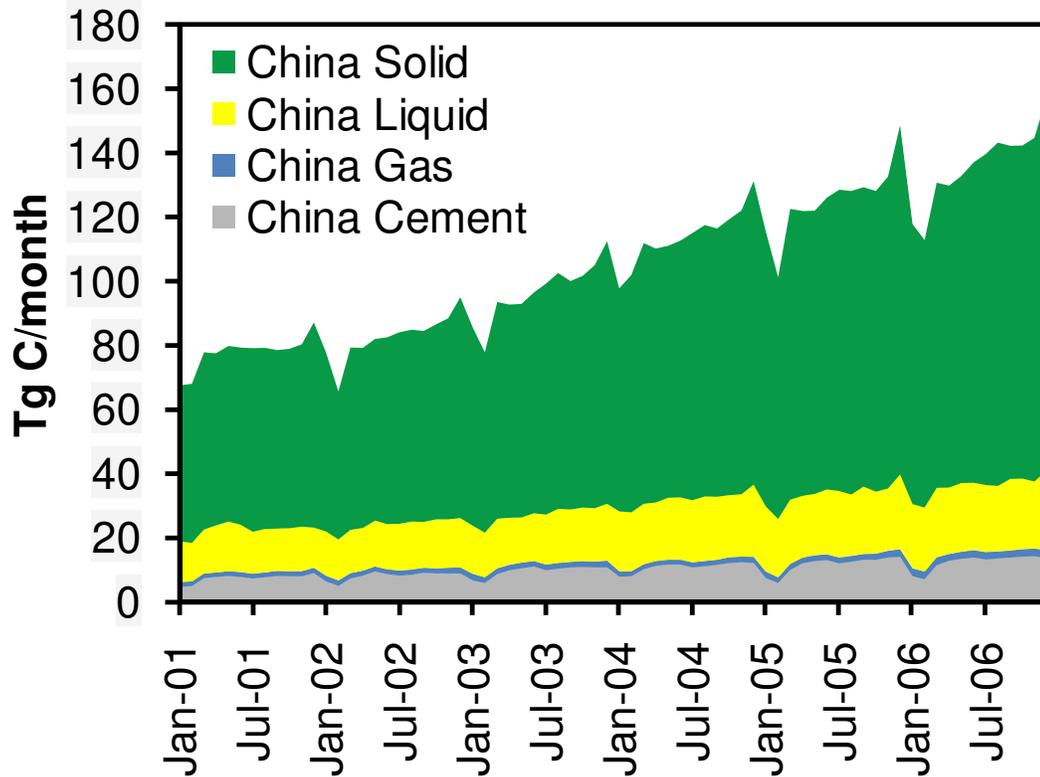


Figure 11. Sources of anthropogenic emissions in China. The majority of emissions are from the combustion of coal.

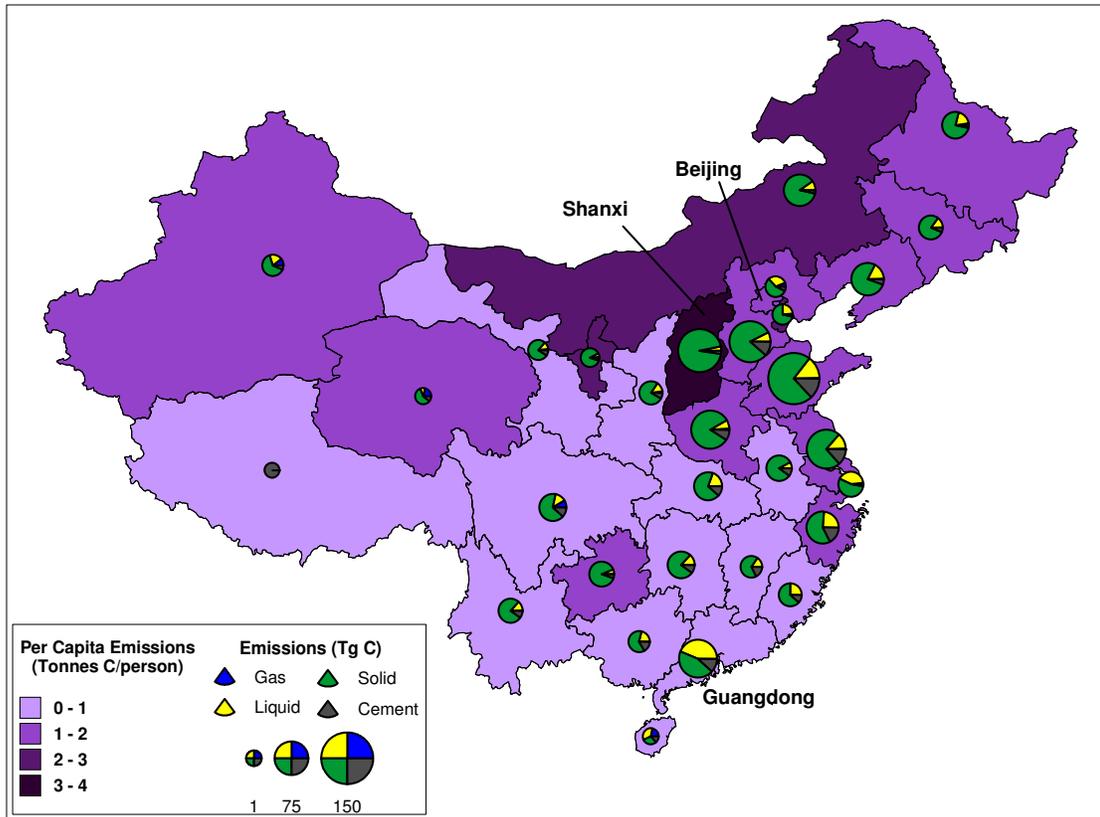


Figure 12. Spatial distribution of CO₂ emissions in China from fossil fuel combustion and cement production. Darker colors depict higher per capita emissions. The relative sizes of the pie graphs indicate absolute emissions and their source. Emissions are higher in the populated areas, and per capita emissions are higher in provinces that produce large quantities of coal. Based on 2005 data.

Discussion

Many analysts have viewed official Chinese statistics with skepticism, and there has been suspicion that politics motivate adjustments in the reported data

(Akimoto et al. 2006; Sinton 2001; Zhang et al. 2007). In this analysis, we have used a multitude of industrial production and energy consumption statistics in an effort to reduce uncertainty and error. Nevertheless, much of the data on which we depend is ultimately from the Chinese National Bureau of Statistics, and any inaccuracies in those energy data will result in inaccuracies in our estimates of CO₂ emissions. One possibility is that the recent growth depicted in the energy consumption statistics may reflect a correction from under-reporting in the late 1990s. According to official statistics, between 1996 and 2000 emissions in China decreased even while electricity generation, industrial output, and GDP continued to increase exponentially (China National Bureau of Statistics 2000-2007). Streets et al. (2001) recognized that errors and uncertainties were increasing in the National Bureau of Statistics during this period and further reasoned that the reductions in coal use reported at the time were not as dramatic as the official data suggested. Logan (2001) suggested that restructuring of the coal industry in 1996 ushered forth a black market that resulted in some coal consumption going unreported in the official statistics. When comparing modeled emissions to remotely sensed tropospheric nitrogen oxide (NO_x) concentrations over this period, Akimoto et al. (2006) concluded that the National Bureau of Statistics had significantly underestimated coal consumption during this period.

Others have raised questions about the GDP figures China has released and have suggested, based on other economic indicators, that growth in the economy has not been as large as reported (Rawski 2001). On paper, the carbon intensity in China

declined sharply in the late 1990s but has recently been increasing as the data indicate a dramatic increase in energy consumption. If it is the case that a large-scale statistical correction is occurring to compensate for underreporting in the 1990s, then our analysis may be overestimating the recent growth rate in emissions.

It is also possible that our analysis may be underestimating the current rate of growth in emissions for China. According to our analysis, emissions are lower in the beginning of the year and higher in December. The mean slope within years (January to December) from 2001-2006 shows an increase in emissions of 23 Tg C month⁻¹, but the average increase in emissions using annual data is only 10 Tg C month⁻¹. Statistically, this difference occurs because of the drop in the emissions estimates from December to January. This could be a result of a ramping up of production to meet annual quotas by the end of the year (or data manipulation to make it appear that way). However, if the annual increase in consumption is actually closer to the reported monthly increase, this would result in us underestimating the annual increase in fossil fuel consumption in China.

Recent adjustments in official Chinese energy statistics have now been incorporated into international energy data sets and have resulted in revisions of estimates of CO₂ emissions. Emissions for 2000, for example, were revised upward by 23% for the year 2000 (Marland, Boden, and Andres 2007). These revisions are in the direction expected but indicate the magnitude of continuing uncertainty in the Chinese data. Based on evaluation of the US data by the US Environmental Protection Agency (EPA) (2006), and on the magnitude of recent revisions of energy

data from China, we estimate that the two-sigma uncertainty associated with the annual total CO₂ emissions estimates is 3% to 5% for the US but could be as high as 15% to 20% for China. It is, of course, not possible to independently evaluate the uncertainty of the Chinese data but the recent adjustments and the incompatibility with satellite NO_x measurements suggest uncertainties in this range. There is no information to suggest an asymmetry in this uncertainty. Taking this uncertainty into account implies that the Chinese emissions could have passed those from the US as early as 2004, or the passing could be as late as 2010 (assuming the current trajectories) (Figure 9).

Designated as a developing country, China was not given Annex I status, and thus is not required to meet emission reduction targets under the Kyoto Protocol. Whether we are observing increasing consumption in developing countries or the export of emissions from developed countries (Munskgaard and Pederson 2001; Rothman 1998; Shui and Harriss 2006), global CO₂ emissions are in a period of rapid growth that is nullifying the mitigation aspirations of international agreements (Auffhammer and Carson Forthcoming). Although there is still concern about energy data from China, it is clear that CO₂ emissions are growing very rapidly; over half of the global growth in emissions is occurring in China. Per capita emissions from China are now at global-average values and are reaching European-average values in some rapidly industrializing areas. This is propelling China into the position as the largest national source of CO₂.

Acknowledgements

Thanks to TJ Blasing for providing data on US cement and gas flaring. Research for this article was conducted at the Library of Congress and was facilitated by the staff of the Asian Reading Room. Research was also conducted at the Chinese Institute for Geographic Studies and Natural Resources Research (IGSNRR), part of the Chinese Academy of Sciences (CAS). Comments and suggestions from two anonymous reviewers have been invaluable in improving this document. Robert J. Andres and Gregg Marland were partially supported by the U.S. Department of Energy, Office of Science, Biological and Environmental Research Programs.

Chapter 4: The temporal and spatial distribution of carbon dioxide emissions from fossil-fuel use in North America

Preface

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In Press:

Journal of Applied Meteorology and Climatology

Abstract

Refinements in the spatial and temporal resolution of North American fossil-fuel carbon dioxide (CO₂) emissions provide additional information about anthropogenic aspects of the carbon cycle. In North America, the seasonal and spatial patterns are a distinctive component to characterizing anthropogenic carbon emissions. The pattern of fossil-fuel-based CO₂ emissions on a monthly scale has

greater temporal and spatial variability than the flux aggregated to the national annual level. For some areas, monthly emissions can vary by as much as 85% for some fuels when compared to monthly estimates based on a uniform temporal and spatial distribution. The U.S. comprises the majority of North American fossil carbon emissions and the amplitude of the seasonal flux in emissions in the U.S. is greater than the total mean monthly emissions in both Canada and Mexico. Nevertheless, Canada and Mexico have distinctive seasonal patterns as well. For the continent, emissions were aggregated on a $5^{\circ} \times 10^{\circ}$ latitude-longitude grid. The monthly pattern of emissions varies on both a north-south and east-west gradient, and evolves through time period analyzed (1990-2007). For many areas in North America, the magnitude of the month-to-month variation is larger than the total annual emissions from land use change, making the characterization of emissions patterns essential to understanding humanity's influence on the carbon cycle.

Introduction

With the increasing concentration of CO₂ in the atmosphere and its implications for global climate (IPCC 2007), there is a growing need for developing a more detailed description of the various components within the global carbon cycle. Scientific inquiries and analyses now call for data on anthropogenic CO₂ emissions at spatial and temporal scales finer than the countries and years at which emissions inventories have traditionally been conducted. Mechanistic understanding of carbon cycling relies on mathematical modeling and on detailed measurements at monitoring

sites (Gurney et al. 2005; Gurney et al. 2002) to determine carbon fluxes among the terrestrial biosphere, atmosphere, and ocean (Wigley 1993). Gurney et al. (2005) revealed that, for example, the lack of detailed information on the seasonal cycle of anthropogenic emissions has a large impact on results derived from inverse modeling of atmospheric concentrations and interchanges with the biosphere. The purpose of this paper is to provide detailed information on the monthly and sub-national spatial distribution of fossil-fuel CO₂ emissions in North America. We know that fossil fuel consumption varies widely across North America and that there are temporal variations with season and with the other patterns of weather change (Blasing, Broniak, and Marland 2005a, 2005b). Recognizing these variations allows us to more accurately describe the biophysical and human processes that are involved in the global carbon cycle.

Nearly four-fifths of global anthropogenic carbon emissions are from the combustion of fossil fuels. There are at least five available data sets for global annual emissions of CO₂ from fossil-fuel use by country: four that cover most countries (EIA 2004; IEA, 2008; Marland, Boden, and Andres 2009; Olivier et al. 2005) and one compilation of individual national reports that covers many of the largest-emitting countries (UNFCCC 2005b). The International Energy Agency (IEA) (2007), the United Nations Framework Convention on Climate Change (UNFCCC) (2005b), and Olivier (2005) datasets contain information on emissions by market sector (e.g., commercial, residential, industrial, etc.). All of these emissions inventories are at the spatial and temporal scale of countries and years. Sectoral emissions data for the U.S.

are maintained by the U.S. Environmental Protection Agency (EPA) (2009). Blasing et al. (2005a; 2005b) have used state level energy data to estimate annual carbon emissions for the 50 U.S. states and for the total U.S. by month, but were not able to estimate emissions by month and state due to limitations in the resolution of the underlying energy consumption data.

Prior studies have utilized various methods to estimate the geographical distribution of emissions at finer levels of detail. Andres et al. (1996), Brenkert (2003), and Olivier et al. (2005) attempted to describe anthropogenic CO₂ emissions on a 1° latitude by 1° longitude grid. In these studies, the CO₂ emissions estimates were only available on an annual time step, at best, so no discernment of seasonal patterns of emissions was possible. Also, the CO₂ emissions estimates in these studies relied mostly on population density to distribute emissions within the political boundaries of each country, the scale at which energy data are traditionally collected. Blasing et al. (2005b) have demonstrated that population density may be a useful first approximation for the distribution of emissions within countries, but that it has serious limitations, strikingly so when, for example, electricity is generated from coal combustion in sparsely populated areas, and then transmitted by a regional electricity grid.

To estimate the seasonal flux in emissions, Erickson et al. (2008) employed a Fourier series on a preliminary subset of the data presented here, and extrapolated to get a rough estimate of global seasonality in fossil-fuel CO₂ emissions. Though the study's conclusions were not based on a comprehensive database of actual emissions,

it nevertheless demonstrated that the temporal variability in fossil-fuel CO₂ emissions typical of mid-northern latitudes could significantly impact estimates of other components of the carbon cycle as well as atmospheric inversion models (Erickson et al. 2008).

The Vulcan inventory (Gurney et al. 2009) contains the most detailed estimates for fossil-fuel CO₂ emissions, produced by combining multiple detailed datasets and in some cases using other pollutant gases as well as road and census data as proxies for CO₂ emissions. The Vulcan inventory includes both point and non-point sources of CO₂, at temporal scales of less than 100 m² on an hourly time step (Gurney et al. 2009). While this approach shows great promise, its scope is currently limited to the conterminous U.S. and is only available for the year 2002.

Gregg and Andres (2008) have developed an approach for capitalizing on a variety of data sources to disaggregate national, annual emissions estimates into a consistent data set of emissions at monthly and sub-national scales. Here, we apply this approach to describe the spatial and temporal distribution of CO₂ emissions among the states and provinces of the United States and Canada, and the temporal distribution of emissions in Mexico as data limitations preclude finer spatial breakdown there. We then discuss the results, their causes as related to energy-use patterns and the global carbon cycle. As of 2004, all three countries in North America are within the top 11 countries in the world in terms of annual CO₂ emissions from fossil-fuel consumption (Marland, Boden, and Andres 2009). Fossil-fuel emissions

from North America comprise roughly a quarter of current global annual emissions and nearly a third of global cumulative emissions since 1751 (Marland et al. 2007b).

Methods

Most countries collect some data on energy production, consumption, and trade. These data can be used, along with data on fuel chemistry and fuel use, to estimate CO₂ emissions from fossil fuels by country and by year. To estimate emissions on finer temporal and spatial scales, ideally, CO₂ emissions would be calculated similarly from complete data on consumption of all fossil fuels from all economic sectors. To do this with accuracy, we would need to know the amount and quality of each fuel consumed, when exactly it was consumed, and what fraction of it is consumed in ways that do not lead to oxidation of the fuel (e.g., incomplete combustion, and sequestration of carbon in products such as asphalt, plastics, etc.), from all countries, provinces and states. For example, using fuel consumption data from the Energy Information Administration (EIA), Blasing et al. (2005a) created a detailed data set of monthly emissions from the U.S. and annual emissions for each state (Blasing, Broniak, and Marland 2005b). However, such an approach requires a level of detail in the underlying energy data that is typically not available for most countries. While these detailed data are available at the annual level for many developed countries, they are typically not available at the sub-national level because collecting and managing detailed monthly energy data by sub-national region is labor

intensive and expensive. Even for the U.S., the data are not available to produce seasonal patterns for each state using this approach.

We have adopted an approach that capitalizes on extensive data collections where they exist but also allows us to estimate emissions at finer spatial and temporal scales even when complete, detailed data on fuel consumption are not available. Lacking data on fuel consumption, close estimates can be made from fuel supply data by calculating “apparent consumption” as the sum of production and imports less the sum of exports and changes in stockpiles (Marland and Rotty 1984). When these data are not available, data on fuel sales for domestic consumption can be used. Fuel sales data can provide a factor for apportioning consumption even if all fuel is not accounted for, although they do raise uncertainties concerning fuel storage between the times of purchase and combustion of a fuel. The apportioning can be done with data on only a fraction of total consumption but will generally be more accurate the greater the fraction of total consumption that is represented.

To estimate emissions sub-nationally, the apportioning methodology uses data on the major energy consumption sectors of a country’s economy to parse annual national emissions both spatially and temporally (Gregg and Andres 2008). In a trial using the U.S. data, this approach was shown to produce estimates that were within the uncertainties of the underlying data (Gregg and Andres 2008). The method was also applied to Brazil, showing its applicability despite gaps in the underlying data (Losey, Andres, and Marland 2006). The basic approach used here is detailed in Gregg and Andres (2008), though in the current study, we have used annual state and

provincial consumption data to better estimate the state-to-state (for the U.S.) and province-to-province (for Canada) spatial distribution of emissions. No sub-national data were available for Mexico.

For this analysis, fuel consumption statistics were taken from the BP Statistical Review of World Energy (2008). Annual carbon emissions estimates, T , for year a and fuel k were calculated as:

$$(T_{a,k}) = \text{Consumption}_{a,k} \text{ (Mtoe)} \times \text{Carbon Conversion}_k \text{ (Tg C Mtoe}^{-1}\text{)} \times \\ \text{Fraction Oxidized}_k$$

(BP reports fuel consumption in Mtoe, or million tonnes of oil equivalent, = 42×10^{15} J.) The carbon conversion rates were based on data from the EIA State Energy Data System (SEDS) (2009). For natural gas and coal, the carbon conversion factors were assumed to be the same for all countries. For petroleum, the weighted average of consumption of petroleum products is calculated for each country to determine the mean national carbon content of all petroleum products consumed. The carbon conversion and fraction oxidized factors are given in Table 8.

Monthly data on fuel consumption or sales were collected for each fossil fuel type from government and industry statistical reports. Table 8 shows the fraction of each country and fuel category represented by the monthly consumption or sales data used. We used the monthly consumption or sales data to parse the national total carbon emissions among states/provinces and months. Missing monthly data points

were filled using an interpolation strategy, which first calculated the discrepancy between the national monthly total and the sum of the existent data points for the given month, and used the difference to fill the missing datum. In months where there were more than one missing datum, the discrepancy was apportioned by the relative weights of the mean values of the corresponding month and states/provinces in previous complete years. The apportioning approach is such that emissions estimates totals are mutually consistent with the annual emissions dataset when summed both spatially and temporally.

Table 8. Summary of data sources, emissions portfolio, and percentage coverage of monthly proxy data relative to the BP annual data (2008) for the years 1990-2007.

	U.S. ¹	Canada ²	Mexico ³
Natural Gas			
Percentage of total annual fossil carbon emissions	19	29	21
Monthly proxy data	Total deliveries	Direct and utility sales	Domestic sales
Percent of annual consumption represented	90	80	56
C Conversion (Tg C/Mtoe)	0.5742	0.5742	0.5742
Fraction Oxidized ⁴	0.98	0.98	0.98
Petroleum			
Percentage of total annual fossil carbon emissions	42	47	71
Monthly proxy data	Sales of gasoline, distillates, jet fuel, and propane	Domestic sales of all refined petroleum products	Domestic sales of all refined petroleum products
Percent of annual consumption represented	77	86	100
C Conversion (Tg C/Mtoe)	0.7798	0.7773	0.7595
Fraction Oxidized ⁴	0.918	0.918	0.918
Coal			
Percentage of total annual fossil carbon emissions	39	24	8
Monthly proxy data	Coal consumption by electric utilities	Conventional steam electricity, weighted by provincial coal based electricity	Flat distribution assumed
Percent of annual consumption represented	84	90, from Stone (2007)	NA
C Conversion (Tg C/Mtoe)	1.0222	1.0222	1.0222
Fraction Oxidized ⁴	0.982	0.982	0.982
Total			
Percent of annual consumption represented	82	85	82

¹U.S. fossil-fuel consumption data are maintained by the EIA (1980-2008; 1984-2008; 1983-2008). Natural gas data includes all deliveries less losses from distribution and processing, representing approximately 90% of the gas consumed in the U.S. (Gregg and Andres 2008). Petroleum fuel data are from average daily sales

of gasoline (all grades), distillate fuels (including diesel fuel), kerosene-type jet fuel and propane. Monthly data on utility consumption of coal is used as a proxy for total coal consumed in the U.S. Annual state fuel consumption data are from the EIA SEDS (2009). The relative state-to-state proportions were projected for the years 2006-2007 based on 2005 data.

²Canada energy data are maintained by Statistics Canada (1992-1996; 2007a; 2007b; 2007d; 2007c). Natural gas data includes the sum of total utility sales and total direct sales of gas for each province by month. For petroleum, monthly provincial domestic sales data for all refined petroleum products are used. Statistics Canada at one time estimated monthly provincial coal consumption data (Statistics Canada 1992-1996). This series was terminated in 1996, now all coal consumption statistics are aggregated to the national level on an annual time step per nondisclosure legal agreements with the Canadian coal industry. Instead, we use monthly provincial data for conventional steam electricity generation (in MWh), scaled by annual provincial data on coal electricity generation. For each province, the mean value for the years 2002-2006 of this scaling factor is applied to the monthly conventional steam electricity time series.

³Mexico sales data for natural gas and petroleum consumption are from Petróleos Mexicanos (PEMEX) (2009). Coal consumption in Mexico is comparatively small; there are only two coal-fired power plants in Mexico, both near the U.S. border, with a total capacity of 2600 MW, representing only 7% of Mexico's total electricity generation capacity (EIA 2002). No monthly data are publically

available for coal consumption at these facilities, and we therefore assume these power plants provide base load power and emissions from coal consumption in Mexico have a uniform temporal distribution.

⁴Fraction oxidized values from Marland and Rotty (1984), available online at CDIAC <http://cdiac.ornl.gov/pns/convert.html>.

Results

The monthly distribution of North American CO₂ emissions

Figure 13 shows the monthly time series of CO₂ emissions for each fuel type for North America, the U.S., Canada, and Mexico for the years 1990-2007. There is roughly an order of magnitude difference between emissions in the U.S. versus emissions in Canada or Mexico, and thus the pattern of total North American emissions (Figure 13a) closely resembles that of the U.S. (Figure 13b). In addition, the rate of annual growth in emissions for the continent is also dominated by the U.S., a large portion of which is from increased petroleum consumption (Table 9). For all three countries, more emissions are from petroleum fuels than from natural gas or coal, but the strong seasonality of the emissions from natural gas and coal in the U.S. and Canada is clearly reflected in the national and regional totals.

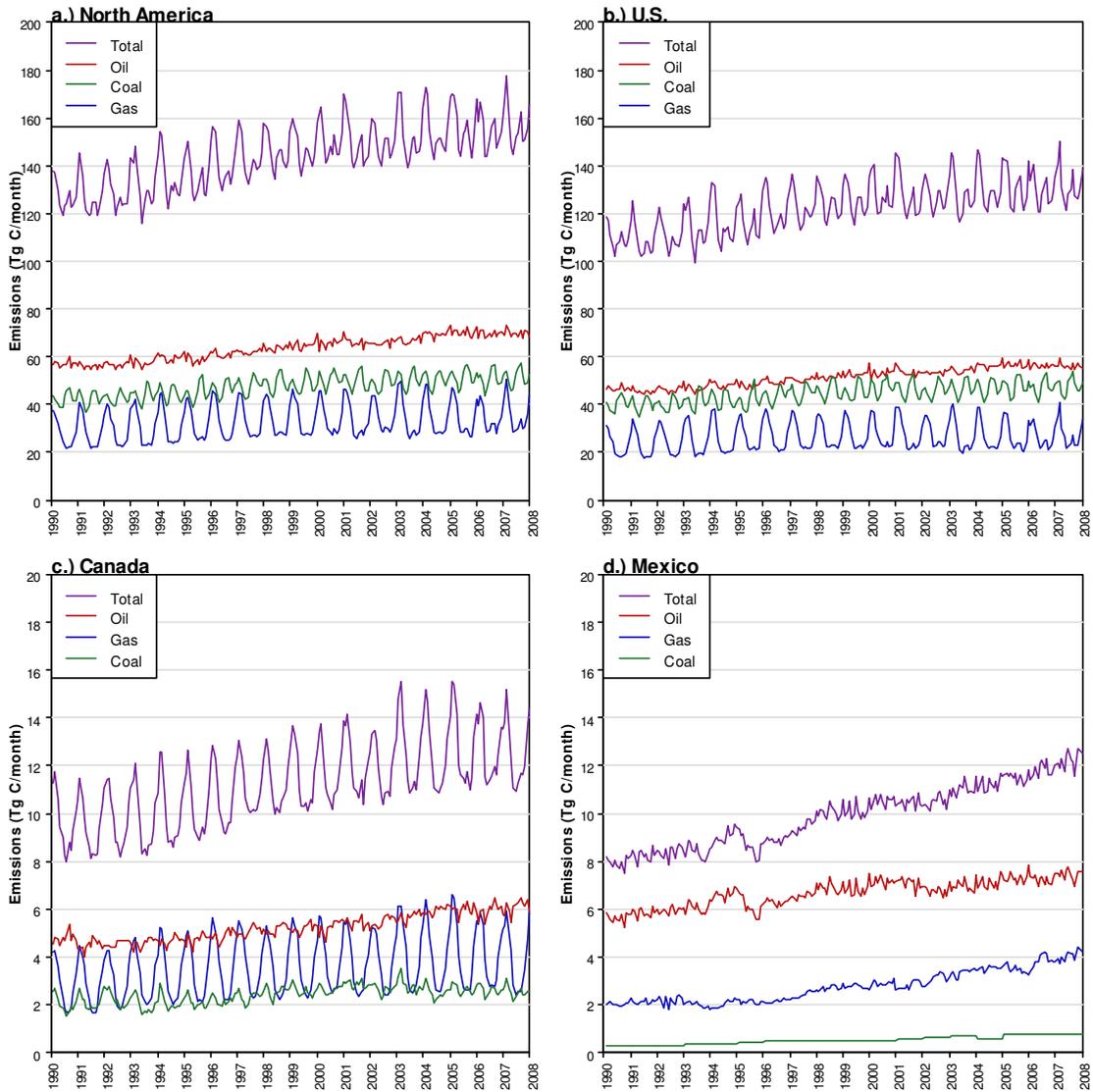


Figure 13. Monthly fossil fuel carbon emissions for a.) North America, b.) the U.S., c.) Canada, and d.) Mexico, by fuel type. Note that the Canada and Mexico plots use a different scale than the North America and the U.S. plots.

Table 9. Mean annual increase (i.e., slope) in fossil-fuel CO₂ emissions, 1990-2007 (Tg C yr⁻²). Columns and rows may not sum exactly due to rounding.

Emissions Source	North America	U.S.	Canada	Mexico
Natural Gas	4.7	2.4	0.9	1.5
Petroleum	10.8	8.4	1.2	1.2
Coal	6.9	6.1	0.5	0.4
Total	22.4	16.8	2.6	3.0

In the U.S., there is a general upward trend in emissions from all fuel types (Figure 13b and Table 9). Emissions from natural gas are the most distinctively seasonal, peaking in the winter. This pattern is evolving through time, with a growing secondary summer peak. Emissions from petroleum consumption are relatively constant throughout the year on the national scale. There is a slight peak in December, in part, from increased gasoline and jet fuel consumption: U.S. gasoline and jet fuel consumption in the US is 9% and 7% (respectively) higher on average in December versus January. Summer driving increases emissions from the transportation sector, but this is offset by a decrease in heating fuel use in the summer (discussed below). The national seasonal distribution of emissions from coal use is bimodal, with peaks in both winter and summer. An analysis by Pétron et al. (2008) has shown a correlation between emissions from power plants and temperature, with a minimum of emissions at approximately 10° C, and increases in emissions with temperatures below 10° C, suggesting increased heating demand, and increasing emissions with increasing temperature, suggesting higher air conditioning use.

Though fossil-fuel-based CO₂ emissions in Canada are considerably lower than those of the U.S., the seasonal patterns of emissions are similar, with higher emissions in the winter than summer (Figure 13c). Like the U.S., petroleum use is not highly seasonal on the national scale. In Canada, approximately 80% (by energy content) of liquid fuel is consumed by the transportation sector (41% gasoline, 31% diesel, 8% aviation fuel). There is a slight summer peak in most areas, though some fuel oil is consumed in the winter. The seasonal distribution of natural gas consumption is also similar to that in the U.S., but with an even greater relative seasonal amplitude. Absent, however, is the small summer increase that has begun to appear in the U.S. emissions patterns for natural gas. Coal represents a smaller proportion of the energy portfolio of Canada and no conspicuous summer peak in emissions from coal combustion is evident. The pattern in coal-based CO₂ emissions is evolving to a flatter seasonal distribution, particularly since the mid 1990s.

Petroleum constitutes the majority of Mexico's energy portfolio, and the total of fossil-fuel CO₂ emissions closely follows the pattern of petroleum consumption (Figure 13d). The mean annual rate of growth in emissions is 2.5% (slope divided by mean emissions, 1990-2007), a rate larger than both the U.S. (1%) and Canada (1.5%); though less than the U.S. in terms of absolute growth (Table 9). Coal consumption is also rapidly increasing (about 6% per year), though absolute emissions from coal consumption in Mexico are still minimal. Electricity generation has been growing steadily in Mexico for the last 30 years, predominately from petroleum and natural gas power plants. As of 2000, Mexico dramatically increased

its use of natural gas for electricity generation, displacing some petroleum use in this sector (IEA 2009).

The spatial distribution of North American CO₂ emissions

CO₂ emissions differ widely among U.S. states and Canadian provinces; in total magnitude, in per capita terms, and in the seasonal distribution. Mexico does not maintain publicly available data that would allow estimation of emissions from Mexico by state.

Figure 14 and Figure 15 characterize (for the U.S. and Canada respectively), the magnitude, per capita distribution, and seasonal variation in CO₂ emissions, by state or province. For the sake of illustration, mean 1990-2007 emissions values are shown for the three winter months (December, January, February) and for the three summer months (June, July, August). The distributions for the three fuel types reflect the magnitude of energy demand, the nature of energy demand, and the access to resources. For example, coal consumption is higher in states such as North Dakota, Wyoming, and West Virginia where coal is mined, natural gas consumption is higher in Alberta, Louisiana and Texas where it is produced. Coal consumption is small in Alaska, Hawai'i and northern Canada where access to this resource is limited. Seasonal distribution for natural gas is more extreme in northern climates where it is used for heating.

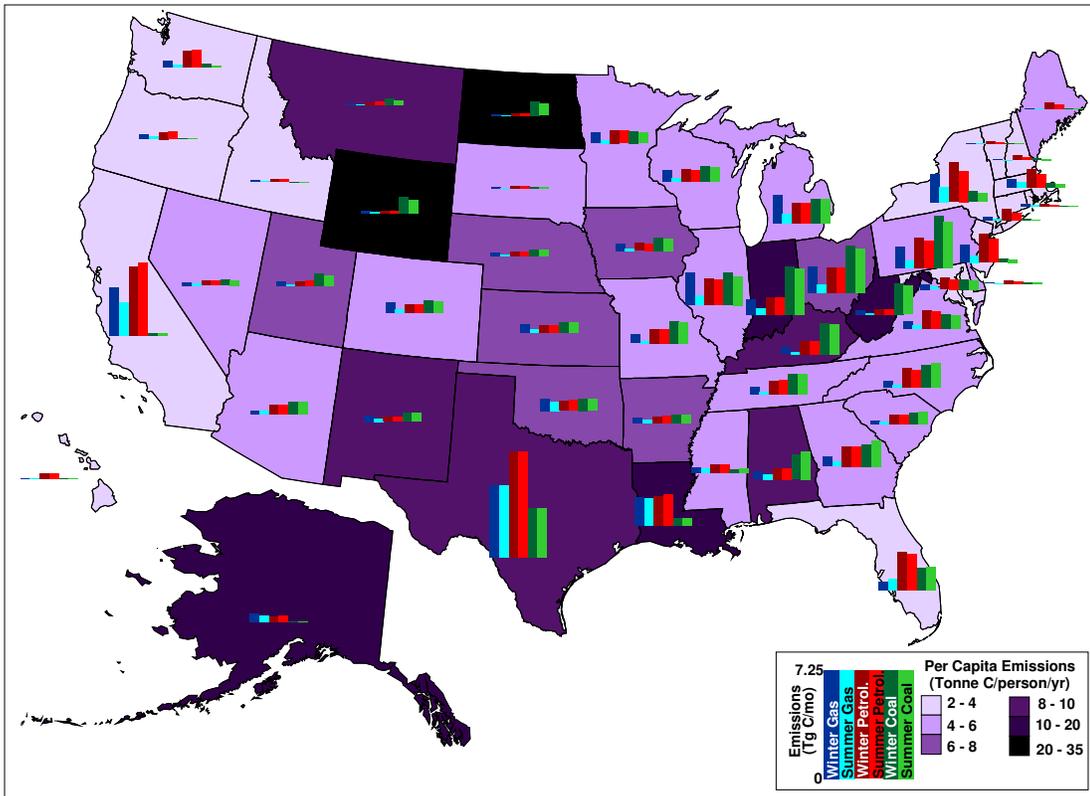


Figure 14. Mean winter and summer spatial fossil-fuel carbon emissions and per capita emissions in the U.S., 1990-2007. Population data are from the U.S. Census Bureau (2001).

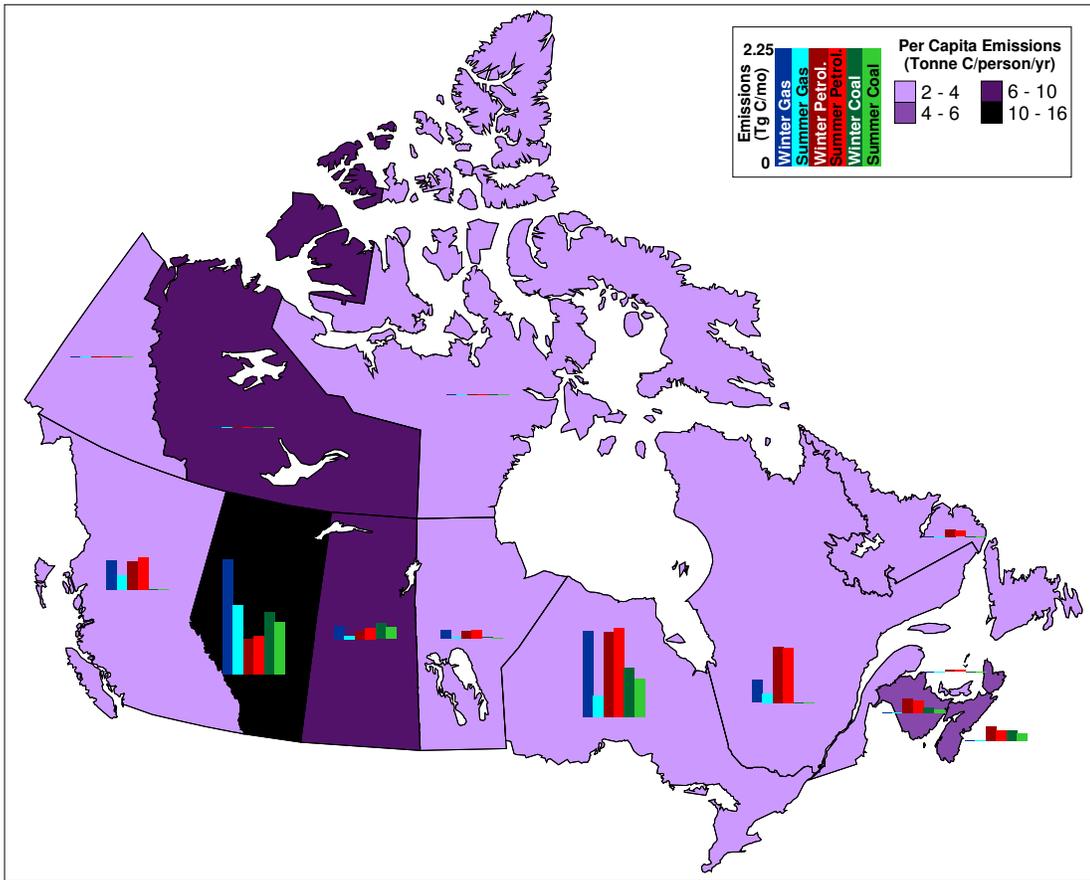


Figure 15. Mean winter-summer spatial fossil-fuel carbon emissions and per capita emissions in Canada, 1990-2007. Population data are from Statistics Canada (2001).

In

Figure 14 and Figure 15, we also calculate the 2000 per capita emissions for each state and province using population data from the U.S. Census Bureau (2001), Statistics Canada (2001) and Instituto Nacional de Estadística y Geografía (INEGI) (2001). While the national annual per capita emissions in the U.S. and Canada are

similar ($5.5 \text{ Mg C person}^{-1} \text{ yr}^{-1}$ and $4.6 \text{ Mg C person}^{-1} \text{ yr}^{-1}$, respectively), Mexico's was much lower at $1.3 \text{ Mg C person}^{-1} \text{ yr}^{-1}$. The spatial distribution of CO_2 emissions from combustion of petroleum fuels is highly correlated to population distribution; of all fuel types, the per capita use of petroleum fuels is the most similar from state to state and province to province. The exceptions to this are states and provinces with large petrochemical and refinery use (Texas, Louisiana, and Alberta), leading to higher per capita emissions from petroleum in these states. Alaska, with a low population and high rate of energy production, also has high per capita emissions from petroleum (as well as gas) consumption. Despite petroleum use being relatively constant on a per capita basis, there is nevertheless high variability from state to state (province to province) due to natural gas and coal availability. For example, total annual per capita emissions are highest in the coal producing states of Wyoming ($33.9 \text{ Mg C person}^{-1} \text{ yr}^{-1}$), North Dakota ($21.1 \text{ Mg C person}^{-1} \text{ yr}^{-1}$). Alberta and Louisiana, where natural gas resources are concentrated, also have relatively high per capita emissions ($15.5 \text{ Mg C person}^{-1} \text{ yr}^{-1}$ and $12.8 \text{ Mg C person}^{-1} \text{ yr}^{-1}$, respectively).

The temporal and spatial distribution of North American CO_2 emissions

In Figure 16 and Figure 17, the monthly patterns of emissions are characterized as a function of latitude and longitude. These were calculated by sorting each U.S. state, Canadian province, and the national total of Mexico into classes based on the nearest latitude and longitude line to the political unit's geographic

centroid (latitude and longitude lines can be seen in Figure 19). Using the values from 1990-2007, the mean (weighted by state and provincial total emissions per fuel type) monthly distribution was calculated for each latitude and longitude group. These were then compared to a hypothetical flat-line distribution to determine the percent error that would result from assuming a uniform monthly distribution for emissions. This gives an approximate spatial distribution of emissions and is done to illuminate and summarize general spatial trends in the emission patterns, though in some areas, the spatial resolution of the underlying data may be greater or less than the corresponding latitude and longitude.

Again, natural gas has the largest seasonal amplitude of all fuel types, varying as much as 85% from a uniform distribution at 50° N. In the most southern latitudes, the annual peak occurs in the summer (for electricity generation) and as one progresses north, the seasonal peak moves to the winter months as the number of heating degree days increases. The amplitude is dampened in northern latitudes, and overall usage is generally lower due to smaller populations and lack of access to natural gas. The seasonal amplitude follows a similar pattern on an east-west gradient, having a more distinct winter peak toward the center of the continent, correlated with climate extremes. The exception to this trend is at 100° W, which is dominated by Mexico and Texas. The mild winters in Mexico and Texas and the large amount of petrochemical use in Texas flatten the distribution at this longitude.

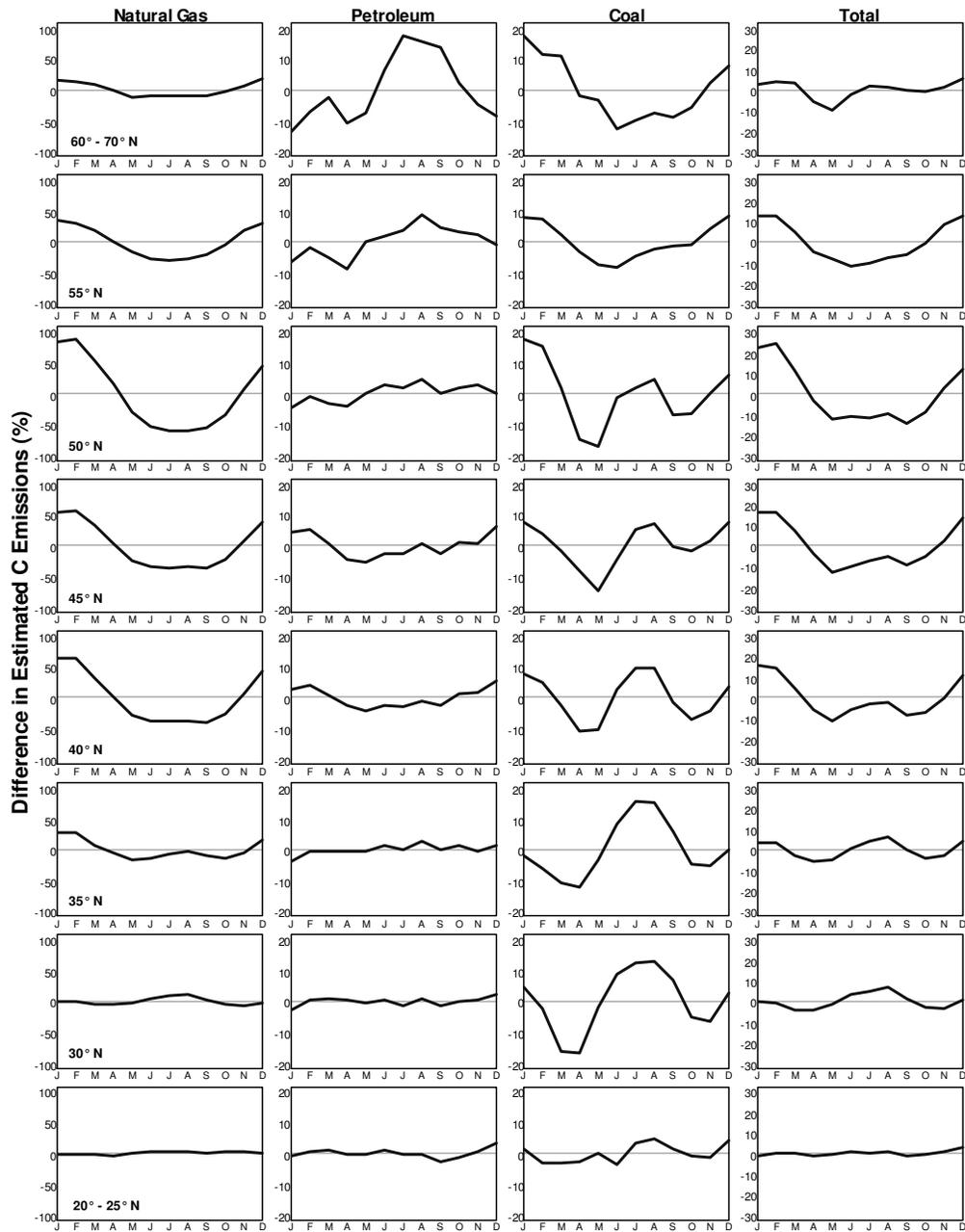


Figure 16. Latitudinal mean monthly distribution of emissions by fuel type. The difference from a uniform distribution (1/12 per month) is shown, given in percent. Scaled months (month = 1/12 year) are used. Note the different vertical scales.

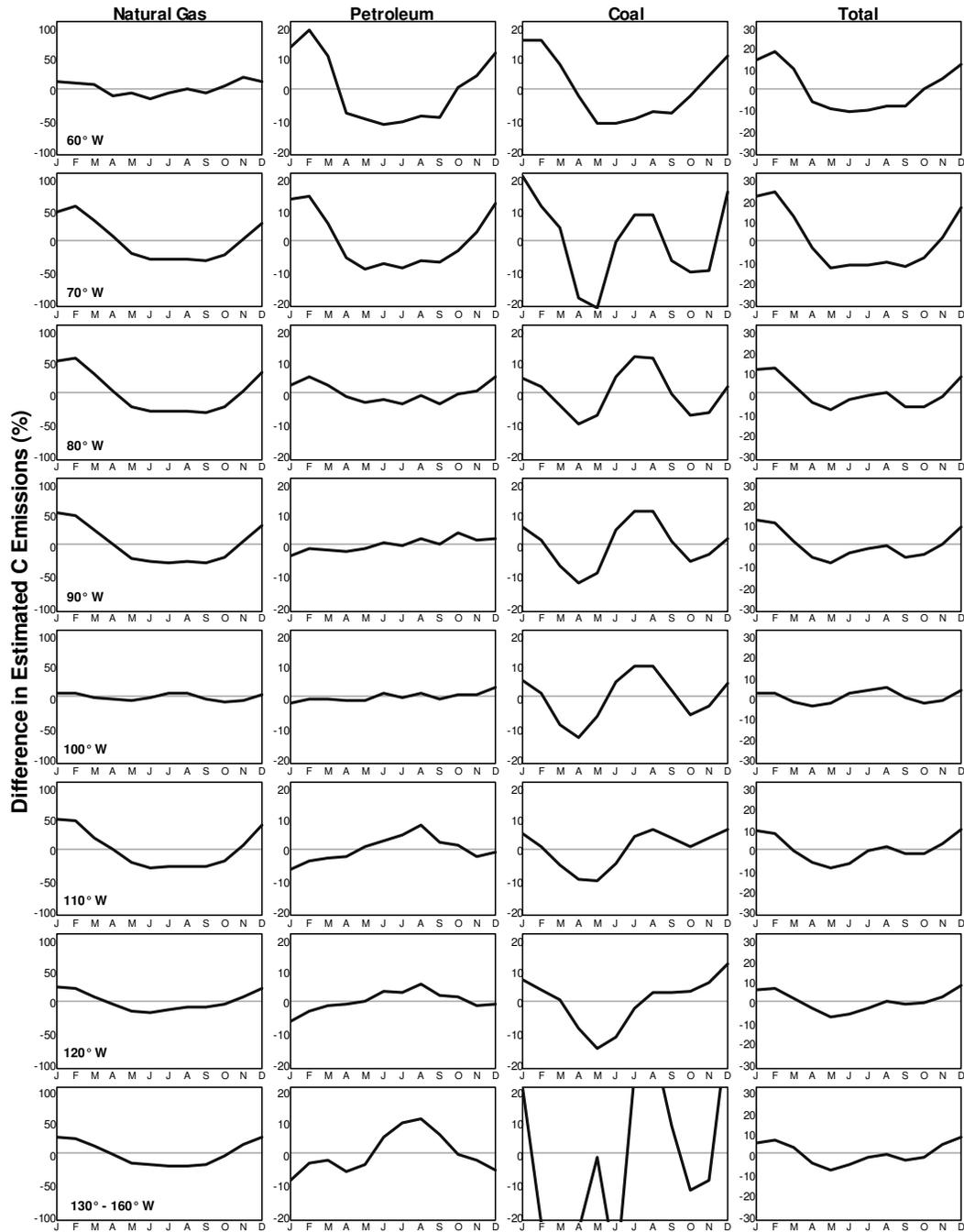


Figure 17. Longitudinal mean monthly distribution of emissions by fuel type. The difference from a uniform distribution (1/12 per month) is shown, given in percent. Scaled months (month = 1/12 year) are used. Note the different vertical scales.

Emissions from petroleum have the smallest amplitude in the seasonal cycle, but can still vary as much as 20% from a uniform distribution. However, contrasting with the emissions from other fuel types, emissions from petroleum have a seasonal pattern that increases in amplitude as a function of latitude and peaks in late summer at 60° - 70° N. This is a result of an increase in transportation in Alaska during the summer months; gasoline and jet fuel consumption in Alaska are on average 50% and 38% higher (respectively) in July versus January. Although the pattern of petroleum emissions in Figure 13 shows little seasonality for North America, emissions from petroleum use are seasonal at both the most extreme east and west longitudes. In the east (60° - 70° W), there is a winter peak due to increases in the use of distillate fuels for heating; the mean increase in distillate fuel consumption January versus July in New England (Connecticut, Maine, Massachusetts, New Hampshire, New York, Rhode Island, and Vermont) is 222% while the mean increase for the rest of the US is only 11%. In the west, (130° - 160° W) there is a distinct summer peak from increased consumption of gasoline in Alaska. When aggregated to the national or continental level, these regional patterns offset each other and the fine scale variability is lost.

Coal produces a pattern of emissions that peaks predominately in the summer at low to mid latitudes, and predominately in the winter at higher latitudes. The seasonal pattern for coal creates differences of up to 20% from a flat line distribution. Emissions from coal consumption are seasonal in North America with the highest

peak at 35° N and 80° W, a distinctive summer peak. These coordinates represent the Appalachian region in the U.S., where coal is used to generate electricity. Moving north, the seasonal pattern changes to peak in the winter months. Similarly, the summer peak diminishes as one moves west. The area 130° - 160° W is dominated by Hawai'i and Alaska, where little coal is consumed and the data are more erratic.

In general, total emissions have the largest amplitude of variability, 25% different than a uniform distribution, at 45° - 50° N, and about 70° - 80° W, (the eastern side of the continent, near the U.S.-Canada border). Erickson et al. (2008) suggested that the amplitude increases from south to north but our observation is that the amplitude decreases at latitudes north of 50° N. Moreover, Erickson et al. (2008) assumed a uniform longitudinal distribution, yet we observe that the pattern of emissions changes on an east-west gradient, with emissions more highly seasonal in the east than in the west.

The pattern of fossil-fuel-based CO₂ can be summarized spatially on a month to month level by using the monthly emissions per fuel type to weight each state's geographic centroid. In Figure 18, the geographic centroid of North American monthly emissions is plotted for each fuel type to depict month-to-month changes in the spatial distribution of emissions. The monthly spatial variation is many times larger in magnitude than annual changes in emissions, thus highlighting the increased value of finer spatial and temporal scale descriptions of fossil-fuel CO₂ emissions for specific applications. The month-to-month variation in emissions is also larger in scale than the year-to-year changes in population demography. The total emissions

centroid for North America emissions lies to the north of the population centroid for the continent (population data are from the U.S. Census Bureau (2001), Statistics Canada (2001), and the Instituto Nacional de Estadística y Geografía (INEGI) (2001)), as per capita emissions are higher in the U.S. and Canada than in Mexico, but has been slowly moving westward, and slightly south from 1990 to 2007, reflecting changes in demographics and economic development. However, the centroid of emissions is still well removed from the population centroid, illustrating both that the distribution of emissions is changing with time and that the distribution of emissions is quite different from the distribution of population. Monthly average centroids for the three individual fuels show that the geographic distribution of emissions changes in systematic ways through the year, with the greatest variability in the natural gas emissions. During the winter months the centroid of emissions from natural gas moves toward the northeast and during the summer it moves back to the southwest in reflection of the changing demand for natural gas. The spatial pattern of emissions from petroleum is seen to change less than for the other fuels and the centroid of emissions from petroleum is closest to that of population, due to the fact that petroleum has the most uniform per capita demand of all three fuel types. Emissions from petroleum shift westward in the summer months, as noted earlier. Electricity generation from coal tends to follow a less distinct pattern, but moves north in the winter and drops to the southeast in the summer.

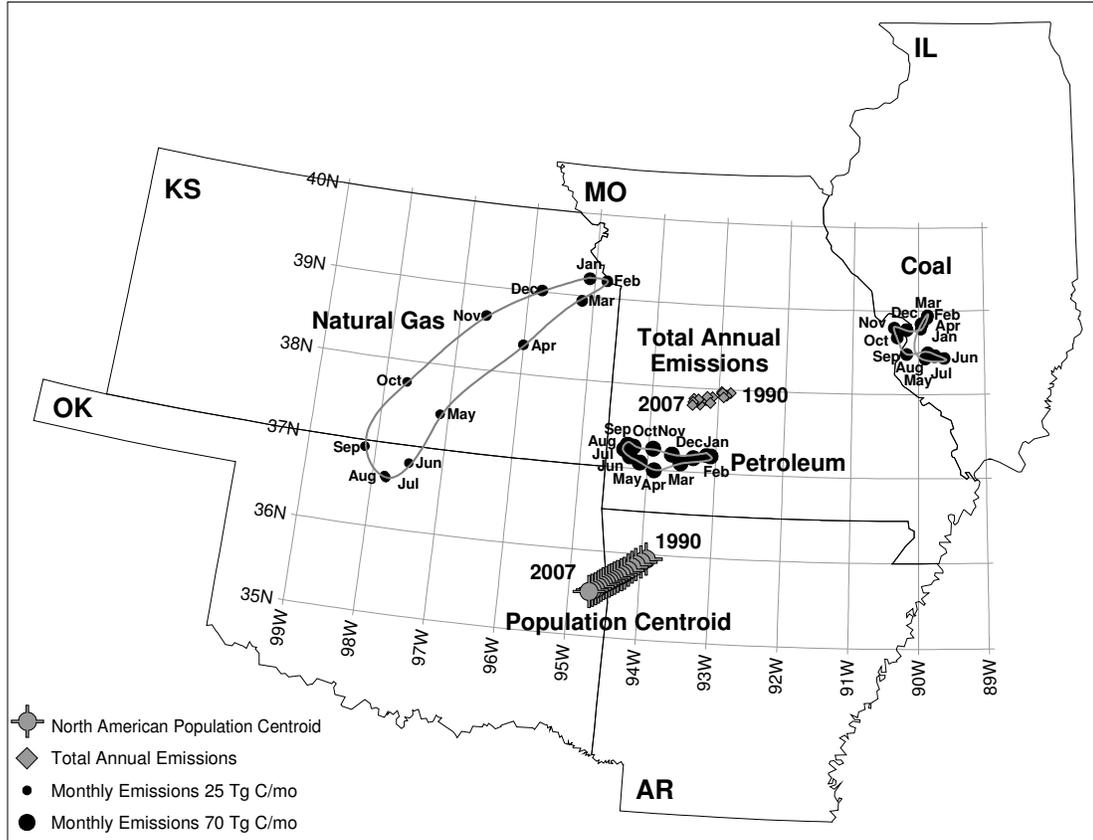


Figure 18. Monthly and annual mean centroid of fossil-fuel CO₂ emissions for North America 1990-2007. Population data are from the U.S. Census Bureau (2001), Statistics Canada (2001), and the Instituto Nacional de Estadística y Geografía (INEGI) (2001).

When carbon fluxes from fossil-fuel combustion are integrated with fluxes from the terrestrial biosphere, the common denominator will be emissions per unit area, not emissions by political unit or emissions per capita. In Figure 19 we show the

monthly variability in emissions from fossil fuel use on a per unit area basis, i.e., the peak to trough difference in monthly fossil-fuel emissions per state divided by the area of the state. The seasonal aspect of emissions in absolute emissions per area is most pronounced in the northern continental interior regions with large populations, in the Midwest U.S. and northern Atlantic states where total emissions are relatively high and the annual cycle is highly seasonal. In terms of magnitude, the seasonal variability in fossil-fuel emissions is similar or even larger than the annual fluxes from land-use activities (discussed below).

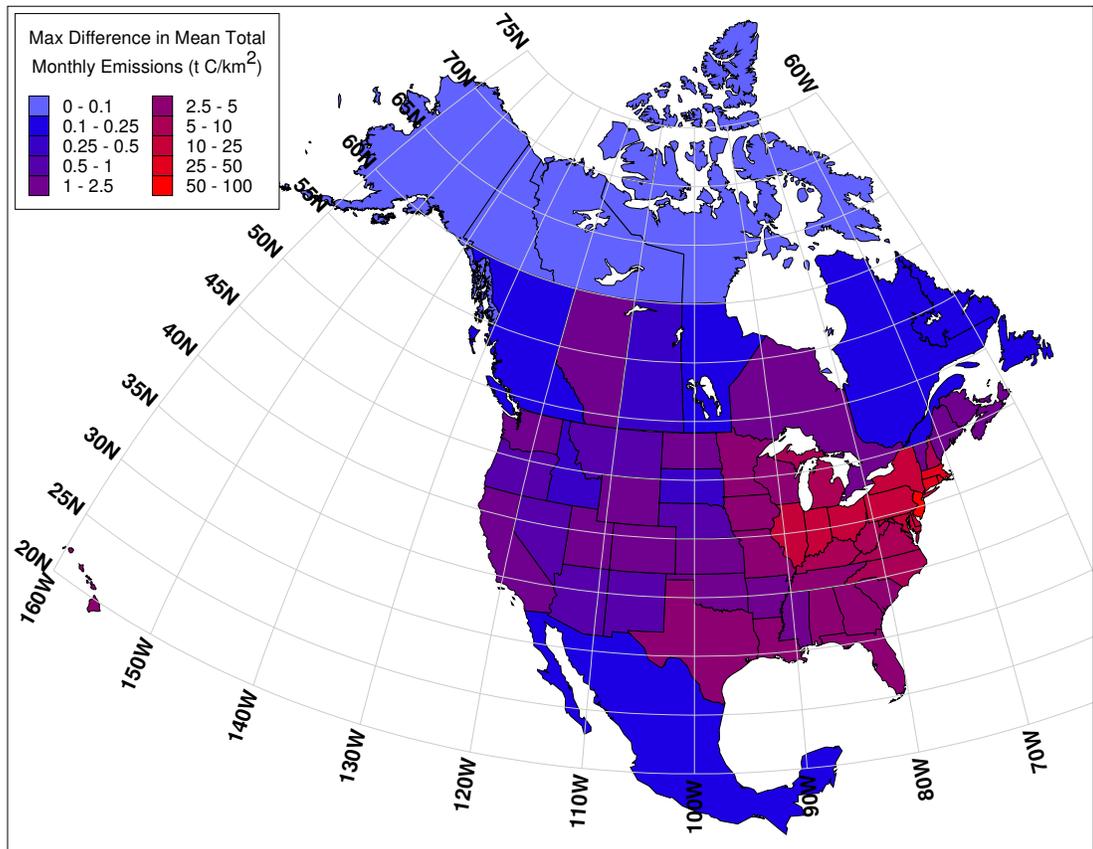


Figure 19. Maximum annual range in fossil fuel CO₂ emissions for North America, normalized by area. Scaled months (month = 1/12 year) are used. 1° × 1° gridded dataset is available.

Discussion

Assumptions and uncertainties

We assume that the seasonal and spatial distributions for non-represented sectors of a given fuel are relatively similar to the subset of consumption sectors chosen to represent the entire national consumption patterns for a given fuel type, and any differences are small enough as to not affect the total estimates significantly. Even though over 80% of total fuel consumption is represented in the underlying data for each country, it is possible to exaggerate or underestimate the seasonal amplitude when only looking at a subset of the entire market for a given fuel. For example, using electricity production as the sole proxy for emissions from coal consumption is likely to slightly exaggerate the seasonal amplitude of emissions, because coal use for steel manufacturing and other industrial purposes is not likely to be as seasonal as heating and lighting.

Another assumption implied by using sales data is that sales are equivalent to combustion and that the effects of storage and stockpiling of fuels are minimal. For example, coal is regularly stockpiled by electrical utilities during periods of lower power generation. Because of this, electricity generation (used here) is a better proxy for the monthly pattern of emissions than is actual coal sales.

However, we do rely on sales data to estimate emissions from petroleum consumption in the US, Canada and Mexico. Because consumers do not immediately combust fuel upon purchase, there is a temporal offset between sales and combustion

for motor fuel. Thus the actual emissions peaks and troughs may lag inferred values. For the US, EIA gasoline consumption data were compared to sales tax data from the Federal Highway Administration (2009) giving a comparison of distributor sales versus end consumer sales. Distributor and consumer sales agree within 10% for the majority of states, and the difference for the U.S. is less than 1%; however, it is unclear how meaningful this comparison is given differing amounts of seasonal ethanol blends per state, uncertainties in both datasets, missing and omitted data, and other differences in accounting. Moreover, this gives no further insight into the amount of time that passes between purchase and the point where the end consumer actually combusts the gasoline in a vehicle. This analysis does not indicate a discernable or consistent lag at the monthly level of analysis, though it would likely be a larger problem at finer temporal scales.

Similarly, motor gasoline sold in one jurisdiction may be actually consumed in adjacent areas. Here we assume that the majority of motor gasoline sold in a state or province is combusted within that state or province, and that the net flux of vehicles at all political boundaries is zero. For fine resolution in cities straddling state lines, where commuters buy gasoline in one state although they may actually live in the other, this assumption will break down, especially as prices may vary across political boundaries through time. The spatial displacement is more problematic for jet fuel that is purchased in airports where the fuel costs are low, but combusted along long flight paths (though the temporal displacement for jet fuel is likely less than that of gasoline and diesel).

When using this approach to estimate emissions, we assume that emissions per unit of energy content of each fuel-type remains seasonally and geographically constant. We also assume that the fraction oxidized remains constant across each country. The fraction oxidized can vary from region to region, particularly in states/provinces such as Texas and Alberta where a larger proportion of petroleum is converted to non-fuel products. The result will be a slight over estimate of emissions from petroleum in these areas. Our approach assumes an average carbon value for all coal combusted. For the U.S., the carbon content for coal combusted varies slightly from region to region. For example, in 1999 this value ranged from 23.7 g C MJ⁻¹ in Oregon to 26.7 g C MJ⁻¹ in Rhode Island and Vermont (EIA 2001).

Finally, because the effectiveness of any method is dependent on the quality of the data inputs, incorrect or missing source data values will diminish the descriptive capacity of any method. Also, changes in accounting procedures in the source data can lead to errors in the results presented here. However, because this approach keeps the total annual national emissions value constant, the input errors are spread over all point estimates and thus this approach is able to produce reasonable estimates even with many missing values and data errors (Gregg and Andres 2008).

Because of the apportioning methodology, errors are not independent, thus the uncertainty surrounding any one specific estimate (emissions in a given state/province for a given month) must be understood in the context of all other estimates for that country. We find that estimates of U.S. annual emissions from this method are within 2% of the estimates given in Blasing et al. (2005a; 2005b) and Marland et al. (2009).

Total monthly emissions for the entire U.S. are within 3% compared to Blasing et al. (2005a) relative to the monthly mean of both datasets (Gregg and Andres 2008). In a comparison to Blasing et al. (2005b), annual state-to-state emissions for the U.S. are within 0.5% (relative to the proportion of national emissions) in the U.S., except for Texas, which is 1.5%. Though different data tables were used in producing the estimates presented in this study, the ultimate source for much of the data used in this study and the previous studies by Blasing et al. (2005a; 2005b) is the EIA, so the studies are not necessarily independent. Error rates are likely higher for Canada and Mexico, but no independent datasets at the monthly time step or provincial spatial scale (for Canada) are currently available.

Fossil-fuel carbon emissions in North America

Though Canada and Mexico occupy positions among the list of the world's top 11 emitters of anthropogenic CO₂, the combined emissions from Mexico and Canada are less than a fifth of that from the U.S. More fossil-fuel-based CO₂ emissions currently come from Texas than from Mexico and Canada combined. Moreover, the amplitude of the seasonal variation in U.S. carbon emissions is greater than the total combined mean monthly carbon emissions from Canada and Mexico. Both Canada and Mexico are net exporters of energy to the U.S.; the U.S., for example, consumes about half of Canada's total annual natural gas production (EIA

2002). The pattern of CO₂ emissions in North America is dominated by emissions occurring in the U.S.

Fossil-fuel emissions are a substantial anthropogenic component of the carbon cycle. In absolute terms, the annual emissions from fossil-fuel are an order of magnitude larger than those of land-use change, for most states and provinces in North America. The 1990-2005 mean annual carbon fluxes from land use change in the U.S. and Canada are respectively estimated at -31.9 Tg C yr⁻¹ and 19.1 Tg C yr⁻¹ (Houghton 2008), whereas fossil-fuel carbon emissions averaged 1460 Tg C yr⁻¹ and 132 Tg C yr⁻¹ over the same period. On average, land use emissions were -3.2 Mg C km⁻² yr⁻¹ and 1.9 Mg C km⁻² yr⁻¹ for the U.S. and Canada, respectively. For many places, such as New England states, the seasonal variability in fossil-fuel carbon emissions is many times larger than the total annual emissions from land use change on a per area basis (see Figure 19). Total 2007 fossil-fuel emissions are 173 Mg C km⁻² in the US, 50 Mg C km⁻² in Mexico, and 17 Mg C km⁻² in Canada. We have shown that these changes in fossil-fuel emissions fluxes are highly variable temporally, and spatially among the states in the United States and provinces in Canada (and likely in Mexico also, although we could not show this). Thus, accurate depiction of the seasonal flux and sub-national spatial distribution of fossil-fuel based CO₂ emissions are essential to understanding the dynamics of the anthropogenic component of the carbon cycle. As compared to national, annual emissions inventories currently available, the higher resolution detailed data presented here allows more accurate determination of terrestrial carbon fluxes where background

levels of fossil-fuel-based CO₂ emissions are needed to calibrate measurements and models. The complete data sets of CO₂ emissions by state/province and month, for the period up through 2007, are available from the Carbon Dioxide Information Analysis Center (CDIAC) (http://cdiac.ornl.gov/trends/emis/meth_reg.html).

Conclusions

The seasonal and spatial patterns of fossil fuel CO₂ emissions in North America are an essential component to a complete understanding of the carbon cycle. Because of the complex pattern of variation in emissions it is difficult to describe either the spatial or temporal pattern in simple ways or to extrapolate from one place to another. Temporal patterns are often sufficiently clear that it may be possible to extrapolate usefully over time, but spatial variations are more difficult to infer which, for the present, precludes finer spatial resolution in Mexico.

We have been able to characterize that the annual cycle of emissions varies most with emissions from natural gas, and most at mid latitudes. Nevertheless, natural gas is not equally available everywhere and there are important variations with longitude and with the balance of economic sectors that characterize energy demand in a given location. While emissions from petroleum use are the most closely tied to population distribution of the three fuel types, there are nevertheless differences in the seasonal distribution of emissions as a function of location. Emissions from coal

consumption are not well-correlated with population, and like petroleum carbon emissions, the seasonal pattern varies with location.

We conclude from this analysis that the spatial distribution of emissions is far different than would be achieved with a uniform per capita distribution and that the temporal variation is large with respect to biospheric net fluxes from land-use change. Recognition of the temporal and spatial variability in fossil-fuel CO₂ is an important component of a detailed, mechanistic understanding of the global carbon cycle.

Acknowledgements

This work was supported by U.S. Department of Energy, Office of Science, Biological and Environmental Research Program. This project was supported by the Climate Change Research Division. Oak Ridge National Laboratory is managed by UT-Battelle, LLC, for the U.S. Department of Energy under contract DE-AC05-00OR22725. A portion of this work was supported by U.S. Department of Energy Grant DEFG02-03ER46030. This paper is intended to supplement the analysis contained in the State of the Carbon Cycle Report (SOCCR) (Marland et al. 2007b) and to aid in the objectives of the North American Carbon Program (NACP).

Chapter 5: Global and regional potential for bioenergy from agricultural and forestry residue biomass

Preface

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In Review:

Mitigation and Adaptation Strategies for Global Change

Abstract

As co-products, agricultural and forestry residues represent a potential low cost, low carbon, source for bioenergy. A method is developed for estimating the maximum sustainable amount of energy potentially available from agricultural and forestry residues by converting crop production statistics into associated residue, while allocating some of this resource to remain on the field to mitigate erosion and maintain soil nutrients. Currently, we estimate that the world produces residue biomass that could be sustainably harvested and converted into over 50 EJ yr⁻¹ of energy. The top three countries where this resource is estimated to be most abundant are currently net energy importers: China, the United States (US), and India. The

global potential from residue biomass is estimated to increase to approximately 50-100 EJ yr⁻¹ by mid- to late- century, depending on physical assumptions such as of future crop yields and the amount of residue sustainably harvestable. The future market for biomass residues was simulated using the Object-Oriented Energy, Climate, and Technology Systems Mini Climate Assessment Model (O^bJECTS MiniCAM). Utilization of residue biomass as an energy source is projected for the next century under different climate policy scenarios. Total global use of residue biomass is estimated to be 25-100 EJ yr⁻¹ by mid- to late- century, depending on the presence of a climate policy and the economics of harvesting, aggregating, and transporting residue. Much of this potential is in developing regions of the world, including China, Latin America, Southeast Asia, and India.

Introduction

Currently, the world consumes nearly 500 EJ (exajoule or 10¹⁸ J) of primary energy annually (BP 2008). Eighty-six percent of this energy is in the form of fossil fuels (EIA, 2007d) (coal, petroleum, and natural gas), resulting in over 8.5 Gt C yr⁻¹ of carbon dioxide (CO₂) emissions (Marland, Boden, and Andres 2009). As global supplies of fossil resources tighten and concerns about climate change mount, interest is growing in biomass energy as a means to replace some part of the energy portfolio currently occupied by fossil fuels. The response to the energy and climate challenges will require a dramatic restructuring of the global energy portfolio, with bioenergy likely to play an increasing role.

Recent years have witnessed a dramatic expansion of bioenergy production, particularly biofuels for the transportation sector, motivated by efforts to increase domestic energy supplies, boost rural agricultural economies, and to reduce greenhouse gas (GHG) emissions by replacing fossil fuels (Kojima and Johnson 2005; Shapouri, Duffield, and Wang 2002; Turhollow and Perlack 1991). Between 1997 and 2007, United States (US) ethanol production [almost exclusively from fermenting corn (*Zea mays* L.)] increased by a factor of 5 (EIA, 2008b); in the European Union (EU), biodiesel production has increased by a factor of 10 (EBB, 2008). Total global bioenergy consumption (including fuel wood and traditional biomass) is estimated to have increased by 70% during 1950-2000 (Fernandes et al. 2007).

In the face of such rapid growth, some have expressed concern over the aggressive expansion of bioenergy production, pointing to many potential negative consequences. Land and resource constraints create economic pressure between the various anthropogenic uses of biomass, the so-called six f's (fuel, food, feed, feedstock, fiber, and fertilizer) (Rosillo-Calle 2007b). Estimates of food price increases due to increased bioenergy demand have been from 2-12% (Ranes, Hanson, and Shapouri 1998) to over 100% (Johansson and Azar 2007). In addition, expansion of biomass production can potentially lead to increases in the conversion of natural areas to agricultural use (Righelato and Spracklen 2007; Wise et al. 2009) and losses in biodiversity (Raghu et al. 2006). Others have expressed concern that the intensification of agriculture that would result from the expanding US bioenergy

economy would bring much of the Conservation Reserve Program (CRP) land back into production, leading to a loss of wildlife habitat (Bies 2006). Fargione et al. (2008) recently introduced the idea of a “carbon debt,” which occurs when virgin lands, particularly in tropical regions, are converted into bioenergy plantations. Expansion of crops into marginal lands could also exacerbate soil erosion (Kort, Collins, and Ditsch 1998), increase consumption of water resources (Berndes 2002), and increase nutrient run-off and eutrophication of riparian and aquatic systems (Hill et al. 2006). Furthermore, a number of life-cycle studies have shown that the energy yield (particularly with corn-based ethanol) tends to hover near a break-even balance when accounting for the energy consumed in the production and processing biofuel energy crops (Shapouri, Duffield, and Wang 2002).

These drawbacks can be assuaged to some degree by utilizing residue biomass; byproducts of practices already taking place. Current research in bioenergy potential has focused primarily on developing novel, dedicated cropping systems and assessing the potential of dramatically expanding energy crop agriculture [see, for example (Hanegraaf, Biewinga, and Bul 1998)]. In contrast, less research is being conducted in utilizing residue biomass, though this resource is already produced. Sources of residue biomass include agriculture residue (stalks, stover, chaff, etc.), forestry residue (tree tops, branches, slash), and mill residue (sawdust, scraps, pulping liquors). Utilization of biomass residues allows for the same land and production practices to produce multiple products, reducing both the resource inputs and the demand for land associated with producing dedicated energy crops.

Considering the global magnitude of agriculture and forestry production, residue biomass is potentially a large and under-utilized resource. However, estimates of the magnitude of future residue biomass utilization vary widely (by as much as a factor of five) due to challenges in accurately defining the resource, the high degree of heterogeneity in feedstock sources, the uncertainties in its technical availability and sustainable recoverability, and challenges in determining the economic viability of residue biomass utilization (Rosillo-Calle 2007b). Ultimately, utilization of residue biomass is an economic decision, influenced by competing crop and energy prices as well as the price of carbon. The difficulty in tracking the various biomass residue streams led Gillingham et al. (2007) to model residue biomass by calibrating to 1990 values and allowing total production to grow as a function of Gross Domestic Product (GDP) in a study that examined the future potential for bioenergy with respect to projected land and energy demand. Other studies have taken a more detailed approach by assuming some fraction of availability for various residue streams to estimate potential global supply of biomass; Fischer and Schrattenholzer (2001) estimated the 2050 potential for bioenergy from agricultural residues to be 35 EJ yr⁻¹, with another 100 EJ yr⁻¹ from forestry (which includes both forestry residues and purpose-grown forest biomass), based on mean biomass productivity rates and global land cover. In a review of several studies, Hoogwijk et al. (2003) estimated the potential for modern primary energy from biomass residues to range between 30 EJ yr⁻¹ to 108 EJ yr⁻¹ by 2050; about 14 EJ yr⁻¹ are produced currently, largely from mill residues. Of note here is that the range for total modern (as opposed to traditional) biomass utilization

(which includes dedicated energy crops) varies widely from 33 EJ yr⁻¹ to 1135 EJ yr⁻¹, depending on assumptions about the availability of land for biomass crop production. In a hypothetical future scenario where all available cropland is used to produce food and fiber, or where converting virgin land for biomass crop incurs an unacceptable carbon debt (Fargione et al. 2008), then residue biomass could provide the major feedstock for expanding bioenergy production. The objectives of this study are to quantify and characterize the current potential supply of residue biomass, and to model the utilization of this resource in the 21st century.

Methods

Current Available Residue Biomass

The framework for projecting the potential for residue biomass functions in two parts. First, the maximum available sustainable supply of biomass residue is estimated based on crop and forestry production statistics and crop-specific parameters. To determine the maximum available supply of biomass residue, national agricultural and forestry production statistics were obtained from the Food and Agriculture Organization of the United Nations (FAO) database (FAOSTAT 2008b, 2008c). For each crop, the harvest index, water content, and residue energy content were estimated from various sources. In addition, for each crop, we estimated a residue retention value- the amount of residue needed to mitigate soil loss and

preserve soil nutrients. Taken together these allow an estimate of the total potential supply of residue biomass.

Second, to project the future production of residue biomass, a market is simulated to estimate the fraction of the maximum sustainable supply of residue biomass that would be collected and utilized. The utilization of residue biomass is simulated and projected with an integrated assessment model for the next century for 14 aggregated regions of the world.

The total amount of residue produced is estimated using harvest index (HI_{dry}) statistics, which represent the dry mass ratio of the harvested crop to the total aboveground biomass, taken from the Environment Policy Integrated Climate (EPIC) model inputs (Williams 1990). For root crops, such as sugar beet (*Beta vulgaris* L.), the harvested crop is below ground biomass, and thus these crops can have reported harvest indices greater than 1. The harvest index, HI_{root} , is adjusted by the following equation for root crops:

$$HI_{root} = \frac{HI_{dry}}{HI_{dry} + 1}$$

Additionally, for orchard and tree crops, we define the harvest index to be the ratio of the harvested crop mass to the sum of the masses of the harvested crop mass and pruned material. Forest and mill parameters were estimated from Perlack et al. (2005) and from a report to the US Department of Energy National Renewable Energy Laboratory by the Antares Group (1999).

Because crop and forestry production statistics are reported on a wet mass basis, the harvest index is adjusted to account for the mass of water in the crop by the following formula:

$$HI_{wet} = \frac{HI_{dry}}{water\ content \cdot (HI_{dry} - 1) + 1}$$

This adjustment allows the determination of residue biomass ratio (dry basis) for every crop in the FAO database by inversion of the HI_{wet} value. The useful form is the *Residue Ratio*, which when multiplied by crop production, gives the total amount of aboveground crop residue:

$$Residue\ Ratio = (HI_{wet}^{-1} - 1)$$

Not all residue is logistically harvestable, however. Moreover, additional residue biomass must remain uncollected to sustain soil nutrients and to prevent erosion. While soil nutrient levels and erosion are a function of local topography, climate, soil, and management practices, here we assume a *Residue Retention* parameter, general crop-specific values in terms of mass of residue per unit area to remain on the field.

$$Residue\ Biomass\ Available = Production \cdot Residue\ Ratio - (Residue\ Retention \cdot Harvested\ Area)$$

The initial values for *Residue Retention* parameter are calculated by taking a percentage of the total available residue based on the mean global 1990 and 2005 FAO yield statistics (FAOSTAT 2008b) and the *Residue Ratio* given above. These crop-specific values are then kept constant for all locations and time periods. This

allows for greater amounts of residue to be harvested as yields increase, and less residue to be harvested if agriculture expands into marginal land that is more susceptible to soil and nutrient loss. For major grain and oil crops [corn, wheat (*Triticum aestivum*), barley (*Hordeum vulgare* L.), oats (*Avena sativa* L.), rapeseed (*Brassica napus* L.), etc.] we assume that the residue retention is 70% of the calculated total available residue (a maximum residue harvest rate of 30%) (Graham et al. 2007; Wilhelm et al. 2007). Likewise, it is assumed that for pruned crops [grapes (*Vitis* spp.), oranges (*Citrus* spp.), tree nuts, etc.] 99% of the estimated residue is recoverable (1% is retained), and for root crops [potatoes (*Solanum tuberosum* L.), sugar beets, etc.] —where the entire plant is harvested—95% of the residue is recoverable (5% retained). For rice (*Oryza sativa*), and all other miscellaneous crops (fruits, vegetables, etc.), it is assumed that 75% of the residue is recoverable (25% retained). These percentages are then converted to crop-specific mass values based on the mean global average 1990 and 2005 production and harvested area statistics in the FAO database (FAOSTAT 2008b) (Table 10). For forests, the residue retention value is estimated to be 2 Mg ha⁻¹, calculated from Pannkuk and Robichaud (2003) (Table 10).

Table 10. Summary of input parameters. Presented here are weighted global means; regional values vary based upon the mix of crop and timber production within groups.

Residue Source	Residue Ratio (dry residue mass/ wet crop mass)	Residue retention (Mg ha⁻¹)	Residue Energy Content (MJ kg⁻¹)	Curve Exponent <i>b</i>	Midprice (2005\$)
Wheat	1.49	2.81	16.22	10.83	2.08
Rice	0.99	0.94	13.55	10.83	2.08
Corn	0.74	2.20	16.86	10.83	2.08
Other Grain	1.02	1.09	15.20	10.83	2.08
Oil Crops	1.28	1.26	13.26	10.83	2.08
Sugar Crops	0.28	1.24	15.21	10.83	2.08
Misc Crops	0.38	0.38	8.17	10.83	2.08
Timber	0.33	2.00	18.93	7.024	2.48
Mill	0.30	0.00	20.00	3.439	1.46

Finally, the net energy content (lower heating values) of residue biomass is estimated on a dry mass basis for each crop (Goswami, Kreith, and Kreider 2000; Tyagi 1989). For all crop and forest parameters, missing values are estimated by using values for similar crops where data are available.

The maximum supply of agricultural residues is thus a function of crop-specific attributes, crop production, and harvested area. For forestry, two residue streams are considered: timber harvesting residue (tree tops, slash, and branches), and mill residue (wood scraps, sawdust, and recovered pulping liquors). As with agricultural crops, the harvest index, milling efficiencies, wood energy content, and residue retention values are used to estimate the total potential supply of forestry residues (Table 10). Specifically, for a given region and given crop type, the total amount of biomass is estimated by the following formulation:

$$\text{Maximum Residue Biomass Energy Available} = [\text{Production} \cdot \text{Residue Ratio} - (\text{Residue Retention} \cdot \text{Harvested Area})] \cdot \text{Energy Content}$$

This formula is used for agricultural, forestry, and mill residue, though for mill residue the residue retention parameter is zero. This formula is employed for all crop types, forestry, and mills, for all countries in the FAO database.

Future Role of Residue Biomass

To estimate the future role for this resource, the economic dimension is added and the previous parameter estimates are aggregated into 14 world regions (Figure 20) and seven crop types by using a weighted mean of the 1990 and 2005 FAO (2008b) crop production statistics (Table 10). The economics of harvesting residue biomass is simulated using data generated for the EIA NEMS (Energy Information Administration National Energy Modeling System), a model developed by the US Department of Energy to forecast US energy markets (supply, demand, prices, etc.) in order to inform energy policy decisions (EIA, 2003). The input data for the EIA NEMS estimates the amount of biomass energy produced per NEMS coal region in the US given a price for bioenergy. The cost curve data represent the delivered cost of residue biomass, assuming the maximum economic distance of transportation to be 50 miles (80 km) from farm gate to processing plant. Transport costs, which are included in the cost curves, are assumed to be between \$10 and \$13 (2005\$) per short ton (\$11 Mg⁻¹ and \$14 Mg⁻¹) (EIA, 2006). For mill wastes, the maximum economic travel

distance is 100 miles (160 km) (EIA, 2006). Cost of transport for these wastes is calculated stepwise for 25, 50, 75, and 100 miles (40, 80, 120, 160 km) with the price being \$0.26 per short ton-mile ($\$0.17 \text{ Mg}^{-1} \text{ km}^{-1}$) (the national average freight shipping rate) and adjusted by state transportation indices (EIA, 2006). Furthermore, the EIA assumes that there is no trading across different US coal districts when deriving the point data (EIA, 2006). Separate EIA NEMS cost curve data are available for agricultural residue, forestry residue, and mill residue. Moreover, the EIA NEMS projected cost curves evolve through time as biomass harvesting is assumed to become less expensive.

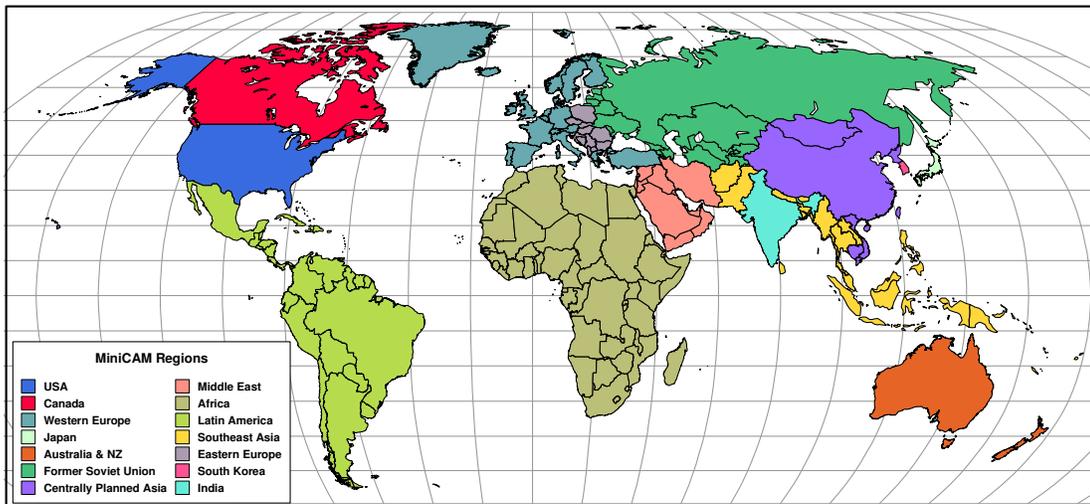


Figure 20. Map of aggregated world regions for the OBJECTS MiniCAM.

For this study, the regional EIA NEMS cost curves are aggregated into a single set of data points for the entire US, by calculating the mean of estimated residue biomass production at each cost increment. These data points are then

converted to relative proportions of the maximum production. A logistic curve is fit to these point data and is defined by the following equation:

$$p = \frac{price^b}{Midprice^b + price^b}$$

where p is the dimensionless proportion, varying from 0 to 1, of the *maximum residue biomass energy* that is supplied to market. The *price* is the independent variable and represents the equilibrium price for biomass energy in 2005 US dollars per GJ. The curve is defined by the *Midprice*, the price where half of the total available is demanded, and b , an exponent controlling the steepness of the curve. Distinct curves are created for agricultural residue, forestry residue, and mill residue (Figure 21). However, for this study, it is assumed that there is no regional variability or evolution in the curves and prices for residue biomass; the same curves are used for all 14 regions of the globe and all time steps. In later years, the NEMS cost curves evolve with expanding biomass production; however, for purposes of this study, only the initial cost curve is used since we directly account for expanding residue biomass production by modeling future agriculture and forestry production with an integrated assessment model, the Object-Oriented Energy, Climate, and Technology Systems Mini Climate Assessment Model (O^{bj}ECTS MiniCAM).

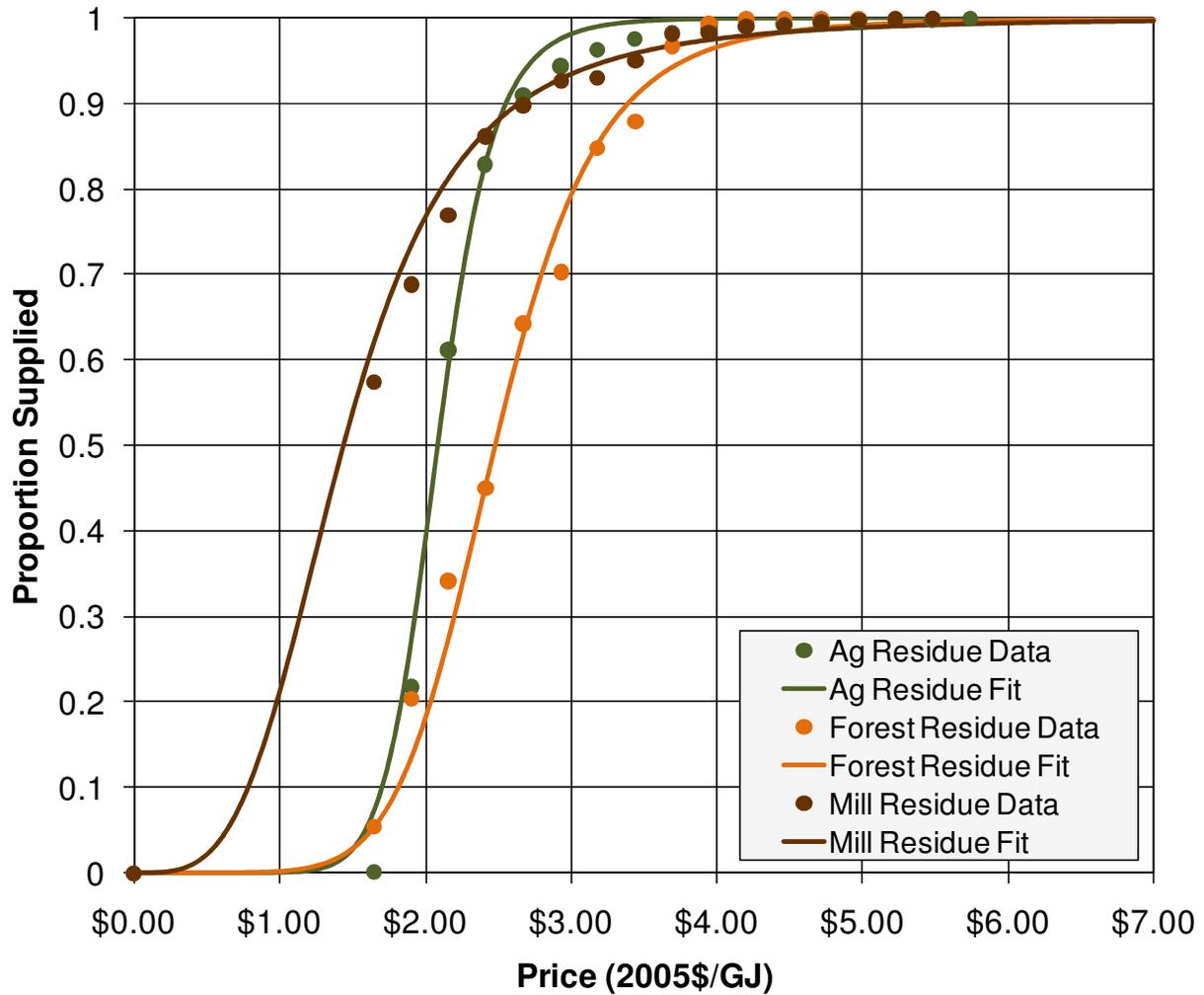


Figure 21. Cost curves for residue biomass from agriculture, forestry, and mills. Data from EIA NEMS (2003). See Table 10 for curve-fit parameters.

Global and regional supply and demand for energy from residue biomass is modeled with the OBJECTS MiniCAM, a modular, object-oriented, partial equilibrium, integrated economic model that simulates long term changes in energy

markets, land use, and greenhouse gas emissions over the next century under various GHG stabilization climate policy scenarios. Development of the general model structure can be found in Edmonds et al. (2004) and is based on Edmonds and Reilly (1985). The model operates on 15-year time steps from 1990 to 2095 for 14 aggregated regions of the world (Figure 20), and uses current aggregated economic, demographic, energy consumption, agricultural, forestry, and land use data to calibrate the historical years of 1990 and 2005. For each region, the model estimates GDP based on assumptions about labor productivity and then estimates energy demand by end use. The model is designed to simulate, under various carbon markets, the integrated interactions between energy production (coal, petroleum, natural gas, nuclear, solar, geothermal, hydro, wind, biomass, and future exotic energy sources), energy transformation (e.g., refining, electricity production, hydrogen production), energy end use (buildings, industry, transportation), agricultural production (corn, wheat, rice, other grains, oil crops, sugar crops, fiber crops, fodder crops, miscellaneous, and biomass crops), forestry and forest production (both for managed and unmanaged forestland), rangeland and animal production, as well as land allocation dynamics. The O^{bj}ECTS MiniCAM employs the Model for the Assessment of Greenhouse gas Induced Climate Change (MAGICC), a simple climate model, which balances equations for sources and sinks of carbon across six reservoirs (ocean, atmosphere, and four terrestrial types) and estimates a range of feedbacks based on modeled temperature changes (Wigley 1993). Both the O^{bj}ECTS MiniCAM and

MAGICC have been used in numerous IPCC reports to develop emissions and climate scenarios.

The projected global economic development and population growth pathways are from Clarke et al. (2007), a scenario similar to the IPCC B2 scenario from the Special Report on Emissions Scenarios (SRES) (Nakićenović et al. 2000). This scenario features a continuously growing population (leveling off at about 9.5 billion people by the end of the century), and intermediate economic growth (Clarke et al. 2007). Global policy GHG stabilization pathways are based on Wigley, Richels and Edmonds (1996) and are designed to optimally reach the target atmospheric CO₂ concentration by the end of the century. For this study, the O^{bj}ECTS MiniCAM is used to simulate the future market for residue biomass under both a reference scenario and a policy scenario that reaches 450 ppm atmospheric concentration of CO₂ by the end of the century.

Results and Discussion

Current Available Residue Biomass

In 2005, if all sustainably collectable residue were converted to energy, it could have supplied over 50 EJ to the global energy market (Table 11), roughly half the annual energy consumption of the US. Table 11 gives the leading countries in terms of total potential residue biomass available in 2005, as well as its source. Major

agricultural producers such as China, India, and the US, top the list, with the potential to produce over 5 EJ yr⁻¹ each from this resource.

Table 11. Potential residue biomass energy for 2005 (EJ). Listed countries could produce over 1 EJ yr⁻¹ of bioenergy from current residue biomass.

Residue Source	Wheat	Corn	Rice	Other Grain	Oil Crops	Sugar Crops	Misc Crops	Forest	Mill	Total
China	1.31	0.76	2.07	0.10	0.92	0.41	0.45	0.88	0.77	7.67
United States	0.49	2.39	0.12	0.15	0.53	0.13	0.10	1.36	1.07	6.34
India	0.45	0.00	1.31	0.08	0.28	1.10	0.24	0.99	0.77	5.21
Brazil	0.00	0.01	0.13	0.02	0.20	1.97	0.30	0.77	0.59	3.99
Indonesia	0.00	0.02	0.57	0.00	1.53	0.14	0.22	0.31	0.24	3.03
Malaysia	0.00	0.00	0.02	0.00	1.72	0.01	0.01	0.08	0.05	1.89
Canada	0.20	0.08	0.00	0.13	0.18	0.00	0.05	0.62	0.49	1.74
France	0.65	0.11	0.00	0.18	0.13	0.02	0.04	0.18	0.14	1.45
Nigeria	0.00	0.00	0.02	0.10	0.32	0.00	0.47	0.21	0.16	1.30
Germany	0.43	0.03	0.00	0.26	0.14	0.02	0.02	0.17	0.12	1.18
Russian Federation	0.03	0.01	0.01	0.02	0.02	0.01	0.07	0.54	0.32	1.02
<i>Global Total</i>	<i>5.58</i>	<i>4.16</i>	<i>6.51</i>	<i>2.01</i>	<i>7.41</i>	<i>6.17</i>	<i>3.89</i>	<i>10.06</i>	<i>7.85</i>	<i>53.64</i>

The relative size of the 2005 potential resource by continent is shown in Figure 22 through Figure 27, though the scales change between figures. In Africa, the majority of the current potential residue biomass is primarily in forestry and mill residues (Figure 22). Nigeria, Ethiopia, and Egypt represent the countries within Africa with the largest estimated amount of residue biomass available, with Egypt the only country in Africa with significant grain residue. China and India are the main countries in Asia with respect to residue biomass availability (Figure 23). This biomass is primarily in the form of wheat stalks and rice stalks in China and

sugarcane bagasse and rice stalks in India. Oil palm residue from Indonesia and Malaysia is also substantial potential residue biomass resource (Figure 23). In Europe, wheat straw is the major component to residue biomass, except in northern Europe and the Russian Federation where forestry residue potentially plays a larger role (Figure 24). France, Germany, and the Russian Federation are the countries with the greatest amount of residue biomass in Europe (Figure 24). In North America, the US is the leading country in terms of the size of the residue biomass resource, primarily in the form of corn stover, and forestry and mill residue (Figure 25). In Canada, the forests and mills provide a larger proportion of residue biomass and in Mexico sugarcane bagasse is a larger potential resource than the US (Figure 25). Sugarcane bagasse is also the dominant source of residue biomass in Australia, the leading producer of residue biomass in Oceania (Figure 26), though the amount of residue available in Oceania is considerably smaller than that of other continents. Residue biomass in South America is also dominated by sugarcane bagasse, primarily in Brazil (Figure 27).

In the 21st century, as global population expands and demand for food and forest products increase, the potential residue biomass supply is expected to also increase. Increasing global crop production is expected to occur through increasing crop yields on current agricultural land, and through bringing more land into production. Assumptions about the future trends of crop yields and agricultural expansion affect the cost and the total future sustainable supply of this resource.

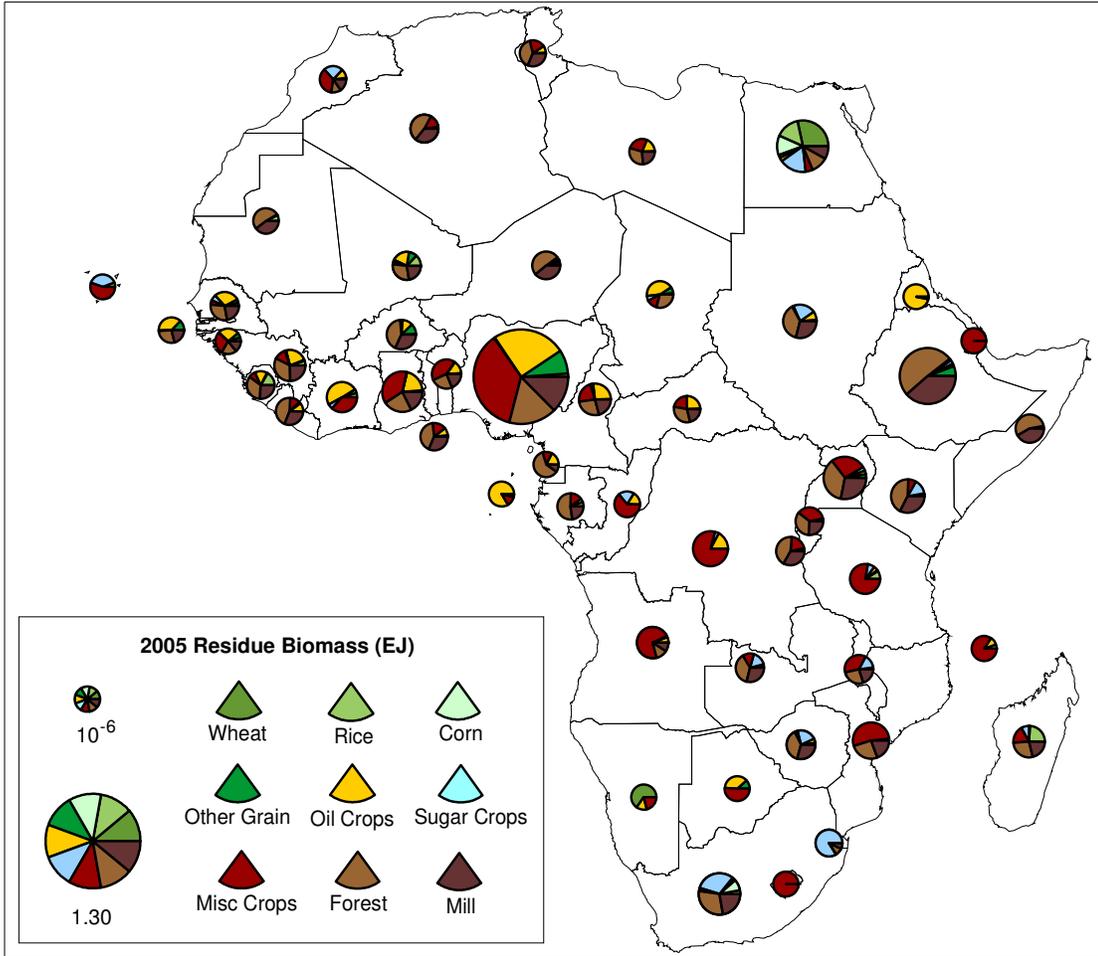


Figure 22. Potential bioenergy from residue biomass sources, Africa, 2005.

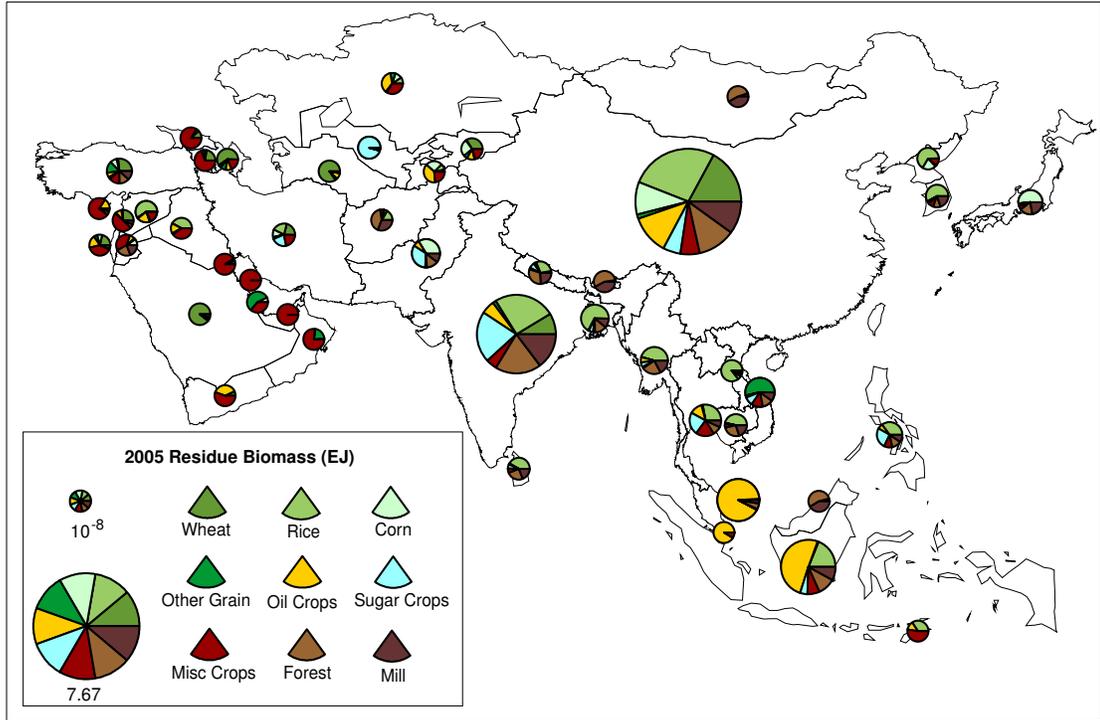


Figure 23. Potential bioenergy from residue biomass sources, Asia, 2005.

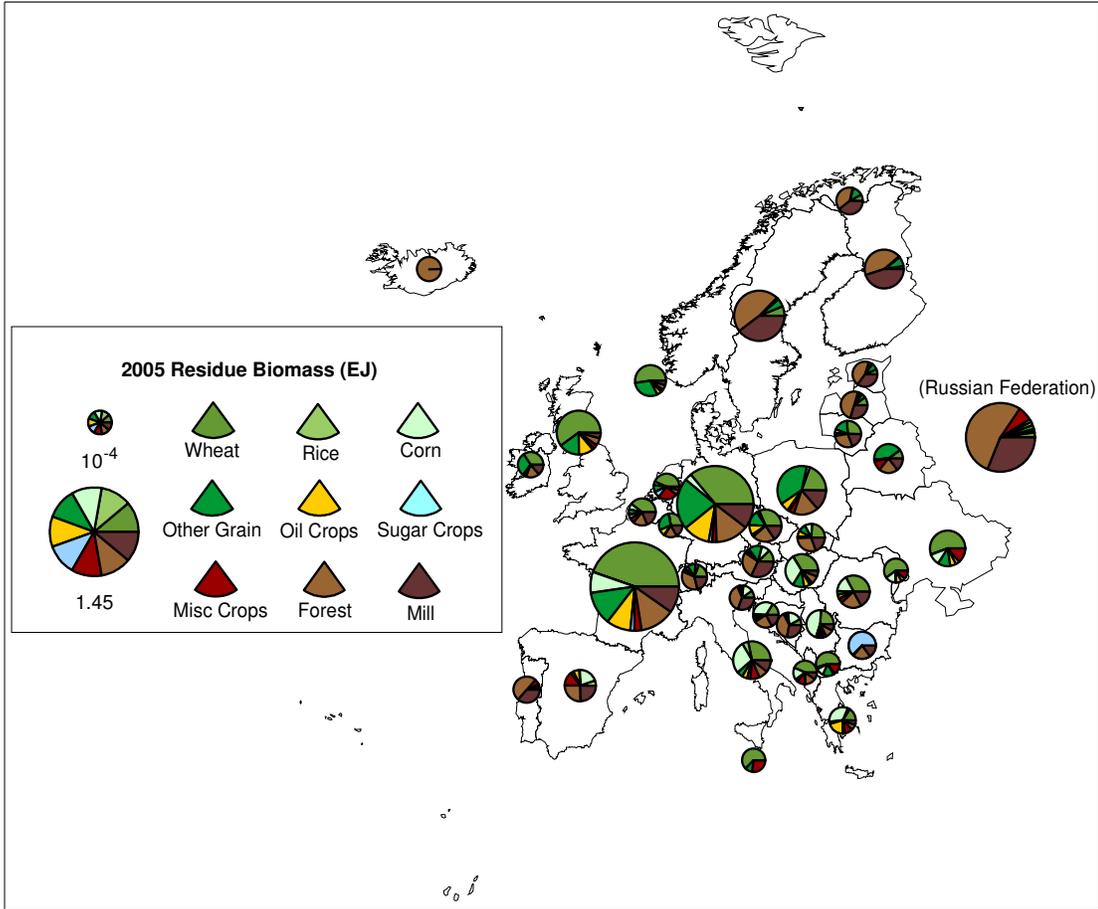


Figure 24. Potential bioenergy from residue biomass sources, Europe, 2005.

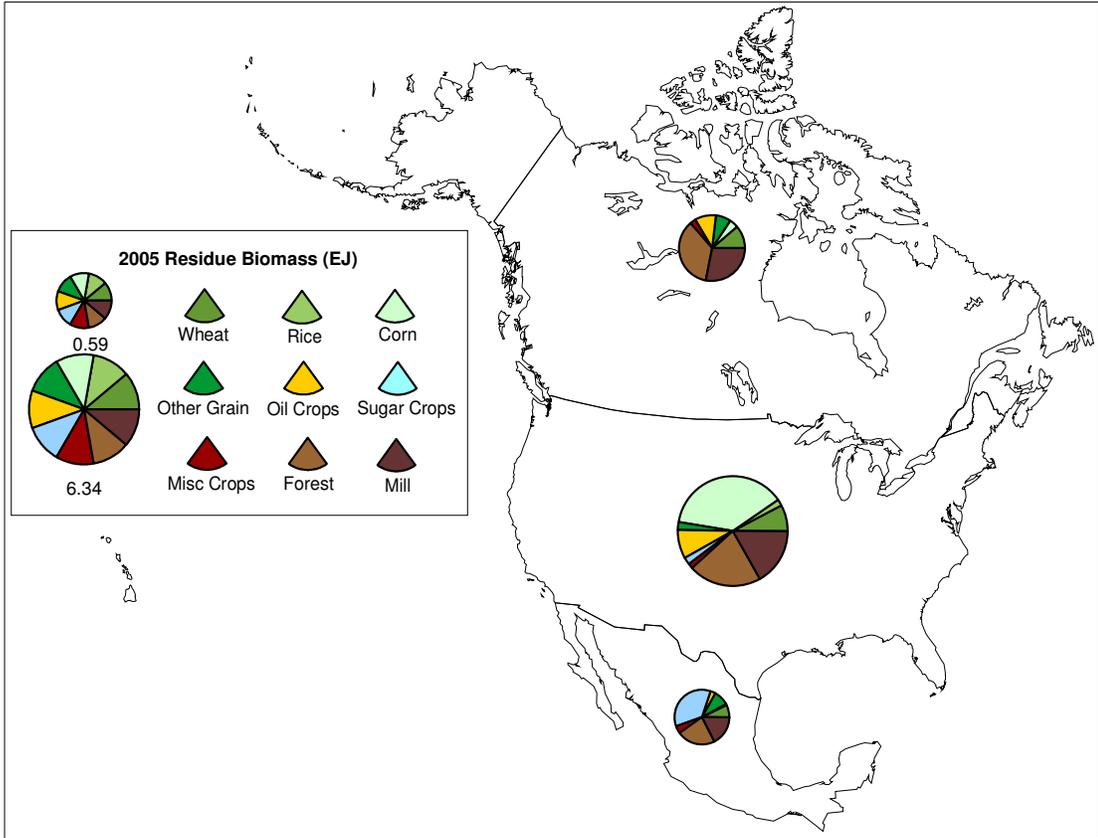


Figure 25. Potential bioenergy from residue biomass sources, North America, 2005.

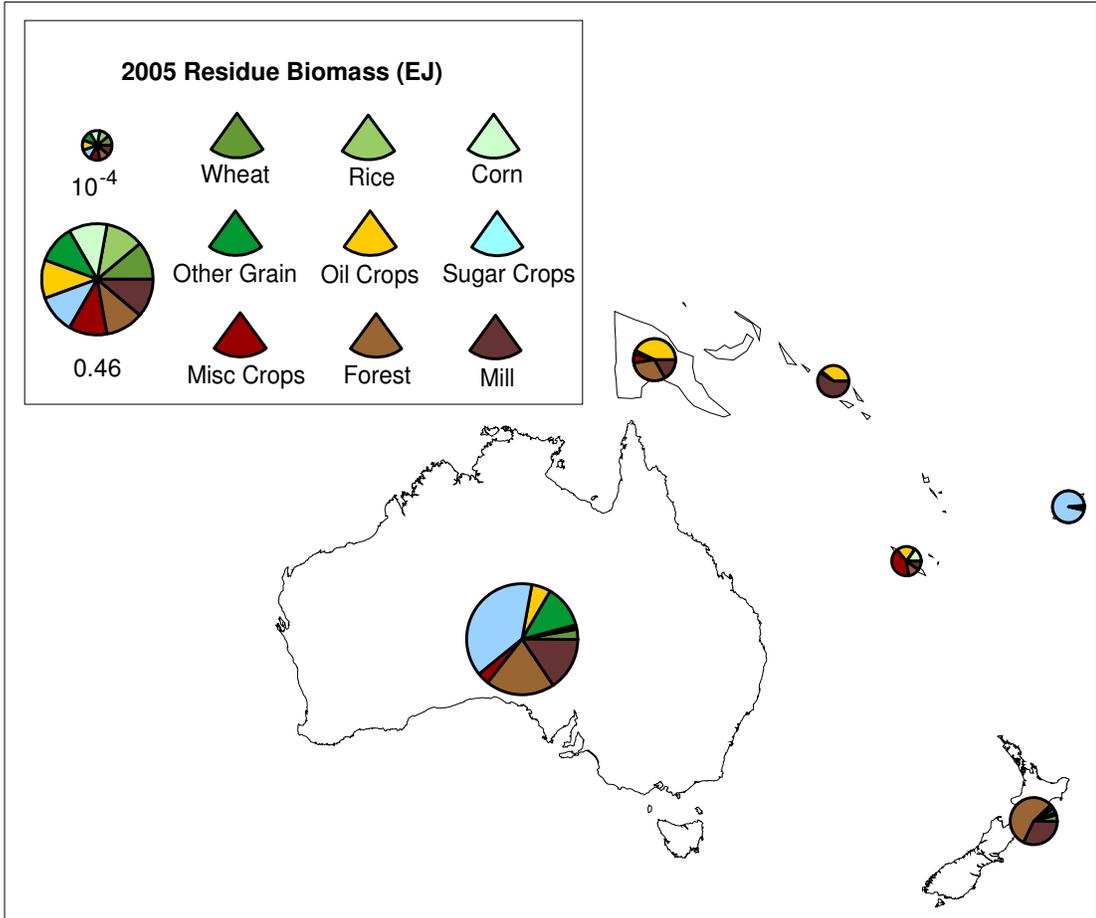


Figure 26. Potential bioenergy from residue biomass sources, Oceania, 2005.

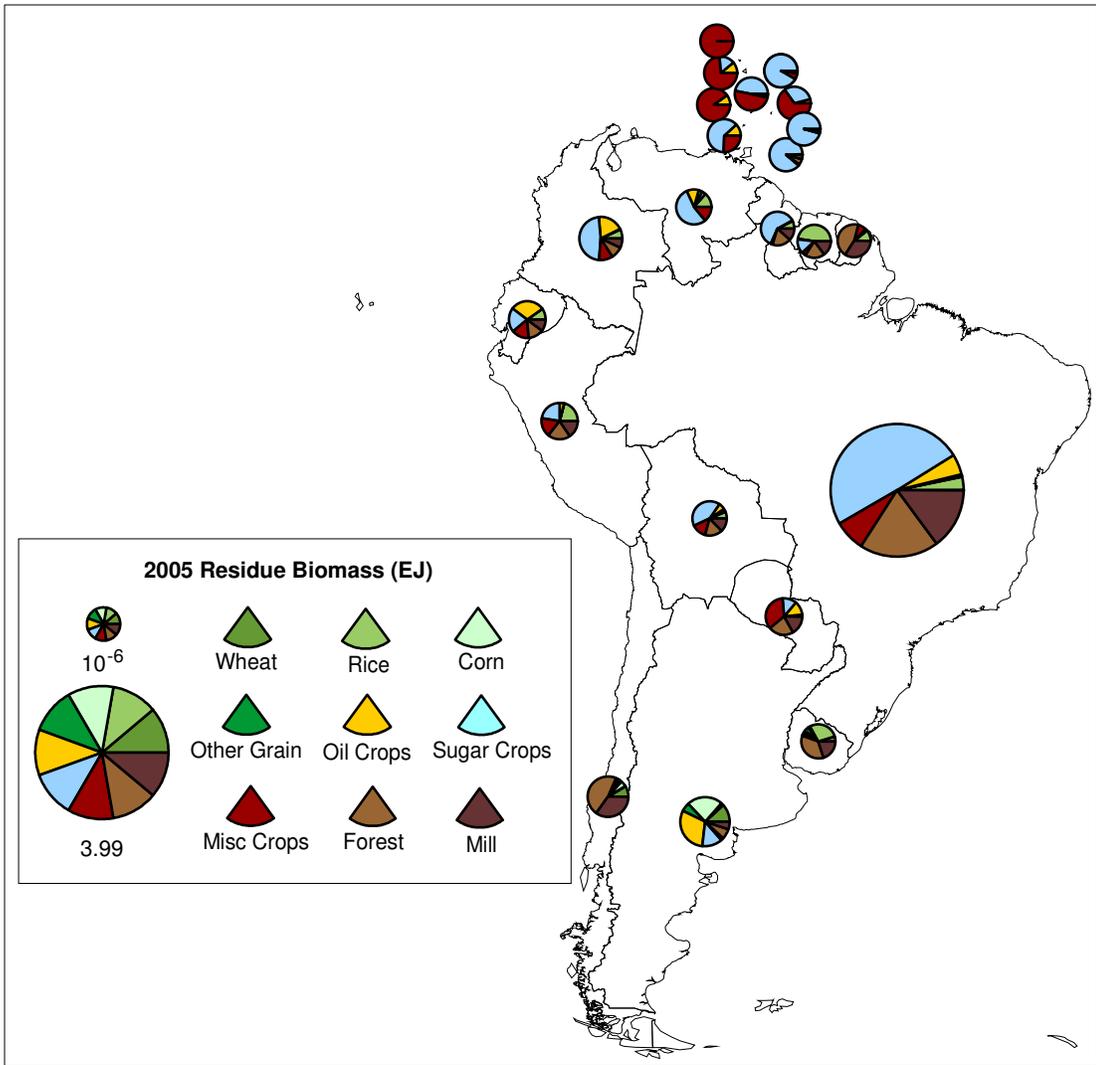


Figure 27. Potential bioenergy from residue biomass sources, South America, 2005.

Future Role of Residue Biomass

To project future scenarios, the price for biomass is computed based on total energy demand and the prices for competing sources of energy. Under a climate policy scenario, fossil energy sources become more expensive; thus demand and price for biomass increase. In the policy scenario, the total available supply of residue biomass (~90 EJ yr⁻¹) is projected to be utilized as energy by mid- to late-century (Figure 28, b and d). In the reference scenario, with no price of carbon, much of the residue biomass is still utilized as energy demand and energy prices increase, reaching a projected global output of approximately 80 EJ yr⁻¹ (Figure 28, a and c).

Centrally Planned Asia (China), Latin America, Southeast Asia, India, and the US have the highest residue biomass production; each of these regions could produce over 10 EJ yr⁻¹ from this resource, particularly under a climate policy scenario (Figure 28b). Though the composition of the residue varies from region to region (China and Southeast Asia are projected to produce more rice straw, Latin America is projected to produce more timber residue, the US is projected to produce more corn and mill residue), globally, mill residues represent the largest utilized resource, followed by forest residue, then oil crops, wheat, sugar crops, rice, miscellaneous crops, corn, then other grains (Figure 28, c and d).

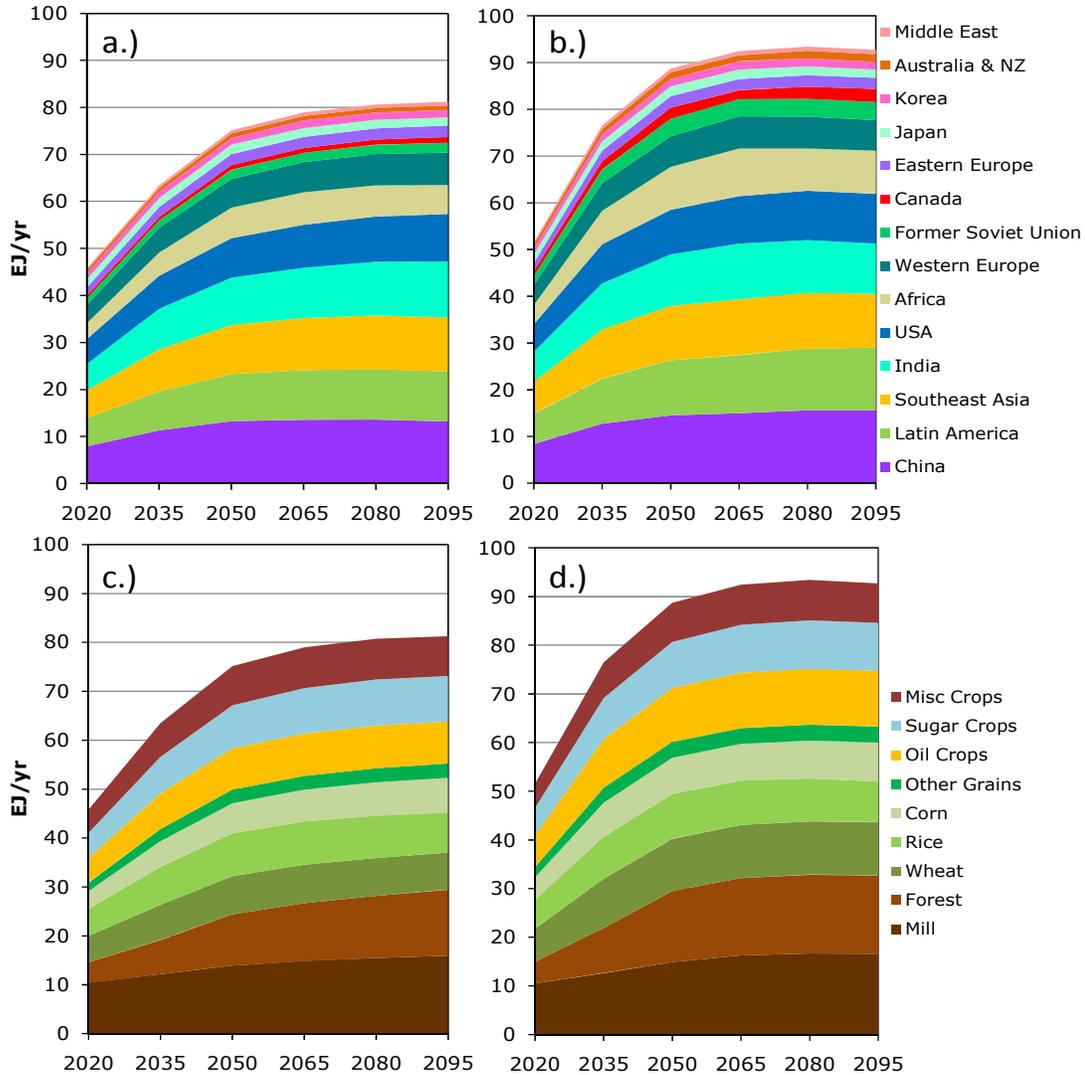


Figure 28. Projected residue biomass energy utilization for the next century. a.) Spatial distribution of residue projected biomass energy distribution, reference scenario (no climate policy). b.) Spatial distribution of projected residue biomass energy distribution, policy scenario (450 ppm atmospheric concentration of CO₂). c.) Composition of projected global residue energy utilized, reference scenario. d.) Composition of projected global residue biomass energy utilized, policy scenario.

Given the cost curve and assumed price structures, residue biomass, which requires no additional land to produce, is more economically competitive than dedicated biomass crops [e.g., switchgrass (*Panicum virgatum*) or hybrid poplar (*Populus* spp.)]. For example, in the 450 ppm CO₂ atmospheric concentration policy scenario, residue biomass meets nearly all the biomass energy demand, nearly three-fourths by mid century, and still nearly two-thirds by the end of the century. Dedicated biomass becomes economical at higher biomass prices when the maximum global output of residue biomass approximates 90 EJ yr⁻¹. Dedicated biomass is projected to contribute an additional 60 EJ yr⁻¹ of energy by the end of the century, but is generally more expensive if land carbon is incorporated in the price of production as assumed in these scenarios (Wise et al. 2009).

While this resource requires little new technology to produce (it is already available as a co-product of farming and forestry), full utilization will depend on technological and economic optimization of processing mixed streams of biomass feedstocks. This could include co-firing for electricity, pyrolysis and gasification (perhaps with biochar returned to the soil), or conversion of cellulose to ethanol. Figure 22 through Figure 27 suggest that biomass processing facilities would receive feedstocks from seasonally consistent sources and thus the engineering processes may be optimized for the local feedstocks and energy markets (i.e., liquid fuels, electricity, etc.)

Sensitivity Analysis

The limiting factors for supply of residue biomass feedstock are both physical and economic. Physical factors include assumptions about future agricultural and forest productivity, and the amount of residue that must be left on the field to mitigate soil erosion and maintain soil nutrients (the residue retention parameter). Economic factors include the cost of collection, aggregation, and transport—captured by the *Midprice* parameter—and assumptions about the shape of the cost curve (*b*) as parameterized in the equations. Because no large-scale energy market currently exists that demands residue biomass, projection of future utilization depends on assumptions about these initial conditions. Therefore, a series of sensitivity tests were conducted to determine the effect these parameters have on the projected utilization or residue biomass.

The largest uncertainty for projecting the amount of energy from residue biomass in the future rests in assumptions about future crop productivity. The default scenarios assume modest increases in agricultural yields for the rest of century, in line with historical increases in crop yields. This reference scenario is based on a FAO report that projects crop yield change to 2030, and assumes yields improve at a slightly faster rate in the developing world, and retain the historical yield increase rate for the developed world. After 2030, we assume the yield changes converge to 0.25% by 2050 for all crops in all regions. Two other scenarios were tested, one in which agricultural yields were held steady at current levels (no yield increases), and one assuming advanced technological developments that increase crop yields dramatically

(approximately double the reference rate) in the next century. Assumptions about future agricultural yields have a large impact on residue availability, more so than the presence of a climate policy (Figure 29a). More residue is expected to be supplied under the reference scenario with high or default yield assumptions than the policy scenario with a low yield assumption (Figure 29a). This is because as yields increase, more residue becomes available (Figure 29b). Thus, more residue can be supplied to as an energy feedstock (Figure 29a). This effect could be more pronounced than modeled here, because if there is more residue available per area, per mass collection costs would likely decrease and the cost curve would change accordingly. The magnitude of the yield effect is non-linear, however, because if future yields are high for all crops, there would be more agricultural land available to grow dedicated biomass, it would be less expensive to produce (because of higher yields), and would therefore be more competitive with residue biomass by the end of the century.

On the other hand, if yields do not increase much in the future, agriculture must expand into less fertile land to produce more food for the growing world population. In this scenario, residue is spread out over larger areas, a higher proportion of the residue must remain on the field to mitigate erosion and less residue biomass is projected to be supplied as an energy feedstock.

The residue retention parameter was also tested by altering the initial default values by 50% and 200%. More stringent residue retention requirements reduce the amount of residue supplied. Varying the residue retention parameter by 50% to 200%

of the default value has about a 25% effect on the global residue supplied by the end of the century (Figure 29c), and little effect on the projected price (Figure 29d).

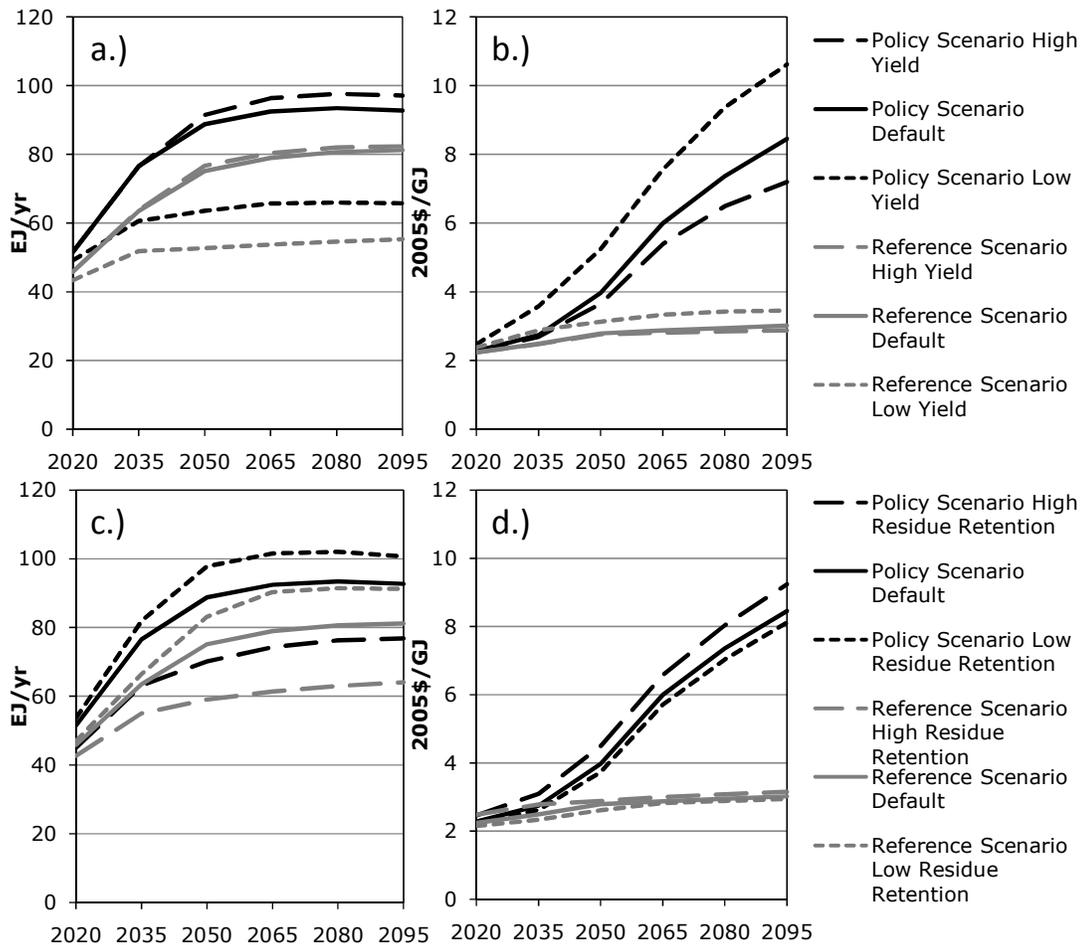
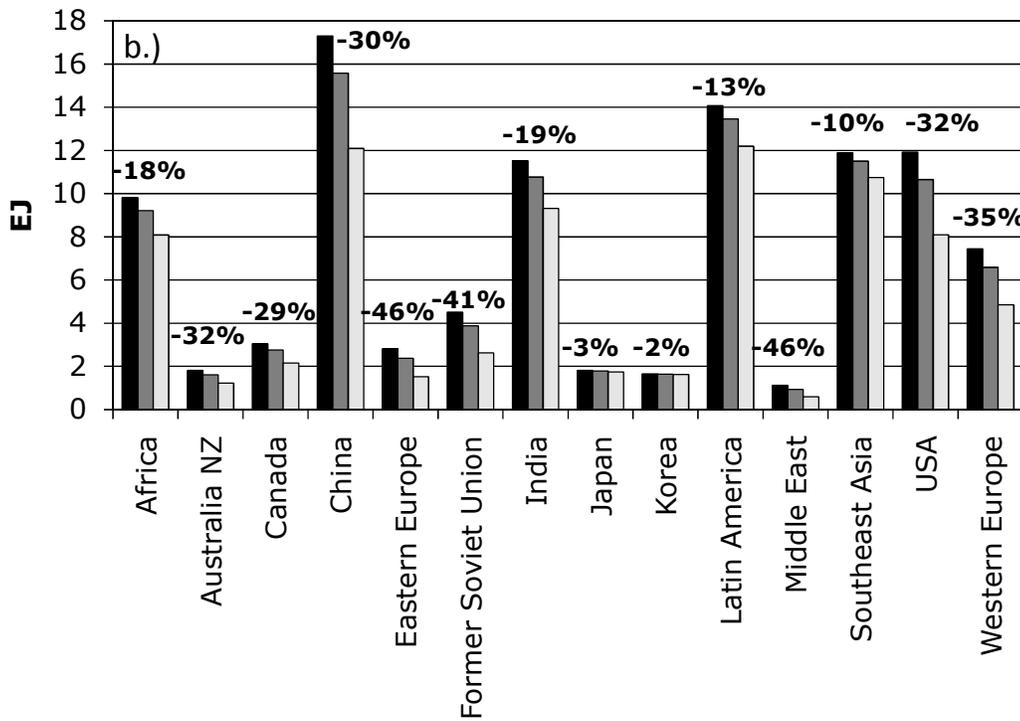
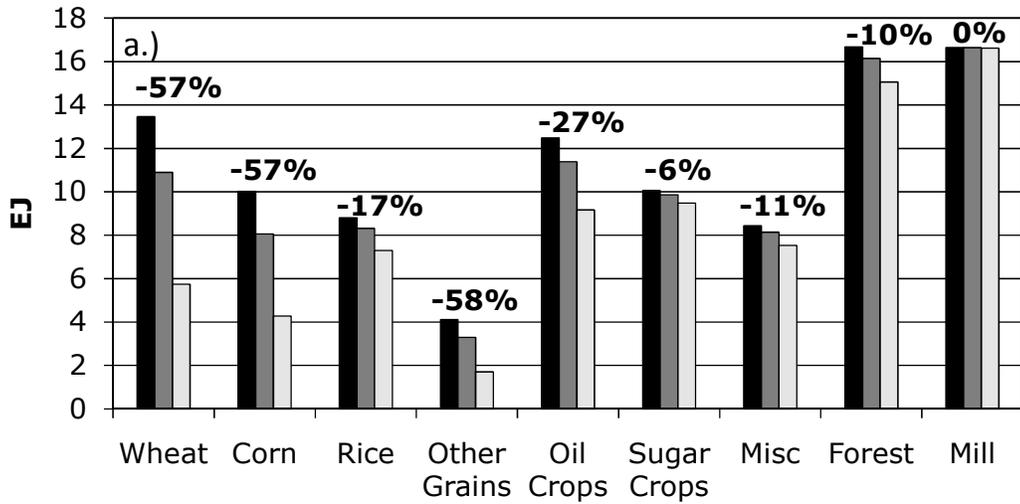


Figure 29. Sensitivity test of physical parameters. a.) Projected residue biomass energy in scenarios where future agricultural productivity is varied from high yield, default yield (continued historical yield increases) and low yield (no increases from current observed yields). b.) Projected biomass price for agricultural productivity scenarios. c.) Projected residue biomass energy in scenarios where the *residue*

retention values are varied from 150% of default values to 50% of default values. d.)
Projected biomass price for *residue retention* scenarios.

Because the various streams of residue are substitute goods, the market demands more residue from sources that do not have strict residue retention requirements, such as rice stalks, forest slash and mill residue. On the other hand, high residue retention requirements significantly reduce the amount of residue available from dry field crops such as corn, wheat, and other grains (Figure 30a). As a result, high residue retention requirements do not affect regions such as Southeast Asia, Korea and Japan (which have a larger proportion of residue in the form of rice, wood, and non-field oil crops) as much as the US, Australia, and Europe (which have a larger proportion of residue in the form of dry grains) (Figure 30b). In the model, residue retention only affects the maximum potential residue supply for each crop; no feedbacks are modeled between unsustainable residue retention, yield, collection price, and chemical inputs. If, for example, unsustainable residue removal were to reduce crop productivity, then the residue retention variable could have a much larger effect on global residue supply, as seen in Figure 29a and Figure 29b when assumptions about future crop yields were varied.



Residue Retention Parameter: ■ 0.5 x Default ■ Default □ 2 x Default

Figure 30. The effect of varying the *residue retention* parameter (from 50% the default value to 200% the default value) on consumption of residue biomass energy by a.) region, and b.) resource. Default residue retention values are crop specific and

given in Table 1. A climate policy scenario was used and end of century (2095) values were compared to closely approximate a situation where maximum potential residue biomass available is utilized. Percentages represent the difference between the high and low values.

In terms of the economic assumptions, the *Midprice*, which would represent the cost of producing and delivering this resource, is projected to either advance (in a scenario where the *Midprice* values are 50% of the default values) or delay (in a scenario where the *Midprice* values are 200% of the default values) the utilization of biomass residue (Figure 31a). This variable makes little difference in the projected total supply delivered by the end of the century, except in the reference scenario with high *MidPrice* (Figure 31a). In that scenario, the equilibrium price for residue biomass never gets above the *Midprice* (Figure 31b) and most of the biomass demand is met by dedicated biomass crops. Thus, the reference scenario is more sensitive to the *Midprice* variable than the policy scenario, because the under the policy scenario, where premiums on carbon-intensive fuels drive up the price for all fuels, the equilibrium price is high enough to accommodate a high value for the *Midprice*.

The EIA NEMS model assumes a steep cost curve (high elasticity of supply) (EIA, 2003; Haq and Easterly 2006), so once the price increases slightly beyond the *Midprice*, the supply of residue biomass is maximized, and as the price continues to rise beyond this point, dedicated biomass crops are necessary to meet demand. Reducing the curve exponent (*b*) to 25% and 50% the default values has a similar

effect to increasing the *Midprice*, in that it takes longer to maximize the utilization of residue biomass (Figure 31c). Altering the curve exponent has little effect on the price of biomass (Figure 31d), but it allows for a more gradual development of both residue biomass and dedicated biomass crops.

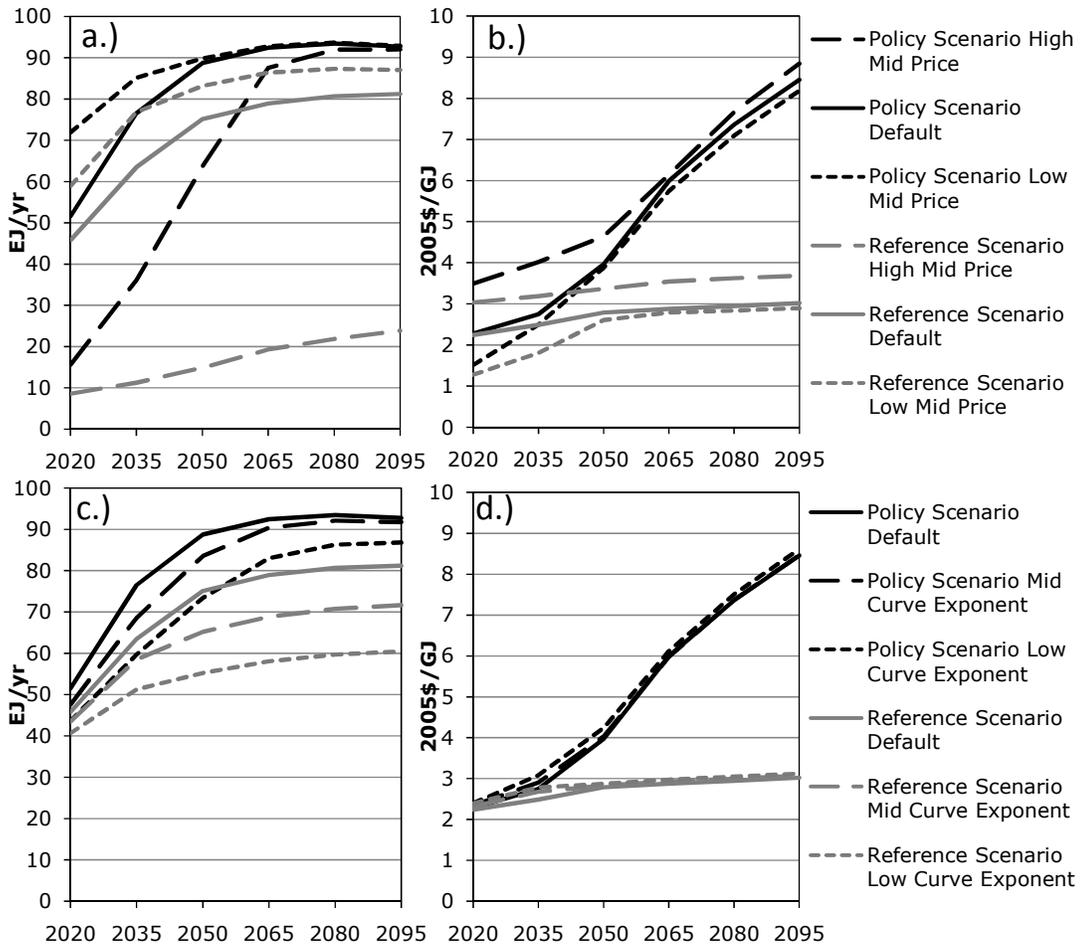


Figure 31. Sensitivity test of economic parameters. a.) Projected residue biomass energy in scenarios where *Midprice* values are varied from 50% of default values to 200% of default values. b.) Projected biomass price for *Midprice* scenarios. c.) Projected residue biomass energy in scenarios where the curve exponent, *b*, values are

varied from 150% default values to 50% default values. d.) Projected biomass price for curve exponent scenarios.

Conclusions

Our global analysis of crop and forestry statistics indicates that approximately 50 EJ yr⁻¹ of residue biomass is currently available on a sustainable basis. The principal source of uncertainty in this estimate is the amount of residue that needs to be left behind to reduce soil erosion. The potential supply of residue biomass increases over the 21st century to perhaps twice the current figure, as the scale of agricultural and forestry activities expand to meet the food and fiber needs of a more affluent and larger world population. In addition to residue retention constraints, the future potential supply of residue biomass depends on the degree to which agricultural yields increase. An increase in yields has a twofold impact on residue supply: increased product yields imply increased residue production, and increased yields mean that more residue per unit area can be sustainably removed without a large increase in erosion and impacts on crop productivity.

The amount of biomass that is actually used depends on the cost of collecting and processing residues, the cost of competing energy technologies, and any environmental incentives. We find that, residue biomass is projected to be increasingly used by mid- to late- century for bioenergy production. In the reference scenario, between 20-90 EJ yr⁻¹ of residue biomass is projected to be produced

globally. This wide variation, for a reference scenario without climate policy, is due to differences in both economic assumptions regarding the cost of residue collection, and physical assumptions regarding the total amount of sustainable residue available. In climate policy scenarios, where a premium is paid for carbon-free energy such as residue biomass, nearly all of the potential residue biomass resource is used for energy with projected use increasing to 65-100 EJ yr⁻¹ globally. Assumptions about collection costs have little impact later in the century on policy scenarios as costs for residue biomass become low relative to alternatives. The primary uncertainty in policy scenarios are the assumptions on residue retention requirements and future agricultural productivity.

The International Energy Agency (IEA) (2006a) estimates that 45 EJ of primary solid biomass was consumed globally in 2005, of which 70% is consumed in the residential sector, primarily in developing countries. It is not clear how much of this is sustainably produced biomass as defined here; it is likely that much, if not the majority, of the biomass used in developing countries is from unsustainable collection and deforestation. The projections presented here, therefore, imply a large increase in the fraction of sustainable residue biomass that is used for energy purposes. This near-term increase is a consequence of the steep shape of the default cost curve used here. In the near-term, the utilization of residue biomass is likely to be highly sensitive to the cost of collection and processing. The cost curves used in this study are specific to the US and further research in this area would be valuable to better elucidate the near-term trajectories for residue biomass consumption.

In the long-term, however, we find that collection and processing costs have little impact once the carbon price, the premium that is paid for carbon-free energy, increases under a climate policy. This is because the carbon price is controlled by the marginal cost of mitigation, that is the cost of the most expensive option that has been put into place. Residue biomass is a low-cost option that is utilized early once climate policy is put into place. At this point, the primary determinants of residue biomass supply are changes in agricultural productivity and constraints imposed due to residue retention for erosion control. For this study a generic crop-specific formulation of residue retention for erosion control in terms of Mg ha^{-1} of residue retained in the field was applied. This may prove to be overly simplistic, and a further elaboration of the tradeoffs in terms of soil nutrients, crop yields, and erosion that come with the removal of crop residues is needed.

Given the potential for bioenergy from residue biomass, further research is needed as to how to most sustainably harvest this resource, and how to most efficiently convert it to energy. Assessing the potential for energy from residue biomass given the economic conditions and the climate policy landscape is essential to forming prudent decisions about our future energy portfolio. Our finding that residue biomass is likely to be heavily utilized under a climate policy implies that policies need to be in place to ensure that residue removal is conducted in a sustainable manner.

Acknowledgements

A special note of gratitude goes to the O^{bj}ECTS MiniCAM development team for their support and troubleshooting efforts. Also, special thanks to R. César Izaurralde for his helpful comments. This work was funded in part with support from the US Department of Energy's Great Lakes Bioenergy Research Center, the Department of Energy's Office of Science, the Electric Power Research Institute, Chevron, and ExxonMobil. Research was conducted at the Joint Global Change Research Institute (JGCRI), a collaboration between the Pacific Northwest National Laboratory and the University of Maryland. The Pacific Northwest National Laboratory is managed by Battelle for the US Department of Energy.

Chapter 6: Effect of crop residue harvest on long-term crop yield, soil erosion, and nutrient balance: trade-offs for a sustainable bioenergy feedstock

Preface

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In Review:

Biofuels Journal

Abstract

Agricultural residues could potentially be harvested to serve as a cellulosic feedstock for bioenergy production. The relationship between crop residue harvest, soil erosion, crop yield, carbon and nitrogen balance was modeled with the Erosion Productivity Impact Calculator/ Environment Policy Integrated Climate (EPIC) using a factorial design. Four crop rotations (winter wheat [*Triticum aestivum* (L.)] – sunflower [*Helianthus annuus*]; spring wheat [*Triticum aestivum* (L.)] – canola [*Brassica napus*]; corn [*Zea mays* L.] – soybean [*Glycine max* (L.) Merr.]; and cotton [*Gossypium hirsutum*] – peanut [*Arachis hypogaea*]) were simulated at four US locations each, under different topographies (0-10% slope), and management

practices [crop residue removal rates (0-75%), conservation practices (no till, contour cropping, strip cropping, terracing)]. In general, harvesting residues is a question of tradeoffs: residue removal increases soil loss at rates that vary with topography and management, decreases yields (100-year mean yields changed -0.07% to -0.08% for every percent of residue mass removed), decreases soil carbon (roughly 40 - 90 kg C ha⁻¹ yr⁻¹ per Mg of residue harvested), and decreases soil nitrogen (roughly 3 kg N ha⁻¹ yr⁻¹ per Mg residue harvested). However, variations in local topography, soil type, and climate have a greater influence on soil erosion than residue harvest rate. In terms of remaining within tolerable soil loss, the sustainable residue harvest rate may vary from 0-75% (the latter being the assumed maximum practical harvest rate). Conservation management is more important when residue removed from corn-soybean rotations than winter wheat-sunflower and spring wheat-canola rotations. Without implementation of conservation practices, residue removal from corn-soybean is practical only on slopes less than 1%. Residue harvest from cotton-peanut rotations is likely impractical if the slope is greater than or equal to 10%, even with conservation practices. Through soil loss prevention measures, adoption of conservation practices can allow for a greater amount of residue to be removed off more erosion-prone land (e.g., land with steeper slopes or highly erodible soil). On average, removal of 75% of crop residue is projected to reduce long-term crop yields by 6%, under both conservation and conventional management practices. The effects of increased residue harvest on crop yield reduction is highly variable depending on local climate and soil erodibility.

Introduction

As the role of renewable fuels increases in the US energy portfolio, bioenergy has garnered much attention as a low-carbon alternative to petroleum, particularly in the transportation sector. In recent years, there has been a dramatic expansion of biofuel production: in the US, ethanol production increased by a over a factor of six in the last decade (1998-2008) to 9 billion gallons per year (34 GL yr⁻¹) (RFA 2007). The Energy Independence and Security Act (EISA), passed by the US Congress in 2007, mandates a further expansion to 36 billion gallons of biofuels per year (136 GL yr⁻¹) by 2022. Of this, 21 billion gallons per year (79 GL yr⁻¹) would be from so-called advanced biofuels—ethanol derived from sources other than corn starch, such as cellulosic ethanol. In addition to ethanol, biomass could also be combusted or co-fired in power plants to produce electricity, displacing coal, or serve as chemical feedstocks to replace natural gas. Alternatively, sequestered residue, e.g., in deep water, could present an option for reducing atmospheric concentrations of carbon dioxide (CO₂).

Meeting the growing demand and legislated targets for increased production of bioenergy will require a dramatic increase in biomass feedstock supply, and some have expressed concern over potential negative impacts this would have. For example, expansion of agriculture into natural areas could result in a loss of

biodiversity (Raghu et al. 2006; Righelato and Spracklen 2007), and may incur an insurmountable carbon debt from emissions from land conversion (Fargione et al. 2008). Intensive farming of land under the Conservation Reserve Program (CRP) or marginal lands could also lead to deer (Bies 2006) and bird (McLachlan, Carter, and Rustay 2007) habitat destruction, increase water consumption (Berndes 2002), exacerbate soil loss (Kort, Collins, and Ditsch 1998), and increase nutrient run-off and eutrophication of riparian and aquatic systems (Hill et al. 2006). Furthermore, economic analyses have suggested that competition for crops could increase food prices (Johansson and Azar 2007; Ranses, Hanson, and Shapouri 1998).

Some of these drawbacks can be mitigated by taking advantage of agricultural residues. Agricultural residues are already produced as a co-product of food and fiber production, and thus have a large potential as a bioenergy feedstock, requiring no new land or technology to produce. Because of this, they are likely to be a much cheaper and a more immediately available source for biomass feedstocks than crops such as switchgrass (*Panicum virgatum*) or hybrid poplar (*Populus* spp.).

However, it is unclear what effect different rates of residue biomass removal will have on soil erosion and crop yields. If large scale exploitation of crop residues results in increased erosion and lower yields, then less residue would be available per unit area of land as time progresses, and aggregation and transportation costs would increase. Additionally, more land would need to be brought into production to feed a growing global population. The question of a sustainable residue harvest rate is a question concerning the conditions under which residue harvest is practical and

environmentally sensible. It is also a question of tradeoffs in terms of yield, erosion, and nutrient balance.

Existent studies on sustainable residue harvest rates typically focus on a specific crop, and often a single field site. Much of the current research on crop residue has focused on corn residue (stover) because of the large amount of biomass the crop produces and because annually, the US produces more corn by mass than all other field crops combined (FAOSTAT 2008b). Based on a series of field studies and limitations of current equipment available, the maximum logistical harvest rate for corn stover is approximately 75% by mass, though the sustainable harvest rate (in terms of erosion control and soil nutrients) is understood to be lower (Graham et al. 2007). A review by Mann, Tolbert, and Cushman (2002) stresses that research is needed to understand the long-term relationship between corn residue removal, erosion, water quality, nutrient dynamics, crop productivity, and management strategies. Hoskinson et al. (2007) examined the economics of replenishing soil nutrients from different levels of stover removal, and recommend a 40-cm cutting height, optimizing removal of the typically drier upper part of the corn stalk and leaving the wet portion to replenish soil nutrients. Graham et al. (2007) concluded that only 28% of stover could be removed under current production practices if soil erosion were to remain below 0.5 Mg ha^{-1} . Graham et al. (2007) also concluded that with improved conservation management practices, such as wide adoption of no-till practices, residue removal rates could approach 50% and up to 100 Tg of stover (dry basis) per year could be produced in the US. Adoption of no-till

practices tend to reduce erosion, retain soil nutrients, and reduce carbon loss over conventional tillage in the upper layers of the soil (Izaurrealde et al. 2007). More stringent erosion control requirements, however, significantly reduce the estimated amount of corn stover available (Graham et al. 2007). For example, even with no-till practices, removal of corn stover has been shown to increase soil bulk density and reduce soil water content in a 1-year field experiment (Blanco-Canqui et al. 2006; Blanco-Canqui et al. 2007). Blanco-Canqui et al. (2007) offer a limit of corn stover harvest at 1.25 Mg ha⁻¹ (roughly a 25% removal rate) for sustaining soil quality, but point out that more research and monitoring is needed to better establish this threshold.

In a US Department of Agriculture (USDA) white paper, Andrews (2006) reviews predicted impacts of residue removal on erosion, soil organic matter and nutrients, and future crop yields; a maximum 30% residue removal rate is given as a general recommendation with the caveat that this number can only serve as a rough guide and site specific research and guidelines need to be developed (Andrews 2006). This value, 30% residue removal, is commonly used in larger modeling studies, such as in studies that consider the potential for ocean sequestration of carbon by sinking crop residues in the deep ocean (Strand and Benford 2009).

However, these optimal residue harvest rates can only be understood as average values to be applied on a large geographic scale. In a literature review on corn stover, Wilhelm et al. (2004) recognized that removal rates would vary depending on local crop yield, climatic conditions, and management practices, and

stressed the need for the development of a procedural tool for recommending a maximum possible amount of corn stover removal to sustain crop productivity. Based on the a modeling study of 10 corn producing counties, Wilhelm et al. (2007) suggested that, on average, approximately 30% of residue could be harvested above a base corn yield of 7-17 Mg ha⁻¹, depending on the tillage system used, but also noted a high degree of local variability. As cellulosic conversion technology progresses, Wilhelm et al. (2007) stressed the need for further study and validation of sustainable residue harvest for multiple locations and cropping systems.

Focusing on a specific crop such as corn [*Zea Mays* L.] gives information about the logistics and dynamics of residue removal at the field level, but results are not necessarily applicable from one crop to another or to long time scales. While corn is the dominant crop in the US, corn is frequently grown in rotation with other crops, and moreover, many local areas specialize in other crops that can also provide a potential source for residue biomass. Residue biomass is substitutable between different feedstocks, for example, stringent erosion control on corn can raise the demand for residue from crops with less stringent erosion control requirements, such as wheat (*Triticum aestivum*). Therefore, an economic and environmental assessment must consider a variety of cropping systems over an extended period of time. The purpose of this study is to apply a biophysical simulation framework to examine multiple crop rotations at multiple locations over a 100-year time frame in order to improve our understanding of the relationships among residue harvest, crop yields, soil loss, carbon and nitrogen balance, and management strategies of sustainable

biomass feedstock systems from agricultural residues. The goal is to determine the sensitivity of crop yield, erosion, and nutrient balance to residue removal through simulating hypothetical agricultural fields.

Materials and Methods

We designed a factorial modeling study to determine the effect of different levels of residue harvest on soil erosion, crop yields, and carbon and nitrogen balance. The Erosion Productivity Impact Calculator/ interactive Environment Policy Integrated Climate (EPIC) model (Williams 1995) was selected to simulate all these interacting effects due to its strength in management details (e.g., crop rotations, tillage, conservation management), erosion processes (i.e., water and wind erosion) and ecosystem nutrient balance (Izaurrealde et al. 2006).

Study Design

The simulations were designed to determine the cross effects of cropping system, location (soil and climate), topography (slope) and to predict under what conditions and management strategies residue harvest would be sustainable. In addition, this allows the prediction of trade-offs in terms of carbon and nitrogen loss. Four crop rotations were considered: winter wheat [*Triticum aestivum* (L.)] – sunflower [*Helianthus annuus*]; spring wheat [*Triticum aestivum* (L.)] – canola [*Brassica napus*]; corn [*Zea mays* L.] – soybean [*Glycine max* (L.) Merr.]; and cotton

[*Gossypium hirsutum*] – peanut [*Arachis hypogaea*]. For each cropping system, four locations were selected (Table 12). The first location selected for each cropping system was the highest producing county (by mass of the lead crop) in the highest producing state. The second selection for each crop rotation was the highest producing county in the second highest producing state. Two additional counties were selected randomly for each crop rotation by assigning probability weights based on the current annual production of the lead crop. This hybrid systematic-random selection process was designed to both ensure spatial variability and also choose characteristic regions where it would be most economic to produce, deliver, and process residue biomass (Figure 32). Soil type, soil layer data, and historical weather data were taken from the National Nutrient Loss Database (Potter et al. 2004). For each location, the dominant eight-digit watershed (by area) was selected, and within each watershed, the dominant soil type (by percentage) (Figure 33) was selected to represent a sample 1-hectare plot (Table 13). This created a total of 16 sample locations (Table 12), dispersed across the US. To account for differences in topography, four slopes (0.1%, 1%, 5%, and 10%) were tested at each location where slope length was assumed to be 100 m.

Table 12. List of location samples by crop rotation, including dominant watershed and soil type.

Crop Rotation	Sample	County	State	Major HUC	Dominant Soil
Winter Wheat & Sunflower	#1 County in #1 State	Sumner	KS	11060005	Detroit
Winter Wheat & Sunflower	#1 County in #2 State	Whitman	WA	17060108	Palouse
Winter Wheat & Sunflower	Weighted Random Sample #1	Cassia	ID	17040210	Declo
Winter Wheat & Sunflower	Weighted Random Sample #2	Grant	OK	11060004	Dale
Spring Wheat & Canola	#1 County in #1 State	Cavalier	ND	09020313	Barnes
Spring Wheat & Canola	#1 County in #2 State	Polk	MN	09020303	Minnetonka
Spring Wheat & Canola	Weighted Random Sample #1	Swift	MN	07020005	Buse
Spring Wheat & Canola	Weighted Random Sample #2	Grand Forks	ND	09020307	Arvilla
Corn & Soybean	#1 County in #1 State	Kossuth	IA	07100003	Kenyon
Corn & Soybean	#1 County in #2 State	McLean	IL	07130009	Drummer
Corn & Soybean	Weighted Random Sample #1	Dawson	NE	10200101	Blendon
Corn & Soybean	Weighted Random Sample #2	Audubon	IA	10240003	Tama
Cotton & Peanut	#1 County in #1 State	Hale	TX	12050006	Acuff
Cotton & Peanut	#1 County in #2 State	Mississippi	AR	08020203	Askew
Cotton & Peanut	Weighted Random Sample #1	Bertie	NC	03010107	Craven
Cotton & Peanut	Weighted Random Sample #2	Darlington	SC	03040201	Eunola

Table 13. Soil data used for simulated trials. AWC= Available Water Content (Field Capacity – Wilting Point), SOC = Soil Organic Carbon. Profile values represent weighted means based on soil horizon mass.

Crop Rotation	Soil	Horizons	Profile Depth (m)	Bulk Density 1st Horizon (Mg/m ³)	AWC 1st Horizon (m/m)	AWC Total Profile (m/m)	Sand Content 1st Horizon (%)	Silt Content 1st Horizon (%)	pH 1st Horizon	SOC 1st Horizon (%)	SOC Total Profile (%)
Winter Wheat-Sunflower	Detroit	4	1.59	1.29	0.18	0.16	24.1	51.4	6.7	2.09	0.49
Winter Wheat-Sunflower	Palouse	3	1.44	1.22	0.20	0.20	11.3	67.7	7.0	1.52	0.88
Winter Wheat-Sunflower	Delco	4	1.52	1.37	0.14	0.12	43.0	39.5	7.9	1.08	0.25
Winter Wheat-Sunflower	Dale	3	1.61	1.30	0.20	0.19	11.4	68.1	7.0	2.02	0.94
Spring Wheat-Canola	Barnes	5	1.51	1.34	0.16	0.12	43.0	39.5	6.7	3.38	0.54
Spring Wheat-Canola	Minnetonka	4	1.50	1.27	0.20	0.17	20.0	49.0	6.5	3.81	1.33
Spring Wheat-Canola	Buse	3	1.51	1.44	0.13	0.11	39.8	37.7	7.5	1.27	0.44
Spring Wheat-Canola	Arvilla	4	1.51	1.49	0.09	0.06	68.2	19.8	7.3	1.54	0.40
Corn-Soy	Kenyon	4	1.50	1.43	0.15	0.11	41.1	36.9	6.5	1.96	0.64
Corn-Soy	Drummer	4	1.50	1.27	0.21	0.18	9.4	67.1	6.7	2.80	0.66
Corn-Soy	Blendon	4	1.47	1.42	0.10	0.09	67.6	20.4	6.7	1.38	0.50
Corn-Soy	Tama	5	1.47	1.30	0.20	0.16	9.2	65.3	6.2	1.76	0.98
Cotton-Peanut	Acuff	4	1.97	1.44	0.09	0.09	52.9	18.7	7.2	0.74	0.30
Cotton-Peanut	Askew	4	1.83	1.43	0.19	0.16	8.7	66.2	6.2	1.12	0.39
Cotton-Peanut	Craven	4	2.01	1.40	0.15	0.14	29.3	53.7	5.1	0.61	0.18
Cotton-Peanut	Eunola	6	1.63	1.52	0.09	0.09	65.9	19.1	5.0	0.62	0.25

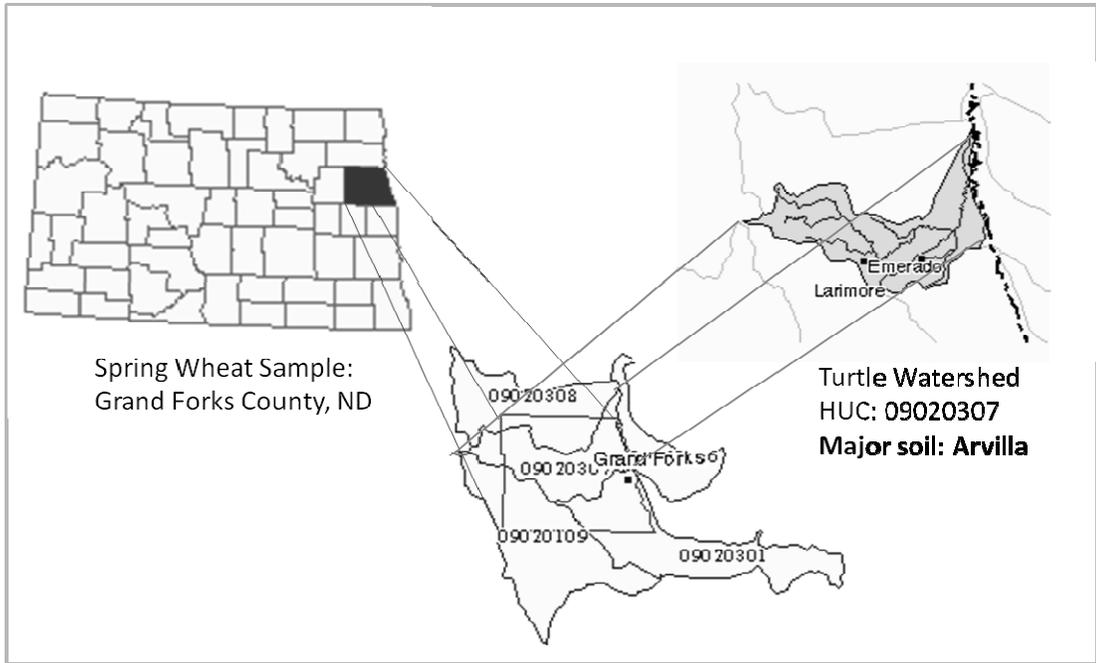


Figure 32. Method for selecting watersheds and soil type from county samples.

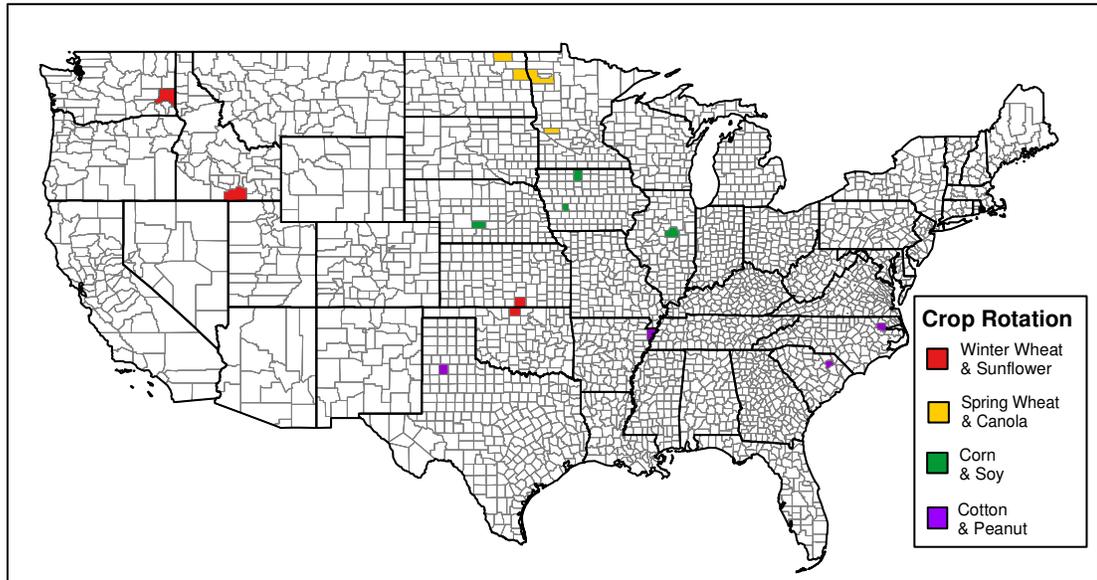


Figure 33. Map of sampled counties by crop rotation.

Crop systems were simulated for 100 years under two contrasting management strategies: the conventional management strategy, using conventional tillage and no conservation measures; and the conservation management strategy, which employed no-till management as well as strip cropping, contouring cropping, and terracing. The conventional management system utilized a tandem disk set to a tillage depth of 75 cm, a field cultivator set to a tillage depth of 50 cm, and a planter set to a depth of 40 cm.

The conservation management system used a no-till system, which retains organic matter and soil cover, as well as below ground biomass, thereby reducing the amount of soil exposure and erosion. Contouring (planting in line with topographic contours) and strip cropping (planting crops in alternating swaths) reduce runoff by creating landscape breaks and slowing water flow. Terracing (building steps into a graded hillside) has the effect of reducing the slope length by the following relationship:

$$L_{terracing} = \frac{0.3 \cdot (X \cdot S + Y)}{\sin(\arctan(S))}$$

where $L_{terracing}$ is the slope length interval between terraces, S is the slope in percent, X is a location-specific constant that varies across the U.S. from 0.4 in the south to 0.8 in the north, and Y is a soil erodibility constant, set to 2.5.

For each location, slope, and management combination, six levels of residue harvested were modeled: 0%, 15%, 30%, 45%, 60%, and 75%, the later representing the theoretical maximum logistical harvest rate. Residue harvest was set up to occur annually, immediately after crop harvest, for all crops within a given rotation. Fertilizer and irrigation were automatically applied based on plant nitrogen and water stress.

Characteristics of the EPIC Model

The EPIC model simulates weather, hydrology, erosion, nutrients, soil temperature, plant growth, plant environment control, tillage, and economic budgets on a field with homogenous soil, weather, and management (Williams 1995). The model was developed in the early 1980s to estimate erosion and crop productivity. In 1985, it was used to estimate erosion for various land areas in the US as part of the 2nd Resources and Conservation Act (RCA). Since then, the EPIC model has been expanded to include aspects such as fertilizer application, crop rotation, and tillage systems. Components of the model have been refined and validated with numerous empirical studies (Gassman et al. 2005), for example, nutrient cycling (Cepuder and Shukla 2002; Chung et al. 2001), water erosion (Bhuyan et al. 2002; Purveen et al. 1997), wind erosion (Izaurrealde et al. 1997; Potter et al. 1998), soil carbon sequestration (Lee, Phillips, and Liu 1993; Roloff, Jong, and Nolin 1998), and crop productivity (Jain and Dolezal 2000; Schaub, Meier-Zielinski, and Goetz 1998).

Currently, EPIC is one of the only models able to simulate both water and wind erosion simultaneously on the same field. Water erosion is simulated in EPIC using the MUSLE (Modified Universal Soil Loss Equation) (Williams 1975):

$$\text{Soil Loss} = a(V \cdot Q_p)^b \cdot K \cdot L \cdot S \cdot C \cdot P$$

where a and b are constants, V is the volume of runoff, Q_p is the peak runoff rate, K is soil erodibility factor, L and S define the slope length and gradient, C is the crop management factor and P is the conservation management factor. Simulation of conservation practices reduce soil loss in EPIC by changing the statistically derived run-off curve regression parameters (frequency and depth of tillage). EPIC also alters the runoff curve parameters based on the number of conservation measures in place, with different values between 0, 1, and 2 or more simultaneous conservation measures. EPIC does not distinguish between the specific conservation measures (strip cropping, contouring, terracing), with the exception that terracing also reduces the slope length, thereby further reducing soil loss.

Wind erosion is calculated with the Wind Erosion Stochastic Simulator (WESS) (Potter et al. 1998). The EPIC model generates weather data stochastically with a fixed random number seed based on historical records. Operations were scheduled using climate data specific to each location and each modeled year.

Simulation Runs and Analysis

The model runs were implemented and executed with i_EPIC, an interactive Windows® based program developed at the Center for Agricultural and Rural Development at Iowa State University to facilitate the management and execution of large simulations with the EPIC model (Gassman et al. 2003). Though EPIC makes calculations at a daily time step, output data were aggregated both at an annual time step for all runs, and at the total simulation length of 100 years. Analysis of the output data (regressions, box plots, etc.) was done with the R environment (GNU S).

Results and Discussion

The relationship between residue harvest rate and residue collected is summarized in Table 14. On average, as the residue harvest rate increases by 15%, residue harvested increases by about 0.9 Mg ha⁻¹ yr⁻¹.

Table 14. Relationship between residue removal rate and mean residue harvested (all locations, all years).

Residue harvest rate (%)	15	30	45	60	75
Mean residue harvested (t/ha/yr)	1.1	2.0	2.9	3.7	4.6
2σ Residue harvested (t/ha/yr)	0.5	0.7	1.1	1.6	2.0

In general, crop erosion increases with increased residue harvest, particularly in the simulations using conventional management. Slope was also a confounding factor; row crops such as corn under conventional tillage were generally more susceptible to erosion with increased slope. However, implementation of conservation management practices allows for more residue to be sustainably removed at greater slope.

In terms of mitigating soil loss, the results suggest that most of the available residue can be sustainably harvested from the winter wheat-sunflower rotations, though with slopes at 10% greater under conventional management, the limit is around 45%, depending on local values for tolerable soil loss (Figure 34). Most residue can also be sustainably harvested from spring wheat-canola rotations (Figure 35) and remain under tolerable soil loss for most locations. The spring wheat-canola fields located in the upper Midwest were highly susceptible to extreme weather events, particularly wind erosion. Reducing residue harvest had little impact in preventing erosion from extreme weather events on these fields, rather, implementation of conservation management reduced soil loss from these events at all levels of residue harvest. For the corn-soybean rotation, conservation management is critical if residue is harvested. Under conventional management, residue harvest is practical only on fields with slopes of less than 1% (Figure 36). Conservation management, on the other hand, allows for most of the residue to be removed up to slopes of 10%, though erosion does increase with increasing residue removal (Figure 36). Conservation management had less effect on cotton-peanut rotations, where 45-

60% residue harvest is sustainable only on slopes of 1% or less under conventional management and perhaps 5% or less under conservation management (Figure 37). In this study, cotton-peanut fields with slopes of 10% could only have 15% sustainable residue harvest under conservation management, and even then, erosion would be within the range of tolerable soil loss. This likely due to the soil disturbance involved with harvesting ground nuts. Thus, residue harvest would likely be impractical from cotton-peanut rotations on steep slopes. Thresholds for sustainable residue removal rates, based on the 75% percentile of annual soil loss in Mg ha^{-1} , are given in Table 15. Conservation management is able to allow a greater amount of residue to be harvested on land with higher slope, particularly for corn-soy rotations. It is less effective at mitigating erosion on cotton-peanut rotations. Moreover, while corn stover has been the focus for much current research in crop residue, Table 15 suggests that other crop residues, such as wheat straw, may be harvested at greater rates more sustainably. The 30% removal rate suggested by the USDA (2006) and frequently employed as a parameter in large scale national studies [see, for example, Strand, et al. (2009)] is a rather conservative estimate in terms of soil loss. Additionally, given the variability with location, topography, and management, it would be problematic to apply any single rate broadly across the entire country.

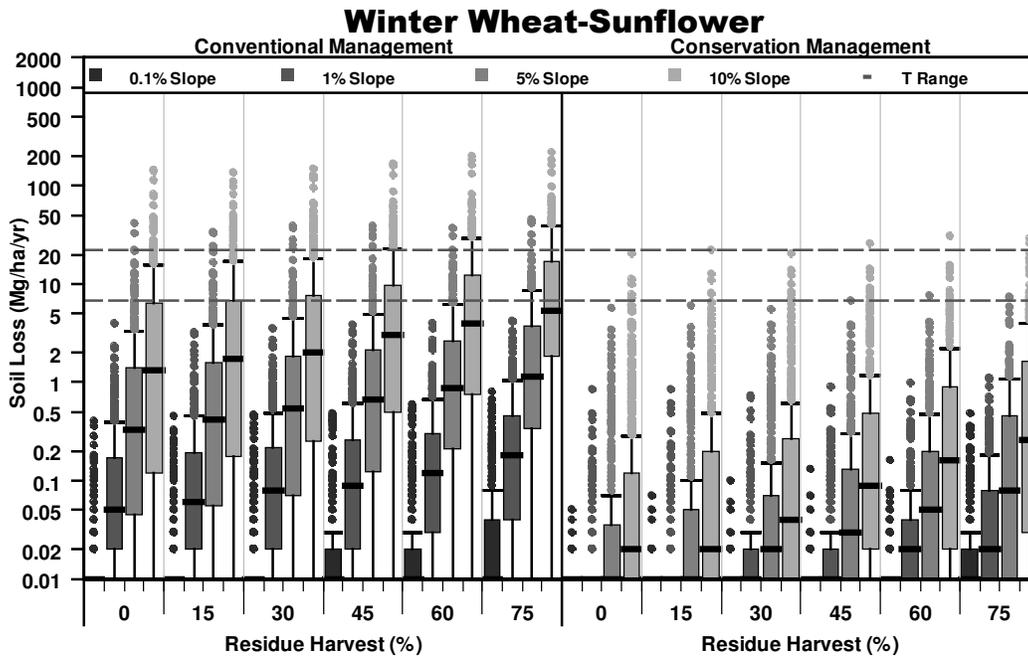


Figure 34. Winter Wheat – Sunflower rotation. Total annual soil loss (sum of water and wind erosion) versus slope and residue harvest rate under both conservation and conventional management. Each point represents total soil loss for a given year at a given location, there are 100 years at 4 locations for a total of 400 points at each level. The bars represent the median values; the box encloses the 25th and 75th percentiles (1st and 3rd quartiles). Whiskers extend to 1.5 times the inter-quartile range for each level. The "T" value represents a range of typical tolerable soil losses, from 3 – 10 tons acre⁻¹ yr⁻¹ (6.7 – 24.5 Mg ha⁻¹ yr⁻¹). Note the logarithmic scale on the y-axis.

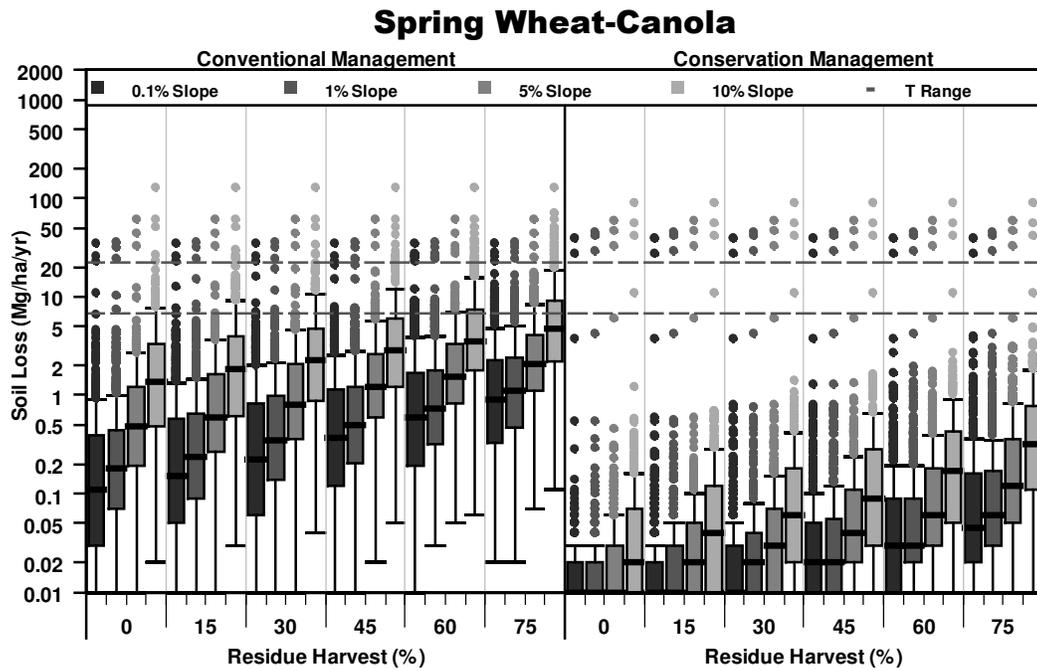


Figure 35. Spring Wheat – Canola rotation. Total annual soil loss (sum of water and wind erosion) versus slope and residue harvest rate under both conservation and conventional management. Each point represents total soil loss for a given year at a given location, there are 100 years at 4 locations for a total of 400 points at each level. The bars represent the median values; the box encloses the 25th and 75th percentiles (1st and 3rd quartiles). Whiskers extend to 1.5 times the inter-quartile range for each level. The "T" value represents a range of typical tolerable soil losses, from 3 – 10 tons acre⁻¹ yr⁻¹ (6.7 – 24.5 Mg ha⁻¹ yr⁻¹). Note the logarithmic scale on the y-axis.

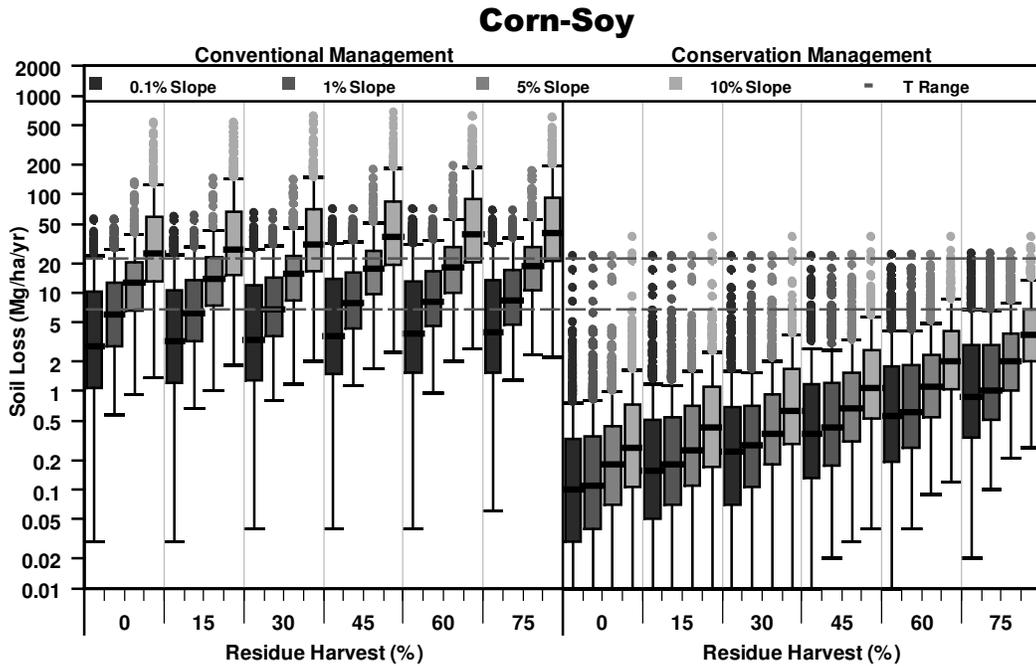


Figure 36. Corn – Soybean rotation. Total annual soil loss (sum of water and wind erosion) versus slope and residue harvest rate under both conservation and conventional management. Each point represents total soil loss for a given year at a given location, there are 100 years at 4 locations for a total of 400 points at each level. The bars represent the median values; the box encloses the 25th and 75th percentiles (1st and 3rd quartiles). Whiskers extend to 1.5 times the inter-quartile range for each level. The "T" value represents a range of typical tolerable soil losses, from 3 – 10 tons acre⁻¹ yr⁻¹ (6.7 – 24.5 Mg ha⁻¹ yr⁻¹). Note the logarithmic scale on the y-axis.

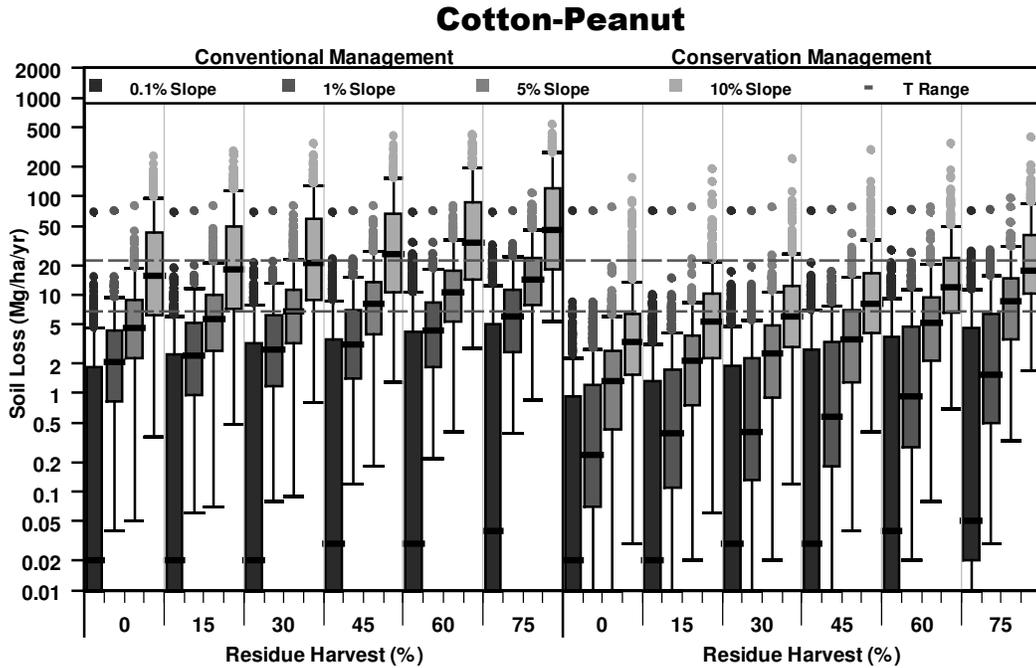
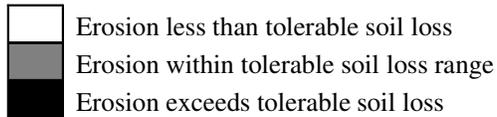


Figure 37. Cotton – Peanut rotation. Total annual soil loss (sum of water and wind erosion) versus slope and residue harvest rate under both conservation and conventional management. Each point represents total soil loss for a given year at a given location, there are 100 years at 4 locations for a total of 400 points at each level. The bars represent the median values; the box encloses the 25th and 75th percentiles (1st and 3rd quartiles). Whiskers extend to 1.5 times the inter-quartile range for each level. The "T" value represents a range of typical tolerable soil losses, from 3 – 10 tons acre⁻¹ yr⁻¹ (6.7 – 24.5 Mg ha⁻¹ yr⁻¹). Note the logarithmic scale on the y-axis.

Table 15. Residue harvest thresholds with respect to tolerable soil loss. Erosion values represent the 75% percentile of annual soil loss in Mg ha⁻¹.

Management System		Conventional					Conservation						
Residue Harvest Rate (%)		0	15	30	45	60	75	0	15	30	45	60	75
Crop Rotation	Slope (%)												
Winter Wheat-Sunflower	0.1												
Spring Wheat-Canola	0.1												
Corn-Soy	0.1												
Cotton-Peanut	0.1												
Winter Wheat-Sunflower	1												
Spring Wheat-Canola	1												
Corn-Soy	1												
Cotton-Peanut	1												
Winter Wheat-Sunflower	5												
Spring Wheat-Canola	5												
Corn-Soy	5												
Cotton-Peanut	5												
Winter Wheat-Sunflower	10												
Spring Wheat-Canola	10												
Corn-Soy	10												
Cotton-Peanut	10												



Though in many fields erosion increased dramatically with increased residue removal, residue removal had only a modest effect on crop yields (Figure 38). In some locations with steep slopes, too much residue removal caused a collapse of the system through excessive soil loss. In particular, the Declo soil in Cassia, ID, consistently saw large declines in yield with increasing residue removal, up to a 22%

reduction at 75% residue harvest rate. Of the four winter wheat-sunflower locations, the Cassia, ID location was under the most water stress and required more irrigation than any other location with this crop rotation. Increased residue removal exacerbated water stress in the Cassia, ID location, reducing crop yield. Grand Forks, ND also experienced large percentage reduction in yields, but less consistently. Overall crop yields in Grand Forks, ND were lower than in other locations for spring wheat-canola, due to the high susceptibility to wind and water erosion.

Residue harvest did not reduce long-term yields as much in other locations; overall, for every additional percent of residue removed, the 100-year mean yield drop for all crops was about 0.07-0.08% (Figure 38). Even at a 75% residue harvest rate, the 100-year average annual yield dropped by approximately 6%, though for most locations the long-term yield reduction was less than 5% at this level of residue harvest. Interestingly, no significant difference was found between the yield drop in conservation versus conventional management strategies. The relationship between mean crop yields and residue removal is heteroscedastic (the variance in yields increases with increasing residue removal). The less residue left on the field, the more susceptible the field is to extreme weather events, increased erosion, and reduced crop yield; but the relationship is probabilistic depending on the local climate.

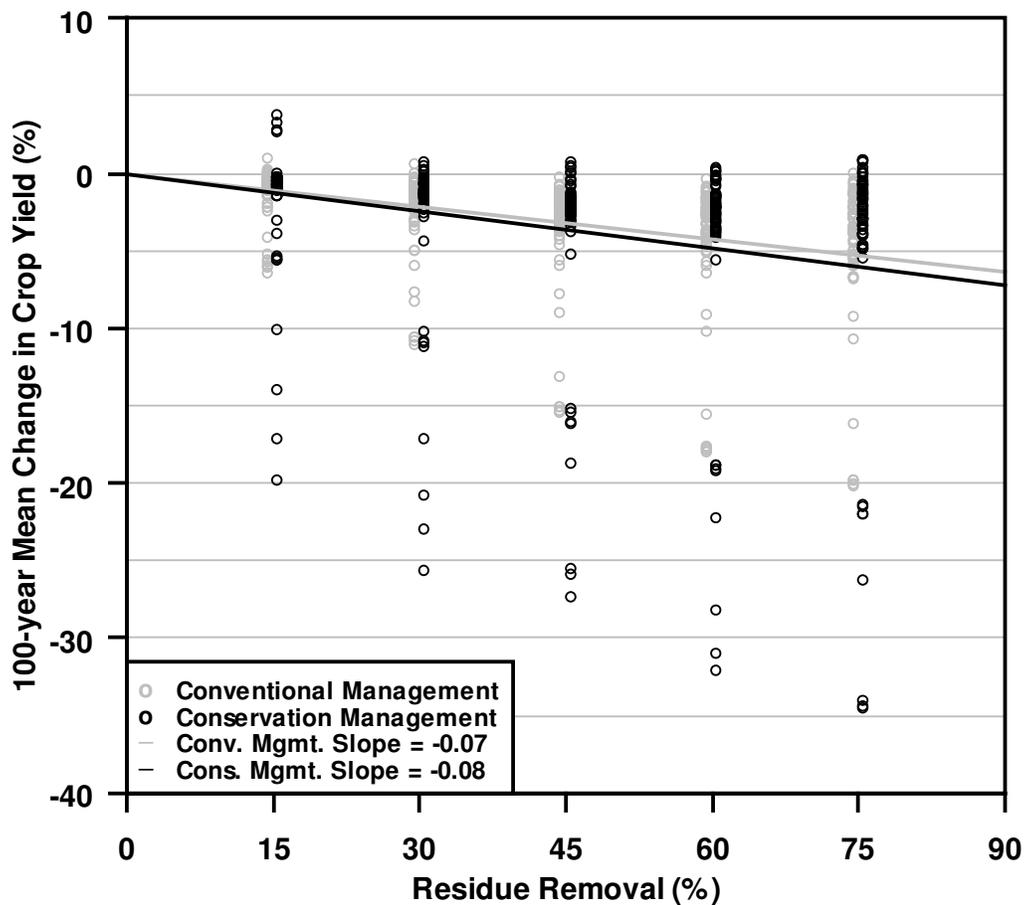


Figure 38. Mean percentage change in yield versus residue harvest rate by management over the base rate. The base rates are the corresponding trials (location, slope, and management) with no residue removal.

In most systems, the total carbon pool of the entire soil profile was reduced by increasing the residue harvest rate. However, since only a small part of the carbon in

the residue left on the field is converted to soil carbon, these reductions were small, especially under conservation (no-till) practices. Mean reductions in total annual carbon per tonne of residue harvested were $90 \text{ kg ha}^{-1} \text{ yr}^{-1}$ under conventional practices and $40 \text{ kg ha}^{-1} \text{ yr}^{-1}$ under conservation practices (Figure 39). Even at 75% residue removal under conventional management, where $2 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ would be removed from the field, the total system would lose on average less than $0.5 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. This suggests corroborates the finding that most of the carbon in agricultural residue decays into the atmosphere; only a small amount accrues in the soil [see, for example, Huggins et al. (2007)]. In general, all soil carbon pools are reduced with increasing residue harvest, though most of this loss occurs in the humus pools (Figure 40). Under conventional tillage, any amount of residue harvest, on average, results in a decrease in the total soil pool, but under conservation tillage, residue removal rates 30% and under, on average, will increase the total soil carbon pool (Figure 40). Therefore using aboveground crop residue for bioenergy would emit carbon that would have mostly decayed into the atmosphere anyway. Assuming a typical residue harvest collection cost of approximately $\$150\text{-}175 \text{ ha}^{-1}$, the price of carbon under a hypothetical carbon policy would only slightly increase the cost of residue harvest, and would not be a significant economic consideration.

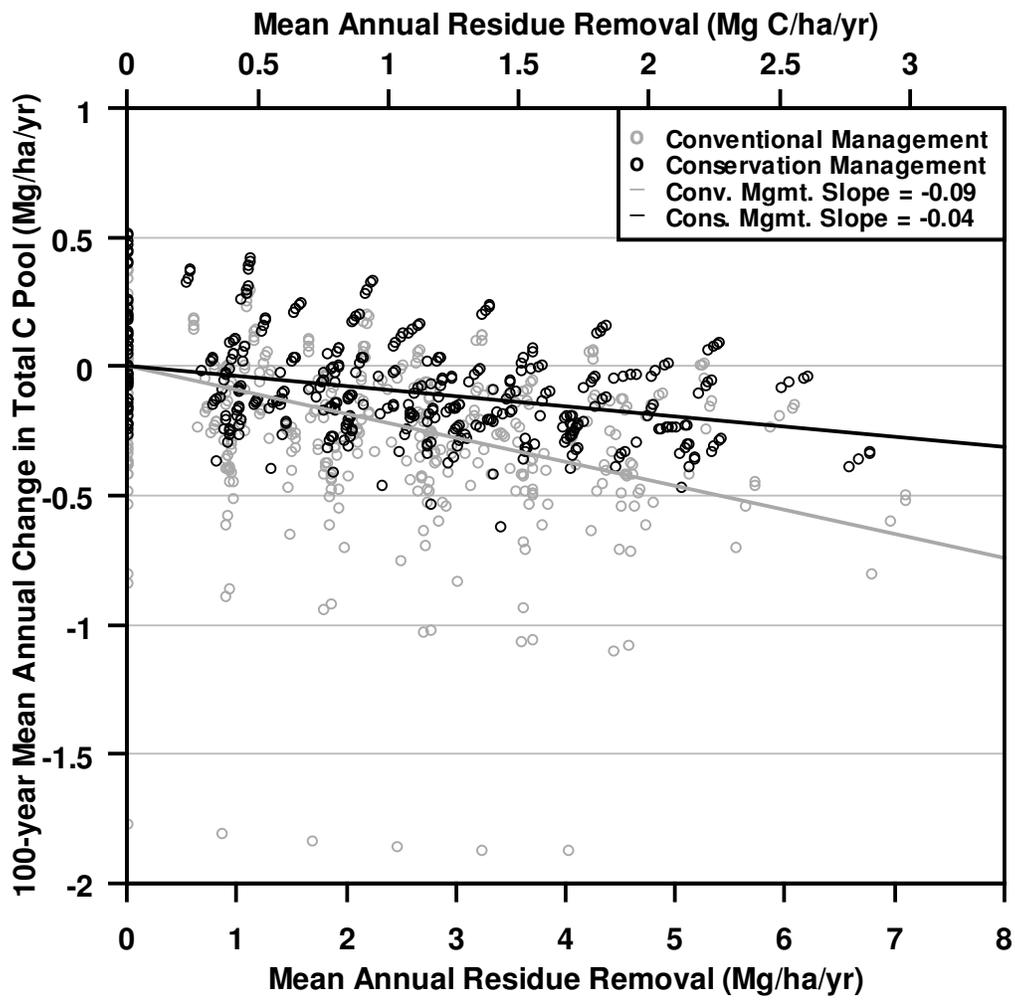


Figure 39. 100-year mean annual carbon loss versus of 100-year mean residue harvested (in absolute mass and mass C), by management. It is assumed that residue is 42% C.

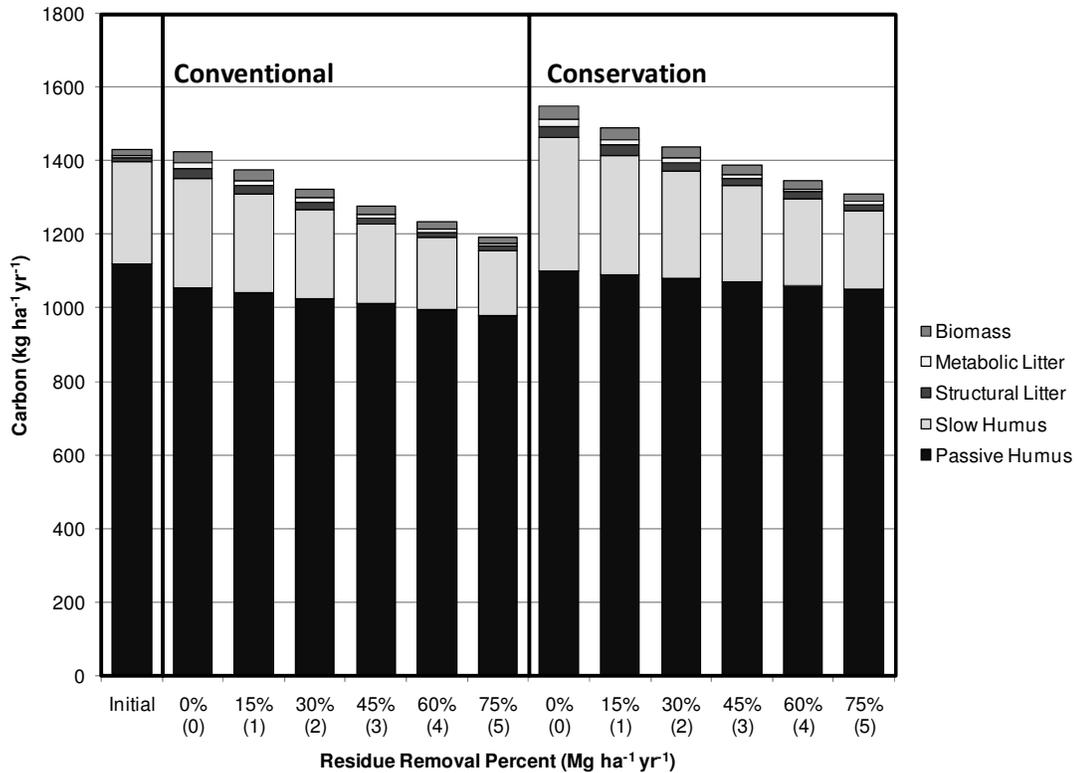


Figure 40. Mean soil carbon balance with different rates of residue harvest and management systems. Columns represent the mean from all slopes and all years.

Residue removal increases loss of soil nitrogen in all pools. Nitrogen loss is particularly dramatic in the slow humus under conventional management, where, on average, up to 15 kg ha⁻¹ yr⁻¹ is lost when high rates of residue are harvested (Figure 41). Again, less nitrogen is lost under conservation management, however conservation management is less effective at retaining nitrogen as it is at retaining carbon. The mean rate of soil nitrogen loss is about 3 kg ha⁻¹ yr⁻¹ per Mg of residue

harvested and would not be a large economic consideration: roughly \$1 per tonne of residue harvested.

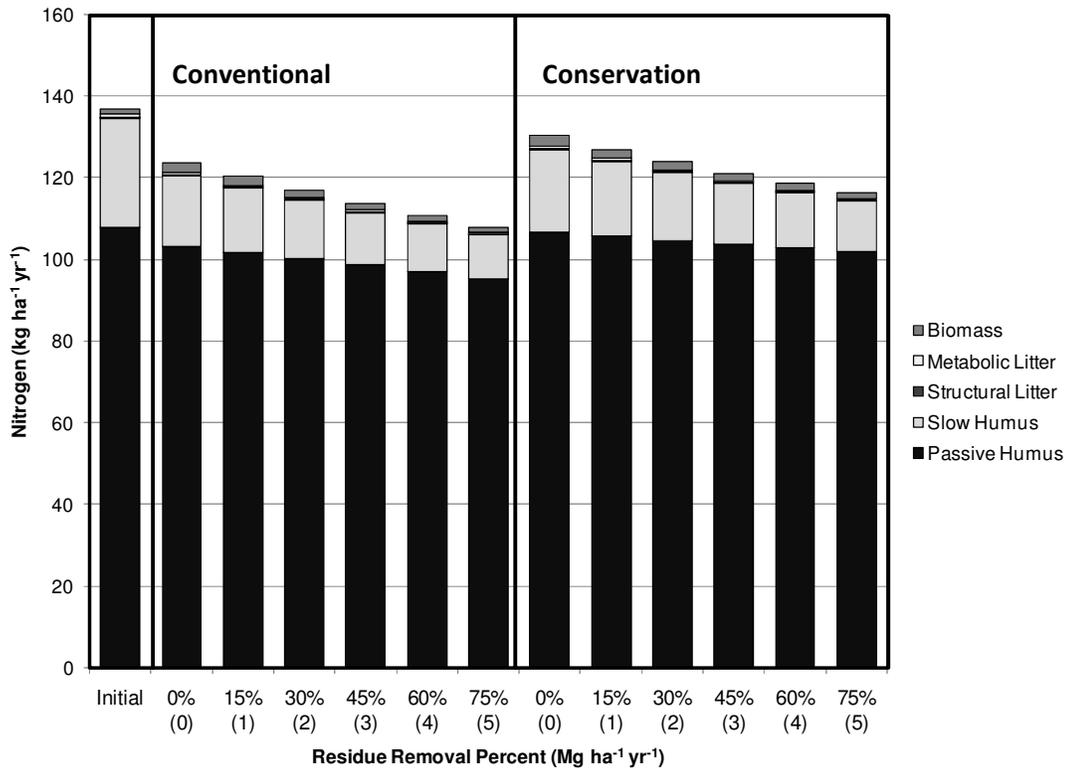


Figure 41. Mean soil nitrogen balance with different rates of residue harvest and management systems. Columns represent the mean from all slopes and all years.

Thus harvesting agricultural residue for bioenergy is a question of tradeoffs. Removal of residue results in increased soil loss, a general reduction of crop yield, and loss of soil nutrients. However, much of these detriments can be mitigated through best management practices. It is possible that sustaining yields while

harvesting residue would increase demand for crop inputs such as irrigated water and fertilizer, though in this study, no significant difference was found in water use or fertilizer between the different levels of residue harvest. However, the base level fertilizer and irrigation rates were set high to produce maximum yields, and large treatment increments and application rates were used. In addition a cap was induced so that the field would not be unrealistically fertilized or watered. All fields simulated in this study were irrigated and fertilized automatically based on plant stress, at large increments and a maximum allowable application rate per year (irrigation: 100-200 mm ha⁻¹ per application, maximum 1500 mm ha⁻¹ yr⁻¹; nitrogen fertilizer: 50 kg ha⁻¹ application, maximum 200 kg ha⁻¹ yr⁻¹). These application rates were designed to approximate practical economic decisions a grower would likely make (e.g., it would not be economical to conduct multiple field passes in a season applying only a small amount of fertilizer each time the crops became slightly stressed). In all trials the base level of irrigation and fertilization was set high enough to ensure maximum potential crop yield and the frequency of subsequent applications of fertilizer and water (as determined by plant stress) was highly variable across location and time.

These results are based on simulations, and therefore are subject to errors and assumptions in the input data and model structure. In this study, there is no direct validation since the fields, topography, management, and crop choices were hypothetical and were designed to isolate the effect of specific parameters. Though EPIC has been validated with field results in numerous studies (discussed above), we cannot expect any model to perfectly predict the future. The principle uncertainties in

these results would fall under two broad categories: future technology, and future climatic conditions.

First, EPIC assumes no changes in agronomic properties of crops (such as increases in yield, changes in harvest index, etc.) and no developments in management strategies as technology and conservation management practices improve. Historically, crop yields have increased in the US with improved technology, precision agriculture, best management practices, and chemical and genetic engineering. Yet it is unclear how these trends will continue into the future.

Second, weather data are based on historical climate data, and therefore do not include regional climatic changes, particularly in the expected increase in extreme weather events. No attempt was made in this study to project the behavior of plant growth under different global climate change scenarios, particularly increased atmospheric CO₂ concentrations; the atmospheric concentration of CO₂ was set at 390 ppm for the duration of all model runs. Holding atmospheric CO₂ concentrations stable for the next century is likely unrealistic, but this was done to isolate long-term trends and for allowing a basic discernment of the sensitivity of erosion, crop yield, carbon and nitrogen to different levels of crop residue harvest without the confounding variable of climate change. These two sources of uncertainty could dramatically affect the sustainability of residue harvest, both positively in the case of improved technology, or negatively in the case of more extreme weather events.

Conclusions

In the search for a single number that represents a sustainable harvest rate, we find that sustainability is highly dependent not only on what crops are grown but also where and what conservation management practices are in place. In terms of remaining within tolerable soil loss, the currently accepted 30% of residue sustainable removal rate is likely a conservative estimate for large-scale national calculations. If conservation practices are in place on relatively flat land, a higher rate of sustainable residue harvest is likely possible. However, all farming is ultimately local, and there is high variability in the sensitivity of erosion and yield to residue removal based on location (soil, climate, topography). Also, the crop rotation also have an effect of the sustainable residue harvest rates. For example, more crop residue could be harvested from wheat rotations than corn-soy rotations. Thus, applying a single residue harvest rate across a broad area (such as the entire US) is likely impractical.

The question of residue harvest is one of trade-offs: removing residue will, in most cases, reduce soil carbon, reduce soil nitrogen, reduce yields, and increase erosion. Nevertheless, with prudent use of conservation management practices, and targeted collection on areas where the slope is modest, it may be possible to harvest a large percentage of crop residue for bioenergy while experiencing only little adverse effect on yield, soil loss, soil carbon and soil nitrogen loss.

Acknowledgements

This work was funded in part with support from the US Department of Energy's Great Lakes Bioenergy Research Center. Research was conducted at the Joint Global Change Research Institute (JGCRI), a joint collaboration between the Pacific Northwest National Laboratory and the University of Maryland, managed by Battelle.

Chapter 7: National and regional generation of municipal residue biomass and the future potential for waste-to-energy implementation

Preface

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In Press:

Biomass and Bioenergy

Abstract

Municipal residue biomass (MRB) in the municipal solid waste (MSW) stream is a potential year-round bioenergy feedstock. A method is developed to estimate the amount of residue biomass generated by the end-user at the scale of a country using a throughput approach. Given the trade balance of food and forestry products, the amount of MRB generated is calculated by estimating product lifetimes, discard rates, rates of access to MSW collection services, and biomass recovery rates. A wet tonne of MRB could be converted into about 8 GJ of energy and 640 kg of carbon dioxide (CO₂) emissions, or buried in a landfill where it would decompose into 1800 kg of CO₂-equivalent (in terms of global warming potential) methane (CH₄)

and CO₂ emissions. It is estimated that approximately 1.5 Gt y⁻¹ of MRB are currently collected worldwide. The energy content of this biomass is approximately 12 EJ, but only a fraction is currently utilized. An integrated assessment model to project future MRB generation and its utilization for energy, with and without a hypothetical climate policy to stabilize atmospheric CO₂ concentrations. Given an anticipated price for biomass energy (and carbon under a policy scenario), by the end of the century, it is projected that nearly 60% of global MRB would be converted to about 8 EJ y⁻¹ of energy in a reference scenario, and nearly all of global MRB would be converted into 16 EJ y⁻¹ of energy by the end of the century under a climate policy scenario.

Introduction

Nearly all the products we create are eventually discarded. Human-appropriated global biomass, estimated to be about a fifth of total global primary productivity (Imhoff et al. 2004; Vitousek et al. 1986), eventually returns to the global ecosystem as the municipal residue biomass (MRB) component of municipal solid waste (MSW). The majority of MRB is collected and aggregated at population centers where energy demands are high, making it a potential non-seasonal bioenergy feedstock, either through incineration or conversion to a liquid fuel (provided cellulosic ethanol technology becomes economically feasible). Other technologies for thermal disposal, such as pyrolysis and gasification, make MRB a potential environmentally sound source of low-carbon energy, chemical feedstocks, and other

value-added products (Malkow 2004). The availability of this resource increases with increasing population and per capita energy consumption (Bogner and Matthews 2003), and using MRB for energy reduces land demands for waste disposal sites near urban areas where land pressures are high (Porteous 2005). Though waste-to-energy facilities currently require large capital expenditures (EIA 2007c), the economics of energy from MRB would be dramatically improved under a carbon market where energy from MRB displaces fossil energy (Consonni, Giugliano, and Grosso 2005b), and where combusted MRB prevents the formation of methane from anaerobic decomposition of landfilled waste.

To determine precisely the potential for energy from MRB, it is necessary to know the quantity of biomass generated by end-users annually, its composition and energy content, the rate at which it is collected, the rate at which it can be recycled, composted, or otherwise recovered, and the economic incentives involved in its disposal (i.e., converting it to energy versus burying it in a landfill). Estimating the potential for MRB energy on national, regional and global scales is challenging because detailed historical data on MSW on the country level are not well archived (Bogner and Matthews 2003; Milke 2008). The majority of studies on waste generation rates and its composition are local, focusing on a set of individuals or households (Pekcan et al. 2006), an local area (Read 1999), or an individual country (Desmond 2006; EPA 2008). Results of studies conducted on different geographical scales are often difficult to reconcile due to differing definitions, methodologies, and the challenges associated with spatial extrapolation.

To analyze MSW trends at national and global scales, it would be desirable to have a national annual database of MSW statistics. Unfortunately, such a database does not yet exist. However, the United Nations (UN) Statistics Division (2009) does maintain a limited dataset on MSW aggregated at the country level for a select countries, based on data from the Organisation for Economic Cooperation and Development (OECD). These data include the rate at which MSW is incinerated, recycled, composted, and buried. Within some countries, the UN Statistics Division also keeps data on the proportion of the population with access to MSW collection services. While the UN Statistics Division give some picture of MSW generation around the world, unfortunately the data are incomplete, are updated infrequently, and have varying degrees of reliability. Also, the UN Statistics Division does not maintain data on the composition of this waste (e.g. proportion of biomass), and thus it is difficult to estimate the energy and carbon content of MSW. The Intergovernmental Panel on Climate Change (IPCC) has produced estimates of waste and its composition by world region for use as default values in a national emissions worksheet (Pipatti et al. 2006), but the data are aggregated into a small number of large world regions, and it is not clear what proportion of the population in each region has access to MSW collection service. Perhaps because of this lack of data, there is a dearth of detailed studies investigating the energy potential from MRB on a global scale.

Therefore, large-scale studies must rely on statistical proxies to estimate the amount of MSW generated in each country. For example, in a study of the technical

feasibility of waste-to-energy, Consonni, Giugliano, and Grosso (2005a) resorted to an "educated guess" about the composition of MSW based on data collected in Italy during the 1990s and professional experience of the authors. In another study that aimed to quantify global methane (CH₄) emissions from landfills, Bogner and Matthews (2003) noted the lack of reliable MSW data from developing countries had lead to shortcomings in the IPCC approach, and therefore adopted per capita energy consumption as a proxy for per capita MSW generation. Global methane emissions databases must resort to the similar strategies; lacking specific annual data on MSW generation, the Carbon Dioxide Information Analysis Center (CDIAC) uses an extrapolation based on GDP growth, with rough adjustments in the slope to account for waste-to-energy in the developed world (Stern and Kaufmann 1998).

Efforts involving integrated assessment models to project future global potential of energy from MRB often assume a growth curve based on Gross Domestic Product (GDP) (Gillingham, Smith, and Sands 2007) or assume per capita MSW generation asymptotically approaches some specific annual rate (e.g., 2.5 Mg person⁻¹ y⁻¹) as per capita GDP increases (Günther Fischer and Schrattenholzer 2001). Further assumptions are made concerning the composition of MSW in order to estimate an appropriate energy conversion factor (Günther Fischer and Schrattenholzer 2001; Gillingham, Smith, and Sands 2007). GDP makes a convenient explanatory variable because statistics are readily available for all countries of the world and are derived in part from the consumption of goods. While using GDP in a simple regression model to estimate MSW generation can provide a first order approximation, it can also lead

to unrealistic results when projected into the future. Across the world, the distribution of GDP is highly skewed and the statistical fit is heavily influenced by developed countries such as the US. Thus, using GDP as an explanatory variable for total MSW generation tends to assume that all countries will eventually have throughputs that resemble the US, and furthermore, the trend will continue on that trajectory into the future. This can result in enormous future estimates for MSW generation in countries with large populations and high future labor productivity projections, such as China. Moreover, a GDP regression on MSW is non-processed based, in that it is disconnected from an analysis based on material throughput of the global economy. Studies have suggested that many of these relationships are non-linear [see, for example, (Schenk, Moll, and Potting 2004)], and a simple GDP regression approach is also unable to capture changes in composition of MSW. The GDP proxy approach neglects the dynamics of product demand, end-user disposal rates, access to MSW service, and development of recycling and composting techniques that increase as economies develop. While overall MSW generation tends to increase with GDP, projecting these patterns into the future requires a more detailed parameterization of the generation and composition of discarded materials, the development of MSW collection service and recovery strategies, and the market potential for energy from MRB.

The purpose of this paper is threefold. First, a throughput approach is used to develop a set of functions that refine the current estimates for the amount of MRB based on domestic consumption of agricultural and forestry products, population, and

per capita wealth. From this, estimates of the collected biomass are calculated for nearly all countries and regions of the world. Second, these statistical relationships are used to project future MRB generation based on projections from an integrated assessment model. Finally, third, this paper examines the effect a hypothetical climate policy would have on the future proportion of MRB converted to energy and the resulting carbon emissions with respect to waste-to-energy implementation. Modeling the life cycle of bio-resources leads to better quantification of how economic and policy decisions affect the global flows of mass, energy, and carbon.

Methodology

Current Municipal Residue Biomass Generation

The bio-material flow into the MSW stream is calculated from production and trade statistics of food, wood and paper. To estimate the amount of collected MRB from a country, first the total domestic agricultural product consumption and total domestic forest product consumption are calculated. The Food and Agriculture Organization (FAO) forest and agricultural products trade statistics (FAOSTAT 2008b, 2008c, 2008d) are used to determine the total product demand: the sum of production and imports, reduced by the sum of exports and changes in stock; similar to a formulation employed for fossil fuels at the Carbon Dioxide Information Analysis Center (CDIAC) (Marland, Boden, and Andres 2005). This gives apparent

consumption of both food and forest products for all countries in the FAOSTAT database.

The apparent consumption of agricultural products is further reduced by the amount used for animal feed, seed, and other uses, yielding the domestic food demand (FAOSTAT 2008b, 2008d, 2008a). All food is converted from mass values (wet basis) to calorific values. It is assumed that the amount of multi-year food storage is small, and that the entire domestic food demand is completely digested or discarded within the year, providing calories to the population while at the same time producing discarded biomass.

For forest products, the apparent consumption is reduced by the amount of firewood and charcoal produced as these are combusted and do not enter the MSW stream. This gives domestic wood consumption, which is converted to a variety of wood and paper products (sawn wood, wood based panels, household and sanitary paper, packaging and wrapping paper, and paper and paperboard). Annual production data are taken from the FAOSTAT (2008c) and are in units of mass, except for sawn wood and wood panels (reported in volume); these were converted to mass using an assumed density of 500 kg m^{-3} for sawn wood and 800 kg m^{-3} for wood paneling. Some fraction (Figure 42) of the wood products are immediately discarded upon consumption as scraps and debris (Buchanan and Levine 1999). This immediate scrap factor is more important for sawn wood and wood panels, products which otherwise have long lifetimes before disposal. For these products, it is assumed that the

immediate scrap rate is 50%, based on calculations from a construction and waste study done by the Georgia Department of Natural Resources (1998).

The remaining fraction of wood products that is not immediately scrapped has an extended lifetime of use before entering the MSW stream. For a given good, the life of use is assumed to follow a gamma distribution (Marland and Marland 2003) (Figure 42), defined by the mean lifetime of each type of good (Figure 42). The wood discarded at any one year is, in addition to the immediate scrap produced, the sum of discrete gamma distributions based on the consumption data from previous 100 years. The FAO ForestStat database begins in 1961, so input data for the gamma distribution for the years 1900-1960 were estimated linearly on a product-by-product basis for each country. This is to provide initial conditions and does not substantially affect the estimates for current generation of wood MRB. For a given year t , the amount of wood discarded each year is:

$$discarded\ wood_t = \sum_{g=1}^5 IS \cdot x_{g,t} + \sum_{t=-100}^0 \sum_{g=1}^5 \frac{(1-IS) \cdot x_{g,t}^{(\gamma_g-1)} \cdot e^{-(1-IS)x_{g,t}}}{\Gamma(\gamma_g)}$$

where x is the consumption of each product g , with mean life γ and immediate scrap fraction IS . The mean life of wood products is estimated to be 40 years for sawn wood (Buchanan and Levine 1999; Marland and Marland 2003), and 3 years for paper products (Marland and Marland 2003). Slight adjustments are made as it is

assumed that the mean life of lumber is slightly longer than wood paneling, and it is assumed that the mean life of household, sanitary, wrapping, and packaging paper is slightly shorter than that of other paper products (Figure 42).

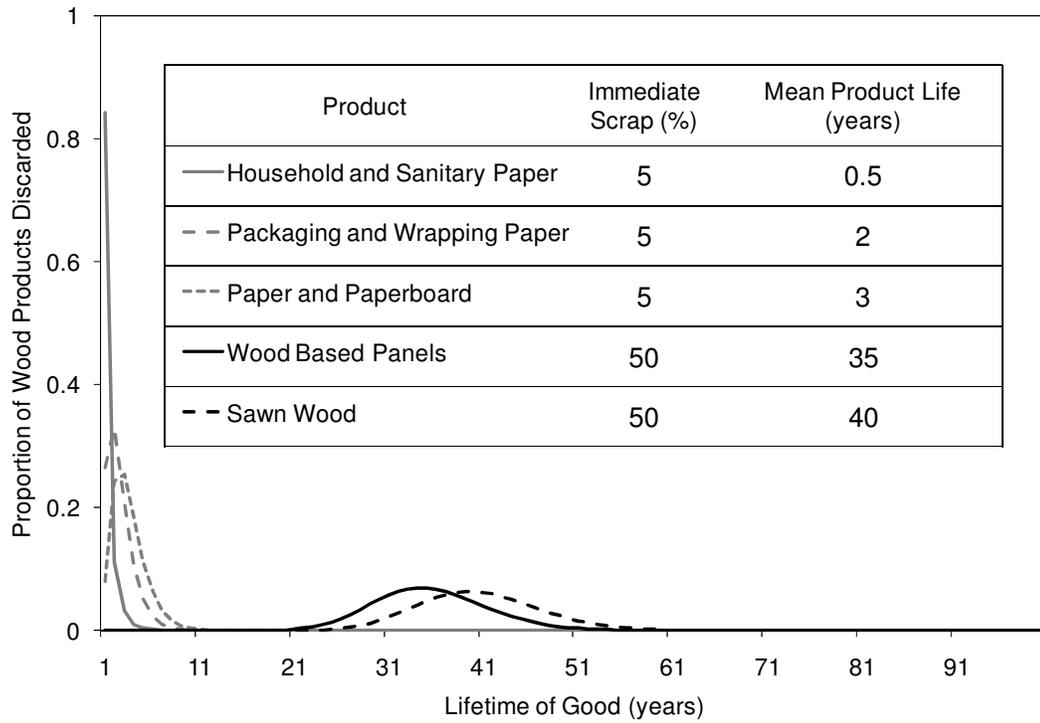


Figure 42. Gamma distribution used to model the life of various wood products, and the parameters used in estimating the amount of wood MRB from domestic wood demand.

Thus, given the apparent consumption of agricultural and forestry products, and the domestic demand for food and wood, the annual amount of discarded biomass can be estimated for each country by applying the wastage proportions. Not all

discarded biomass is available for energy, however. First, only some proportion of a country's population has access to MSW collection services. Second, some portion of the discarded biomass never enters the MSW stream: it is lost, littered, or recovered by the consumer. Third, a portion of the discarded biomass that enters the MSW stream is composted, recycled, or otherwise recovered by the municipality. Therefore, the discarded biomass is multiplied by a series of proportions varying from 0 to 1.

These proportions are implemented using a series of logistic functions, based on per capita Gross Domestic Product (GDP) in terms of Purchasing Power Parity (PPP) in inflation- adjusted international dollars. The general form of this function is given as:

$$p = \min + \frac{(\max - \min) \cdot PPP^b}{midPPP^b + PPP^b}$$

where p is the proportion of interest, PPP (the independent variable) is per capita wealth in Purchasing Power Parity, \min and \max are the *a priori* bounds to the proportion, $midPPP$ is the per capita wealth level where half of the potential is realized, and b defines the shape of the logistic curve. The curves are fit such that they minimize the Root Mean Squared Error (RMSE) between predicted and country-level reported data.

The proportion of country's population that has access to MSW collection service is estimated from 1998-2005 data from the UN Statistics Division (2009). To

estimate the food wastage rate, total discarded food calories were divided by the sum of the discarded food calories and the dietary intake calories for the years 1980-2003 in the FAOSTAT (2008a) food balance database. Wood wastage rates are from the Confederation of European paper Industries (CEPI) (2008) and the International Energy Agency (IEA) (2007). The total recovery rate is estimated from the sum of recycling and composting rates of all MSW data from the UN Statistics Division (2009). The apparent wood product recycling rate is computed as the ratio of recovered paper to apparent paper consumption from the FAOSTAT (2008c) forest trade statistics database. It is assumed that the maximum recovery rate is 81% (Schenk, Moll, and Potting 2004), due to the breakdown of useful fiber and contamination of discarded wood products. Economic data (per capita GDP PPP) are from the World Bank (2008).

After fitting these various parameters to the data, they are multiplied by the total discarded food and wood products, to obtain an estimate for collected biomass. The total collected food or wood biomass potentially available for energy conversion is:

$$\text{Collected MRB} = \sum_{(food, wood)} \text{Discarded Biomass}_{(food, wood)} \cdot p_{service} \cdot p_{MSW}(food, wood) \cdot (1 - p_{recover}(food, wood))$$

where $p_{service}$ represents the proportion of the population with MSW collection service, p_{MSW} is the proportion of discarded biomass that enters the MSW stream, and

$p_{recover}$ is the proportion of MRB that is recovered (reused, recycled, composted, etc.). The collected MRB is converted to per capita rates by dividing by population statistics from the World Bank (2008).

The total potential amount of energy from MRB is determined by converting the collected MRB into energy units based on the caloric content of the food (roughly 7 GJ Mg^{-1} , wet basis, on average; this varies regionally based on diet), and by converting the volume of wood MRB into energy units using an average density of 0.5 Mg m^{-3} and a net heating value that assumes a water content of 15% (approximately 17 GJ Mg^{-1} , wet basis) (Rosillo-Calle 2007a).

Future Municipal Residue Biomass Generation

To project future MRB generation, carbon emissions, and biomass feedstock demand, output from the Object-Oriented Energy, Climate, and Technology Systems Mini Climate Assessment Model (O^{bj}ECTS MiniCAM) is used (Kim et al. 2006). O^{bj}ECTS MiniCAM is a modular, partial equilibrium economic integrated assessment model that projects changes in regional energy portfolios, housing, industrial and transportation sectors, and land use, agriculture, and forestry markets under different climate policy scenarios (Kim et al. 2006). Here, it is used to project future regional consumption of agricultural and forest products, global prices for biomass energy feedstocks, and the global carbon price under the presence of a 450 ppm atmospheric CO₂ concentration stabilization policy. Future population and GDP PPP roughly

follow the B2 Scenario of the IPCC Special Report on Emissions Scenarios (Clarke et al. 2007). The carbon and energy prices are multiplied by the regional MRB generation estimates, and waste-to-energy utilization is projected based on maximizing profit.

Using OBJECTS MiniCAM output, projections of future consumption of agricultural and forestry products are obtained. However, estimating wood MRB by summing across multiple gamma distributions is computationally intensive and impractical for future projection using an integrated assessment model, which is designed to handle flows rather than stocks. Therefore, to project future forest product MRB generation, a regression relationship is used: from the historical data and the summed gamma distributions, wood MRB in a given year is approximately 62% of the domestic wood product consumption (Figure 43). The strong correlation between MRB and consumption is due to the relatively stable proportions between types of products consumed, and the assumption that the product lifetimes and immediate scrap percentages (the gamma distribution parameters) have not changed through time. From here, the statistical relationships between *PPP* and MSW service, *PPP* and discard rates, and *PPP* and recovery rates are applied to each region of the world to estimate future collected MRB.

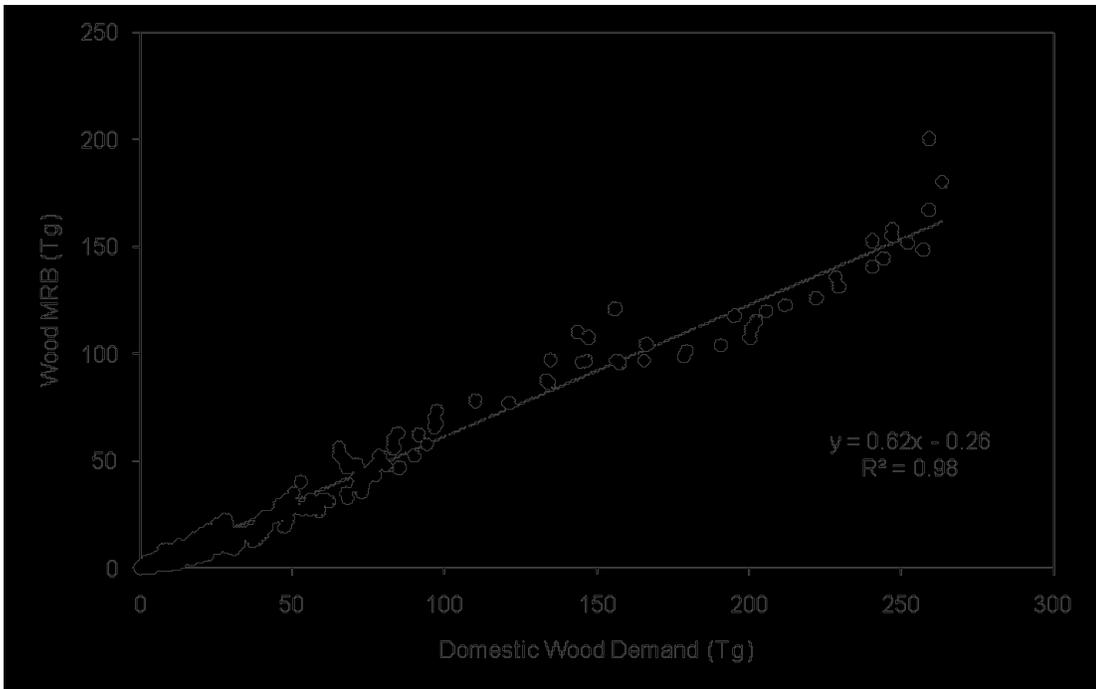


Figure 43. For each year 1980-2006, the estimated wood waste (total mass of wood products at their end of useful life) is plotted against the domestic demand of wood. The amount of wood waste in a given year is approximately 62% of the domestic demand of wood products.

The decision to convert MRB to energy is fundamentally an economic one, based on urban land pressures and energy prices. The dynamics of urban land prices occur on a scale much smaller than what can be modeled in a global integrated assessment model, and thus regional biomass energy prices and global carbon prices are used as a proxy to model future economic decisions regarding MRB energy utilization. A cost curve for biomass feedstocks is fit to data generated for the

National Energy Modeling System (NEMS) developed by the Energy Information Administration (EIA) (2003). These data project an amount of energy output at a given biomass price, but for this study, have been scaled from 0 to 1 to represent the proportion of the total potential biomass supplied at a given price. The cost curve also takes the form of a logistic function, ranging from 0 to 1, and given a biomass price, it is possible to estimate the proportion of the biomass converted to energy versus the total energy potential from all collected MRB in a region. It is fit to the discrete data points of EIA NEMS output, where the midpoint is the energy price at which 50% of all collected MRB is converted to energy. The same cost curve is applied globally, as country-specific values are not available. However, the EIA NEMS data are used only to determine the shape of the cost curve. The specific values are not applied in this study; rather, the logistic function is scaled to each region to more accurately reflect the size of the hypothetical market in each area.

Under a climate policy, the cost of carbon emissions affects the economic decision to utilize biomass, both in terms of emissions from combustion and avoided emissions from landfills. To calculate greenhouse gas emissions, wood MRB is estimated to be 85% dry matter, and food MRB is estimated to be 40% dry matter by mass (Pipatti et al. 2006). Furthermore, wood MRB and food MRB are estimated 48% and 38% carbon, respectively (dry basis) (Pipatti et al. 2006). If MRB is converted to energy, the carbon is assumed to be converted to CO₂ with a 99% combustion rate, similar to that of fossil fuels (Blasing, Broniak, and Marland 2005a). On the other hand, MRB buried in a landfill decomposes into CH₄ and CO₂. It is

estimated that, globally, approximately 25% of the carbon in wood products in landfills is converted to greenhouse gases, based on a consumption-weighted mean of rates for various wood products multiplied by coefficients estimated by Skog and Nicholson (1998). Food decays more easily than wood as there is no lignin to protect the cellulose, hemicellulose, and carbohydrates; thus it is assumed that 90% of the carbon in food is converted to carbon dioxide (CO₂) and methane (CH₄). Of the carbon emitted from landfills, approximately 40% is in the form of methane (CH₄), and 60% decays into CO₂ (Skog and Nicholson 1998). Thus, every kg of food (dry basis) that enters a landfill eventually produces 0.18 kg CH₄ and 0.75 kg CO₂, and every kg of wood (dry basis) that enters a landfill eventually produces 0.07 kg CH₄ and 0.27 kg CO₂. Given a global mean composition of food and wood MRB, a kg of MRB (dry basis) buried in a landfill decomposes to 0.17 kg CH₄. This is in line with the 0.10 kg CH₄ per kg of dry solid waste estimated by Bingemer and Crutzman (1987) using a temperature-based model and the theoretical maximum of 0.18 kg CH₄ per kg of dry solid waste predicted by Halvadakis et al. (1983) using an empirical equation. Though emissions from landfills occur many years after a given amount of MRB is buried, it is assumed for economic calculations that the carbon price for the estimated landfill emissions would be paid in the year that the MRB is buried and would not be discounted over a long time span.

Therefore projected future implementation of waste-to-energy is simulated using two economic incentives: the energy market and the carbon market. A wet tonne of MRB (given current mean global proportion of food versus forestry products

consumption) can be converted to approximately 8 GJ of energy, resulting in 640 kg of CO₂ equivalent emissions (in terms of global warming potential). On the other hand, if this tonne of MRB is buried in a landfill, no energy is generated, and nearly 1800 kg of CO₂ equivalent emissions are eventually released into the atmosphere.

Results

Current Municipal Residue Biomass Generation

Figure 44 contains the prediction parameters from the fitted logistic curves. MSW service tends to become available rather early as an economy develops; by the time PPP GNI reaches \$2,500, MSW collection can be expected to serve nearly half a country's population. In the developed countries, nearly the entire population has access to MSW service. In general, food discard rates are lower for developed countries, as they have more sophisticated distribution and tracking systems, and more access to technology that prevent spoilage, such as advanced packaging, processing, and refrigeration. Wood discard rates, in contrast to food, increase with wealth. Rates for total MRB recovery (recycling and composting) increase with increasing wealth, but generally occur in countries where nearly all the population has access to MSW collection service. Recovery of forest products tends to occur at a higher *PPP* than the average of all products, and occurs in countries where essentially all the population is covered with MSW collection service.

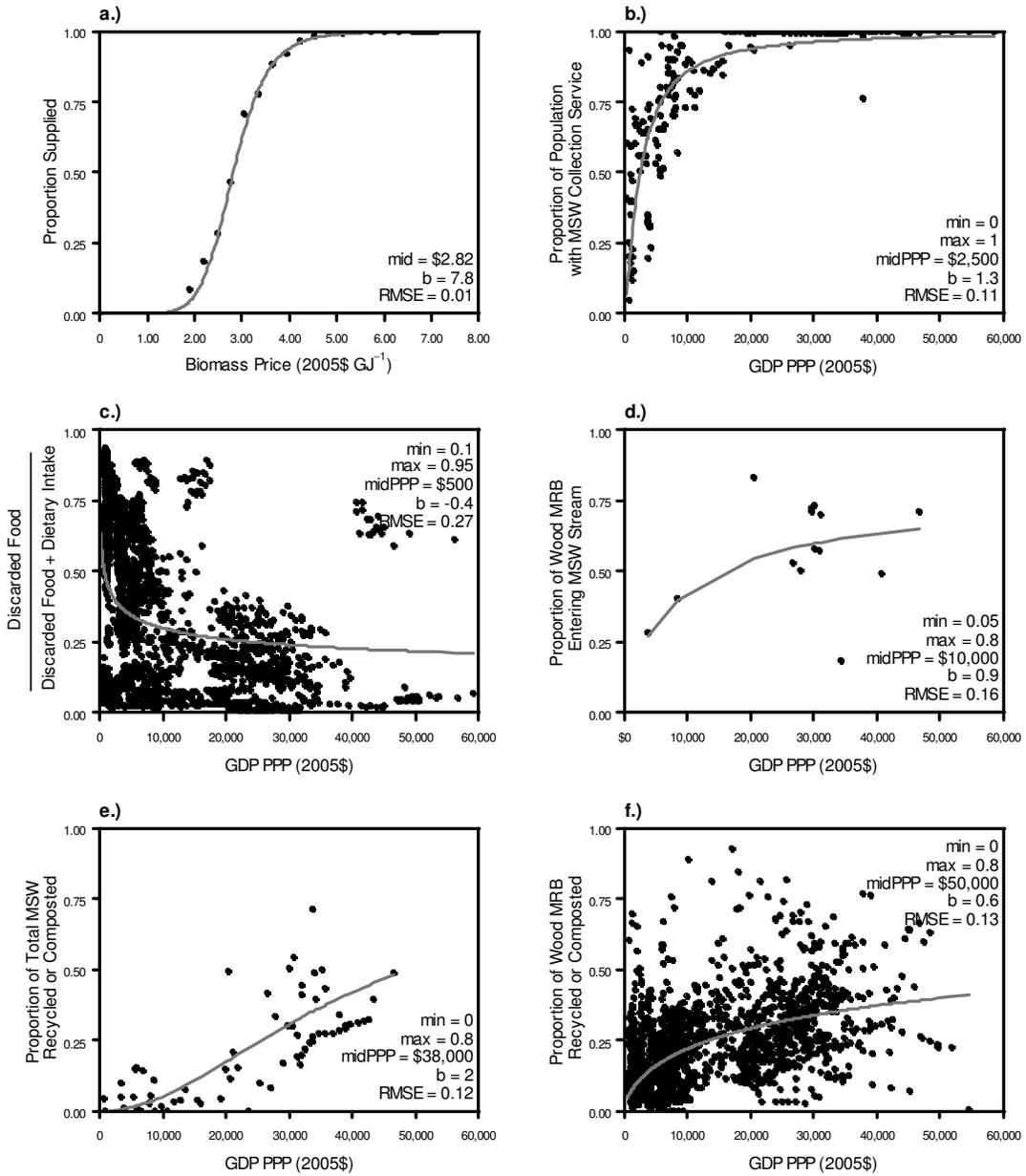


Figure 44. Logistic regressions on the components of MRB generation. a.) The cost curve for biomass fit to data from the NEMS (EIA 2003). b.) The proportion of a country's population that has access to MSW collection service fit to 1998-2005 data from the UN (2009). c.) Food discard rates based on FAO food balance sheets. For

the years 1980-2003, the estimated discarded calories were divided by the sum of the discarded calories and the dietary intake calories in the FAO (2008a) food balance database. d.) The discard rate for wood products (paper) fit to data from the CEPI (2008) and the IEA (2007). e.) The total recovery rate (recycling and composting) of all MSW, fit to data from the UN (2009). f.) The apparent recycling rate for wood products, computed as the ratio of recovered paper to apparent paper consumption from the FAO (2008c) trade statistics database. Per Capita GDP PPP data from the World Bank (2008).

Multiplying the series of logistic equations yields the estimated collected MRB proportions (Figure 45). The curves represent point estimates based on the regressions in Figure 44. There is a steep increase as MSW service develops, then a leveling off as increased efficiency (lower discard rates) and increased recovery technologies are implemented. Multiplying the apparent national consumption of food and wood products by the collected MRB proportions gives a predicted amount of collected MRB per capita (Figure 46). While developed countries generally have lower discard rates (Figure 45), they nevertheless generate more discarded biomass per capita because overall consumption food and wood products is higher (Figure 46). For example, North America, Europe, and Oceania have the lowest predicted discard rates and highest predicted recovery rates, but also highest rates of per capita throughput of biomass, and thus produce the largest amount of per capita collected MRB. Globally, it is estimated that 1.5 Gt of MRB (wet basis) are currently collected

annually, an average of roughly $220 \text{ kg person}^{-1} \text{ y}^{-1}$ (wet basis). It is estimated that 2005 global CH_4 emissions from biomass in landfills were approximately $90 \text{ Tg CH}_4 \text{ y}^{-1}$. This a bit higher than the emissions estimated in the CDIAC database (Stern and Kaufmann 1998), which would be $46.2 \text{ Tg CH}_4 \text{ y}^{-1}$ in 2005 (extrapolated using their stated methodology). In addition to the reasons discussed below, the source of this discrepancy is in part due the accounting assumption that all emissions are counted at the point of burial in the estimates produced by the methodology presented here, whereas this is not the case in the CDIAC estimates. Furthermore, the CDIAC database ends in the year 1994, and had to be projected over 10 years. Nevertheless, spatially, these emissions are estimated to be higher in developing countries with large populations. Because garbage is expensive to transport, the spatial distribution of the emissions within countries are expected to follow the population distribution and be aggregated near urban centers.

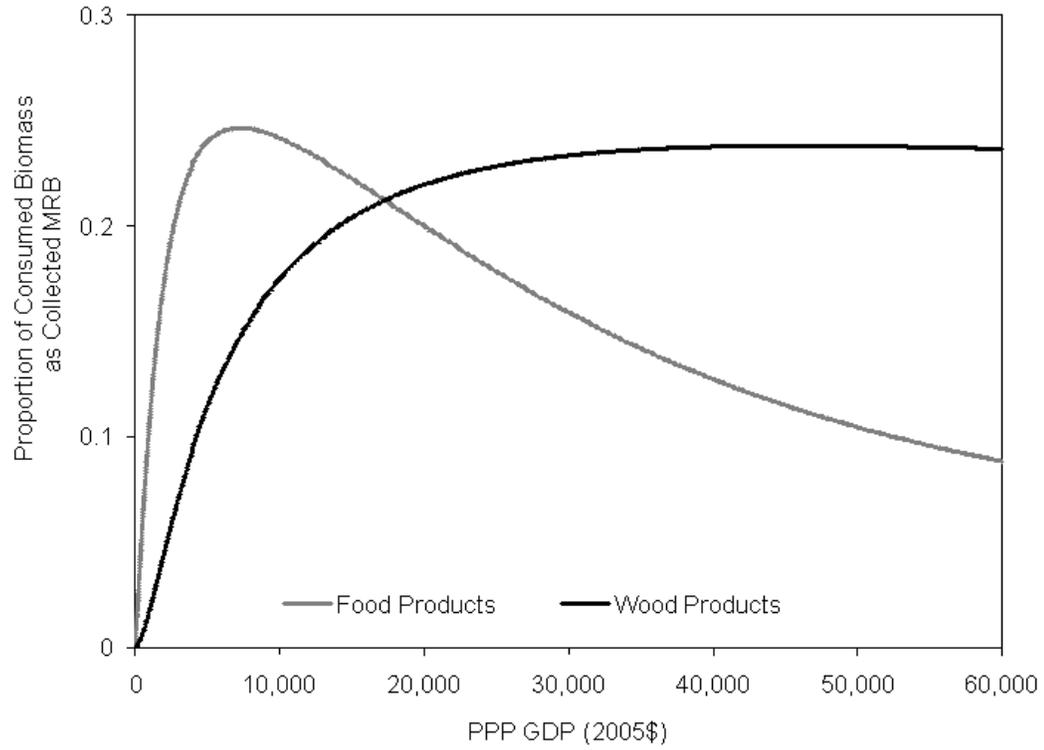


Figure 45. Estimated waste proportions based on GDP PPP. The proportion represents the amount of collected non-recovered MRB to the total biomass (food or wood products) consumed by a country.

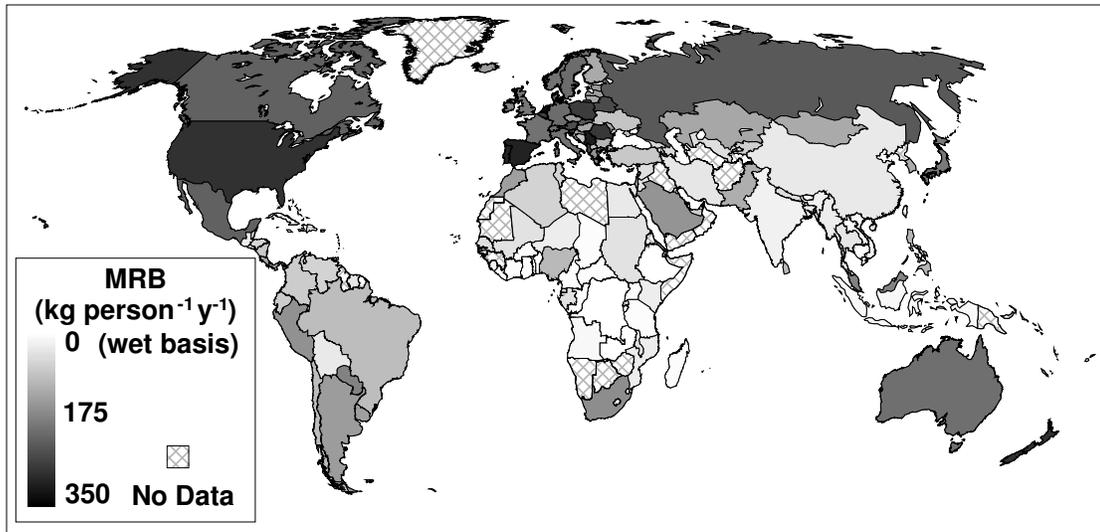


Figure 46. Global distribution of per capita MRB by country for the year 2000. While developed countries have higher waste recovery rates, they also have higher consumption of biomass and more access to MSW collection service, thus more MRB is collected per capita in developed countries. Cross hatched countries have insufficient food balance and forest product trade data to estimate collected MRB. Figure generated using MapViewer™ (Version 6.2.25) by Golden Software Inc., based on calculations described in the Materials and Methods section.

Future Municipal Residue Biomass Generation

In a future scenario without a climate policy, there is no cost for avoided landfill emissions, but there is demand for energy, and thus some of the MRB is still expected to be converted to energy, approaching nearly 8 EJ y⁻¹ globally over the next

century. With the presence of a climate policy, nearly all MRB is converted to energy, generating up to 16 EJ y^{-1} globally by midcentury, and then falling off as recycling and composting rates increase (Figure 47). In terms of carbon emissions, Figure 48 is constructed with the two extreme bounds: one scenario where the all the collected MRB is buried in a landfill, and one scenario where all collected MRB is converted to energy. In between these bounds the reference scenario (with no climate policy) and the policy scenario (a 450 ppm atmospheric CO_2 stabilization) climate policy in place. In the reference scenario, MRB energy develops slowly, approaching about 60% by the end of the century. In the climate policy scenario, MRB energy develops early; by mid-century nearly all annual collected MRB that is not otherwise recovered is converted to energy.

By the end of the century, without a climate policy, landfill greenhouse emissions are projected to be cut by about 50%, with developed countries being first to implement the MRB energy option. With a climate policy, greenhouse gas emissions from landfills are projected to be nearly eliminated, as MRB energy is implemented worldwide.

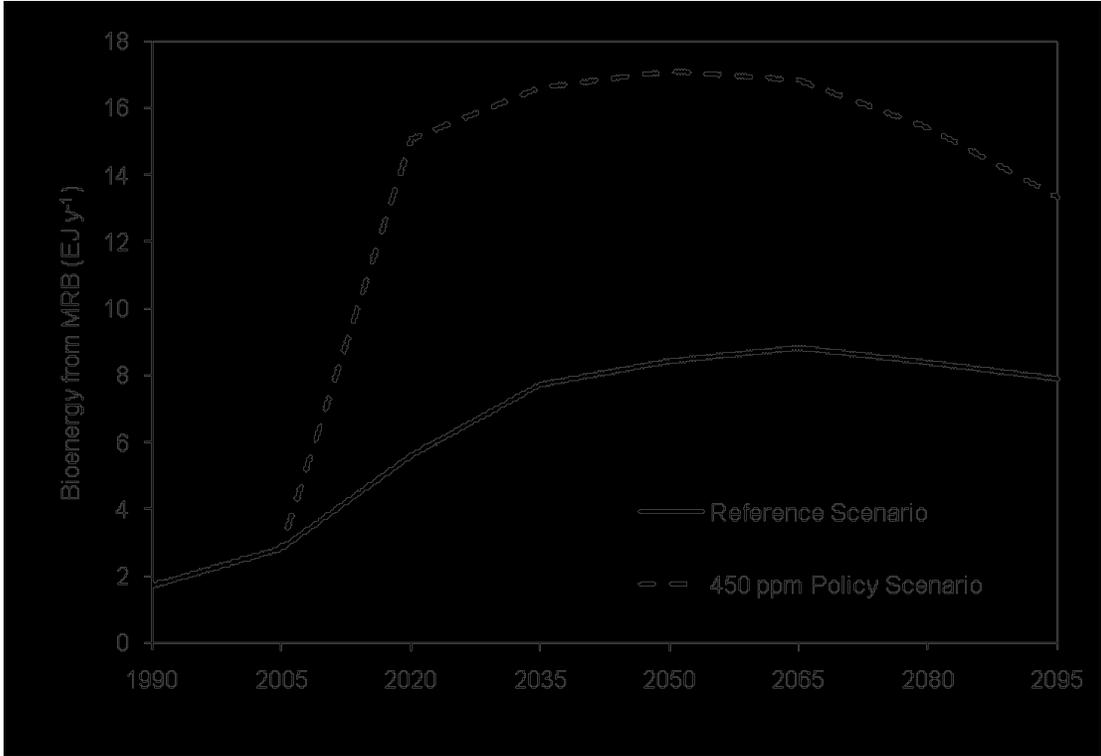


Figure 47. Bioenergy generated from MRB and the influence of a climate policy that prices carbon emissions.

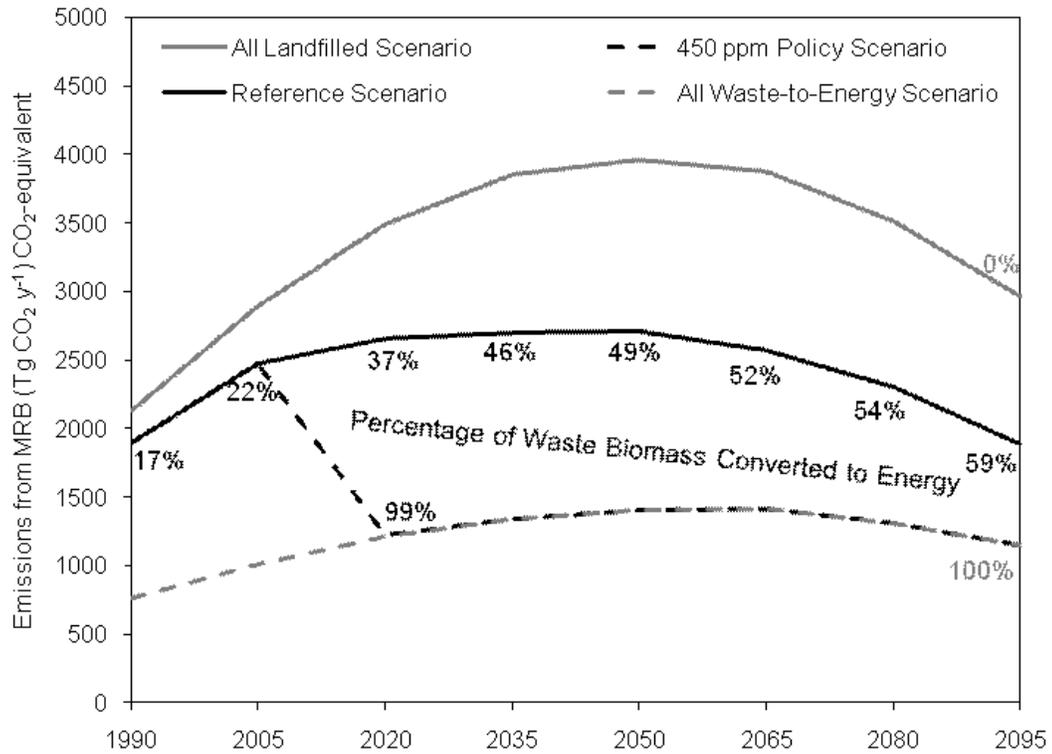


Figure 48. Carbon emissions from the management of MRB. Under a carbon policy that prices carbon, the economic incentive is to convert biomass into energy to avoid landfill greenhouse gas emissions, and all of MRB is utilized to that end. Without a carbon policy, the price for biomass energy is the only economic incentive for utilizing MRB. For the reference scenario (no climate policy) the percentages shown are the amount of MRB converted to energy (energy content basis).

Discussion

Ideally, the predictions based on the biomass throughput approach developed above would match the MSW statistics that estimate what is coming out of the economy; the two ends should meet up. In Figure 49, the predicted results from the throughput approach developed in this paper are compared with the estimates of MRB generation rates for various regions of the world from the IPCC (Pipatti et al. 2006). While regional aggregation hides much of the noise, and there are clear discrepancies, MRB generation across regions nevertheless has the same general distribution and magnitude in both estimates. The RMSE between the two estimates is 45 kg person y⁻¹ (wet basis) or about 20% of the estimated global average MRB generation rate of 220 kg person⁻¹ y⁻¹.

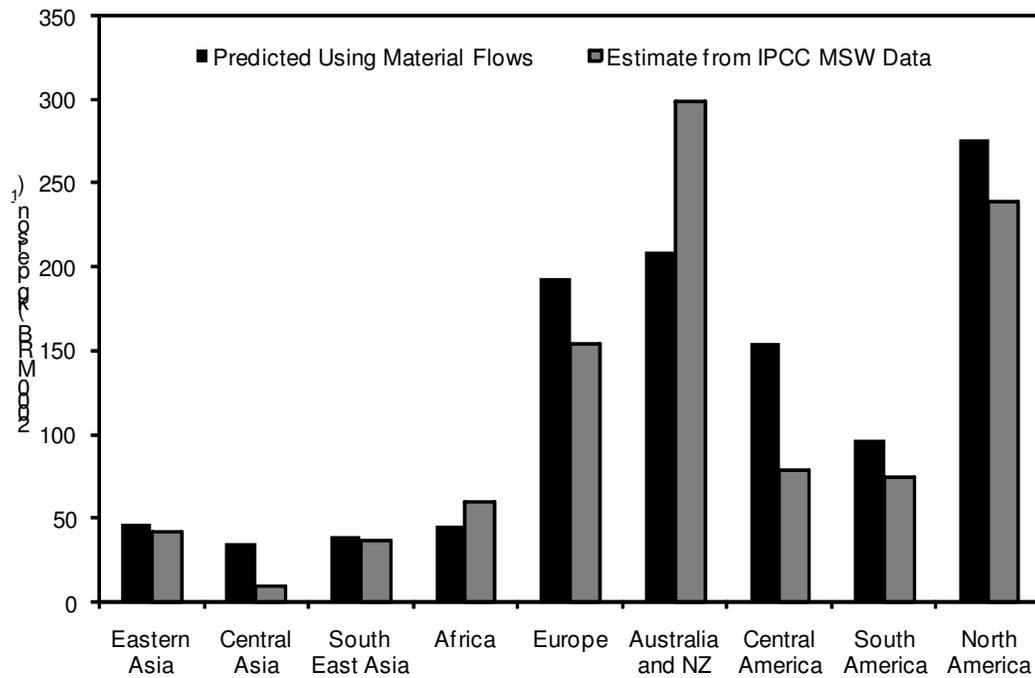


Figure 49. Comparison of per capita MRB rates for various regions of the world between the estimates created by employing the material throughput approach developed in this paper and the IPCC MSW estimates (Pipatti et al. 2006).

It is instructive to examine the reasons such discrepancies exist, which can serve to guide future research. Uncertainty in the estimates produced by throughput method developed here enter in at each step of the process. There is some uncertainty associated with the FAOSTAT trade balance statistics, particularly in the developing countries. For some countries, records are incomplete or missing, creating an inherent selection bias. There are very little data on the proportion of discarded biomass that actually enters the MSW stream, particularly for food. In the throughput approach,

food discard rates are estimated from the FAOSTAT food balance sheets, though in real life, not all this non-digested food necessarily is discarded into the MSW stream: some may be composted by the consumer, or given as feed to pets or livestock, or simply littered. No country-level data are available to account for these variations. This can lead to an overestimate of MRB by the throughput approach developed in this paper. The water content of the biomass is also highly variable, based on local climate and MRB composition. Variation in water content affects the estimates in mass, energy conversion, and CH₄ production from MRB decomposition. Finally, there are societal and cultural differences that are not captured by the per capita wealth. For example, Japan has much lower per capita discard rates than the US, possibly due in part to cultural differences and differences in population density. Thus, there is spread in the logistic fits on many of the parameters, seen in Figure 44. Assuming the errors are independent, the uncertainty in any one given country for any one year is $\pm 31\%$ for the mass of food MRB and $\pm 24\%$ for the mass of wood MRB.

There are also large uncertainties in the underlying IPCC data used for comparison. This is particularly the case with estimates of the percentage of population that has collection service. It is assumed that developed countries have complete MSW collection service, while developing countries only have MSW collection in urban areas; a standard approach taken [see, for example, Bogner and Matthews (2003)] This can lead to an over-estimate in the IPCC defaults for developed countries with low population density, such as Australia. Within developing countries, this raises the question of distribution of wealth and

consumption. For example, if half the population of a given developing country is urban and therefore half the population is expected to have access to MSW service, it does not necessarily follow that half of the given country's MRB is collected, because urban areas tend to be wealthier and have higher per capita rates of consumption. Furthermore, a large fraction of MSW in developing countries is classified as "unknown" in the IPCC worksheet, and thus is not counted as biomass in the IPCC estimates shown in Figure 49. This leads to an underestimate of MRB in developing countries in the IPCC worksheet.

Globally, there are also many uncertainties in greenhouse gas emissions from MRB. In the developing world, MSW is often disposed in so-called "non-engineered landfills" or open dumps (Bogner and Matthews 2003). In these locations, organic decay occurs aerobically, and thus favors the production of CO₂ over CH₄ (Bogner and Matthews 2003). In terms of greenhouse gas emissions, open dumps are favorable to landfills, though the former are certainly less preferable in terms both ecological and sociological considerations. Therefore, the approach taken here assumes that all non-recovered, collected MRB is either converted to energy or buried in a landfill. Open dumps are considered an unsustainable option for future MSW disposal, and in order to more accurately model economic incentives, the option of open dumping has to be made unavailable in the future scenarios envisioned here. In effect, this assumes that as economies develop, regulation will prohibit open dumping in favor of engineered landfills and MRB energy installations. This assumption would create errors in direction seen in the disagreement with the database maintained by

CDIAC (Stern and Kaufmann 1998). Nevertheless, in the economic framework used in this study, it is a necessary economic assumption, as open dumping would be unrealistically incentivized over both waste-to-energy development and MSW burial if only carbon emissions and biomass price are considered.

Given that the choice of managing MRB is constrained to either energy generation or landfill disposal, MRB energy can be an effective method of producing energy and significantly reducing global carbon emissions. Full implementation of MRB energy essentially eliminates landfill gas emissions, while producing up to 16 EJ y^{-1} of energy by mid-century. While this is large amount of energy, it is rather small when compared to the current global primary energy consumption of nearly 500 EJ y^{-1} (BP 2009). Nevertheless, if we assume that this biomass energy were to displace energy from coal combustion, then the net reduction in greenhouse gas emissions (including avoided CH_4 emissions) by 2035 under a climate policy would be roughly 240 kg CO_2 GJ^{-1} (CO_2 equivalent) or approximately 4000 Tg CO_2 y^{-1} (CO_2 equivalent), roughly two-thirds the 2005 fossil-fuel carbon emissions from either the US or China (Gregg, Andres, and Marland 2008).

In a carbon constrained world, there will be increased demand for bioenergy and biomass feedstocks. If MRB biomass can displace biomass crops grown on newly cleared land, then there is an added benefit in terms of carbon, depending on the value of the standing carbon stock in unconverted land. Utilizing the MRB relieves some of the pressures for growing biomass, and circumvents the many of the debates surrounding the bioenergy issue, including land conversion (Fargione et al. 2008),

food versus fuel (Johansson and Azar 2007; Ranses, Hanson, and Shapouri 1998), net energy balance (Pimentel 2003; Shapouri, Duffield, and Wang 2002), and biodiversity loss (Raghu et al. 2006).

Finally, it must be stressed that any policy incentivizing MRB energy expansion must also contain regulations for pollution control. While converting MRB to energy reduces carbon emissions, the process can create other pollutants, such as dioxins, Nitrous Oxides (NO_x), Volatile Organic Compounds (VOC's), Carbon Monoxide (CO), and particulates and ash with high metal content. However, these can be substantially mitigated by using scrubbers, filters, and electrostatic precipitators at the incineration or cellulosic biofuel production plant. It is also imperative to have effective MSW sorting operations as part of any MRB energy strategy to make energy conversion more efficient and prevent combustion of toxic and hazardous waste. These measures increase the capital costs of MSW processing facilities, which could, in theory, be offset in some part with carbon emissions credits. Recycling of other non-combustible goods (metals, plastics, etc.) would likely coincide with energy recovery and could also provide additional potential revenue.

Conclusions

When considering the throughput flow of biomass through national economies, MSW management strategies follow a predictable development path as per capita wealth increases. This study demonstrates that the per capita MRB

generation rate is non-linear, and moreover does not increase indefinitely as per capita wealth increases. MSW service develops rather early as an economy develops. Recycling comes later, with the municipalities generally developing programs to recycle wood products after developing strategies recover other forms of MSW. Per capita collected MRB rates first increase (as MSW collection service becomes more available), then decrease (as discard rates decline and recovery rates increase) as wealth increases. Still, overall throughput is the main factor that determines the absolute amount of per capita MRB in each country.

Given pollution is effectively regulated, it is sensible to convert this MRB into energy rather than bury it in a landfill. Doing so provides non-seasonal electricity or fuel to urban areas, reduces net greenhouse gas emissions, and reduces land pressure for waste disposal sites and for land to grow other biomass energy feedstocks. Utilization of MRB is one of the least expensive methods of reducing carbon emissions as it cuts methane emissions from landfills while at the same time generates energy that can serve as an alternative to energy from fossil fuels. However, a climate policy that prices carbon is necessary to reach full implementation of MRB energy. The energy value alone only provides enough economic incentive to reach about 60% utilization by the end of the century, although this estimate used a single global cost curve – regional values are needed for an improved estimate. The added incentive of a carbon price from a global climate policy encourages MRB energy implementation to increase rapidly, reaching full potential within the next few decades.

Acknowledgements

This work was funded in part with support from the US Department of Energy's Great Lakes Bioenergy Research Center, the Department of Energy's Office of Science, the Electric Power Research Institute, Chevron, and ExxonMobil. Research was conducted at the Joint Global Change Research Institute (JGCRI), a joint collaboration between the Pacific Northwest National Laboratory and the University of Maryland, managed by Battelle.

Chapter 8: Conclusions and Future Perspective

On June 26, 2009, the United States (US) House of Representatives passed bill HR2454: American Clean Energy and Security (ACES) Act; the so-called "Climate Change Bill". Currently, the bill is still being debated in the US Senate. This bill aims to cut greenhouse gas emission by roughly 80% by 2050. It would establish a Climate Registry, maintained by the Environmental Protection Agency (EPA) to monitor and record emissions from the various industries and other entities within the US. Verification of emissions data, particularly at the level of detail required for a cap-and-trade mechanism to function fairly, will be a formidable challenge.

It is perceived that the passage of ACES will improve the likelihood that the US would become a signatory to the successor of the Kyoto Protocol to the United Nations Framework Convention on Climate Change. This international agreement functions similarly to the US ACES bill, in that it seeks to reduce emissions through a cap-and-trade mechanism. Most signatory countries have committed to emissions reductions of roughly 5-8% under the baseline 1990 levels. However, using national energy inventories, greenhouse gas emissions inventories have, at best, uncertainties within this range. For other countries, such as China, the world leader in industrial greenhouse gas emissions, the uncertainty is much higher (15-20%, as reported in Chapter 3). Globally, we are now in a period where the uncertainty in global emissions is increasing, as developing countries (which generally do not have the level of detail in their national energy statistics that developed countries do) comprise

an ever larger portion of the total share of global emissions. While these countries are currently not under an obligation to reduce emissions under the Kyoto Protocol, this provision is being reconsidered in drafting Kyoto's successor. Without reliable, consistent, independent, honest, and verifiable emissions inventories, legislation and international agreements seeking to reduce anthropogenic greenhouse gas emissions become untenable.

Unfortunately, on February 29, 2009, the Taurus rocket carrying the Orbiting Carbon Observatory (OCO) failed to achieve orbit. This was a National Aeronautics and Space Agency (NASA) mission to monitor carbon emissions from space. While the satellite likely would have not provided a level of detail necessary for monitoring entity-specific emissions, it would have provided an independent check at aggregate scales via a process similar to the ones used in Chapters 2-4 of this dissertation.

Regardless of emissions accounting methodologies, part of the strategy to reduce emissions is transforming the energy sector to low carbon sources of fuel. The US government has set aggressive targets for production of cellulosic ethanol by 2022 in the Energy Independence and Security Act (EISA) of 2007. The feedstock for this likely will be made in large part by agricultural residue, as it is currently much cheaper to produce than dedicated bioenergy crops. In addition, residue biomass requires no conversion of land, which when counted in greenhouse gas inventories makes dedicated bioenergy crops much less attractive in terms of carbon balance.

If the price of enzymes continues to decrease, cellulosic ethanol technology will be economically feasible in the near future. This being the case, it is likely that

biorefineries will be optimized to take advantage of local residue feedstocks and will perhaps seasonally adjust the enzymes to most efficiently convert residue biomass into ethanol. Growers likely will employ simulation models such as EPIC to determine the most sustainable management practices and residue removal rates for their particular fields. A market for agricultural residues may encourage greater adoption of no-till and conservation practices, as farmers attempt to not only maximize crop yields, but maximize a sustainable residue harvest as well. Better use of municipal waste streams and more efficient waste-to-energy systems would also be a way to provide a source of energy.

Together, residue biomass could provide roughly 25-120 EJ yr⁻¹ of sustainable energy globally by mid century, based on the analysis above. The presence of a climate policy, by placing an additional price on carbon intensive fuels such as fossil fuels, and incentivizing bioenergy from residue biomass has a large impact on this estimate; utilization of residue biomass is projected to be much higher with a global climate policy that seeks to reduce greenhouse gas emissions. Much of this bioenergy would be in developing countries where emissions from fossil fuels are currently increasing most rapidly.

Certainly, with current global energy consumption at approximately 500 EJ yr⁻¹ (90% of which is fossil fuels) and with large-scale biofuels still more expensive than fossil fuels, bioenergy from residue biomass and municipal solid waste is not the sole solution to the energy-environment-economy trilemma (Figure 3). Nevertheless, bioenergy from residue biomass and municipal solid waste represent a possible option

for a sustainable energy alternative to fossil fuels. How we power our future economy will depend not only on science and technology, but also upon what policy decisions and priorities we make today.

Yes there are two paths you can go by

But in the long run

There's still time to change

The road you're on.

-Robert Plant (1948-)

(from 'Stairway to Heaven', 1971)

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