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Sustainable management of industrial capital: LCA and spatial analysis in decision making for beneficial use of industrial by -products

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SUSTAINABLE MANAGEMENT OF INDUSTRIAL CAPITAL: LCA AND SPATIAL ANALYSIS IN DECISION MAKING FOR BENEFICIAL USE OF INDUSTRIAL BY-PRODUCTS

BY

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Dedication

This dissertation is dedicated to my father, Ray Carpenter, who has been an inspiration to me my whole life. He is the one who has inspired my desire to do work that can make a difference. He also taught me to work hard and be passionate about my work and that quitting is not an option. I would not have endeavored to achieve this goal if it weren't for him.

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There are so many people who have believed in me and supported me through this time. They have all been critical in my success, each contributing in their own way.

First I would like to thank my committee members: Kevin Gardner, Jenna Jambeck, Jeffrey Melton, Scott Matthews, and John Halstead. They have kept me on track, providing guidance, and kept me from wandering off course and ending up in the research wilderness. I would also like to thank Jason Fopiano who let me borrow his Hydrus analysis and Mike Routhier who helped me sort of figure out how to make my GIS analysis work the way I needed after many months of attempting to learn how to do it myself and realizing that the DIY method was not working.

Next, I would like to thank all the friends I have made during my tenure in New Hampshire: Deana, Eileen, Buster, Scott, Tom, Linda, Irina, Sheila, Sally, and my Boston Outrigger Racing Association fellow paddlers (Jeff, Laura, Will, Joe, Scott and all others). I would like the especially thank Deana for helping me to develop a better ashtanga practice and for introducing me to Yoga East and all the wonderful people who practice there and whose positive energy has helped me stay centered and limber (or bendy as my nephew, Adam, would say).

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I want to thank all my other wonderful friends who I've collected over the years and who have never doubted my ability to do what I set out for: Anne F. has been my star cheerleader. Debbie, Kristen, Grace, Joan, Katy, Sherry, Norma. I have been lucky to have known them and to have their support.

I don't know that any family is normal and my family is no exception. However, I would not trade my family for any other in the world. They are my biggest fans, my best friends, my support, my life line: Mom, Dad, Julie, Lisa, Josh, Terry and Joe and extended families. Their support and encouragement and belief in me has been incredible and taken me through many rough times.

My advisor, Kevin Gardner, deserves special thanks. Thank you for your friendship, your mentoring and your support during my down times. Thank you for believing in my ability to do good work, for encouraging me to reach beyond myself, for all the opportunities to take incredible journeys and to meet a wide variety of experts while presenting my research.

Finally, I would like to thank Rufus, who started this journey with me, but passed on before I could finish. I miss him dearly and he will always hold special place in my heart.

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Abstract

SUSTAINABLE MANAGEMENT OF INDUSTRIAL CAPITAL:
LCA AND SPATIAL ANALYSIS IN DECISION MAKING FOR BENEFICIAL USE OF INDUSTRIAL BY-PRODUCTS

BY

Alberta Carpenter
University of New Hampshire, May 2009

The goal of this research was to broaden understanding of multiple impacts in assessing materials for construction. Life cycle assessment was used to this effect to understand the impacts from the use of industrial by-products for different applications on different spatial scales. The first two studies looked at applications in highway construction for a single project scenario and for a regional management scenario. The third study considered life cycle impacts for the management of construction and demolition (C&D) wood debris to include combustion for energy recovery. The fourth chapter reviews the literature for life cycle energy impact for building materials.

The first study found the use of bottom ash in place of crushed rock, on a regional scale, would result in a reduced energy and water consumption, reduced air emissions, reduced mercury and lead emissions and a reduced non-cancer human toxicity potential (HTP), but an increased HTP cancer due to contaminants leaching from the bottom ash into the groundwater. A fate and transport analysis however indicated that in this scenario these contaminants would not reach groundwater for over 200 years and in levels far below maximum contaminant levels.

The second study found the use of industrial by-products in combination with virgin aggregate in a regional management plan for roadway sub-base construction has lower life cycle impacts than the use of virgin aggregate alone, with the exception of HTP cancer. The HTP cancer values are highly conservative, not accounting for availability or fate and transport through sub-surface materials.

The third study indicated that combusting construction and demolition (C&D) wood for energy recovery has fewer environmental impacts than landfilling. A comparison of combustion of C&D wood versus virgin wood found the C&D wood scenario more favorable for all impacts with the exception of lead air emissions. However, lead air emissions for C&D wood still resulted in a reduction in emissions compared with the Northeast power grid.

The fourth study looked at the life-cycle energy (focused on the phases through manufacturing) of building materials, considering energy implications of recycling and material substitution, transport, and energy use compared to pre-use.

Chapter 1

Introduction

"The major clash between economics and ecology derives from the fact that nature is cyclical, whereas our industrial systems are linear. Sustainable patterns of production and consumption need to be cyclical, imitating the cyclical processes in nature."

Fritoj Capra
The Web of Life

"In an age when the speed, intensity, and complexity of change increase constantly and exponentially the ability to shape change – rather than being its victims or spectators – depends on our competence and willingness to guide the purposeful evolution of our systems, our communities, and our society."

Bela H. Banath
Designing Social Systems in a Changing World

"The world we have created today as a result of our thinking thus far has problems that cannot be solved by thinking the way we thought when we created them."

Albert Einstein

Objectives

There is increasing research being done on the beneficial use of industrial by-products. Applications have been found for the use of coal combustion by-products, foundry by-products, recycled concrete, recycled asphalt, asphalt shingles, and other secondary materials in construction applications. The primary concern in their use has revolved around groundwater contamination due to leaching of heavy metals from the materials and regulations regarding their use have been singularly focused on the leaching impacts. However, there are many other important impacts that should be considered and would ultimately result in trade-offs when considering the use of secondary materials, with reductions in some impacts and increases in other impacts. Additionally, the impacts will occur at different spatial scales (i.e. some will be local and others will be remote or regional or global) that are also not typically considered. Again, regulations typically only consider impacts that are local, however, the regional and global impacts are still occurring, damaging the ecosystems and affecting populations. This represents one of the biggest challenges in the permitting of recycled materials use: stewards of groundwater have a singular mission, as do those designing a roadway or permitting air emissions. The goal of this research was to provide a holistic approach to assessing the beneficial use of secondary materials that considers a broader range of impacts across different scales.

Dissertation organization

This dissertation is a compilation of three separate self contained articles that have been submitted for publication to different professional journals and a literature review. Each article and the review is presented in separate chapters covering a life cycle assessment comparison of materials in a specific roadway scenario (chapter 2), a regional level life cycle assessment comparison of virgin aggregate and a combination of regionally available industrial by-products (foundry sand, foundry slag and coal combustion products) (chapter 3), the life cycle assessment of the use of construction and demolition wood in combustion as an energy source (chapter 4) and a white paper considering the embodied energy of building materials (chapter 5).

Chapter 2 uses a life-cycle assessment (LCA) framework to characterize comparative environmental impacts from the use of virgin aggregate and recycled materials in roadway construction. This study concludes that the trade-offs can potentially be heavily in favor of the use of secondary materials. Additionally, while groundwater emissions can be significant, these are based on the very conservative toxicity characteristic leaching procedure (TCLP). When incorporating a more detailed consideration of contaminant fate and transport, the groundwater emissions from the use of secondary materials can be insignificant, but will vary from one location to another depending on several different variables such as precipitation rate, soil type, infiltration rate, and depth to groundwater.

Chapter 3 broadens the analysis from Chapter 2 to the regional level, modeling the impacts of regionally-managed use of secondary materials and, impacts on the materials flow of the region, impacts from transportation and the cost differential. On a regional level, transportation can be minimized by utilizing secondary materials that are generated within the region reducing the demand on virgin materials, making them more available for other demand applications and reducing demand from outside the region. Disposal costs for secondary materials can also be minimized with respect to transportation and landfilling.

Chapter 4 considers the life cycle impacts associated with the management of construction and demolition (C&D) wood debris for energy recovery. This analysis compares the various New Hampshire C&D wood management scenarios which look at combustion, recycling and landfilling along with assessing different power grid offsets, transportation distances and landfill gas management alternatives. A comparison of combustion of C&D wood against virgin wood is also made. When including the entire life cycle of the management processes, which include impacts that are both locally and regionally generated, the option with the least impacts becomes apparent.

Chapter 5 is a white paper that reviews the literature looking at the embodied energy (which includes extraction of raw materials, transport to manufacturer, processing into an end product and transport of the end product to the distribution center) of building materials. It considers the impacts of the use of recycled materials, material substitution, material reuse, transportation and the difference between the embodied energy (pre-use) and the operational energy. Overall, operational efficiency is where the greatest savings will occur, however, with increases in building efficiencies; the embodied energy of the materials will become more important. The use of recycled materials or alternative materials with lower embodied energies can be significant. In addition, the transport of these materials can also be significant, especially when the materials are being transported from across the globe.

Chapters 2, 3 and 4 have been published or submitted for publication in peer-reviewed journals. The appropriate citations are as follows:

Chapter 2:

Carpenter, A.C., K.H. Gardner, J. Fopiano, C.H. Benson and T.B. Edil. Life cycle based risk assessment of recycled materials in roadway construction. *Waste Management*, 2007, 27, 1458-1464.

Chapter 3:

Carpenter, A.C. and K.H. Gardner. Use of industrial by-products in urban transportation infrastructure: argument for increased industrial symbiosis. Submitted to *Journal of Industrial Ecology* September 2008.

Chapter 4:

Carpenter, A.C., J. Jambeck, K.H. Gardner and K.A. Weitz. Life-cycle assessment of construction and demolition derived biomass/wood waste management. Submitted to *Environmental Science & Technology* February 2009.

CHAPTER 2

LIFE CYCLE BASED RISK ASSESSMENT OF RECYCLED MATERIALS IN ROADWAY CONSTRUCTION

ABSTRACT

This paper uses a life-cycle assessment (LCA) framework to characterize comparative environmental impacts from the use of virgin aggregate and recycled materials in roadway construction. To evaluate site-specific human toxicity potential (HTP) in a more robust manner, metals release data from a demonstration site were combined with an unsaturated contaminant transport model to predict long-term impacts to groundwater. The LCA determined that there were reduced energy and water consumption, air emissions, Pb, Hg and hazardous waste generation and non-cancer HTP when bottom ash was used in lieu of virgin crushed rock. Conversely, using bottom ash instead of virgin crushed rock increased the cancer HTP risk due to potential leachate generation by the bottom ash. At this scale of analysis, the trade-offs are clearly between the cancer HTP (higher for bottom ash) and all of the other impacts listed above (lower for bottom ash). The site-specific analysis predicted that the contaminants (Cd, Cr, Se and Ag for this study) transported from the bottom ash to the groundwater resulted in very low unsaturated zone contaminant concentrations over a 200 year period due to retardation in the vadose zone. The level of contaminants predicted to reach the groundwater after 200 years was significantly less than groundwater Maximum Contaminant Levels (MCL) set by the U.S. Environmental Protection Agency (US EPA) for drinking water.

Results of the site-specific contaminant release estimates vary depending on numerous site and material specific factors. However, the combination of the LCA and the site specific analysis can provide an appropriate context for decision making. Trade-offs are inherent in making decisions about recycled versus virgin material use, and regulatory

frameworks should recognize and explicitly acknowledge these trade-offs in decision processes.

Keywords: recycled materials, roadway construction, life cycle assessment, contaminant transport, risk determination.

INTRODUCTION

There are approximately 6.4 million km of roadway in the U.S. that are being repaired every 2-5 years and replaced every 20-40 years (Transportation of the United States, 2006). The U.S. uses approximately 1.2 billion Mg of natural aggregate every year, 58% of which is used in roadway construction (Ewell, 2004). Approximately 90% of the aggregate used in roadways is virgin (636 million Mg). This equates to approximately 99 Mg of aggregate per km of roadway. While the U.S. is not currently suffering from a lack of natural aggregates, there are regions of the U.S. where natural aggregates are not as readily accessible and where the cost is higher due to transportation requirements. Furthermore, it is becoming harder to open new quarries, which increases the cost and transportation requirements for virgin aggregate.

The U.S. annually generates approximately 88 million Mg of coal ash (bottom and fly) of which 41% are recycled or reused in a wide variety of applications from concrete, structural fill and pavement to waste stabilization (American Coal Ash Association 2004). The remaining 53 millions Mg of coal ash are landfilled. Aside from the cement and concrete applications, coal combustion products (CCPs) can be used for structural fills or embankments, soil stabilization, stabilization of waste materials, flowable fill and grouting mixes, and mineral filler in asphalt paving (American Coal Ash Association, 2003). A recent survey revealed that a primary reason that recycled material use in the US is limited is concern over environmental impacts (ASTSWMO, 2000). This manuscript explores the environmental impacts from the use of coal ash, and puts these impacts in the context of other systemic impacts that result from the choice to use or not use a recycled material to replace a virgin material.

One significant aspect influencing the economic and environmental impact of high-volume material use is transportation from place of generation to application. The majority of power plants are generally located in areas of high population density, where there is an increased electricity demand. Figure 2.1 demonstrates that in the state of Wisconsin, the majority of the population lives in the southeastern portion of the state, and there is a strong correlation between population density and power plants (U.S. Census Bureau, 2005). This suggests that the majority of coal ash will be generated in areas of higher population density and higher infrastructure demand.

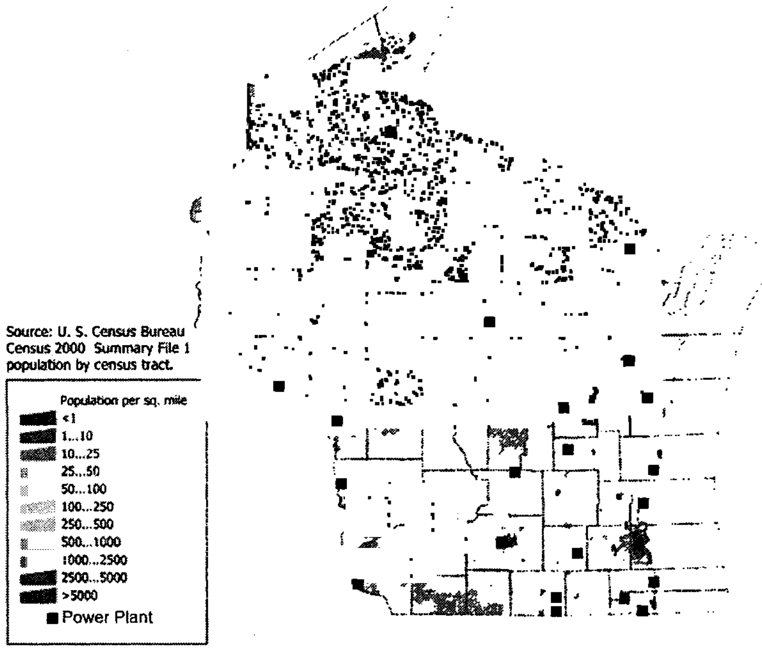

Figure 2.1: Location of power plants with respect to population densities in Wisconsin.

Virgin aggregate, aside from being a non-renewable resource, is energy intensive to produce and has significant associated environmental impacts. The use of the industrial by-product in place of virgin aggregate, aside from reducing aggregate mining and associated environmental impacts, reduces the need to landfill industrial by-products, which can be costly due to tipping fees and utilization of landfill space. The Robinson et al. (2001) study of the Mid-Atlantic region indicated that the greatest deficiency (deficient is defined as not being able to meet 2/3 of the aggregate needs of the region) of aggregate materials occurs in high population density regions, possibly due to resulting higher infrastructure needs. This results in a need to transport aggregate from a source outside that county or region equating to a significant transportation requirement. For Wisconsin, almost every county in the state has some level of sand and gravel or crushed stone production (Ewell, 2004). However, as the Robinson study proved, the higher density regions do not have the natural aggregate production capacity to meet their needs. These aggregate needs could potentially be supplemented or replaced by recycled materials.

The use of coal ash in place of natural aggregates is common in concrete construction and is accepted as having minimal risks by regulators in this consolidated state. There are commonly used American Association of State Highway Transportation Officials (AASHTO) and Association of State and Territory Solid Waste Management Officials (ASTSWMO) specifications established for its use in concrete. The use of coal ash in unconsolidated fill is still a point of concern due to potential impacts from leaching of contaminants out of the recycled materials into the groundwater. The US EPA recommends using precautionary measures when utilizing coal combustions products (CCPs) in the unconsolidated form, to ensure that there are no adverse impacts on ground or surface water (U.S. EPA, 2005).

Modeling tools have recently been developed to predict contaminant transport associated with the use of secondary materials in the highway environment (Apul et al. 2005). Through the application of these tools in regional, state or site specific scenarios, risk analyses can be performed and put into context with other existing or occurring contaminant transfer situations that can assist regulators in making realistic determinations of the risk in using the secondary materials. Information from a life cycle assessment (LCA) can also be useful to consider how impacts differ from use of recycled materials compared to virgin materials. The combination of a life cycle impact assessment, which can be viewed as a macro-scale (regional/national) assessment of environmental costs and benefits related to recycled materials use, and a micro-scale (site-specific) risk assessment can provide a unique perspective that may be useful in considering trade-offs associated with recycled material use. The question for a regulator may then become "which impacts provide a greater risk to human health, the regional or national scale impacts or the site-specific scale impacts?" The answer can help regulators to make better informed decisions regarding the use of recycled materials and allow them to explicitly consider off-site impacts in their decisions.

ISO (1997) defines LCA as "studying the environmental aspects and potential impacts throughout a product's life cycle (i.e. cradle-to-grave) from raw material acquisition, through production, use and disposal" (figure 2.2). In each phase of the life cycle an

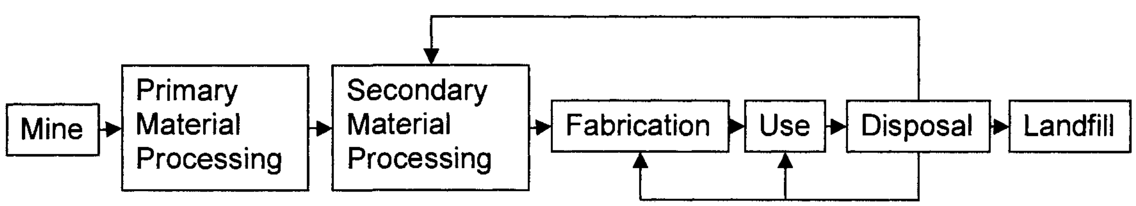

Figure 2.2. Generalized life cycle process flow diagram (Hendrickson et al., 2006).

energy and materials balance is conducted to determine all the inputs and outputs for the product. ISO 14040 (1997) developed a framework for how to conduct LCAs (figure 2.3). The goal scope and definition include the boundaries of the study as well as the functional unit, the impacts to be assessed and how they are determined. The inputs and outputs (emissions and resources) for all the life stages are quantified in the inventory analysis. These inputs and outputs include resource use (materials, energy, water) and

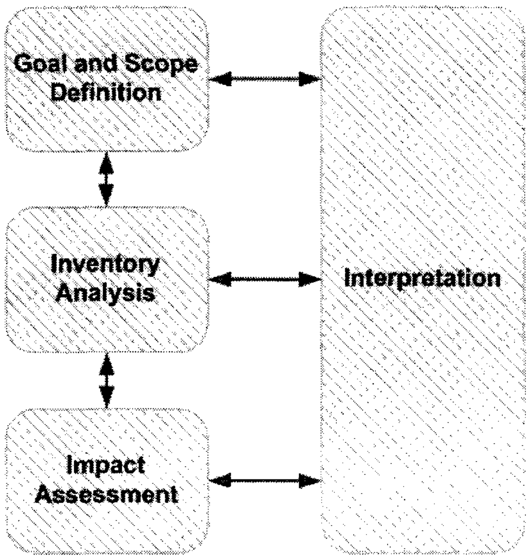

Figure 2.3: ISO 14040 framework. (ISO 1997)

emissions (to air, water and land) as well as the products or by-products of any of the processes. The impact assessment phase included the characterization, normalization, and weighting of the aggregated inventoried data. The interpretation phase allows for the determination of whether the LCA study goals were met, any sensitivity and uncertainty analysis and to determine what can be learned from the study.

LCA allows for the analysis of the environmental impacts for a product or process on a larger scale to determine environmental and economic costs and impacts from cradle to grave. While the most obvious advantage of this type of analysis is to see the most apparent cost savings over the entire life cycle of a product or process, the other advantage is that the environmental impacts of a product or process can be assessed.

Based on these impacts the product or process can be modified to reduce the impacts; or a separate product can be compared to determine which has a lower cost or fewer or less severe impacts. The scope of the LCA can be defined to fit the type of analysis desired. Roth and Eklund (2003) define four levels of system boundaries to define an LCA specifically for road construction: 1) the material level; 2) the road environment; 3) the road environment plus transport and pre-treatment of materials; and 4) industrial system level. The industrial system level is comprehensive to include mining and production of materials, material processing, transportation, manufacturing of necessary equipment, administrative processing, product assembly, distribution, sale, use, repair, and ultimate disposal and looks at overall environmental impacts. This is a very data intensive and complex analysis. The road environment level allows the comparison of environmental performance of different materials. Using LCA for analysis of materials in roadway construction, the immediate impacts may be more of concern and this will allow the user to narrow down the scope of the LCA to those aspects that have an immediate affect on the local area. This would include the road environment and transport and could be of use to local regulators who need to assess the local impacts from a particular roadway construction and the use of the recycled materials. The transport factor would be included in this assessment since it can have impacts on the surrounding community.

Several studies are available that utilized LCA for roadways (Mroueh el al. 2001, Stripple 2001, Park et al. 2003, Birgisdottir 2005, Olsson et al. 2006, Carpenter et al. 2007). Mroueh et al. (2001) conducted an LCA of the use of industrial by-products in roadway construction and included the life phases that were relevant to the comparison of the different materials (excluding use and maintenance of the roadway). The generation of the IBP was excluded as the materials were considered waste and had no economic value. The impacts assessed were resource use (energy, natural materials, IBPs), air emissions (CO_2, NO_X, SO_2, VOCs, PM), emissions to the ground (heavy metals, chloride, and sulphate) and other loadings (noise, dust and land use). Stripple (2001) conducted a pilot study to compare asphalt and concrete roadways. The study included different methods of roadway construction, low emission and normal vehicle comparisons as well as the disposal (removal or reuse of materials) of a roadway over 40 years. The impacts considered were energy use and NO_X, SO_2 and CO_2 emissions. However, no alternative materials were considered in this study. Olsson et al. (2006) conducted an LCA on a roadway utilizing MSW incinerator bottom ash as a replacement for aggregate in the sub-base of the roadway and followed the boundary guidelines recommended by Mroeuh et al (2001). Additionally, Birgisdottir (2005) conducted an extensive LCA of roadways incorporating the use of MSW ash as an alternative material in Danish roadway construction. This study assessed a range of environmental impacts (potential for global warming, acidification, nutrient enrichment, stratospheric ozone depletion, photochemical ozone formation, human toxicity, ecotoxicity and stored ecotoxicity and included landfilling impacts, roadway repairs and maintenance. These studies provide useful information on the life cycle impacts of specific lengths of different types of roadways in full construction.

MODELING TOOLS

The purpose of this paper is to demonstrate the utility to decision makers of conducting LCA alongside site-specific risk characterization. In order to accomplish this task for a road construction scenario, two modeling tools were used. Pavement Life Cycle Assessment Tool for Environmental and Economic Effects (PaLATE) considers materials, designs parameters, equipment and maintenance and cost inputs and provides a full life cycle costs and environmental assessment. It can be considered a semi-industrial system level analysis (it does not include the impacts from generating the recycled materials) and provides estimates of life cycle air emissions, contaminant releases, water and energy consumption and cancerous and non-cancerous human toxicity potentials (HTP). PaLATE is a hybrid model utilizing the U.S. Department of Commerce census data based EIO-LCA (CMU-GDI, 2002) and process data from a range of sources (USEPA, OECD, equipment manufacturers, and research from leaching studies and transportation studies). Data sources are listed on a reference tab in the excel program (available on-line at http://www.rmrc.unh.edu/tools/tools.asp) (Horvath, 2004). As the U.S. Department of Commerce data is highly aggregated based on the sectors of the economy, this does add some uncertainty to results based on that data.

HTP is a normalized risk factor reflecting the potential harm that a chemical can cause when released into the water or air environment, based on its toxicity and the potential dose (Hertwich et al., 2001). It is a mid-point indicator that aggregates emissions from toxic chemical releases into the environment, assesses the potential fate and transport of the chemicals through different exposure pathways and environmental compartments.

HTP is not an indicator of actual effects, but rather potential effects as a scientific basis for comparison of products (Guine'e and Heijungs, 1993); it determines potential impact to human health in terms of benzene equivalents emissions for cancer and toluene equivalent emissions for non-cancer. There is significant uncertainty still in this assessment method due to uncertainty in the data regarding potential dose and toxicity parameters. The methods for calculating the HTPs are provided in detail in Hertwich et al. (2001).

HYDRUS2D, a finite element modeling program for simulating the movement of water, heat, and multiple solutes in variably saturated media, was used to model the site-specific impacts of the use of recycled materials (Simunek et al., 1999).

SCENARIO

The scenario used in this paper is based on portions of a field scale project, constructed along a highway in Lodi, Wisconsin, that used multiple industrial by-products for roadway stabilization (Edil et al., 2002). The project constructed several sections of roadway using different recycled materials in the road sub-base as well as a control section using crushed rock. The recycled materials used in the project were coal fly ash, coal bottom ash, foundry slag and foundry sand; the physical description of the roadway scenario is described in table 2.1 and figure 2.4. This paper analyzes only the effects of using bottom ash (obtained from Alliant Energy's Columbia Power Station, Columbus, WI), since the leached metals concentrations were higher for this material than the other recycled materials. Each section of roadway had two equally sized (3.5 m X 4.75 m) lysimeters (one on the shoulder line and one at the center line) underneath the test sections to determine the quantity and concentration of leachate being generated (Edil et al., 2002).

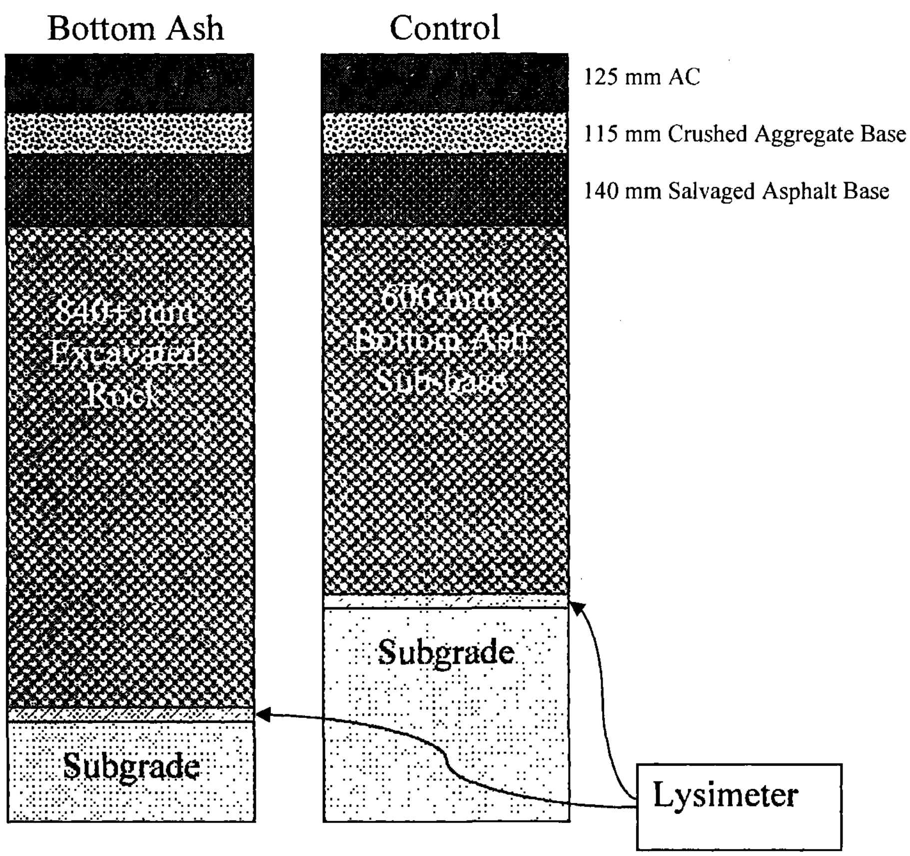

Figure 2.4: Physical description of roadway scenario.

Length	305 m
Pavement width	10.4 m
Shoulder width	1.5 m
Base and stabilized subgrade width	13.4 m
Depth of vadose zone	6 m

Table 2.1: Physical description of roadway scenario (Edil et al. 2002).

The scenario parameters were entered into the PaLATE and Hydrus2D programs to predict long term impacts from the use of bottom ash in the sub-base of a road. The PaLATE program evaluated the impacts from the use of bottom ash to replace crushed rock in the sub-base, and the material source distances were varied to observe the relative significance of the impacts from transportation.

The Hydrus 2D simulations used the average concentrations of Cd, Cr, Se and Ag in the leachate collected from the bottom ash section of the University of Wisconsin project for Monitoring and Analysis of Leaching from Sub-bases Constructed with Industrial Byproducts (Sauer et al., 2005). Using the Hydrus2D default parameters for silty loam, US EPA partition coefficients for metals analyzed, and the infiltration rate observed by the University of Wisconsin project team (table 2.2), the model predicted transport through the sub-grade to groundwater assumed to be located 5 meters below the test sections, over a range of time up to approximately 200 years. The sub-surface material was assumed to be a silty loam, based on USGS reports.

Infiltration rate	0.026 cm/day
Cd K_d	501.2
Cr K_d	6.3
Se K_d	20
Ag K_d	398.1
Depth of vadose zone	5 m

Table 2.2: Hydrus2D model parameters

PaLATE RESULTS

In comparing the PaLATE results for virgin material (crushed rock) with bottom ash at equivalent source distances, in almost all impact categories, bottom ash has significantly less impact than crushed rock (table 2.3). The exceptions are SO_2, with negligible difference, and HTP Cancer, where crushed rock has significantly less impact than

Material (transportation distance)	**Bottom Ash (80km)**		**Virgin Material (80km)**		**Virgin Material (160km)**	
	Mat Prod	Trans	Mat Prod	Trans	Mat Prod	Trans
Energy [GJ]	1,299	606	2,684	686	2,684	1,315
Water [kg]	234	103	427	117	427	224
CO_2 [Mg]	56	45	154	51	154	98
NO_X [kg]	581	2,413	778	2,731	778	5,239
PM_{10} [kg]	409	470	1,815	532	1,815	1,021
SO_2 [kg]	36,269	145	36,365	164	36,365	314
CO [kg]	139	201	268	228	268	437
Hg [g]	1	0	1	0	1	1
Pb [g]	43	20	72	23	72	44
RCRA HazW Gen [1000 kg]	8.1	4.4	9.7	4.9	9.7	9.5
HTP (Cancer) (1000)	258	13	154	15	154	28
HTP (Non-cancer) (1000)	581	15,933	146	18,035	146	34,592

Table 2.3: LCA results from PaLATE analysis for material production (mat prod) and transportation (trans) impacts for use of bottom ash (source distance = 80 km) and virgin materials (source distance = 80 and 160 km).

bottom ash (figure 2.5). Figure 2.5 also presents impact ratios for the case when virgin materials have twice the haul distance; increased transportation distane has the greatest effect on HTP Non-cancer and NO_X emissions. The impact ratios for these factors decrease significantly, indicating an increase in HTP Non-cancer and NO_X emissions with the increase in transportation distances (figure 2.5). SO_2 emissions show negligible impact from transportation and all other factors show slight decreases in impact ratio. For this scenario, with the exception of HTP Cancer, the impacts due to bottom ash are less than the impacts from the use of virgin materials. The HTP Cancer impacts, conversely, are approximately 38% greater for bottom ash than for virgin materials. This

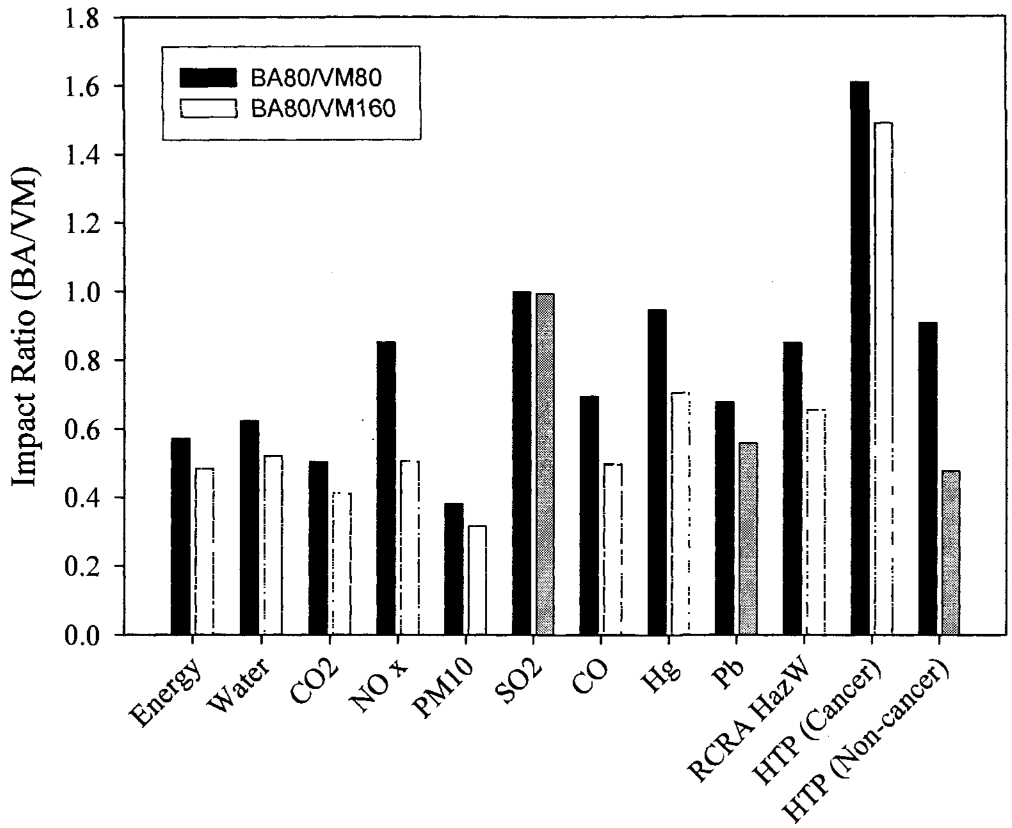

Figure 2.5: Ratio of impacts from use of bottom ash (BA) in roadway construction compared to virgin materials (VM): BA source at 80 km, VM source at 80 and 160 km. Ratios less than 1.0 indicate that impacts due to virgin material are greater than impacts due to bottom ash. The black bar indicates the ratio of impacts for materials sources at equal distances. The grey bar indicates the ratio of impacts for materials with the source for virgin materials being twice that of the bottom ash.

increase is due to potential impacts from heavy metals in the bottom ash leaching into groundwater. For this specific case, the virgin material is crushed dolostone rock, which has a negligible potential risk to groundwater. The HTP cancer levels calculated by PaLATE indicate that some virgin materials, such as limestone, siliceous gravel and siliceous sand have equivalent HTP cancer levels as bottom ash. This is primarily due to the concentrations of arsenic in these materials. Arsenic is the main contributor to the HTP cancer for the water compartment and these materials contain similar concentration levels of arsenic (Sauer, et al., 2005).

HYDRUS RESULTS

Because of the bottom ash scenario had significantly greater HTP cancer levels for the calculated by PaLATE, a closer examination of risks associated with this pathway was warranted. Hydrus 2D simulations were run to predict contaminant transport through the subsurface material (vadose zone) to the groundwater. The simulations assumed the same type of engineered highway with the specifications of the Wisconsin case considered previously. The results the indicated that Se and Cr leached from the bottom ash used in the sub-base of the road will not reach the groundwater located 5 meters below the surface even after 200 years. Figures 2.6 and 2.7 show the Hydrus2D simulations for Cr and Se transport from beneath the bottom ash layer through the vadose zone to the groundwater table located 5 meters below. The figures demonstrate that the aqueous concentrations of Cr and Se drop dramatically over time and with depth. Simulations for Cd and Ag (not shown) predicted several orders of magnitude less concentration than for Cr and Se. The simulations predict that none of the contaminants will achieve significant concentrations (relative to the US EPA MCL concentration) in the groundwater after the 200 years (see table 2.4). It is important to note that the significant vadose zone depth at this particular site has a significant influence on the modeling results; the impact of the groundwater table can be seen in figure 6 by observing concentrations at shallower depths.

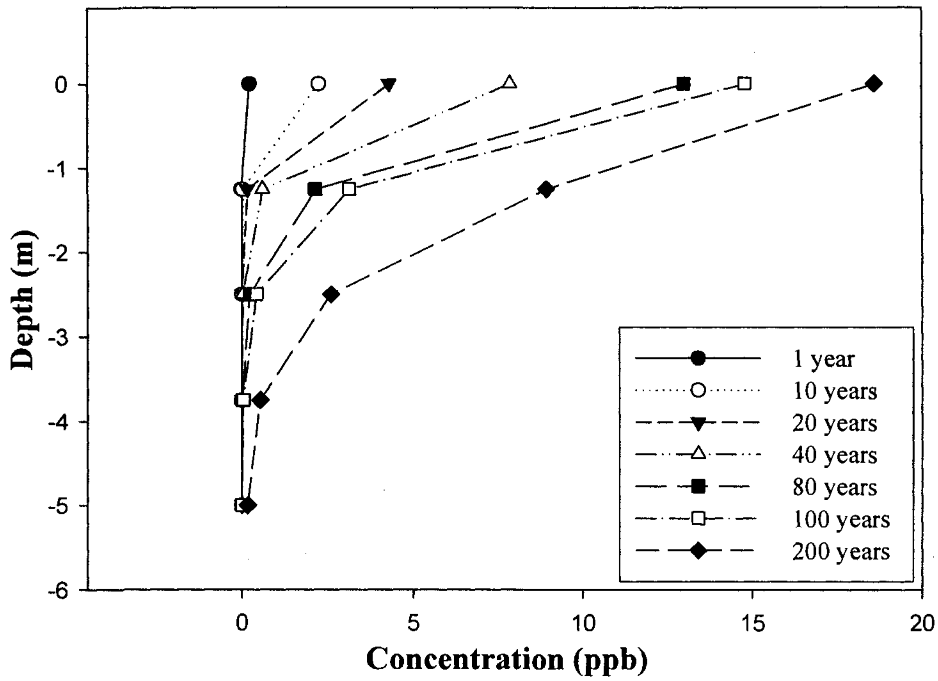

Figure 2.6: Hydrus2D simulation for transport of Cr from beneath the recycled materials layer in the road sub-base to groundwater (5 meters below the recycled materials layer) over 200 years.

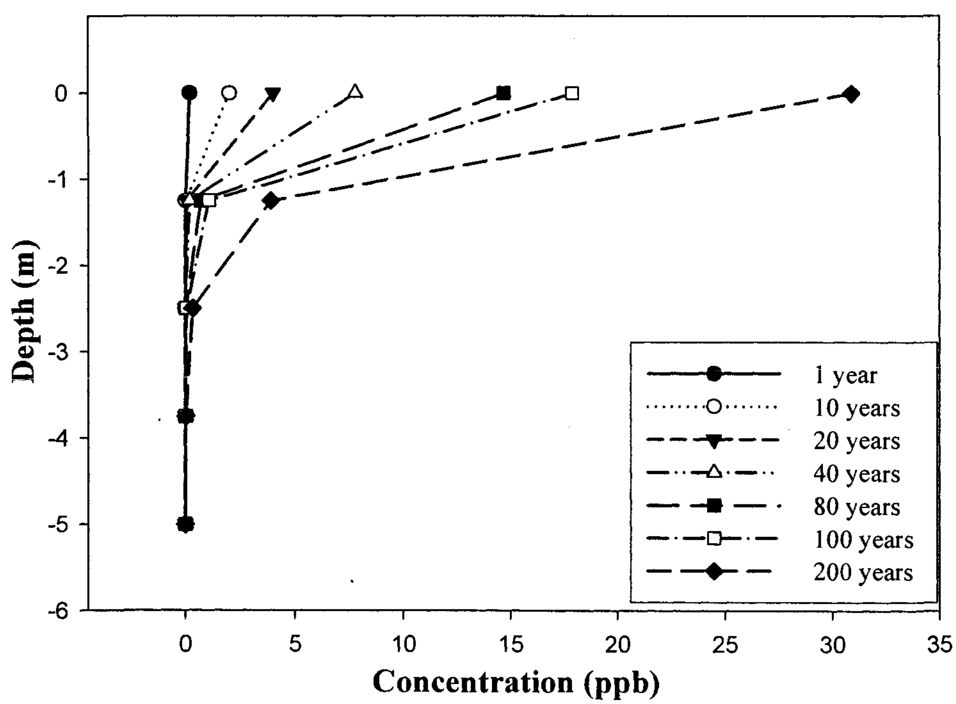

Figure 2.7: Hydrus2D simulation for transport of Se from beneath the recycled materials layer in the road sub-base to groundwater (5 meters below the recycled materials layer) over 200 years.

Metal	PaLATE (Morse, 2001)	UWisc data (Sauer et al, 2005)	MCL for groundwater (USEPA, 2003)	Hydrus 2D prediction - 200 yrs
Cd	< 1.0 ppb	21.2 ppb	5 ppb	2.60e-10 ppb
Cr	10.60 ppb (sd 4.34)	15.1 ppb	100 ppb	0.171 ppb
Se	<25.0 ppb	41.2 ppb	50 ppb	2.24e-3 ppb
Ag	None	11.8 ppb	100 ppb*	2.60e-10 ppb

Table 2.4: Metal leaching concentrations from bottom ash. Secondary MCL standard (USEPA, 2003)*.

COMPARISON OF RESULTS

HTP results from PaLATE are derived from leaching potential of materials and average heavy metal concentrations. Table 2.4 provides tabulated data for the metal concentrations used in the PaLATE program calculations, data collected by the University of Wisconsin project, the US EPA MCLs and the concentrations predicated by the Hydrus2D simulations. The data used in the PaLATE program came from a study by Morse et al. (2003) using materials collected in southern United States (NM, TX, OK, and LA). The Morse study metal concentrations were determined by synthetic precipitation leaching procedure (SPLP) (EPA SW-846 Method 1312). Data collected by the University of Wisconsin and used in the Hydrus 2D simulations are greater than the data collected by Morse et al., and furthermore, simulations were conducted with a constant flux boundary condition, meaning that leachate concentrations were assumed to be constant over the 200-year period. Both of these indicate a certain level of conservatism, as studies have shown decreases in leachate concentrations over time (Sauer et al., 2005).

HTP values are based on the potential leaching concentration of the metals in the materials and does not account for the retardation of contaminants in the sub-surface materials, which acts to prevent significant transport to the groundwater over very long time frames and which reduces peak concentrations reaching the groundwater. Further research would be required to quantify the extent of the uncertainty associated with the HTP values. The Hydrus 2D simulations do account for transport through the sub-surface and the chemical and physical reactions that occur to reduce contaminant flux, the resulting degradation of groundwater resources and associated human health risks.

The predictions for contaminant concentrations in the groundwater below a 5 meter vadose zone after 200 years are shown in table 2.4. The maximum concentration just above the groundwater table after 200 years is 0.171 ppb for Cr and 0.002 ppb for Se, both significantly below the groundwater MCLs for those metals (table 2.4) (U.S. EPA, 2003).

DISCUSSION

The PaLATE results have a degree of uncertainty associated with it due in part to the highly aggregated nature of the EIO-LCA data source. Uncertainty also arises from the fact that the data is from 1997 and is strictly based on only the U.S. economy. Due to the overall uncertainty associated with the PaLATE results (as illustrated in figure 2.5), results with ratio difference greater than 50% should not be considered significant. However, the results do provide a reasonable indicator of the potential impacts associated with the scenarios.

The two simulations combined indicate that using bottom ash in place of crushed rock, on a regional or national scale, would result in a reduced energy and water consumption, reduced CO, CO_2, NO_X, SO_2 emissions, reduced mercury and lead emissions and a reduced non-cancer HTP. It would, however, result in an increased cancer HTP due to contaminants that may leach from the bottom ash into the groundwater. Making a direct comparison of HTP cancer to HTP non-cancer would require some subjective valuation which can vary depending on the population, the location and the existing environmental conditions.

The HTPs calculated by PaLATE for this scenario are a summation of risk factors for all the contaminants in a material in water and the potential harm that can be caused when all of the contaminants leached from a recycled material reach the groundwater. The Hydrus2D simulations, however, indicate that the contaminants leached from the recycled material might never reach the groundwater at any significant level, suggesting the risk associated with this particular use is quite small from this exposure pathway.

In the United States, a regulatory body currently is likely to only consider the potential impact to the groundwater. However, in the case study provided, trade-offs associated with coal ash use are significant, particularly in comparison with predicted groundwater impact. National or regional level regulators may use this type of analysis to encourage the use of bottom ash; in the case study shown, an increase in cancer HTP could be considered a reasonable trade-off for a reduction in energy and water consumption, air emissions, mercury and lead emissions and non-cancer HTP. The Hydrus 2D results reveal that the HTP impacts, which are specific to the locality, would not be realized for well over 200 years, and at levels that would still be significantly below groundwater MCLs.

There are additional factors that may be considered important to consider in this type of analysis that were not considered here. For example, using the recycled materials saves non-renewable resources and disposal of recycled materials in landfills has real environmental and economic costs, additional trade-offs not considered in this analysis.

The analysis conducted here demonstrates the importance of considering a broad range of environmental and economic impacts when establishing policies and regulations. Regulations in the US are segmented, sometimes referred to as "stove-pipes" for their lack of ability to mix with other types of regulations. Explicit consideration of environmental and economic trade-offs associated with a policy or decision requires the ability to consider how a decision or policy may influence other, perhaps seemingly disconnected, areas of the environment or economy. Environmental regulations may be broadly described as being designed to protect the environment. The analysis provided in this paper shows that a more holistic and multi-scale analysis may be most appropriate for determining whether decisions or policies accomplish that. In the case study described, it is clear that significant environmental trade-offs and small risk reduction rewards would result from a decision allowing recycled materials use in place of virgin aggregate.

Appendix: PaLATE parameters

Input parameters:

For initial construction and maintenance:

For wearing course 1/2/3, sub-base 1/2/3 and embankment/shoulder:

- For asphalt pavement materials, concrete pavement materials, and sub-base & embankment construction materials
    - volume (yd^3)
    - density ($tons/yd^3$)
    - one-way transport distance (miles)
    - transportation mode (dump truck, tanker truck, rail, barge)
- Equipment selection (drop down menu selection) for:
    - concrete paving
    - asphalt paving
    - cold in place recycling
    - full depth reclamation
    - hot in place recycling
    - rubblization
    - milling
    - concrete demolition
    - crushing plant
    - excavation placing and compaction
    - tire recycling
    - glass recycling
    - HMA production

Output Environmental parameters:

For Initial Construction and Maintenance:

For Materials Production, Material Transportation and Processes (Equipment):

- Energy (MJ)
- Water consumption (kg)
- CO_2 (Mg) = GWP
- NO_x (kg)
- PM_{10} (kg)
- SO_2 (kg)
- CO (kg)
- Hg (g)
- Pb (g)
- RCRA Hazardous Waste Generation (kg)
- Human toxicity potential (cancer)
- Human toxicity potential (non-cancer)

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CHAPTER 3

USE OF INDUSTRIAL BY-PRODUCTS IN URBAN TRANSPORTATION INFRASTRUCTURE: ARGUMENT FOR INCREASED INDUSTRIAL SYMBIOSIS

ABSTRACT

The incorporation of roadways into a region's industrial ecology may be an efficient method of managing some of the industrial by-products (IBPs) that are generated. Current management of these industrial by-products is through beneficial use (for certain types of materials), but also stockpiling or landfilling, which have economic and environmental implications. This article considers the Pittsburgh urban regional aggregate demand for both vertical and horizontal infrastructure, and compares the use of IBPs (e.g. coal ash, foundry sand and slag) and virgin aggregate with virgin aggregate alone for use as base material for roadway construction in an optimization analysis to minimize the transportation impacts. The life cycle impacts associated with the choice of material (virgin or IBP) are also evaluated in this article, and it is shown that IBP usage results in lower life cycle impacts in almost all categories. Additionally the transportation costs are 25% less for the combined IBP and virgin aggregate usage than for the use of virgin aggregates alone due to the closer proximity to the source materials. The combination of reduced economic and environmental costs provide a strong argument for state transportation agencies to develop symbiotic relationships with large IBP producers in their regions to minimize impacts associated with roadway construction and maintenance with the additional benefit of improved management of these materials.

INTRODUCTION

The concept of industrial ecology "requires that an industrial system be viewed not in isolation from its surrounding systems, but in concert with them. It is a systems view in which one seeks to optimize the total materials cycle, from virgin materials, to finished material, to component, to product, to obsolete product, and to ultimate disposal. Factors to be optimized include resources, energy, and capital" (Graedel and Allenby, 1995). In order for human industrial systems to be sustainable, they need to be modeled after natural systems, in which waste is all reusable. Industrial symbiosis is directly related to industrial ecology and is concerned with the flow of energy and materials through regional economies; collaboration opportunities offered by geographic proximity is important and allows the user to avoid the high costs and impacts of transportation (Chertow, 2000). The by-products from one industry should be able to serve as a resource for another, ideally adjacent, industry (figure 3.1). Roadways are an integral part of any region's infrastructure and in this context can be considered an industry, albeit a dispersed one. Roads are needed to move supplies and people and as an industry, the construction and maintenance of roadways is highly resource intensive. The demand for roadways increases in high density regions and with the increased roadway demand comes an increased demand for resources and a concomitant deficiency in regional natural resources to build and maintain them (Robinson et al., 2001). The most transparent impacts from roadways are due to the materials required (mining and processing), the construction, the transportation required to import materials and the use of the roadway over its lifetime. Some impacts that are not so apparent are utilization and reduction of non-renewable resources and end of life disposal. Indirectly, landfilling of industrial by-products that are not reused can also be considered an impact.

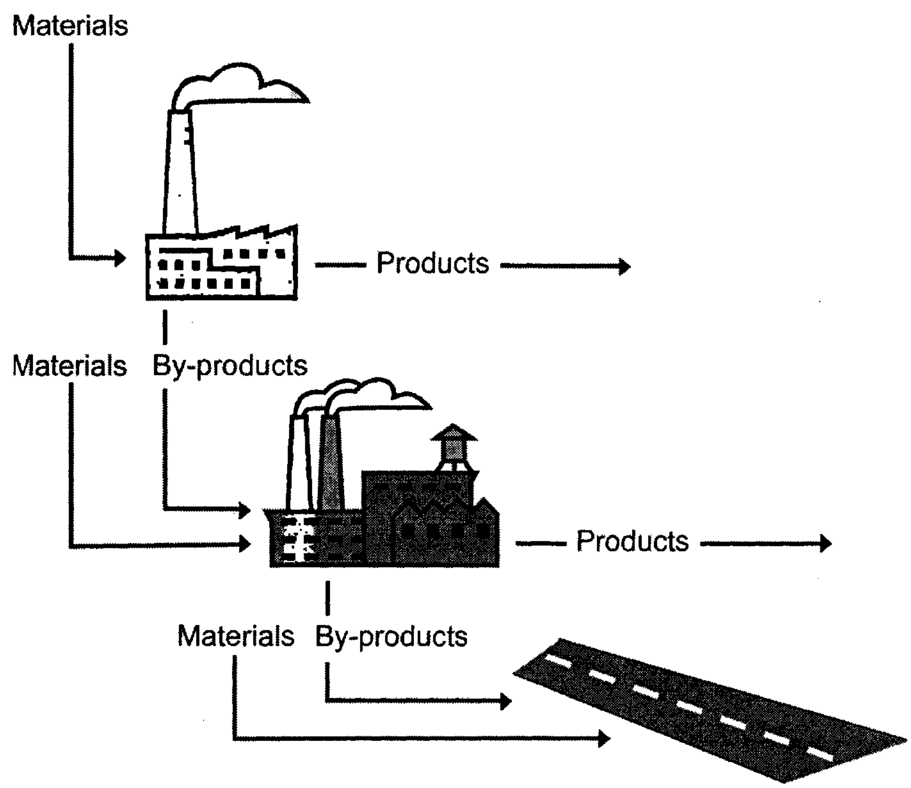

Figure 3.1. Simplified ideal material flows within the industrial ecosystem should attempt to utilize the by-products from one industry as the source material for another. Roadways can be considered an industry within the industrial ecosystem, utilizing the by-products of adjacent industries as a source material.

Incorporating roadways into the industrial ecology of a region requires shifting it from an open loop system that utilizes virgin resources and then disposes of them at the end of their life cycle, to one that utilizes secondary materials for maintenance and reconstruction. In road construction and maintenance, some use of virgin resources may always be necessary, but the aim should be to minimize the amount required, and with that the impact from their use. Utilization of industrial by-products (IBPs) helps to minimize impacts from mining and processing of virgin materials and disposal of IBPs. Additionally, as industry is generally located in urban regions, the by-products are closely located to areas with higher roadway infrastructure needs and transportation of building materials can be minimized. Different types of roadways will have different lifetimes, therefore the type of roadway will determine how long the IBP material will potentially be in place; as of this point, there does not appear to be any data to indicate that the use of IBP in roadway structures reduces the longevity of the structures.

Research has been conducted to investigate not only the physical properties of secondary materials for roadway construction, but also the leaching properties (Kosson et al., 2002; Carpenter et al., 2007). Some EU countries have maximized their utilization of recycled materials by using landfill and resource extraction disincentives and other initiatives. The U.S. has more recently adopted these practices and in some regions is just beginning to utilize secondary materials for roadway construction (table 3.1). Without the pressures of minimizing landfill waste, or reduced access to virgin materials, there has not been a significant driving force to increase recycling rates farther in the U.S. Without regulatory incentives for beneficial use of IBPs or disincentives to landfilling in the U.S., there currently is only a market driven incentive. This means the materials will be used if the material is more readily accessible than virgin materials, if users are experienced and comfortable with handling the materials, and if they are proven to be an acceptable risk to the environment.

	USA		Germany		Denmark		Netherlands	
	% Reused	Qty Reused (MMT)	% Reused	Qty Reused (MMT)	% Reused	Qty Reused (MMT)	% Reused	Qty Reused (MMT)
BFS	90%	11.40	100%	8.30	n/a	n/a	100%	1.20
Steel Slag[1]	67%	8.70	92%	4.40	100%	0.06	100%	0.50
CBA	31%	4.00	96%	2.70	100%	2.00	100%	0.08
CFA	27%	13.20	87%	2.70	100%	1.06	100%	0.85
C&D waste[2]	25%	31.00	n/a	n/a	n/a	n/a	100%	9.20
MSW ash	n/a	n/a	69%	1.80	n/a	n/a	100%	0.80
RAP	81%	30.00	55%	6.60	100%	0.48	100%	0.10
RCM	n/a	n/a	n/a	n/a	81%	0.86	n/a	n/a

Table 3.1: Quantity and percentage of recycled material usage for the USA, Germany, Denmark and the Netherlands. [1] USA values estimated from 2005 USGS Minerals Yearbook and American Iron and Steel institute 2006 Statistical Report. [2] USA values based on USEPA estimates of 2.3 lbs/day per cap (1998) and 2005 population census. Recycling rates estimated at 20-30% by USEPA.
Note: BFS = Blast Furnace Slag; CBA = Coal Bottom Ash; CFA = Coal Fly Ash; C&D = Construction and Demolition; MSW = Municipal Solid Waste; RAP = Recycled Asphalt Pavement; RCM = Reclaimed Concrete Material; n/a = data not available.

By-products from road maintenance and construction have been used in road construction and an extension of this is to generate roads from waste of other industrial processes thereby including the construction of roads in a larger industrial ecosystem. This allows roadways to minimize their demand for natural capital (virgin materials).

This article evaluates the incorporation of roadways into a regional industrial ecosystem and compares the combined use of recycled materials (IBPs in this study) and virgin aggregate to virgin aggregate alone in the construction of the roadways. The study includes aggregate demand from not only roadways, but also from vertical infrastructure (i.e. buildings) demand. In this study, the vertical infrastructure demand utilizes only virgin aggregate. A spatial analysis was conducted to simulate the use of the materials for "projects" (simulated roadway construction sites) in the closest proximity to the source and to compare the life cycle impacts as well as the transportation costs; the distribution of the vertical infrastructure demand was assumed to be the same as for the roadway demand.

Several studies are available that utilized life cycle assessment (LCA) for roadways (Mroueh el al., 2001, Stripple, 2001, Park et al., 2003, Birgisdottir, 2005, Olsson et al., 2006, Carpenter et al., 2007). Mroueh et al. (2001) conducted an LCA of the use of industrial by-products in roadway construction. This article included the life phases that were relevant to the comparison of the different materials (excluding use and maintenance of the roadway). The generation of the IBP was also excluded as the materials were considered waste and had no economic value. The impacts assessed were resource use (energy, natural materials, IBPs), air emissions (CO2, NOX, SO2, VOCs, PM), emissions to the ground (heavy metals, chloride, and sulphate) and other loadings (noise, dust and land use). Stripple (2001) conducted a pilot study to compare asphalt and concrete roadways. The study included different methods of roadway construction, low emission and normal vehicle comparisons as well as the disposal (removal or reuse of materials) of a roadway over 40 years. The impacts considered were energy use and NO_X, SO_2 and CO_2 emissions. However, no alternative materials were considered in this study. Olsson et al. (2006) conducted an LCA on a roadway utilizing bottom as from a municipal solid waste incinerator. The MSW ash was utilized as a replacement for aggregate in the sub-base of the roadway. The Olsson study followed the boundary guidelines recommended by Mroeuh et al (2001). Additionally, Birgisdottir (2005) conducted an extensive LCA of roadways incorporating the use of MSW ash as an alternative material. This study assessed a range of environmental impacts (potential for global warming, acidification, nutrient enrichment, stratospheric ozone depletion, photochemical ozone formation, human toxicity, ecotoxicity and stored ecotoxicity. These studies provide useful information on the life cycle impacts of specific lengths of different types of roadways in full construction. This current article differs in that it is not connected to a specific length of roadway and it but instead looks at the regional level use of aggregates (natural or alternative) for sub-base construction in roadways. The scope of the LCA fairly narrow, but allows the focus to remain on the aspect of regional management of IBP materials.

RECYCLED MATERIALS AND APPLICATIONS

A wide variety of recycled materials may be used in roadway construction. Reclaimed asphalt pavement (RAP) and reclaimed concrete material (RCM) are the most widely used recycled materials. Other recycled materials used include: slag, coal combustion products, foundry sand, asphalt shingles, reclaimed concrete aggregate, amongst a variety of other products. Their use is dependent upon their material properties and the environmental conditions; some standards have been developed by ASTM and AASHTO to ensure the quality of the products is adequate. Guidelines have been developed for the use of different industrial by-products in different applications and provides information on specifications and material qualities (RMRC, 2008). The materials cannot be used in direct substitutions as their material properties are not exactly the same. Primarily the use of IBPs is dependent upon whether the material properties meet the material specifications (i.e. plasticity, shear strength, compaction, drainage and durability) (USDOT, 2004). Different states within the U.S. have different regulations concerning the type of uses allowed for industrial by-products (ASTSWMO, 2006). Whereas the quantity of RAP, and steel and iron (BFS) slag that are recycled is high (table 1), that is not the case for other materials. This analysis considers the environmental and economic impacts from the use of slag, coal ash and foundry sand available in the greater Pittsburgh urban region. Pennsylvania allows for the use of these industrial by-products for varying applications (ASTSWMO, 2006).

Slag: A wide variety of slag is generated in the U.S., such as steel furnace slag and iron slag (also known as blast furnace slag or BFS) that can be air-cooled, granulated or palletized, and lead, copper, bottom boiler, phosphorus, zinc and foundry slag. Slag has been used in a variety of engineering applications for over a century (NSA, 2007). Some of the uses include aggregate substitution, fill material, railroad ballast and Portland cement replacement. The optimal use depends on the type of slag (typically steel or iron slag) and the process in which it was produced. The weathering process of the slag also affects the physical properties of the slag. The USGS reports that between 19 - 26 million metric tons of iron and steel slag were sold or used in the U.S. in 2006 (van Oss, 2006), however, unused portions end up being stockpiled or landfilled. In the greater Pittsburgh region alone, the Pennsylvania Department of Environmental Protection (PA DEP) recorded close to 1.9 million metric tons of material generated and stockpiled for disposal for 2003-2004 (PA DEP, 2004).

Coal Combustion Products: Coal combustion produces a variety of ash products, including coal fly ash (CFA), coal bottom ash (CBA), and flue gas desulfurization (FGD) products. Approximately 64 million metric tons of CFA is produced each year in the U.S., 15 million metric tons of CBA, and 2 million metric tons of boiler slag. Of this, approximately 25 million metric tons of the CFA, 7.2 million metric tons of the CBA, and 1.8 million metric tons of the boiler slag are recycled (40%, 47%, and 90% recycling rates, respectively (ACAA 2004)). The Pittsburgh region generates over 5 million metric tons of CCPs biannually (PA DEP 2004). The highest value use is for the high calcium CFA, which is mostly used as an additive in Portland cement concrete. Additional uses of CFA include structural fills and embankments, stabilization of soils, flowable fill and grouting mixtures. CBA and boiler slag may be used as road base material, structural fill material, for snow and ice control, and as an aggregate in asphalt pavement (more frequently in base courses).

Foundry sands: Foundry sand is a recyclable material from the metal casting industry. Production in the U.S. is approximately 5.6 to 9 million metric tons per year. The majority of foundry sand is composed of silica sand with smaller amounts of organic additives and binders (with bentonite being the most common binder). The majority of foundry sand in the U.S. comes from iron and steel casting; sands from brass, bronze, and copper foundries are generally not suitable for recycling because of their metal leaching properties. The primary use of foundry sand for construction of transportation related facilities is for construction of embankments and structural fills. It is also suitable for use in road base applications and for the stabilization of sub-base materials. It is an excellent material for use as flowable fill aggregate and hot mix asphalt (FIRST, 2006).

TRANSPORTATION COMPONENT

The Eno Transportation Foundation has tracked trucking costs from 1960 – 2001 (Eno 2002). The cost per tonne-km in 2001 was 38.3 cents/tonne-km (44.6 cents in 2007 dollars, not accounting for increase in fuel cost). Fuel costs have increased by 45% from 2001 to 2006 which would result in an overall cost of 45.6 cents/tonne-km in 2006. IBPs are generally assumed free on board (FOB) and therefore the cost is on the generator to transport the materials to the market (or to the landfill, plus tipping fees). The market for the materials must be close enough to make it more cost effective to transport the materials to a customer than to take it to a landfill due to a lack of incentive structure for reducing disposal. Roadway infrastructures exist in all areas and could provide a beneficial use application that would be close to the point of production of the IBPs. In a regional context, municipality, city and state governments could look to optimize the use of the IBPs in their region, thus allowing them to minimize the cost and environmental impact from the mining and extraction of virgin aggregates that would otherwise be utilized for their roadway construction, repair and maintenance projects. Some virgin aggregate would likely still be required as the IBP generators would likely not be able to generate sufficient quantities to meet the region's entire aggregate demand (Robinson et al., 2001). Additionally, some IBPs are usable for certain applications (i.e. Portland cement concrete) (RMRC, 2008). For sub-base applications, as used in this study, depending on the supply, some regions may be able to meet all their roadway needs.

In comparing the two types of materials (virgin aggregate and IBPs), the virgin aggregate supplier would typically mine and process the aggregate and have it available at the processing site. The user would pay for the aggregate, plus transportation to the construction site. For IBPs, the suppliers provide the IBP material and transportation to the construction site (at some maximum distance) or pay to transport and dispose of the IBP at a landfill; typically, the IBP users save on cost of the material as well as cost of transporting it to the construction site.

Lack of experience in the use of the IBP materials can be a deterrent as it brings uncertainty for the user. Contractors have to understand the physical properties of each of the different materials available in their region and how to handle and apply them to their projects. Use of IBPs may require different techniques during construction. Additionally, lack of readily available information on the quantities and properties of IBPs being generated in a particular region prevents users from taking advantage of the potential cost saving of the use of the IBP materials. Additionally, some IBPs have unique engineering properties that must be understood if the user is to obtain maximum performance and thus maximize values. An IBP/recycled materials exchange for different urban regions would help to inform the market of the availability of the materials.

METHODOLOGY

This study looks at the relation between urban aggregate demand for vertical and horizontal infrastructure and industrial by-product availability for Pittsburgh. The 2007 Pennsylvania Department of Transportation (PENNDOT) aggregate demand for the Pittsburgh region was obtained. The Pittsburgh region (figure 3.2) was defined as a square block extending 50 miles out from the Pittsburgh downtown (defined as the city center for this analysis). The portion of the block extending outside of the Pennsylvania state line is excluded.

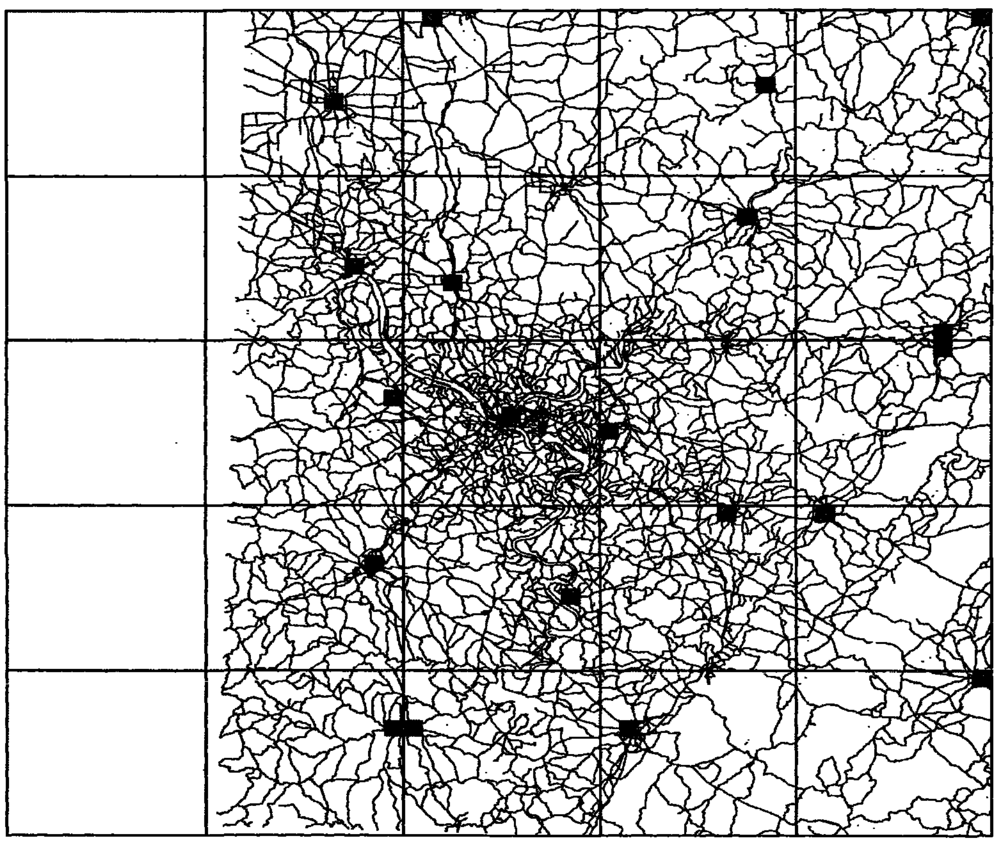

Figure 3.2. Highway density map of greater Pittsburgh urban region, excluding out of state roadways. High density points are marked as black squares for a grid of twenty 20 X 20 mile blocks (PENN DOT, 2008).

Using GIS data for PENNDOT roadway systems (PENNDOT, 2008), a grid was overlaid onto the Pittsburgh region and broken into twenty 32X32 km large blocks. Each large block was further broken down into 3.2X3.2 km small blocks and the road density for each small block was calculated. The small block with the highest road density was designated as the high density "point" for each large block. The total roadway density for each large block was also calculated and the 2007 PENNDOT aggregate demand was allocated into each large block based on the block's total roadway density. Locations and aggregate generation rates were found for the PENNDOT approved aggregate sources (PENNDOT, 2008). Sources where the generation rates were not available were assumed to be 180,000 tonnes/year (based on quantities being generated by other sources). IBP sources were determined from the Pennsylvania Residual Waste Report (PA DEP, 2004), which provided locations and quantities for residual waste generators, to include generators of coal ash, foundry sand and slag. Appropriate "sources" of aggregate were determined for each large block that minimized the required distance to transport aggregate materials from "source" to "project" ("project" is high density point for each large block). This was done using virgin aggregate only (PENNDOT approved sources) for one case scenario and using a combination of IBP and virgin aggregate sources together for a second case scenario. The combined IBP and virgin aggregate usage scenario allowed for the use of whichever material was closest to the "project" and thus minimized transportation requirements. A portion of the virgin aggregate was allocated for vertical infrastructure demand for each block, thus restricting it from use for roadways. This restriction forces the analysis to assume a farther transportation requirement for materials. The scope of the assessment includes extraction, processing and production for the virgin materials, post use processing for the IBPs and construction for both types of materials. The extraction and initial use of the IBPs are not included because at this point they are considered a waste product with no economic value. If they were to develop an economic value then allocation of the impacts from their extraction and initial processing would need to be assessed. The life span of the different scenarios is considered to be equal as there is no current evidence to indicate a significant difference.

Incorporation of the vertical demand (residential and industrial construction) was based on the U.S. Census data (2002; 2006). Census payroll data for the Pittsburgh Metropolitan Statistical Area (MSA) was available for 2006 for the construction NAICS 236 (Building Construction – vertical infrastructure) and 237 (Heavy and Civil Engineering Construction – horizontal infrastructure). Value of work for these NAICS was not available for the Pittsburgh MSA, but was available on the national level for 2002. A relationship between payroll and value of work was determined based on the national level data, and used to calculate value of work for the Pittsburgh MSA for NAICS 236 and 237. Horizontal construction (NAIC 237) value of work was 51.1% of the total construction value of work. This value of work was then entered into the EIO-LCA tool (CMU-GDI, 2002) and an economic output for stone quarrying was calculated for each NAIC. USGS minerals yearbook data (Ewell, 2002) was used to disaggregate the stone quarrying economic output to provide just the sand and gravel (S&G) economic output. The value of construction gravel was $5.05 per ton (Ewell, 2002) and allowed for the calculation of the required tonnage of aggregate for horizontal and vertical construction.

The vertical demand impact was included in the transportation cost calculation, but was not used to provide environmental impacts.

The material was assumed to be used for sub-base coarse aggregate only and all other factors (construction processes, longevity, traffic loading) in a roadway design were assumed to be the same. The construction processes for the different materials are not significantly different as the materials are required to have certain material properties to meet the roadway construction standards. At this point no evidence is available to indicate that the use of IBPs reduces the life spans of roadways. The Pavement Life Cycle Assessment Tool for Environmental and Economic Effects (PaLATE) program was utilized to assess life cycle impacts for this study. The boundaries of the life cycle assessed by PaLATE include material extraction (virgin material only), material processing, transportation and construction as illustrated in figure 2.3. Data for the material quantities, type and distance transported was entered into the PaLATE program to determine differences in the LCA impacts between using virgin aggregate and IBPs in roadways on a regional level. The PaLATE program was developed by the Recycled Material Resource Center (RMRC) to assess life cycle impacts for roadways. It is hybrid model that includes economic input-output data as well as process data (Horvath, 2004). The model has not yet been formally validated but is the only existing model for U.S.

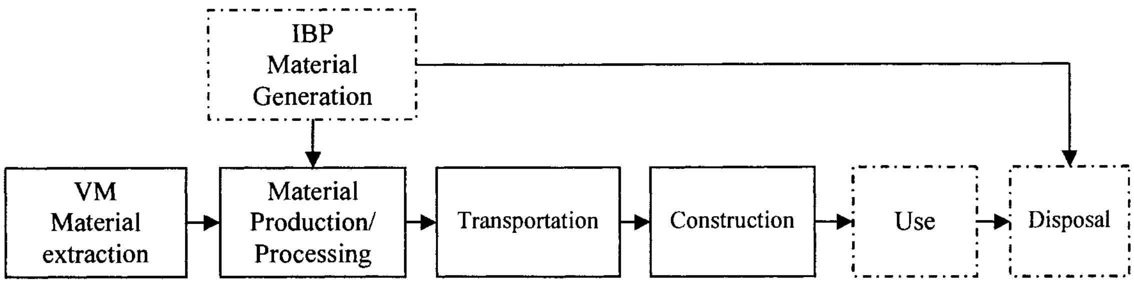

Figure 3.3. Life cycle processes and flow chart for roadway study. The dotted line processes are not included in this assessment. IBP materials are currently considered a waste product, not a co-product, therefore the initial material generation impacts are allocated only to the initial use. The use phase is not considered relevant to this study; the disposal phase is relevant, but no data is available to be included at this point.

roadways to include IBP materials. As such, it was determined to be the most appropriate tool for this assessment. The tool allows users to provide input on the specific type of roadway that is being constructed (type of materials, number and thickness and width of wearing courses and base course and embankments). The tool also allows the user to input information on the type of maintenance that may be performed. For the purposes of this study, the focus was on the impacts from the sub-base. The number and type of wearing courses were assumed to be the same for both scenarios. Additionally, the maintenance portion of the tool was not utilized as the maintenance for both scenarios was also assumed to be the same. This reasonable as the use of IBPs is permitted is different states in the U.S. and no data is currently available to indicate that the use of IBPs in the sub-base reduces the longevity of roadways. As the other parts of the roadway (embankments, wearing courses) were the same, the maintenance required for them also were assumed to be the same.

PaLATE considers materials, design parameters, equipment, maintenance and cost inputs and provides a full life cycle costs and environmental assessment on a semi-industrial system level (impacts from generating the recycled materials are not included) based on the U.S. Department of Commerce census data. The PaLATE analysis estimates impacts from energy, Global Warming Potential (GWP), carbon monoxide (CO), sulfur dioxide (SO_2), nitrogen oxides (NO_X), Particulate Matter – 10 Micron (PM_{10}), mercury (Hg), lead (Pb), Resource Conservation and Recovery Act (RCRA) Hazardous Waste, human toxicity potential (HTP) cancer and HTP non-cancer (Horvath, 2004). These impacts were selected as they were available from the EIO data. More extensive impact categories as utilized in the Birgisdottir (2005) study would be useful to provide a more comprehensive assessment; however that inventory data is not currently available for the U.S. These impacts assessed were compared for the usage of different IBPs throughout the region with the use of virgin aggregate. Person equivalents (PE) were also determined for all impacts (WRI, 2007; UNSD, 2004; USEPA, 1999; USEPA, 2005) except the HTPs (no information was available to make valid conversions for HTPs). Tonne-kilometers were also calculated for each case and the transportation cost was calculated based on 45.6 cents/tonne-km (Eno, 2002).

RESULTS

The results from data entered into PaLATE (table 2.2) indicate the use of virgin aggregates in the base course for roadway construction generates greater impacts in all the categories calculated except HTP cancer which is about 10% greater for the combined IBP and virgin material usage than for virgin material alone. The HTP cancer impacts for the IBPs are based on the leaching potential of the materials that PaLATE has allocated to the material production process. The HTP calculations, however, are highly

		Virgin Materials				Industrial By-Products & Virgin Materials			
Impact	units	Mat Prod	Mat Trans	Proc (Equip)	Total	Mat Prod	Mat Trans	Proc (Equip)	Total
Energy	TJ	274.8	73.1	7.9	355.8	131.4	38.3	8.7	178.4
Water	Mg	38.3	12.4	0.8	51.5	18.3	6.5	0.8	25.7
GWP	Gg	19.5	5.5	0.6	25.5	9.3	2.9	0.7	12.8
NO_X	Mg	39.2	291.2	12.8	343.3	18.8	152.7	14.1	185.5
PM_{10}	Mg	278.9	56.8	2.2	337.9	133.3	29.8	1.6	164.7
SO_2	Mg	19.1	17.5	0.8	37.4	9.1	9.2	0.9	19.2
CO	Mg	25.6	24.3	2.8	52.7	12.3	12.7	3.0	28.0
Hg	g	0.7	52.8	5.7	59.3	0.3	27.7	6.3	34.3
Pb	kg	5.6	2.5	0.3	8.3	2.7	1.3	0.3	4.3
RCRA HW	Mg	319.4	526.9	56.9	903.2	152.7	276.2	62.5	491.5
HTP cancer	million	33.2	1.6	0.0	34.8	38.0	0.8	0.0	38.8
HTP non- cancer	billion	467.1	1.9	0.0	469.0	272.5	1.0	0.0	273.5

Table 3.2: Regional impact values per process (material productions, material transportation and process (equipment)) and totals for the use of virgin materials and industrial by-products in roadway sub-base construction for the greater Pittsburgh urban area

conservative and do not account for availability of elements for release or sorption of the contaminants in the soil layer as the leachate moves through the vadose zone (Carpenter et al., 2007). For this study, seven of the twelve impact categories (energy, water, GWP, PM10, Pb, HTP cancer and non-cancer), the majority of the impacts are due to materials processing. The impact from equipment processes are minimal ranging from 0 – 18% of the total emissions for an impact. The NO_X and Hg impacts are mostly due to transportation, while SO_2 and CO impacts are about the same for materials processing and transportation. Knowing the primary contributor can be important to look at when trying to target certain type of impact reduction.

The PE impacts from GWP and SO_2, CO and Hg emissions are depicted in figure 3.4.

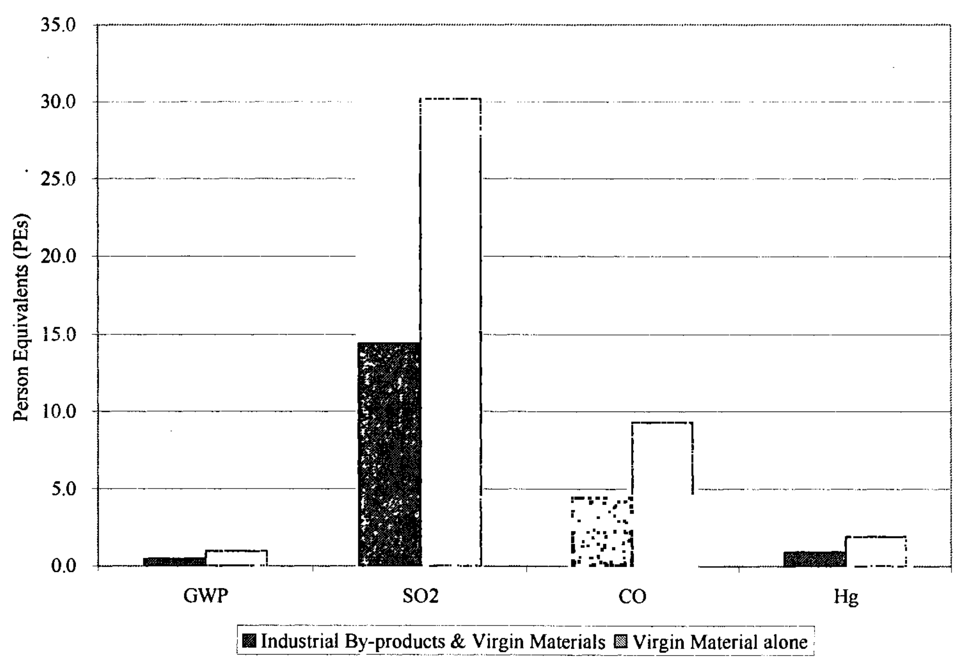

Figure 3.4: GWP, SO_2, CO and Hg emissions impacts in PEs comparing the use of virgin aggregate with a combination of mixed industrial by-products (CBA, Foundry Sand and Foundry Slag) and virgin material in roadways for the greater Pittsburgh urban area.

Impacts are greater in all categories for the scenario using virgin material alone, approximately doubling the PE impacts for the combined IBP and virgin aggregate usage scenario.

The energy consumption, NO_X, PM_{10} and Pb emissions and RCRA Hazardous Waste generation PE impacts are depicted in figure 3.5 ranging from 500 PEs (energy) to 7,700 PEs (RCRA Hazardous waste generation). Again, the impacts from virgin aggregate usage alone is approximately double that of the combined IBP and virgin aggregate usage.

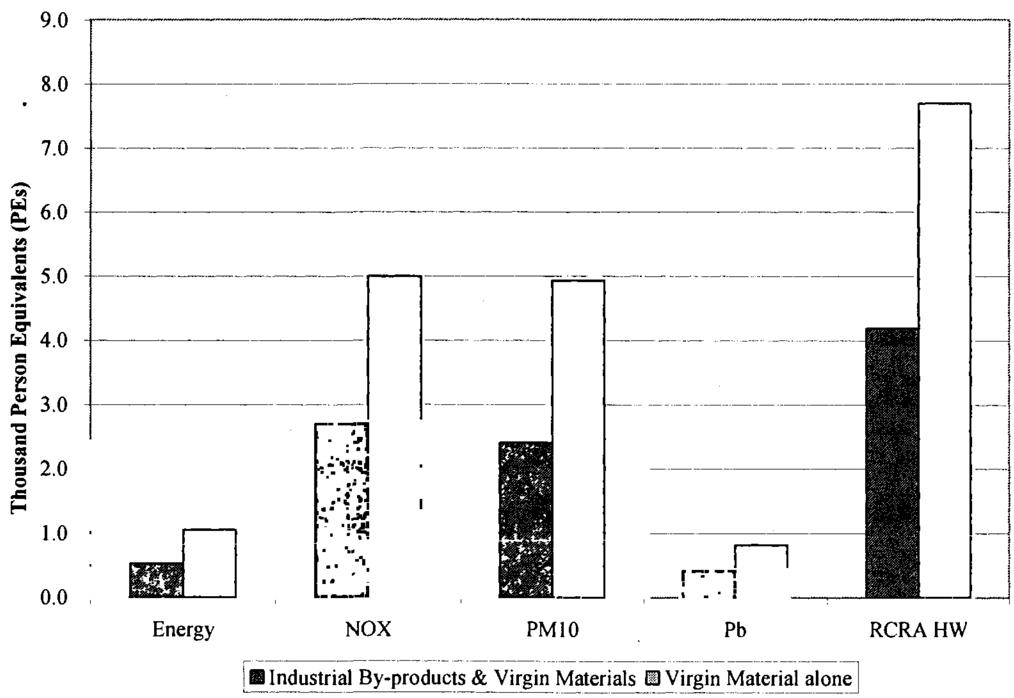

Figure 3.5: Energy consumption, NO_X, PM_{10} and Pb emissions and RCRA Hazardous Waste generation impacts (in thousand PEs) comparing the use of virgin aggregate with a mix of industrial by-products (CBA, Foundry Sand and Foundry Slag) and virgin aggregate in roadways for the greater Pittsburgh urban area.

The transportation component of this study includes a simple cost analysis based on ton-miles. The virgin aggregate scenario requires the transportation of almost 36 million tonne-km more than the combined IBP and virgin aggregate scenario for vertical and horizontal infrastructure construction (figure 3.6). At the adjusted transportation rate of 45.6 cents/tonne-km, this increased ton-mile requirement costs PENN DOT (and the tax-payers) almost $9 million over the transportation cost for the combined IBP and virgin aggregate use. When accounting for the assumed FOB delivery of IBPs, the transportation cost savings increases to $15 million.

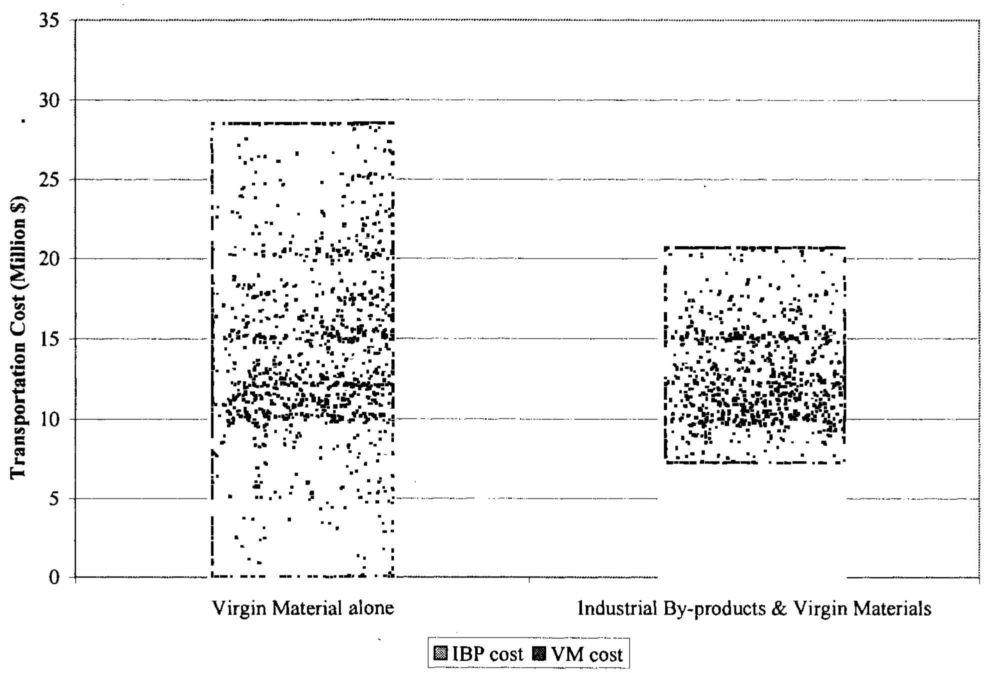

Figure 3.6: Transportation costs in millions of dollars for the use of virgin aggregate and industrial by-products in building and roadway construction for the greater Pittsburgh urban region. *Note*: IBP = Industrial By-product; VM = Virgin Material

DISCUSSION

The annual aggregate demand for the Pittsburgh region was 2.3 million tonnes, 51% of which was utilized for horizontal construction. For the mixed IBP and virgin aggregate scenario, 26% of the total aggregate demand was met by using IBP materials and the rest was using virgin aggregate.

The use of IBPs in combination with virgin aggregate for roadway sub-base construction has lower life cycle impacts than the use of virgin aggregate alone, with the exception of HTP cancer. The HTP cancer values are derived from total content of elements in the materials to groundwater and are highly conservative, not accounting for availability or fate and transport through sub-surface materials.

Comparison to previous roadway LCAs are difficult as the boundaries of the different studies vary with functional units closely tied to specific lengths of roadway. This study is regionally oriented around aggregate demand, but not connected to any specific length of roadway. Additionally, it does not include construction of other components of the roadway (embankments, wearing courses, base courses) nor does it include the use and maintenance phases. The impacts considered are similar however to other studies, with the exception of the RCRA Hazardous Waste impact. This is an impact that is specific to the U.S.

This study has several limitations that carry some uncertainty. The analysis attempts to account for vertical infrastructure aggregate demand. The data used was from sources for different years and assumes there is not a significant change from year to year. The vertical infrastructure aggregate demand data required some disaggregation and translations that also add to the uncertainty. The study also does not consider aggregate for concrete or asphalt applications in roadways. The total PENNDOT demand, however, is only 25% of the availability of IBPs as indicated in the PA Residual Waste

Report (PA DEP, 2004). The demand would be greater if all other beneficial use applications were included; however, the availability of IBPs is much higher than what was required by PENNDOT for roadway construction. In the case of the state providing incentives for the use of IBPs for state funded projects, a scenario that utilizes IBPs alone would be more applicable. The virgin aggregate scenario would still be limited as those sources have a much greater demand throughout and outside the region. The analysis done here can be considered conservative in that the virgin aggregate would be less available and would potentially have to be extracted from sources farther away.

The life cycle impacts of landfilling or stockpiling of the IBPs that are not beneficially used (defined as residual waste by the state), is not accounted for and therefore again makes this analysis conservative. The cost of transporting the IBPs to their designated disposal facility (PA DEP, 2004) would be almost $64 million (2007 dollars) on top of the cost to transport the virgin materials used in place of the IBPs. The PaLATE analysis sheds more light on the environmental impact from the use of IBPs in roadways on a regional level that can lead to expanded beneficial use of the materials. This would likely still require state incentives calling for the use of the IBP materials in state and federally funded projects, up to date information on the location and quantity of materials available and an increase in working experience in handling the IBP materials in different applications. The most ideal scenario in terms of transportation costs may include a combination of both virgin aggregate and IBPs to meet the total demand of the region for all project types, servicing projects that are in the closest proximity to the material source points. The impacts these incentives might have on the market, supply chains and relevant organizations are not covered in this study.

Urban policy makers should consider the potential benefits of recycling in conjunction with the land-use and development issues. The recycling climate can be improved by imposing increased fees for landfilling, providing economic incentives for recycled materials use, increasing the market for IBP use, educating the public to the benefits of IBP use, expanding specifications for IBP use in different applications, increasing research and development for increasing the quality of IBP for reuse, and providing information to consumers for the optimum use of IBP as an alternative for virgin materials. Establishing a program to increase the use of IBPs for state funded projects would also help to increase utilization of the IBP materials as well as provide contractors experience in handling of the IBP materials.

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Chapter 4

LIFE-CYCLE ASSESSMENT OF CONSTRUCTION AND DEMOLITION DERIVED BIOMASS/WOOD WASTE MANAGEMENT

ABSTRACT

To provide assistance in quantifying trade-offs for the management of wood derived from construction and demolition (C&D) debris in New Hampshire, a life-cycle assessment of various management options using the U.S. Environmental Protection Agency's Municipal Solid Waste Decision Support Tool was conducted. Seven different management scenarios were considered based upon the annual production of C&D debris in the state of New Hampshire, and one scenario was used to compare the combustion for energy production of virgin wood from northern New Hampshire with locally produced C&D wood. The scenarios included transport distance and various management options for C&D wood (combustion, recycling, and landfilling) as well as different types of offsets for energy production (Northeast power grid and coal combustion). Impacts were obtained for energy consumption, carbon emissions, criteria air pollutants, ancillary solid waste produced, and organic and inorganic constituents in water. These impacts were normalized by person equivalents and then ranked with each impact given equal weighting. In the ranking, all scenarios with C&D debris recycling coupled with wood waste combustion and energy recovery had lower net impacts than the others. The C&D debris recycling-only scenarios resulted in less overall impact than the disposal-only scenarios. For the disposal scenarios, the landfill gas (LFG)-to-energy scenario had fewer impacts than when the LFG is flared. The lowest impact scenario included C&D debris recycling along with local combustion of the C&D wood derived product with energy recovery providing a net gain in energy production of over 7 trillion BTUs per year, and up to a 130K tons per year reduction in carbon emissions.

INTRODUCTION

In 2003 the U.S. generated 164 million tons of building related construction and demolition (C&D) debris (U.S. EPA, 2006a). Of the total C&D debris generated, approximately 34% was wood debris (55.7 million tons per year)(Jambeck, 2004). In 2006, New Hampshire generated 702K tons per year of C&D debris of which 40% was wood (NH DES 2007) (figure 4.1). In order to quantify the environmental impacts

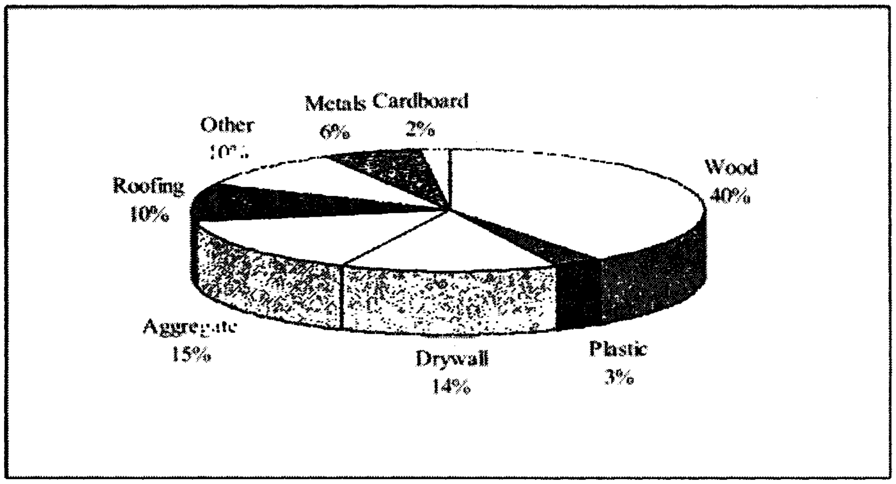

Figure 4.1. Characterization of C&D debris in New Hampshire (NH DES, 2007).

associated with the various options for management of C&D waste wood in New Hampshire, an analysis of various management scenarios was conducted. The U.S. Environmental Protection Agency's (EPA) Municipal Solid Waste Decision Support Tool (MSW DST), which employs a life cycle assessment approach, was used to analyze the impacts associated with the different scenarios. There are several options for managing wood waste material: reuse, recycling, combustion with energy recovery, and landfilling. The EPA hierarchy states generally that waste should be managed in the following order when possible (most to least preferred): 1) Source reduction and reuse, 2) recycling/ composting, 3) combustion with energy recovery and 4) landfilling and incineration without energy recovery (U.S. EPA, 2007). New Hampshire has also codified this waste management hierarchy (NH Statute 1997). For the purposes of managing wood waste materials, this hierarchy allows for the conservation of landfill space by reducing the amount of waste generated, maximizing recycling and composting, and reducing volume by combustion.

Life cycle assessment (LCA) is a method of assessing environmental impacts associated with a product or process over its entire life, from "cradle to grave". The method entails compiling an inventory of inputs and outputs for separate systems of a product or process (materials extraction and processing, production, transportation, use, disposal, etc) and combining them for a more holistic analysis. The potential environmental impacts associated with the analysis of the inputs and outputs are evaluated and interpreted relative to the objectives of the study. The LCA method is standardized in ANSI/ISO 14040 (ANSI, 1997); it allows for quantification of impacts from, and trade-offs between, various waste management options.

With carbon emissions and climate change as significant contemporary issues, the waste management hierarchy has far reaching effects beyond landfill boundaries. The Intergovernmental Panel on Climate Change (IPCC) has released its Fourth Assessment Report which justifies concern for greenhouse gas (GHG) emissions and provide some mitigation strategies. IPCC Working Group III (IPCC, 2007) has recommended the waste sector examine methane recovery from landfills, combustion with energy recovery, recycling and waste minimization. There is potential for a combination of these options to provide for greenhouse gas mitigation.

The generation of carbon dioxide (CO_2) from wood (in either a landfill or combustion facility) is considered "carbon neutral;" as CO_2 emissions are released, forests are taking up similar amounts of CO_2 (on a shorter time scale than needed to produce fossil fuels from carbon sources). This relatively quick cycling of carbon means the CO_2 emissions from the combustion or landfilling of wood are acknowledged, but not often counted in climate change calculations. According to the most recent EPA GHG report (U.S. EPA, 2006b) the GHG emissions from the combustion of wood and wood-based fuels are biogenic, so they are not included in the national emissions totals. It is assumed that any emissions from these activities are recouped with the growth of new forests and crops. The IPCC also does not count CO_2 emissions from wood combusted for energy or produced from landfills for inventory proposes (IPCC, 2006). With the carbon neutral characterization of wood combustion, wood combustion with energy recovery and carbon capture would provide a method of reducing global atmospheric carbon levels (IPCC, 2007).

Preservation of landfill space is also a benefit to landfilling wood ash as opposed to non-combusted wood. Ash takes up 80 – 90% less space in landfills, which means less leachate production and more airspace preserved for future waste disposal. Land use and conservation also has carbon offset implications as forested and undisturbed land sequesters carbon. In Europe, disposal of any wood waste in landfills is prohibited; all wood waste must either be reused or incinerated (Krook et al, 2004; Peek, 2004). In addition, the European Commission recommended that chromated copper arsenate (CCA) treated wood be subject to separate collection as a household hazardous waste and disposed via incineration (Genedbien et al., 2002).

The growth of crops/forest specifically for combustion is a form of wood fuel; however, for C&D wood, the wood is processed for construction use and either becomes scrap or used in a building's structure prior to being combusted. Carbon is stored in the wood product while it is in use. Combustion of the wood can take place at the end of the wood's life cycle (figure 4.2). Additionally, the BTU value (by mass) is greater for C&D

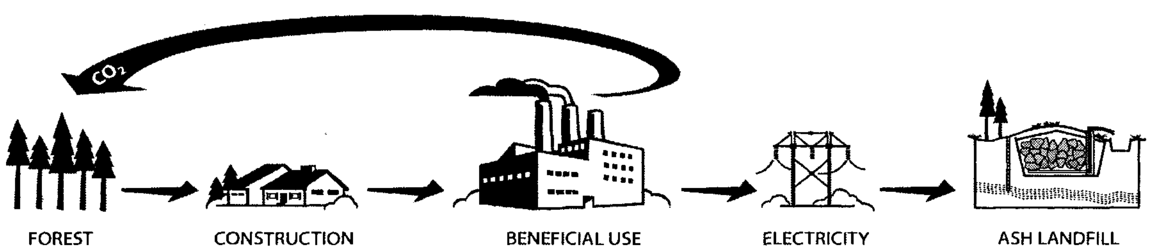

Figure 4.2. The life cycle of wood for construction can potentially be as follows: growing in the forests where the wood is harvested, processing and use in construction, incineration at a combustion facility where it is turned into electricity and ash (which is landfilled). The CO_2 that is produced during the incineration process is then ideally utilized by new trees growing in the forest.

derived biomass than for virgin wood (green timber) with C&D derived biomass having an energy value of 7,400 BTU/lb versus 2,100 (as collected) – 4,200 BTU/lb (dried) for green timber (Tchobanoglous et al., 1993). In 2004, the Northeast Sun Grant Region produced *A Strategic Roadmap for the Northeast Region on Biobased Energy and Product Technologies Fueling America's Future*, which highlights C&D wood as a current and future source of biomass energy to meet future energy needs in a more sustainable fashion (SGRI, 2004).

One of the primary concerns with management of C&D wood is that some of the wood used in various construction applications is treated with preservatives or coatings containing toxic chemicals. Historically, preservatives have included pentachlorophenol (PCP), creosote, and CCA. With the ban on the use of CCA for residential applications in 2003, other preservatives have been developed; new wood preservative treatments include Alkaline Copper Quaternary (ACQ) and Copper Azole (CBA). In addition to preservative treatments, C&D wood may also contain painted wood, which from historical applications could include lead-based paint (LBP). The percentages of this type of treated or painted wood in the C&D wood are not known on a national scale. Studies in Florida, where climate requires much of the wood to be treated aggressively, estimated the percentage of CCA-treated wood in C&D wood ranged between 8 – 22% (Tolaymat et al., 2000; Solo-Gabriele et al., 2004; Jacobi et al., 2007). It can be hypothesized that in areas of older construction, LBP percentages are higher than in areas of newer construction (LBP wood percentages will continue to decrease in the waste stream as homes are renovated or demolished). Treated and painted wood are often removed from the waste stream prior to arriving at (e.g., LBP abatement) or upon arrival at C&D processing facilities, so the percentage of treated and painted wood in the processed C&D wood itself is often less than that in the bulk collected wood. In addition, the state of Maine has fuel quality standards for C&D wood with regard to fractions of noncombustibles (<1%), plastics (<1%), CCA treated wood (<1.5%), fines (<10-20%) and asbestos (<1%) (ME DEP 2006).

The presence of various chemicals in C&D wood has been a concern when landfilling non-combusted C&D wood due to leaching of the metals into landfill leachate and/or groundwater (Jambeck et al., 2008, Jang et al., 2003; Townsend et al., 2005; Weber et al., 2002). The same contaminants that impact landfills also are concerns with utilizing C&D wood for an alternative energy source because of air emission concerns. C&D derived wood is comprised of primarily wood (greater than 85%), with other inert materials contributing a small fraction. As best available control technologies (BACT) exist for both coal combustion and municipal solid waste (MSW) combustion, preliminary investigations indicate that air emissions from combustion of C&D wood for energy meets national air emissions standards. The air emissions data from these investigations were used for inputs to the MSW DST (Humphrey, 2005; NESCAUM, 2006; Atkins, 1995).

Additional studies have looked at polychlorinated dibenzo-p-dioxin and polychlorinated dibenzofuran (PCDD/F) emissions from the combustion of C&D wood. Tame (2007) reviews different studies looking at the combustion (both industrial and domestic) of both C&D and virgin woods and the resulting formation of PCDD/F. The Tame study concluded that domestic (uncontrolled) combustion of both copper-based treated wood and virgin wood resulted in significant and equivalent levels of PCDD/F formation released in air emissions and contained in the ashes Wasson et al. (2005) conducted an open burn CCA – treated wood emissions test and found PCDD/F production was 1.7 ng Toxic Equivalent/kg of treated wood burned, a value typical for virgin wood combustion. The Freeman study (2000), which looked at C&D wood being co-combusted with coal in high temperature industrial incineration, found no increases in PCDD/F emissions over coal-only combustion. Additionally, Humphrey (2005) found that the copper from CCA-treated wood did not promote an increase in PCDD/F formation upon combustion of C&D derived biomass fuel based upon emissions tests conducted at Livermore Falls and Stratton facilities in Maine. Humphrey also summarized the results of modeling studies conducted by the Maine Department of Environmental Protection (MDEP) for their Livermore Falls and Stratton facilities. The model results indicated that the dioxin emissions would average 0.39% of the dioxin maximum ambient air guideline (MAAG) for the Livermore Falls facility and 0.22% of the dioxin MAAG for the Stratton facility.

The combustion of C&D wood evokes some concern amongst both the public as well as regulatory authorities due to concerns about impacts from air emissions. However, when making environmental policy decisions, it is important to consider all life cycle impacts. This manuscript reports an LCA conducted for New Hampshire-specific C&D wood management scenarios which was aimed at more fully understanding the impacts from various management scenarios as they are influenced by factors including landfill gas (LFG) to energy recovery, C&D wood processing, transportation and energy offsets.

METHODS

The MSW DST is a linear programming – based decision model to aid in identifying environmentally and economically efficient strategies for integrated MSW management (Solano et al., 2002a and 2002b). The U.S. EPA's National Risk Management Research Laboratory (NRML) in cooperation with RTI International and North Carolina State University (NCSU) developed the MSW DST. LCA and full-cost accounting are used to estimate the environmental and economic aspects for hypothetical integrated solid waste management alternatives (Weitz et al., 1999). The entire waste management system including waste collection, transportation, recycling, treatment, and disposal are considered in the emissions calculations which include cost and emissions of CO_2 (both wood and fossil fuel derived), nitrogen oxides (NO_X), carbon monoxide (CO), sulfur oxides (SO_X), total particulate matter (PM), carbon equivalents (MTCE), energy consumption and metals released into the environment. For recycled materials, offsets are calculated to determine savings compared to the use of virgin materials (Thorneloe and Weitz, 2004). The MSW DST model inventory data was developed by the University of Wisconsin, the Environmental Research and Education Foundation and the model and all documentation went through stakeholder review, external peer reviews as well as quality assurance and U.S. EPA administrative review (Ham and Komilis, 2003; Komilis and Ham, 1999 and 2000; Ecobalance, 1999).

The MSW DST contains life-cycle environmental data for waste collection, transport, recycling, composting, waste-to-energy combustion (WTE) and landfilling; for the production and consumption of energy for the U.S. national and regional grids; and for the production of aluminum, glass, paper, plastic, and steel (Thorneloe and Weitz, 2004). The MSW DST has a very broad scope which makes it practical for comparing the environmental impacts resulting from the management of wood waste in New Hampshire. Since the model is not yet commercially available, input data, based on the 2006 quantity and characterization of C&D debris generated in New Hampshire (NH DES, 2007) was supplied to RTI International. To better model the combustion of C&D wood, specific data related to C&D wood characterization (C&D wood composition, metal content and BTU values) were used to be able to provide output that more accurately reflected the New Hampshire based scenarios (tables 4.1 and 4.2).

Metal	Default Value in MSW DST (used for virgin wood) (lb/ton combusted)	Total Metal Content for C&D Wood (lb/ton)[1]	Value used for C&D Wood (lb/ton combusted)[2]
Arsenic	3.17E-06	7.39E-02	1.30E-04
Boron	5.45E-04	NA	NA
Cadmium	2.68E-04	1.29E-03	1.57E-04
Chromium	2.61E-04	1.10E-01	1.10E-01*
Copper	1.93E-05	6.44	6.44*
Mercury	3.94E-04	2.61E-04	1.28E-04
Nickel	3.66E-04	NA	NA
Lead	6.51E-03	5.17E-01	2.72E-02
Antimony	6.87E-05	NA	NA
Selenium	1.50E-07	BDL	BDL
Zinc	5.75E-03	NA	NA

Table 4.1. Metal Content of Virgin and C&D Wood used in the MSW DST (Quantities before air pollution efficiencies are applied) [1]The total metal values for C&D wood are the metal content based upon 10 samples from NH obtained and analyzed by the consulting firm Green Seal Environmental, 2007. [2]Some metal content would not volatilize. Volatilization percentage based upon method outlined in internal RTI document, see https://webdstmsw.rti.org/ for details. *No volatilization factor available, so total metal content used. NA=Not Available, BDL=Below Detection Limit: virgin wood values used

	Default Value in MSW DST (used for virgin wood)	Value used for C&D Derived Biomass[2]
Moisture (%)	45%	12.4%
Carbon (%)	23.3%	43.3%
Hydrogen (%)	2.9%	4.75
Nitrogen (%)	0.9%	0.4%
Sulfur (%)	0.2%	0.2%
Ash (%)	10.1%	6.9%
Oxygen (%)	17.5%	32.0%
BTU/lb	4,500[1]	7,380

Table 4.2. [1]Average reported by Tchobanoglous et al. (1993). [2]Values based upon 10 samples from NH obtained and analyzed by the consulting firm Green Seal Environmental, 2007

The metal content values for the virgin and C&D wood (Table 4.1) are not the actual emissions produced by the waste to energy (WTE) facilities, but the input values to which the air pollution control efficiencies are applied. Values for moisture, carbon, hydrogen, nitrogen, sulfur ash, oxygen content and BTU/lb for virgin and C&D wood are listed in table 2. The WTE plant heat rate was obtained from Public Services New Hampshire (PSNH) for their wood burning Schiller plant. Using the default values for branches (i.e., "clean" wood) for criteria pollutants in the model is based on the literature (*NESCAUM, 2006*).

NEW HAMPSHIRE LCA C&D WOOD WASTE MANAGMENT SCENARIOS

All scenarios begin with the assumption of a 25 mile (local) transport distance to the processing or disposal facility (no other collection is considered). All of the scenarios model management of C&D debris, except for the last scenario, which models the transport and combustion of virgin wood from northern New Hampshire for a comparison with the energy recovery combustion of C&D wood. The waste management processes that are included in the study are transport to a recycling facility, processing at the recycling facility, transportation of the C&D wood fuel, combustion at an incineration facility, transportation of ash residue to a landfill and landfilling of C&D waste not combusted. The recycling facility includes separation of wood as well as other recyclable fractions from the C&D debris (figure 4.1). The scenarios are summarized below and in table 4.3 and include some combination of these processes including generalized distances for transport.

Scenario 1 models the impact of processing C&D debris at a mixed C&D recovery facility, allowing the recovery of the wood component. The wood fraction is then transported to combustion at energy recovery facilities in Maine or Canada (with an average transport distance assumed to be 140 miles). The energy generated by the combustion of the C&D wood offsets energy otherwise generated by the NE power grid and the wood ash generated is disposed of in an ash landfill.

Scenario 2 is identical to scenario 1 except that the combustion with energy recovery is local to New Hampshire and assumed to be located at a 25 mile transport distance from the C&D recycling facility. At 115 miles less than scenario 1 and assuming 25 tons per trip transporting 280,000 tons of wood per year, the difference from scenario 1 would equate to a savings of 2.6 million miles per year. Furthermore, assuming 6 miles per gallon of diesel fuel at $3 per gallon, it could equate to a savings of approximately 429,000 gallons per year of diesel fuel and $1,300,000 per year.

Scenario 3 is identical to scenario 2 except the energy generated offsets 100% coal power only, instead of the power distribution of the NE power grid. The NE power grid has a power generation distribution as depicted in figure 4.3. Offsetting 100% coal represents the energy recovery combustion of the C&D wood offsetting the power generated at the coal power fired plants in New Hampshire.

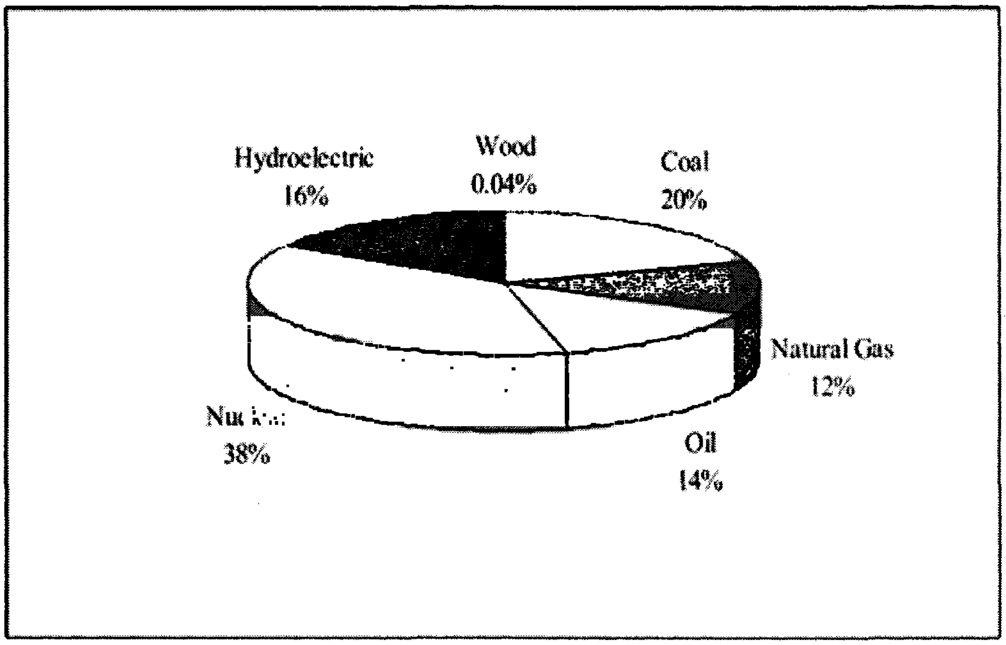

Figure 4.3. NE power grid distribution (Dumas, 1999)

Scenario 4 models the impact of processing C&D debris at a mixed C&D facility as in scenarios 1-3. The wood fraction is then transported and disposed in a local (25 miles) landfill along with the residuals. The LFG potentially generated by the C&D debris (wood fraction is the only fraction to produce methane) decomposing in the landfill is flared.

Scenario 5 is identical to scenario 4 except that the LFG generated by the C&D wood is used for energy production.

Scenario 6 models the impacts of no C&D debris separation, processing or recycling. All C&D debris is disposed in a landfill 25 miles away. The LFG generated from the C&D debris is flared.

Scenario 7 is identical to scenario 6 except that the LFG generated by the C&D debris is used for energy production.

Virgin Wood Scenario. The final scenario looks at the use of virgin wood. This scenario models the combustion with energy recovery of virgin wood collected from harvesting operations in northern New Hampshire (at a distance of 150 miles). The energy generated is offset from the NE power grid and the ash is used for some beneficial use application. In this case, to compare to a ton-to-ton basis for C&D wood, the tonnage put through the model is 280,000 tons, equal to the amount of C&D wood generated annually in NH. This scenario was also compared to the combustion with energy recovery and landfilled portions of C&D wood management, which consisted of 280,000 tons, for consistency. Neither the energy used nor the environmental implications of the production of the virgin wood (logging, chipping) is considered in this analysis.

Scenario	Transport (mi)	Recycling/ Transfer	Wood Fuel Transport (mi)	Treatment	Offset	Disposal
1	25	All materials sent to C&D recycling facility [1]	Maine/ Canada (140mi)	Combustion with energy recovery (C&D wood only)	North- east Grid	Ash LF (10mi)
2	25	All materials sent to C&D recycling facility [1]	Local (25mi)	Combustion with energy recovery (C&D wood only)	North- east Grid	Ash LF (10mi)
3	25	All materials sent to C&D recycling facility [1]	Local (25mi)	Combustion with energy recovery (C&D wood only)	100% Coal	Ash LF (10mi)
4	25	All materials sent to C&D recycling facility [1]	Local (25mi)	None (all landfilled)	None	C&D Wood – landfilled with flare
5	25	All materials sent to C&D recycling facility[1]	Local (25mi)	None (all landfilled)	None	C&D Wood – landfilled w/ energy recovery
6	25	None	None	None (all landfilled)	None	All landfilled w/ flare
7	25	None	None	None (all landfilled)	None	All landfilled w/ energy recovery
Virgin Wood (VW)	None	None	Northern NH (150mi)	Combustion with energy recovery	North-east Grid	Beneficial use

Table 4.3. Summary of scenarios input in MSW DST for NH C&D Wood Waste LCA. Collection route not considered since C&D not regularly collected like municipal solid waste. 25 mile transport distance considered from incoming C&D debris to recycling facility. [1]83% recycled and 17% landfilled with flare.

RESULTS

Since the 2006 C&D debris tonnage was used as an input into the MSW DST, the impacts and offsets associated with the results are total quantities generated on an annual basis and weighted by person equivalents (PEs; the impact annually generated per capita in the U.S. (U.S. EPA, 2002 and 2005; U.S. DOE, 2004)); national level PE data was utilized as no data was available for the state or regional level for New Hampshire. The negative values in the figures presented in this section are benefits and positive values mean energy is consumed and/or emissions are produced (i.e. a net reduction in lead emissions will be reported as a negative value).

Energy

The scenarios that include C&D debris recycling and wood combustion with energy recovery (scenarios 1 - 3) have the greatest energy savings, with slightly greater energy savings from the offset of 100% coal (figure 4.4). The principle processes contributing to the energy savings were the recycling and the wood waste to energy (WTE) processes. When the wood fraction of the C&D debris is landfilled, the recycling of the non-wood C&D debris contributes to about half of the potential energy offsets, demonstrating the energy benefits to recycling C&D debris (scenarios 4 - 5). Landfilling of all the C&D debris consumes energy with the consumption 580 PEs less when LFG-to-energy is implemented (scenarios 6 - 7).

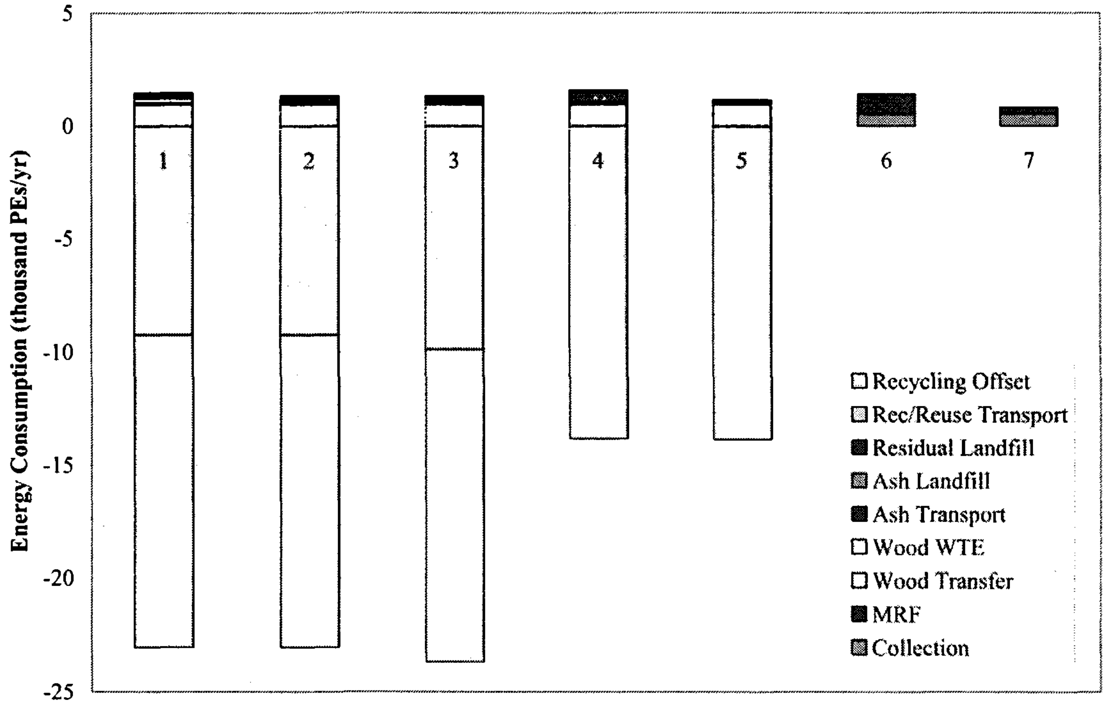

Figure 4.4. The total annual energy consumption for each management scenario is broken out for each process analyzed in this study (thousand person equivalents per year).

Carbon Emissions

Recycling and C&D wood combustion with energy recovery (scenarios 1 - 3) have the most carbon reductions. The 100% coal combustion offset (scenario 3) has the highest carbon offset (figure 4.5). Offsetting of the Northeast (NE) power grid (figure 4.3) (scenarios 1 – 2) has smaller offsets because of the lower carbon intensity of the NE power grid. The recycling and wood WTE processes offered the greatest carbon equivalent emission PE savings. Minor savings in carbon emission PEs are gained for the transportation difference between scenarios 1 and 2 (115 miles) and for the energy recovery from LFG compared to gas flaring in the 100% disposal scenarios (scenarios 6 - 7).

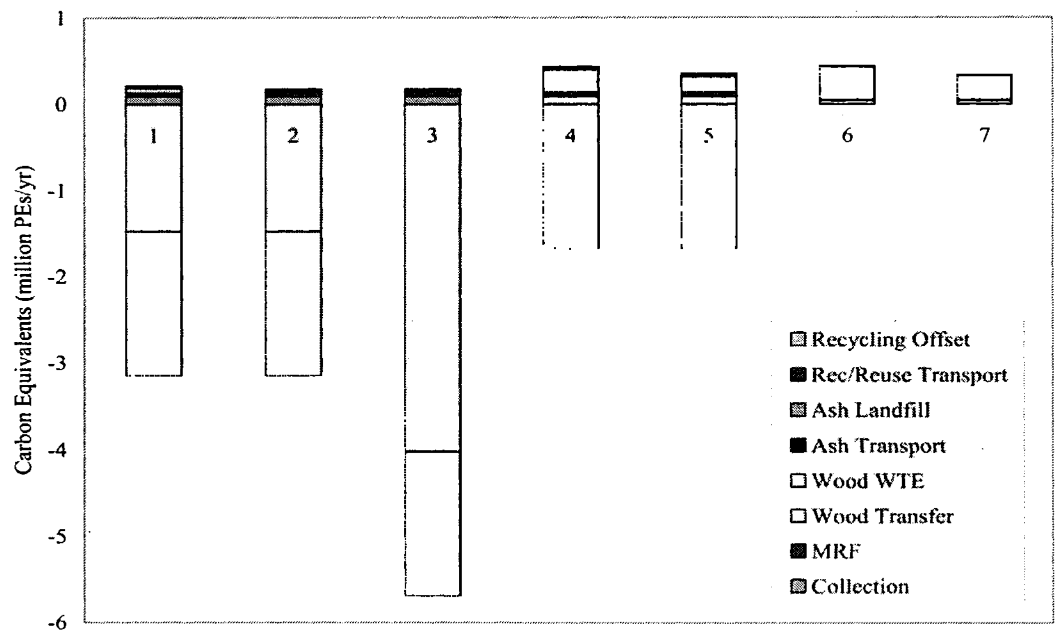

Figure 4.5. The total annual carbon equivalents air emissions for all management scenarios are broken out by process (million person equivalents per year).

Priority Air Pollutants

The greatest reduction in priority air pollutants (PM, NO_X, SO_X and CO) occurs in scenarios 1, 2 and 3 (figure 4.6). For PM emissions, combustion of the wood with energy recovery provides some additional benefit through offsetting the NE power grid (scenario 1 and 2); however a much larger benefit is shown when 100% coal is offset (scenario 3). The 115 mile transportation difference between scenarios 1 and 2 and the difference between flaring and energy recovery for the LFG (scenarios 6 and 7) is negligible. Landfilling of the C&D wood increases the NO_X emissions by 2500 PEs while all the other management scenarios have reduced NO_X emissions ranging between 900 - 1380 PEs.

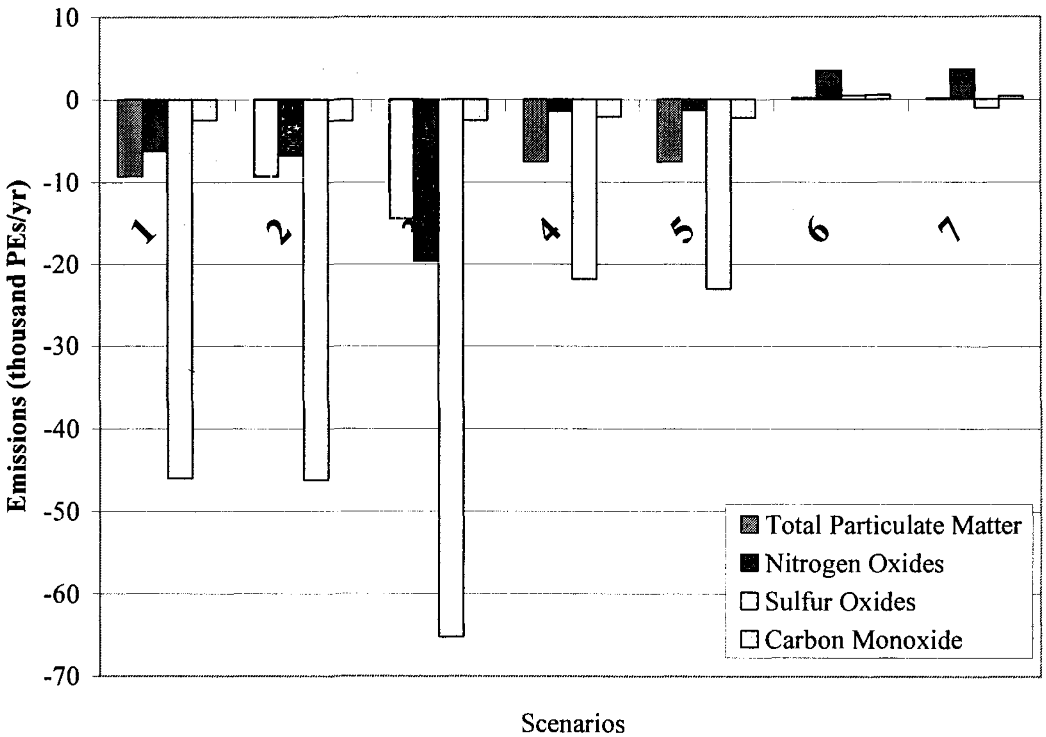

Figure 4.6. Total annual priority air pollutant emissions for all management scenarios (thousand person equivalents/year).

The C&D wood combustion scenarios (1 – 3) offer the greatest reductions in SO_X emissions with the 100% coal offset (scenario 3) providing the greatest reduction. The 115 mile transportation difference between scenarios 1 and 2 and the difference between flaring and energy recovery of LFG when all waste is disposed (scenarios 6 and 7) offers less that 1000 PE SO_X emission savings.

The recycling component of the scenarios (scenarios 1 - 4) provides the biggest reduction (1600 – 2100 PEs) in CO emissions when compared to landfilling all the material (scenarios 6 - 7) (figure 4.6). For landfilling scenarios, energy recovery instead of LFG flaring provides saving of 100 PEs of CO emissions.

Lead air emissions

There are Pb air emissions savings of between 4300 – 6500 PEs per year in the scenarios which include recycling and C&D wood combustion with energy recovery (scenarios 1-3) (figure 4.7). Smaller offsets also occur with recycling only (500 – 1000 PEs per year) (scenarios 4 -5) and LFG to energy (~300 PEs per year) (scenario 6). Pb air emissions occur in all the scenarios during collection. Scenarios 1 – 5 also have positive Pb air emissions from upstream energy production activities for the material recovery facility (MRF) activities. In these scenarios however, the positive emissions are more than offset by the recycling and WTE activities.

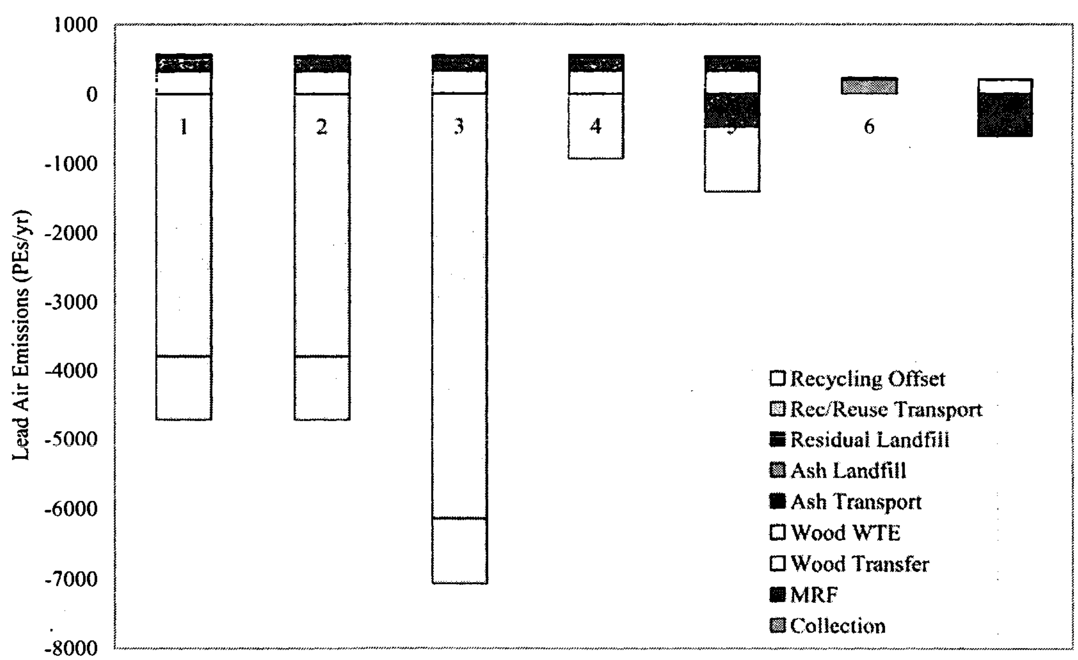

Figure 4.7. The total annual lead air emissions are broken out by process (person equivalents per year) with the greatest offsets coming from wood WTE.

Heavy Metal Water Emissions

There is a consistent trend for all heavy metal emissions to water with reduced net emissions occurring in the C&D wood combustion scenarios (1 - 3), some increase occurring in the recycling scenarios (4 -5) and the most emissions occurring in the disposal scenarios (6 – 7). The emissions for Zn, Cd and Cr (figure 4.8) are three orders of magnitude higher than for As, Se and Pb (figure 4.9). Hg emissions were also

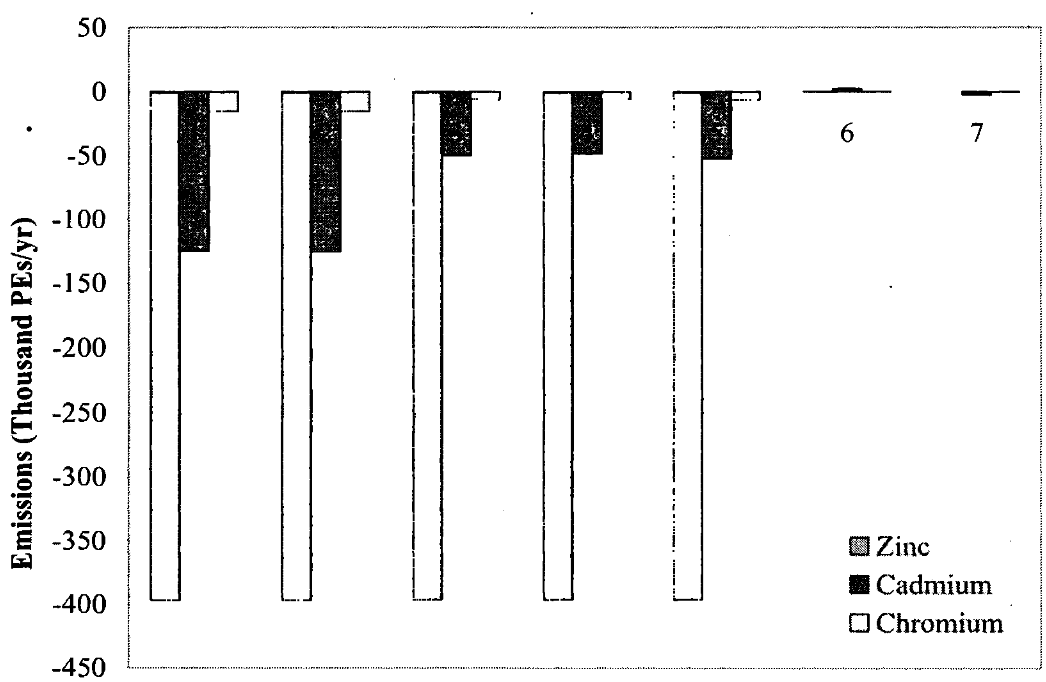

Figure 4.8. Total zinc, cadmium and chromium water emissions for all management scenarios (person equivalents/year).

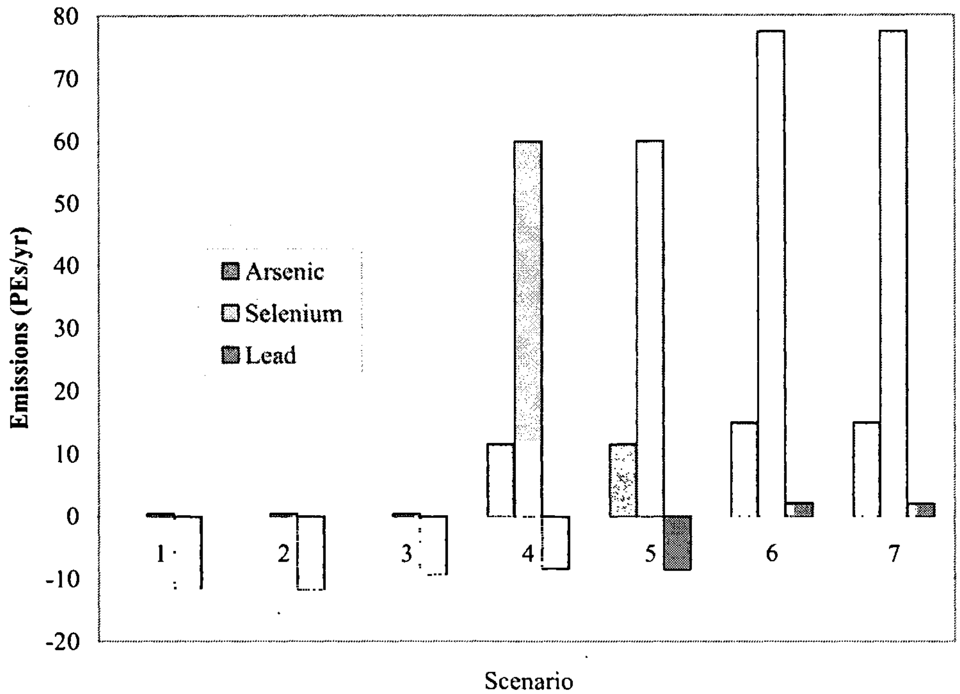

Figure 4.9. The total annual arsenic, selenium and lead water emissions (PEs per year) for each scenario indicate that scenarios 1 – 3 with the wood WTE and the recycling processes have the lowest impacts for these categories.

calculated for all scenarios and resulted in offsets of 7K – 8K PEs per year for scenarios 1–4, which is primarily due to the recycling component of the scenario life cycle. The landfill only scenarios (6 and 7) resulted in net Hg emissions of 70K – 110K PEs per year. Cu emissions were also analyzed and for all scenarios were determined to be negligible at less than 1.2E-6 lb per year.

Other Water Emissions

Projections for life cycle water emissions for total dissolved solids (TDS), total suspended solids (TSS), biological oxygen demand (BOD), chemical oxygen demand (COD), oil, phosphate, sulfuric acid and ammonia were determined. The C&D wood incineration scenarios (1 - 3) generated offsets for all the impacts except BOD and were lower than the landfilling scenarios (6 - 7).

Consolidated impact analysis

As detailed in the sections above, recycling followed by combustion of wood with energy recovery offsetting either the NE energy grid or 100% coal results in a multitude of benefits and offsets. In order to determine which scenarios had the highest total impact, the total PEs were summed for each waste management scenario. This method does not include any weight analysis to account for varying toxicity potentials of the wide array of impacts outlined above for both humans and the ecology. Rather the summation of the data simply represents how many U.S. PEs of each impact are being generated by each scenario (Hauschild and Wenzel, 1998).

Figure 4.10 shows that C&D recycling with local combustion of wood with energy recovery offsetting the NE energy grid for combustion at a local recovery facility or at a facility in either Maine or Canada (scenarios 1 and 2) has the lowest total PE impacts, with net negative total PEs representing impact offsets. The 100% coal offset scenario (3) is 50K PEs higher, followed by the C&D recycling-only scenarios (4 – 5) all of which still have net negative total PEs representing impact offsets. The top three largest PE offsets come from reduction in carbon emissions, cadmium emissions to water and sulfur oxide air emissions. Both the disposal scenarios (6 and 7) have net positive total PE impacts with the flared LFG scenario (6) having the largest impact.

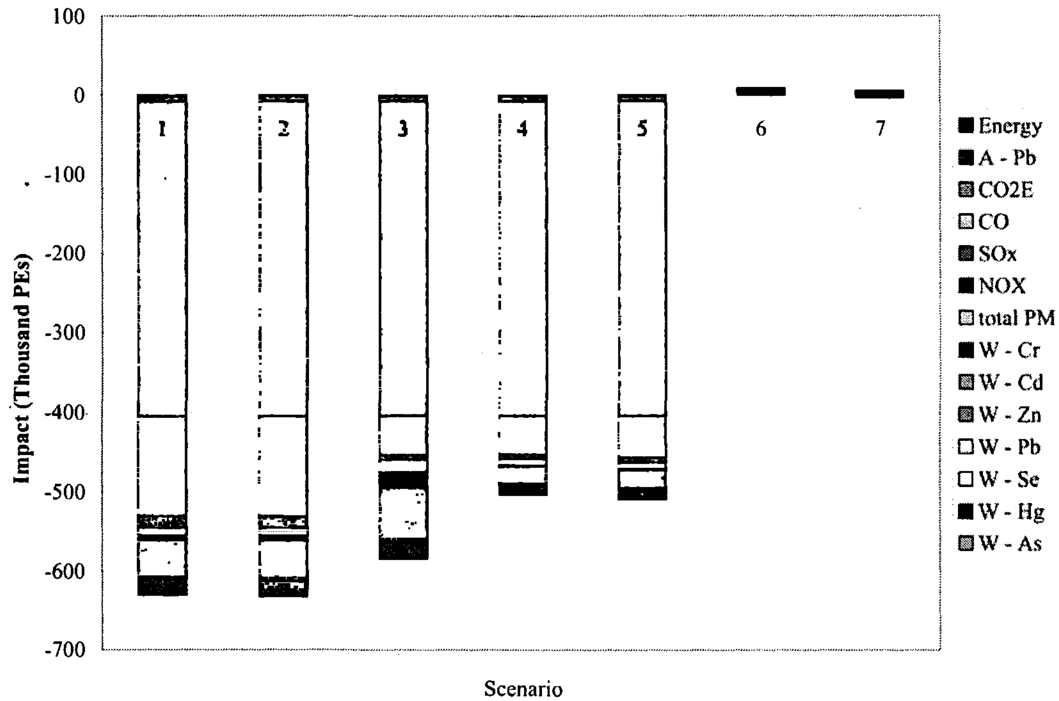

Figure 4.10. The total person equivalents (PEs) for each waste management scenario representing energy, carbon equivalents (CO_2E), water emissions (As, Hg, Se, Pb, Zn, Cd and Cr) and air emissions (PM, NO_X, SO_X, CO, Pb) impacts are illustrated here with almost all impact categories providing some level of offsets for scenarios 1 – 5 (wood WTE and/or recycling) and the greatest offset coming from scenarios with wood WTE.

Virgin Wood Comparison

In order to compare the combustion of C&D wood with the virgin wood currently being combusted in New Hampshire, the MSW DST was used to model the combustion of 280K tons of virgin wood (equivalent to the mass of C&D wood generated in 2006 in New Hampshire) harvested in northern New Hampshire (at a distance of 150 miles) in a combustion facility. The impacts from energy consumption, carbon, Pb and priority air pollutant emissions were considered in this comparison. The analysis also examined the impacts from different stages of the life cycle (wood transport, WTE, ash transport and ash landfill). The energy generated offsets the NE power grid (figure 4.3) and the ash is used for some beneficial use application (not landfilled). This scenario was compared to the same quantity of C&D wood combusted with energy recovery at a local facility (25 miles away) and offsetting the NE power grid with the ash from the combustion facility being landfilled.

While combustion of virgin wood and C&D wood both produce energy, the C&D wood combustion produces over 1.2 trillion BTUs/year more energy for the same mass of wood when compared to virgin wood combustion (figure 4.11). The difference in energy is due

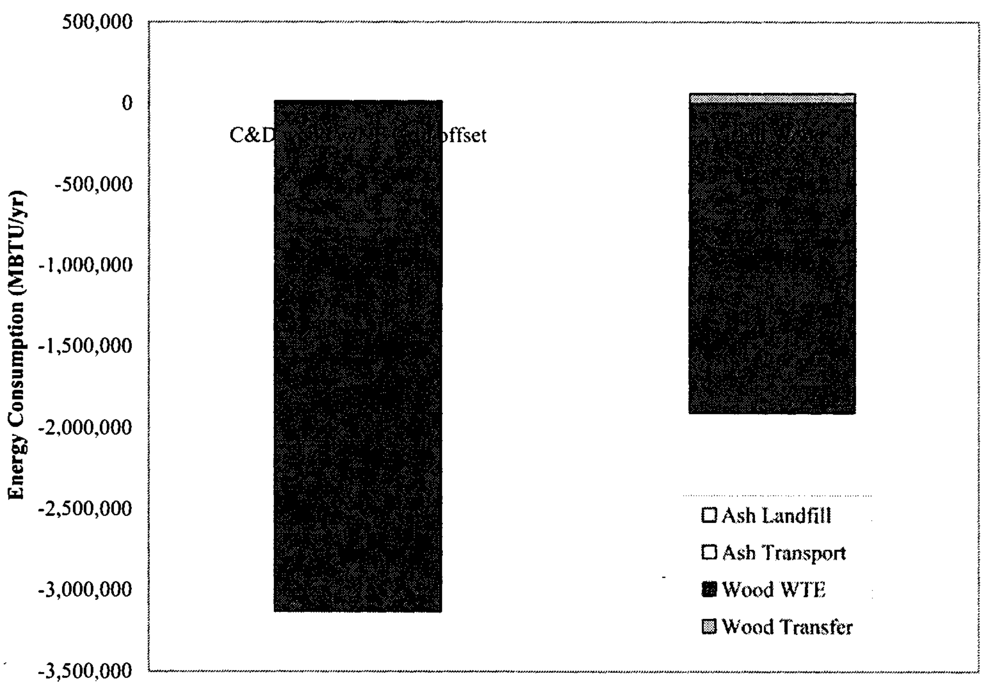

Figure 4.11. Energy consumption for C&D wood combustion offsetting the NE energy grid versus virgin wood combustion (MBTU per year) allocated into the different life cycle phases of the scenarios.

to the high water content of virgin wood (4500 BTU/lb) which has 45% moisture versus 12% moisture for C&D wood (7380 BTU/lb). The offsets from the wood WTE phase are much larger than the impacts and/or offsets of the other phases. The carbon emissions for the C&D wood offsetting the NE grid is also greater than for virgin wood use, by 17,000 MTCE per year. This is again attributed primarily to the wood WTE phase of the life cycle (figure 4.12). This does not include the initial manufacturing and treatment of the C&D wood material. C&D wood debris is currently considered a waste product and has no value. Were that to change and a value be assigned, it would be prudent to allocate some of the harvesting and manufacturing impacts to the combusted C&D wood.

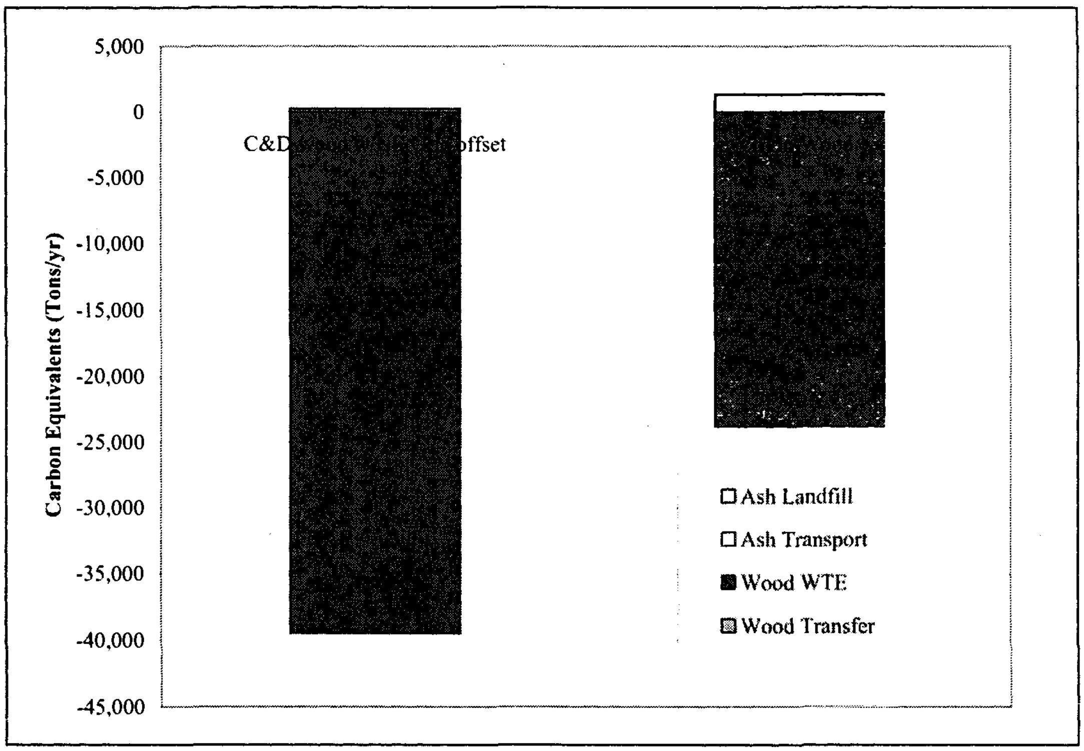

Figure 4.12. Carbon emissions for C&D wood combustion offsetting the NE energy grid versus virgin wood combustion (Tons per year) allocated into the different life cycle phases of the scenarios.

Based upon best available air pollution technologies and a wood quantity of 280K tons per year, the C&D wood used for energy production had reduced emissions in PM, NO_X, SO_X and CO (figure 4.13). The Pb emissions (attributed to the wood WTE portion of the

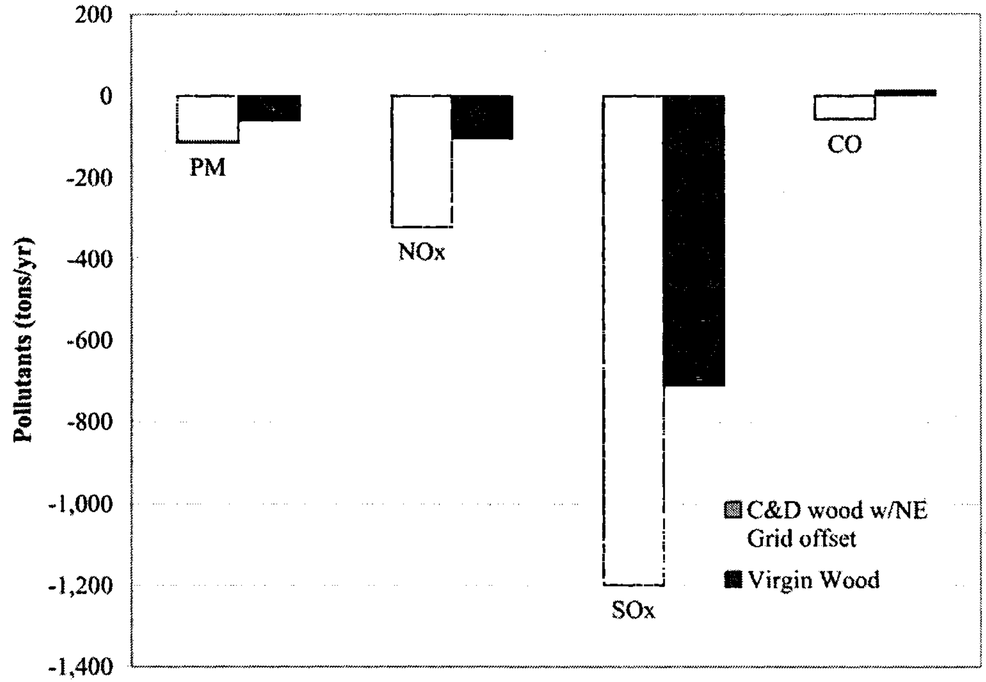

Figure 13. Particulate matter (PM), nitrous oxides (NO_X), sulfur oxides (SO_X) and carbon monoxide (CO) air emissions for C&D wood combustion offsetting the NE energy grid versus virgin wood combustion (lbs per year).

life cycle for both scenarios) are greater for C&D wood than for virgin wood by 1.5 lbs, but this still represents a 9 lb offset from the NE energy grid (figure 4.14). The reason for the reduction in emissions associated with the combustion of C&D wood is because the BTU/lb value of C&D wood is greater than for virgin wood. Therefore, more electricity from fossil fuels is offset for C&D wood combustion than electricity produced by virgin wood combustion. Consequently, even if there is ash to be landfilled or a slightly higher metal content in C&D wood, the greater electricity production offsets these differences.

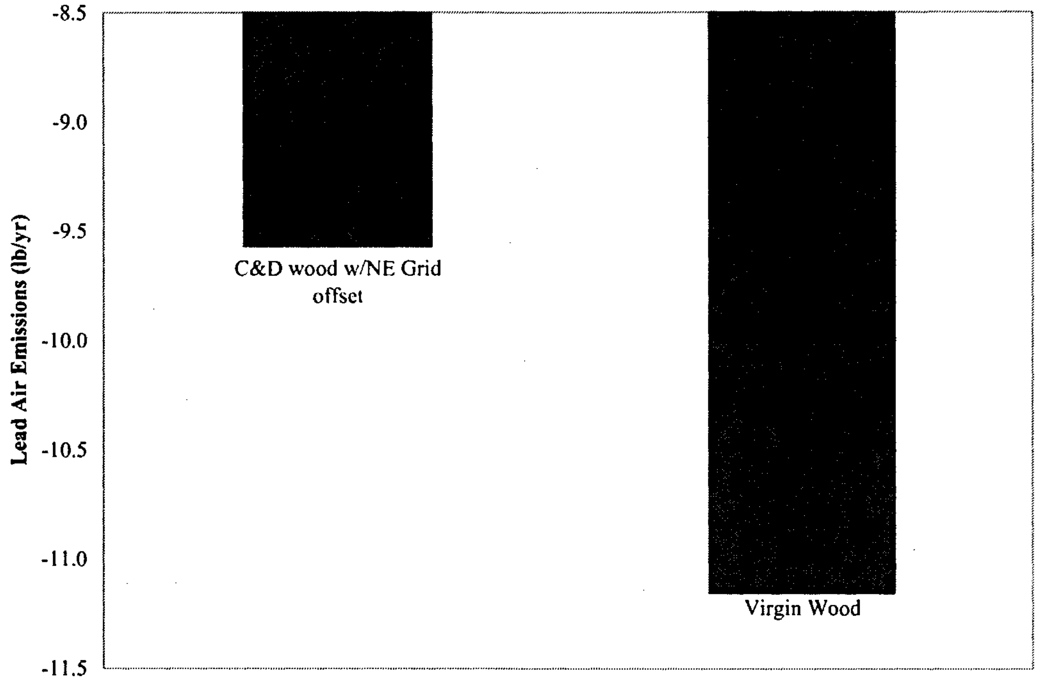

Figure 4.14. Lead air emissions for C&D wood combustion offsetting the NE energy grid versus virgin wood combustion (lbs per year).

The cost to use virgin wood combustion for energy recovery is over $9M dollars more than to use C&D wood (figure 4.15). The greatest difference comes from the transport of the wood materials. The virgin wood is being transported from over 150 miles away, and the C&D wood is being generated locally. The cost per BTU of energy of combusting the virgin wood is over double that of combusting C&D wood.

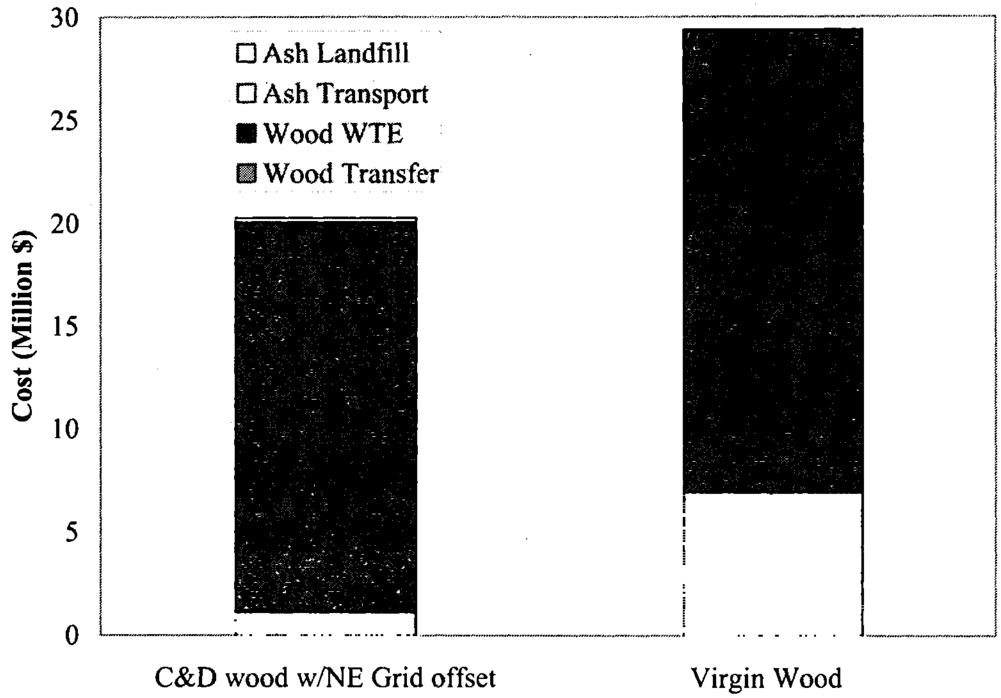

Figure 4.15. The cost of utilizing C&D wood to offset the NE power grid compared to utilizing virgin wood, broken out by process.

DISCUSSION

C&D wood potential

The U.S. generates approximately 30 – 40 MMT per year of C&D wood debris (Sandler, 2003). In areas with appreciable quantities of C&D wood waste, as well as appropriate infrastructure and management systems in place, these materials could be used as an energy source producing up to 650 trillion BTUs per year of energy. This does not include C&D wood waste generated from disaster debris management. Use of C&D wood debris as a source of alternative energy could be significant in the management of C&D debris generated by hurricanes and other major storm events. Hurricane Katrina generated 12 million cubic meters of C&D wood (Dubey et al., 2007); this equates to 93 trillion BTUs of energy. However, this could be difficult to achieve given that disaster debris is commingled and separating materials for reuse can be logistically challenging and an expensive task given the constraints of a disaster situation. Use of C&D wood for energy recovery as a potential component of disaster management is worth considering, with the need for more detailed consideration of logistical challenges associated with this practice.

Land Impacts

The comparison of virgin wood to C&D wood does not account for land impacts. Depending on the perspective, aside from the landfill space impact, it can be considered that C&D wood does not have a land impact beyond the recycling facility and that the land impacts are all allocated to the initial use of the material. This is because C&D wood is still considered a waste product with no market value. If post-consumer C&D wood developed a monetary value, then the initial logging and processing impacts could be allocated to that use. In this scenario, the land impact from C&D wood would be increased, but by how much would depend on how the impacts are allocated between its initial used (some 20 – 30 year old building, wall, deck, etc) and its post consumer power generating capacity. With regards to the landfill space, C&D wood ash takes up 95% less landfill space than non-combusted C&D wood (Jambeck et al., 2007).

Local Impacts

Looking at the local scale impacts, virgin wood incineration does produce fewer emissions per BTU, when compared to C&D wood incineration (table 4.4 and 4.5) when other life cycle aspects associated with C&D management are ignored. This can be an

		Incineration Emissions		Person Equivalents	
Parameter	Units	C&D Wood	Virgin Wood	Conversion factor (unit/cap)	Difference (C&D - Virgin)
Energy Consumption	MBTU	8,284	497	328,517	0.024
Air Emissions					
PM	lb	15,474	214	128	119
NO_X	lb	310,880	2,814	128	2,411
SO_2	lb	50,299	333	99	506
CO	lb	55,853	706	693	80
GWP	MTCE	3,317	11	0.03	122,472

Table 4.4. Facility site impacts (discounting offsets) for combustion of C&D wood and virgin wood for energy consumption and air emissions (PM, NO_X, SO_2, CO and GWP). Conversions to person equivalents (PE) are included.

	initial total Pb content (lb/ton)	BTUs	Lead emissions (lb)	Lead emissions (lb/BTU)	Criteria Air Pollutants (lb)	Criteria Air Pollutants (lb/BTU)
Coal [1]		2.04E+12	6.3E+01	3.1E-11	2.09E+07	1.0E-05
C&D Wood [2]	2.72E-02	2.07E+09	8.1E+00	3.9E-09	4.33E+05	2.1E-04
Virgin Wood [2]	6.51E-03	1.26E+09	1.1E+00	8.9E-10	2.28E+05	1.8E-04

Table 4.5. Local scale emissions. [1]The coal values are based annual emissions (TRI). [2]The C&D wood and virgin wood values are from MSW DST calculations based on 280K tons.

argument against the use of C&D wood incineration as an energy source instead of virgin wood. Virgin wood's relatively low BTU value (compared to coal and C&D wood) and the necessity of trucking it 150 miles from northern New Hampshire (for this scenario and as accounted for in the larger analysis that includes offsets) may make it a cost prohibitive option for New Hampshire as well as all the NE states in the near future. This will be especially true if the transportation energy requirements for virgin wood exceed its own inherent energy. Combined utilization of C&D wood and virgin wood in place of the currently utilized coal for the Schiller Station facility would reduce the local impacts, but would not generate the equivalent BTU power as coal. There is not enough C&D wood, nor virgin wood waste generated on an annual basis to completely supplant the need for coal. The incineration of the New Hampshire generated C&D wood in the Schiller facility would produce 8.1 lbs of lead emissions and a combined 432 thousand lbs of criteria air pollutants, which are 13% and 2% respectively of the emissions currently generated by coal combustion. As a management tool for New Hampshire, incineration of C&D wood has the least overall impacts of all the other management scenarios considered in this study.

Results of the MSW DST model indicate that combusting C&D wood for energy recovery has fewer environmental impacts than to landfill it. Furthermore, recycling C&D materials in general, even if the C&D wood is not combusted for energy, is still more favorable than to landfill all of the C&D materials. In the comparison of combustion of C&D wood versus virgin wood in this paper, the C&D wood scenario is preferable to the virgin wood scenario with respect to all impacts with the exception of lead air emissions. However, the lead air emissions for C&D wood are still less than emissions from the NE power grid.

The benefits afforded by C&D recycling and use of the recovered wood fraction for energy production have significant ramifications. For example, recycling C&D debris and use of C&D wood in energy recovery facilities produces a net gain in energy production of over 7 trillion BTU per year, 7.6% of the total annual New Hampshire residential energy consumption (U.S. EIA, 2006), which is enough to power close to 45 thousand homes in New Hampshire. In addition to the reduction of 70,000 - 130,000 tons per year of carbon emissions eliminated, criteria air pollutants are significantly reduced when combusting C&D wood with energy recovery producing 600 tons per year less PM, 430 tons per year less NO_X, 2,300 tons per year less SO_2, 890 tons per year less CO, and 10 lbs less Pb (with NE energy grid offset) when compared to landfilling.

Most of the offsets outlined in this section come from the fact that the C&D wood is an available source of energy and it can offset traditional energy sources when it is used for energy production. The use of alternative energy sources will continue to increase and this analysis illustrates that C&D wood waste, readily available in the solid waste stream, can contribute to an integrated alternative energy portfolio.

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Chapter 5

EMBODIED ENERGY OF BUILDING MATERIALS – A REVIEW

EMBODIED ENERGY

The construction industry is a large part of the U.S. economy. The design and construction of a new building is finance and resource intensive. In 2007, 1.35 million new homes were built in the United States, of which approximately 1.05 million were single detached dwellings (NAHB, 2007). Household energy consumption equates to approximately 21% of the total U.S. energy demand (101 quadrillion BTUs (DOE, 2006)).

A 2001 study of a standard single family home (Keoleian et al., 2001) determined that its 50 year life cycle total energy demand came to 16,000 GJ (15 trillion BTUs). The embodied energy of a standard house ranges from 0.4 – 15% of the total life cycle energy. For the 1.05 million new single family homes this equates to between 1.3 – 47.7 quadrillion BTUs per year and 1.3 – 47% of the total US annual energy consumption (for 2007).

There are various definitions for embodied energy, but generally they tend to all include:

- extraction of the raw material
- transport to manufacturer
- manufacture into end product and
- transport of the end product to the distribution outlet.

The embodied energy of a particular material can vary depending on the location of end use with respect to the extraction point, the technology being used to extract the material and manufacture of the end product.

A profusion of studies on the embodied energy of building materials has been published over the last ten years with case studies comparing different materials, energy in the product's use phase versus the embodied energy, different house types, and looking at specific components of houses for different climatic regions. The literature covers the energy use breakdown of different types of buildings or components of buildings, investigates methods of increasing the efficiency of buildings, compares different building materials for both Standard Homes (SH) and Energy Efficient Homes (EEH) and investigates the significance of the building materials impact for the life cycles of the houses. Many studies in the literature are individual case studies for specific buildings.

Scheuer, et al. (2003) looked at the embodied energy of the materials used in the construction of a campus building at the University of Michigan. The quantities of each type of material used were tracked as was the embodied energy of the materials. Electric arc furnace (EAF) steel, cement in concrete and sand contributed to over a third of the total embodied energy of the building. Steel and cement are known to be high energy materials, however sand is a unit material with a very low unit energy. The quantity of sand required for the building however was large enough to make it a major contributor to the total embodied energy of the building. Table 5.1 lists the top ten materials that contribute almost three quarters of the total embodied energy.

Material	kg	MJ/m2	% of total M
Steel, EAF	478,536	806	13%
Cement (in concrete)	1,341,120	680	11%
Sand	8,158,480	671	10%
Polyamide/nylon, primary	30,480	522	8%
Aluminum, primary	15,240	432	7%
Steel, cold rolled	85,344	337	5%
Steel, Galvanized	77,216	324	5%
Kraft paper	61,976	320	5%
SBR latex	31,496	302	5%
Cast iron	49,784	224	3%
Top 10 TOTAL	10,329,672		
Building TOTAL	14,151,864		
% of total bldg	73%		

Table 5.1. Data from Scheuer et al. (2003) study for a campus building at the University of Michigan representing the materials used that had the highest embodied energy.

The table in the appendix provides some reported values for the embodied energy of a wide range of different building construction materials.

On a building component level, the EE of the substructure and the frame of a building can account for about 45% of a building's life cycle EE (Yohanis and Norton, 2006; Chen et al., 2001; Lawson, 1996). The walls, roof, floor and windows together can account for 30%, finishes 13% and heating 10% of the life cycle EE (Yohanis and Norton, 2006). For energy efficient houses in Sweden the embodied energy accounts for 40% of the total building energy due to the low energy use required for heating and cooling (Thormark, 2002, 2006).

RECYCLING/MATERIAL SUBSTITUTION

A study conducted by Chen, et al. (2001) in Hong Kong determined that the energy embodied in steel and aluminum ranks as the first and second largest energy use and may account for more than three-quarters of the total embodied energy in a residential building, despite concrete being the largest quantity of material. This would indicate that a significant savings can be achieved through the use of recycled aluminum and steel and that increased usage through standard and innovative design uses can be beneficial. Scheuer et al. (2003) also found in a simple analysis that through the use of secondary material, the embodied energy of the building was lowered by over 25% of the original energy requirement (not accounting for availability or transportation issues).

Thormark (2002, 2006) also considered the recycling and reuse potentials of the building materials, which were 15% and 17% of the total energy over a 50 year life cycle; 90% of the energy recovered was through recycling and combustion. The studies found the following to be important: 1) the buildings have been designed for deconstruction, 2) the energy intensity of the materials be considered, 3) recycled materials are utilized and 4) materials are not cross contaminated during deconstruction. A German study (Quack, 2001) that assumed 100% recycling of materials after building demolition resulted in a potential energy savings of 12% of the building's total embodied energy; a Japanese study (Gao et al., 2001) found that the embodied energy of a building that was constructed using recycled materials was reduced by 25% compared to one using conventional materials. Using a process analysis, Gao et al. found that most materials provide better energy returns when compared to their primary counterparts, but the energy savings will vary depending on the type of structure and the materials being used. The energy savings for reusing a building's structure was also analyzed. This afforded the greatest savings, but can be limiting depending on the function of the structure and whether it is compatible with the structure replacing it. It would be useful to be able to look at the use of recycled materials and designing for deconstruction and energy efficiency together and to understand how they contributed to the overall life cycle energy.

Thormark (2006) looked at the impact of material substitutions for a baseline house and two alternative designs that allowed for minimizing and maximizing the embodied energy, detailing specific components and materials that would be used for each design. The minimum design provided a 15% reduction in the embodied energy, and increased the recycling/combustion potentials by 6% and the reuse potential by 9%. The maximum design resulted in a 6% increase in embodied energy, but a 13% increase in the recycling/combustion potential and an 18% increase in the reuse potential. In a net energy analysis, the minimum design alternative provided the lowest energy demand. A study by Lippke et al. (2004) looked at the difference in embodied energy between wood frame housing and steel/concrete framed housing in warm and cold climates. A 15-16% increase in embodied energy requirements for steel/concrete framed houses in warm/cold climates respectively, compared to wood framed houses.

Chani et al. (2003) conducted a study in India comparing the embodied energy of different wall materials (different types of brick and concrete units). The results showed that the traditional bricks commonly used throughout India were the most energy intensive, not accounting for their use of topsoil, which is a precious commodity for Indian agriculture. The use of clay flyash and sand lime bricks in addition to concrete blocks proved to be the most energy efficient alternatives for this study with embodied energy rates of between 31 – 60% of the rate for traditional bricks.

Pierquet, et al. (1998) compared 11 different building wall materials and compared them to a base case (wood 2x4 constructions) for a cold climate scenario (Minneapolis, MN) (table 5.2). The total embodied energy, thermal performance and long term energy savings were calculated. The strawbale wall system was the only system to have a lower embodied energy than the 2x4 wood stud wall base case. All of the wall systems would be more energy efficient than the base case over a 20 year period, but the life time energy savings is minimal for the systems with more than a 5 year payback period. This analysis does not consider the material transportation energy.

Cole (1999) looked at the energy differences between wood, steel and concrete structural assemblies with respect to construction energy (to include on-site equipment use, equipment and materials transportation and worker transportation). The concrete assemblies had the highest construction energy, up to 6 times that of the wood and steel assemblies. Despite having the lowest embodied energy of the three materials (between 0.5 – 3.4 MJ/m^2 for concrete, 2.5 – 28.5 MJ/m^2 for wood and 8.2 – 42.0 MJ/m^2 for steel), when the analysis included direct energy from construction was included, the concrete assemblies were highest ranging from about 250 – 900 MJ/m^2, where the steel and wood assemblies ranged from 100 – 700 MJ/m^2 and 50 – 300 MJ/m^2 respectively. The concrete assemblies have an order magnitude greater overall energy demand due to the labor intensive nature of the construction. This energy demand ends up negating any benefits from it being a lower embodied energy material.

Wall no.	Description	R-value	EE (GJ)	One Season Heating (GJ)	Energy Use 20 yr (GJ)
1	Base case, 2 X 4 stud walls	15.8	127.8	9.7	322.9
2	Wall 1, Upgraded to 2 X 6 walls	22.2	131.8	7.4	280.4
3	Wall 2, with insulating foam sheathing	25.7	129.9	6.8	266.8
4	Wall 2, plus interior strapping & insulation	29	139.3	6.4	267.7
5	Double wall	40.3	139.3	5.4	247.2
6	I-beam wall studs	38.6	142.8	5.7	255.8
7	Foam core insulated panels	26.5	149.9	6.8	285.2
8a	Plastered strawbale construction	44.8	125.1	5.4	232.3
8b	Plastered strawbale construction	34.1	125.1	6.0	244.4
8c	Plastered strawbale construction	23.2	125.8	7.2	269.4
9	Cordwood masonry construction	20.5	130.5	7.8	287.0
10	Insulated concrete forms (Greenblock)	22.8	165.3	7.3	311.3
11	Wall 2, with 28% recycled content steel framing replacing wood	22.1	143.5	7.5	292.5
12	Autoclaved cellular concrete	21.2	166.6	7.7	319.9

Table 5.2. Wall systems considered in Pierquet, et al. (1998) cold climate study, associated R-values, embodied energy of building using wall system (EE), heating energy for a single season and energy use of a 20 year life span.

Material to material comparisons of energy consumption for buildings do not necessarily provide enough information to determine which material is most sustainable. The material needs to be put into the context of a system that represents its end use and then a system to system comparison can be made. This type of comparison will better reflect which material not only has the lowest embodied energy, but also which system also requires the least amount of energy to construct, maintain and operate over its full life cycle.

Looking at demolition of old inefficient buildings compared to refurbishment, there is not a clear cut answer as to which option will have lower energy demand. There are arguments for both sides: inefficient buildings have very high operational energy requirements; construction of new buildings has high embodied energy demand. There is potential to reuse some of the existing structure to increase the efficiency of the building without having to start completely from scratch. Power (2008) analyzes this problem based on studies conducted in both the UK and Germany and considers the energy implications as well as social and environmental implications. There were not enough data available to fully understand the impacts of demolition, new construction, renovations and materials together to make a definitive assessment. There are cases where the refurbishment option has greatly reduced the operational energy demand without having to expend the embodied energy of new materials. The refurbishment options provide considerable energy savings in the short term (without major negative social and environmental impacts), although for the long term the demolition and new construction option provides better energy savings (Power, 2008).

TRANSPORT

The transportation component for building materials is minimal compared to the total life cycle energy (Adalberth et al., 1997). Some regions may rely heavily on imported building materials (i.e. cities). Table 5.3 lists data on the transportation energy intensities of different building products in Hong Kong and table 5.4 lists data on the energy use of different modes of transportation.

Building Products	**Energy Use (MJ/kg·km)**	**Building Products**	**Energy Use (MJ/kg·km)**
Acrylic paints	1.04	Homogenous floor tiles	0.77
Aluminum	0.84	Plasterboard	3.36
Asphalt	2.34	Plywood	0.88
Ceramic tiles	0.77	Steel	0.94
Concrete	1.2	Steel (galvanized)	1.77
Concrete roofing tiles	0.98	Steel (zinc sprayed coated)	0.79
Extruded polystyrene	3.97	Timber	0.83
Glass mosaic tiles	3.67	Unglazed vitreous mosaic tiles	1.55
Granite wall facing slabs	2.38	UPVC	0.77

Table 5.3. Intensities of energy use (MJ/kg km) in transporting common building products in Hong Kong (Chen et al., 2001).

Method of transportation	**Energy use (MJ/(kg·km))**
Deep-sea transport	0.216
Coastal vessel	0.468
Truck	2.275
Class railroads	0.275

Table 5.4. Energy use in different modes of transportation. (Chen et al., 2001; Sperling and Shaheen, 1995; UNEP, 1995; Tillman, 1991)

Venkatarama (2001) claimed the energy required to transport high energy materials is negligible when compared to energy to manufacture, however this would depend on how far the material is being transported. Looking at data from Chen et al., the embodied energy of timber ranges from 2.5 – 28.5 MJ/kg, while the transportation energy is 0.83 MJ/(kg·km). The embodied energy of aluminum ranges from 8.1 (recycled) – 207 (primary) MJ/kg while the transportation energy of aluminum is 0.84 MJ/(kg·km). The embodied energy of aluminum is much higher than that of wood, but the energy to transport it is the same.

Considering the case study conducted by Asif, et al. (2007) for a Scottish dwelling, if the 5,725 kg of timber were brought in from China, instead of using local timber, that would require transport across ~40,000 km. Trucking transportation energy for timber is around 0.83 MJ/(kg km) (Chen et al., 2001); transportation for shipping ranges from 0.037 MJ/(kg km) (Horvath, 2004) to 0.216 MJ/(kg km) (Chen et al., 2001). Timber from within the UK might have to be trucked between 150 – 1,000 km. Timber from outside the UK (i.e. China) would have to be shipped some 40K km (worst case) and then trucked from the nearest port (say 150 km). The transportation energy for the local case (timber from Scotland) ranged from 0.7 – 4.7 million MJ, while the foreign timber (timber from China) transportation energy ranged from 9.2 – 50.2 million MJ for a single Scottish dwelling. The total embodied energy of the materials for the dwelling in the study could range between 14.3 – 163 GJ, two to three orders of magnitude less than the transport energy for the local timber and four to five orders of magnitude less than the foreign timber. Looking at a more energy intensive material (i.e. 1 kg alumimum) being transported from China to Scotland, the embodied energy of the aluminum requires 207 MJ. Transport of that aluminum from China to Scotland (port only – 40K km) would require 1,480 MJ of energy and transport within country (150 km) would require an additional 124 MJ of energy. The transportation energy may only be considered negligible if the material is being used within 20 km of the processing site. The USGBC LEED certification allowed regional source credit for materials produced within 500 miles (806 km) of a building site. While 800 km certainly requires less energy than 40K km, for the aluminum, it would require three times its embodied energy to transport it that distance.

Significant differences occur between the amount of energy and greenhouse gas emissions associated with the construction of alternative wood\ steel and concrete structural assemblies with concrete typically involving order of magnitude higher quantities. Transportation of workers to and from a jobsite accounts for over one third of the total energy expenditure for concrete structures which is more labor intensive than steel or wood structures (Cole, 1999).

LIFE CYCLE ENERGY – USE VS PRE-USE

Many building life cycle energy (LCE) assessments focus on energy efficiency and energy use during the use phase of a building, neglecting the indirect "external" costs. Including the embodied energy of materials into the analysis provides a wider view of the source of energy expenditures, both internal and external to the physical building. Several studies have considered embodied energy in connection with the buildings total life cycle energy (LCE), looking at the energy of the materials in comparison with the energy consumed through operation (heating, cooling, lighting, etc) over the life of the building (30 – 50 years).

The goal of a low energy building is typically to increase the operational efficiency thereby decreasing the operational energy. Active and passive technologies are used to achieve this through increased insulation, better performing windows, reduction of infiltration losses (passive) and solar thermal collectors, solar photovoltaic panels and biomass burners (active). The use of these active and passive technologies can result in an increase in the embodied energy of a house. A question of whether the increased embodied energy counteracts the decreased operational energy can not generally be answered without considering the context (Sartori and Hestnes, 2007). Studies considering specific cases have shown that increased operational efficiency does increase the embodied energy, but depending on the technology and materials used, can have payback periods ranging from 0 – 18+ years (Pierquet, et al., 1998).

The Early Design Model (EDM) is a design tool that allows the engineers and architects to optimize the design of the building to minimize the life cycle energy consumed and is based on varying inputs that contribute to the operational energy, embodied energy and capital cost. Factors contributing to the operational energy include daylight factors, hourly ambient luminance, non-lighting gains, fabric heat loss, average daily comfort temperature, ventilation heat loss, solar gains and cooling load. The embodied energy and capital cost calculations are based on room and window dimensions, elemental cost data and embodied energy coefficients. The model is not intended to be able to provide a specific cost evaluation, but rather to provide a framework for comparative analysis between different building options and scenarios, allowing the user to see where and which components of the building design affect the greatest energy savings (Yohanis and Norton, 2006).

Through the use of the EDM, Yohanis and Norton (2006) determined that the embodied energy of a building is almost negligible compared to the operational energy over a 30 year life span. Embodied and operational energies therefore could potentially be considered separately and building materials and components with low embodied energy could be selected as long as they didn't impact the operational energy. Operational energy reduction may be currently more significant, but with improvements in building efficiencies, embodied energy savings will become more important.

Keoleian et al. (2001) conducted a study looking at the saving for a standard house (SH) compared to an energy efficient house (EEH) and found that there were significant savings for the EEH. The total life-cycle energy consumption of the SH was determined to be 16,000 GJ (equivalent to 2,614 barrels of crude oil). In contrast, the total life-cycle energy of the EEH was 6,400 GJ (~1,046 barrels of oil). A 60% reduction in life-cycle energy was achieved with the EEH model. However, this did not necessarily translate to economic cost savings and the study was somewhat limited in that the objective was to not alter the layout or look of the building. A study that seeks to minimize both life-cycle cost and life-cycle energy would have utilized different improvement strategies. Hence, the context of the analysis is very important.

Operational energy is the largest part of a buildings life cycle energy cost and can be reduced through improved insulation, configuration, size, location, more efficient

Life cycle phase	Suzuki et al. (1998)	Adalberth et al. (1997)
Construction	15.1%	11 – 12%
Operation	81.5%	83 – 84%
Renovations	2.6 %	4 – 5%
Demolition	0.8%	0.3 – 0.5%

Table 5.5. Energy consumption for building life cycle phases from studies by Suzuki et al. (1998) and Adalberth et al. (1997):

systems, etc. At 11 – 15% of the total LCE, the construction component can offer some significant saving, however the bulk of the savings come from improved efficiencies for the operation/use phase of the buildings life cycle. With the increased operational efficiency generally comes an increase in the embodied energy (Satori and Hestnes, 2007).

Studies have shown that the embodied energy of a building can contribute between 0.4 - 15% of a standard home (SH). That percentage can jump up to 40-60% for more energy efficient homes (EEH) or for building in areas with low operational energy requirements (temperate/warm climates). The embodied energy of buildings, even if just a small percentage, when analyzed on a national or global scale becomes quite significant. If a SH requires approximately 2,610 barrels of crude oil worth of energy over its 50 year life cycle and an EEH requires only 1,050 barrels of oil (Keoliean et al., 2001), there is potentially a 40% savings in energy. The energy savings on the U.S. nation wide scale could equate to 2.1 trillion barrels of oil. Utilization of energy efficient technologies or materials with lower embodied energies can be significant when considered on a national and global scale. As the operational energy demand decreases for more energy efficient buildings the contribution of embodied energy of building materials will become more significant.

Cole and Kernan (1996) looks at the relative importance of the initial embodied energy compared to the operating and recurring energy. Intuitively, one could ascertain that the longer the building is used, the less significant the initial EE will be compared to the other energies. Cole and Kernan's study determined that while for a 25 year life span the initial EE markedly outweighs the recurring (maintenance) energy, this is not the case once the building reaches a 50 year life span. Over a typical 50 year building life, the initial embodied energy of the structure represents a relatively small portion of life-cycle embodied energy (between 0.4 - 15%) (Keoloian et al., 2001) and, as a consequence, the distinction between wood, steel and concrete systems is also less marked. In looking to reduce the embodied energy of a building, focus needs to be given not only to the materials used, but also to their longevity as the recurring maintenance and repair energies can be significant.

Typically, LCE studies will utilize a process based analysis that considers all the process associated with a product over its entire life cycle and all the inputs and outputs of those processes. It is a very data and labor and time intensive process, which usually results in the narrowing down of the study's scope and boundary. Alcorn and Baird (2006a) propose a hybrid analysis that utilizes statistical analysis, process based analysis and input-output based analysis together to best represent the embodied energy of building materials. The hybrid process allows the user to incorporate more of a product's processes in the study on a more aggregated level, to utilize the available statistical data on the processes and then to utilize the process based analysis to focus in on the more significant processes. The net result was achieved more quickly and accurately than with other analysis methods (Alcorn and Baird, 1996a). The differences in embodied energy values when using a hybrid analysis compared to a process analysis can be quite significant and can contribute to the variations in EE values in different studies (Baird et al., 1997)

CONCLUSIONS

While this paper does focus only on the energy impacts of building materials, there are certainly other life cycle impacts (water consumption, air and water emissions, human toxicity potential, ecological toxicity potential, social, etc) that also need to be considered; such consideration may result in a building that has higher energy intensity, but with less social and environmental impacts.

There is a consensus in the studies that the best way to reduce the energy intensity of buildings is to increase the efficiency. While this is not a simple or inexpensive task, there are increasing available new and old technologies that can be utilized to provide significant energy savings.

Material choices can, on a large scale, also provide significant savings, but should not do so at the expense of energy efficiency. Use of recycled materials in place of primary materials (specifically for aluminum and steel) has the potential for great savings in embodied energy.

Transportation can be a significant contributor to the overall energy of buildings, depending on where the materials originate. It is inherently obvious that if a building material can be sourced close to the site, obtaining the material from farther (and likely cheaper) sources would result in an unnecessary increase in energy that has a potential of increasing the embodied energy of the materials by several orders of magnitude.

APPENDIX

Material	MJ/kg	Material	MJ/kg
ALUMINUM		Lime	5.63
Aluminum, primary	166 - 237	Mineral spirits	5.5
Aluminium (recycled)	8.1 - 17.3		
		MORTAR	
Argon	6.8 - 7.0	Mortar	<0.1 - 1.9
		Cement mortar	2
ASPHALT			
Asphalt	50.2 - 51.0	PAPER	
Asphalt (paving)	3.4	Paper	16.2
Asphalt shingle	14.6	Paper, secondary	6.9
Bitumen	44. 1	Kraft paper	12.6 - 37.7
		Paper, building	25.5
Brass	62 - 239		
		PAINT	
BRICK		Paint	90.4
Brick	2.5 - 4.5	Water based, paint	76.0 - 88.5
Brick (glazed)	7.2	Solvent based, paint	98.1
Soil cement pressed brick	0.4	Xylene (paint, waterproofing)	60.2
CARPET		Particleboard	3.9
Carpet, wool	106	Oriented-strand board	3.2
Acrylate lacquer (carpet grout)	30.8	Plaster board	6.1
Cast iron	32.8	POLYAMIDE	
		Polyamide resin (PA)	137.6
CERAMIC / CLAY		Polyamide/nylon, primary	125
Ceramic 5	20.5	Polyamide, secondary	<0.1
Ceramic and quarry tile	2.5 - 5.5		
Ceramic Plasterboard	6.1	PLASTIC	
Clay (fire proofing)	32.4	Polyethylene	79.5 - 87.0
Kaolin (ceiling tiles)	1.3 - 5.47	high-density polyethylene (HDPE)	87.5

CONCRETE / CEMENT		polymethylmethacrylate (PMMA)	207.3
Concrete	1.6		
Block, concrete	0.86 - 0.94	Polypropylene	75.0 - 83.8
Brick, concrete	0.97		
Paver, concrete	0.5 - 1.2	Polystyrene	94.4 - 105.0
Roofing tile, concrete	0.81	Acrylonitrile butadiene styrene (ABS)	112.2
30 Mpa, concrete	1.3 - 1.4		
Ready mix 17.5 Mpa, concrete	1	Polyvinyl chloride (PVC)	60.7 - 77.4
Pre-cast concrete	2		
Glass-reinforced, concrete	3.4	Rubber	143.0 - 150.0
Flyash (in concrete)	< 0.1	Ethylene propylene diene monomer (EPDM) rubber	183
Cement	5.85-7.8	SBR latex	70.0 - 70.8
Cement (fireproofing)	3.7	Styrene butadiene rubber (SBR) 7	70.8
Cement (in concrete)	3.7		
Fiber board, cement	13.1	Starch	15
LP cement	2.33		
		STEEL	
Ethylene glycol	85.1	Stainless steel	8.2 - 16.3
		Steel, primary, general	32.0 – 42.0
Felt underlayment #15 (roofing)	41.2	Steel, Reinforcing, section	8.9
		Steel, cold rolled	28.0 - 28.8
FORMALDEHYDE		Steel, EAF	12.3
Formaldehyde resin	72.1	Steel, Galvanized	30.6 - 34.8
Uurea formaldehyde	78.2	Steel, secondary	8.9 - 14.1
Phenol formaldehyde	87		
		Titanium dioxide	73.8
GLASS			
Glass	6.8 – 25.8	Toluene	67.9
Float, glass	14.9 - 15.9	Toluene diisocyanate	101
Glass fiber, secondary	11.9		
Glass fiber, primary	17.6 - 30.3	Vinyl	11.8

Laminated glass	16.3	Vinyl resilient flooring	50.8
Glass, toughened	25.3		
		Waxes	
GRAVEL / ROCK / AGGREGATE			
Granite	0.1	WOOD	
Gravel	0.2 - 0.9	Wood	10.8 - 28.5
Stone, dimension	0.79	Rough saw wood	5.2
Aggregate, general	0.1	Plywood	18.9
General, virgin or river rock	0.04 - 0.1	plastic-wood composite 6	5.1
Sand	0.6	plywood	8.3
Limestone	0.1	Timber, glulam	4.6
		Timber, kiln dried, dressed	2.5
GYPSUM		Timber, medium density fibreboard	11.9
gypsum	3.8 - 8.6		
Gypsum, primary	0.9	OTHER METALS	
Gypsum, synthetic	< 0.1	Lead	35.1
HCFC 22	33.7	Silver	128.2
INSULATION		Copper	48.7 - 70.6
Insulation, cellulose	3.2 - 4.4	Copper tube	65.8
Insulation, wool	16.1	Copper, primary, extruded	71.6
Polyisocyanurate (PIR)	70.0 - 70.6		
Polyester	53.7		
Bauxite ore (fireproofing)	0.6		
Glass wool	14		

Summary of manufacturing energy (extraction of raw material, production and transportation of semi-manufactures, heating of manufacturing facilities and production of final material product), (Suzuki et al., 1995; Keoleian et al., 2001; Baird et al., 1997; Baird and Chan, 1983; Alcorn and Baird, 1996b; Harris, 1999; Adalberth, 1997b; Cole and Rousseau, 1992; Howard, 1995).

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