A New Screening Metric to Benchmark the Sustainability of Municipal Solid Waste Management Systems

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Abstract

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Typically, evaluations of the environmental performance of municipal solid waste (MSW) management systems involve crude measures that are subjective in nature. The most common such measure is the recycling rate – the percentage of MSW that is separated from the waste stream for materials recovery. While useful as a rough estimate of the performance of cities, materials recycling is not the only way to conserve resources. Energy recovery – from waste to energy (WTE) power plants or landfill gas to energy systems – also contributes to improved environmental performance. The principal objective of this thesis was to quantify the environmental performance of municipal waste management systems by introducing and utilizing a new metric, the resource conservation efficiency (RCE). RCE measures the lifecycle energy efficiency of materials and its value depends on the recyclability of the material as well as the method cities choose to dispose of non-recyclable materials.

To verify the validity of RCE as an environmental metric, a life cycle impact assessment (LCIA) was performed. It was shown that the cumulative energy demand for different materials correlates strongly with EcoIndicator 99 scores, a common LCIA environmental evaluation tool. This correlation demonstrates the validity of using the RCE not only as a measure of energy efficiency, but of "environmental efficiency" as well.

In addition to the development of RCE, a new method of acquiring and organizing US national MSW data was developed and described. The resulting Municipal Solid

Waste Database allows for tonnage-based waste flow measurement that is used to analyze important US states such as California, where it is discovered that – though recycling rates have steadily increased over the years, disposal rates have remained relatively constant after an initial decline. Data collected in the development of the database is used to perform a case study comparing two American cities' (Honolulu and San Francisco) waste management systems using RCE. The results show that a combination of recycling and WTE is the most environmentally efficient way to manage MSW. More specifically, San Francisco would be much better served by building a WTE plant to handle its non-recycled waste than by expanding recycling. Honolulu's best course of action, however, would be to increase recycling to be commensurate with San Francisco's rates.

Additionally, in the search for an adequate life cycle metric, the statistical entropy method for quantifying the extent to which metals are diluted or concentrated in waste management systems was extended to account for carbon. This proved to be a useful measure of the global warming effects of different waste management strategies, but not as effective as the RCE metric, as an overall systems assessment tool.

Table of Contents

List of Figures

List of Tables

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1. Introduction – The Need for Better Waste Data and Performance Metrics

In many respects, municipal solid waste (MSW) can be thought of as a "gateway" environmental problem – because all of us deal with waste every single day of our lives, there is greater awareness of issues around MSW than for many other environmental concerns. Perhaps this can also be attributed to the tangible nature of the problem – as opposed to, say, carbon dioxide emissions, which are invisible and whose effects are not immediately apparent – we all see piles of garbage on curbsides; or pass by incinerators with mysterious plumes pouring out of the stacks; or live near the vast mountains of garbage that are modern landfills.

Whatever the reasons for the attention it receives, it is also clear that waste treatment systems and their environmental effects are not very well understood. Where attempts are made to quantify or otherwise objectively evaluate treatment systems, experts are often required to decipher the results and it is still frequently unclear which options are "better" and which underperform.

This thesis is an attempt to address that problem – the goal is to develop and introduce a rational, robust system of measuring environmental performance that can be used to level the debate and leave stakeholders, decisionmakers, and the lay public feeling surer of their footing as they influence important choices about what will be done with our waste for the next several decades.

There are many influences that led to the ultimate system presented here as a solution to these problems. Professional experience played a strong role; mentors encountered along the way surely affected the viewpoints presented here. But perhaps the strongest factor of all is location…

As Columbia University is located in New York City, and I have spent the majority of my life in and around New York, this ultimate metropolis stands out as an example of the need for more objective analysis of waste problems. Opposition to anything that isn't explicitly "recycling" in the narrowest sense of the term is passionate – albeit from a vocal minority of activists. This has led to the death of otherwise rational approaches to the waste problem here and to the current export based system. Figure 1, from an article in *The New York Times*, shows just how far New York City's waste travels – and how expensive it is to get it to these places (Lipton 2003).

The opposition to waste to energy is firmly entrenched in New York's environmental community (Sexton 1994). It is likely that this opposition has its roots in the dirty old incinerators that could be found operating in the basements of many New York City highrise apartment buildings up until the late 1980's (Miller 2000). These devices had no environmental controls whatsoever, and were proven health hazards that may have proved to be galvanizing forces for groups opposed to incineration on any grounds.

The problem of exporting waste is not confined to New York City Proper – Long Island is facing similar problems as old incinerators reach the end of their useful lives and opposition to newer facilities grows (Rather 2005). Nor is the problem limited to the New York area. California's opposition to WTE is well-known (Melosi 2005). While The Golden State leads the way in many environmental initiatives, the passion of the well-organized environmentalist community there can perhaps get in the way of otherwise rational needs. Indeed, as will be seen in Chapter 1 California, despite its great advances in recycling, still disposes of far more waste than any other state in the country.

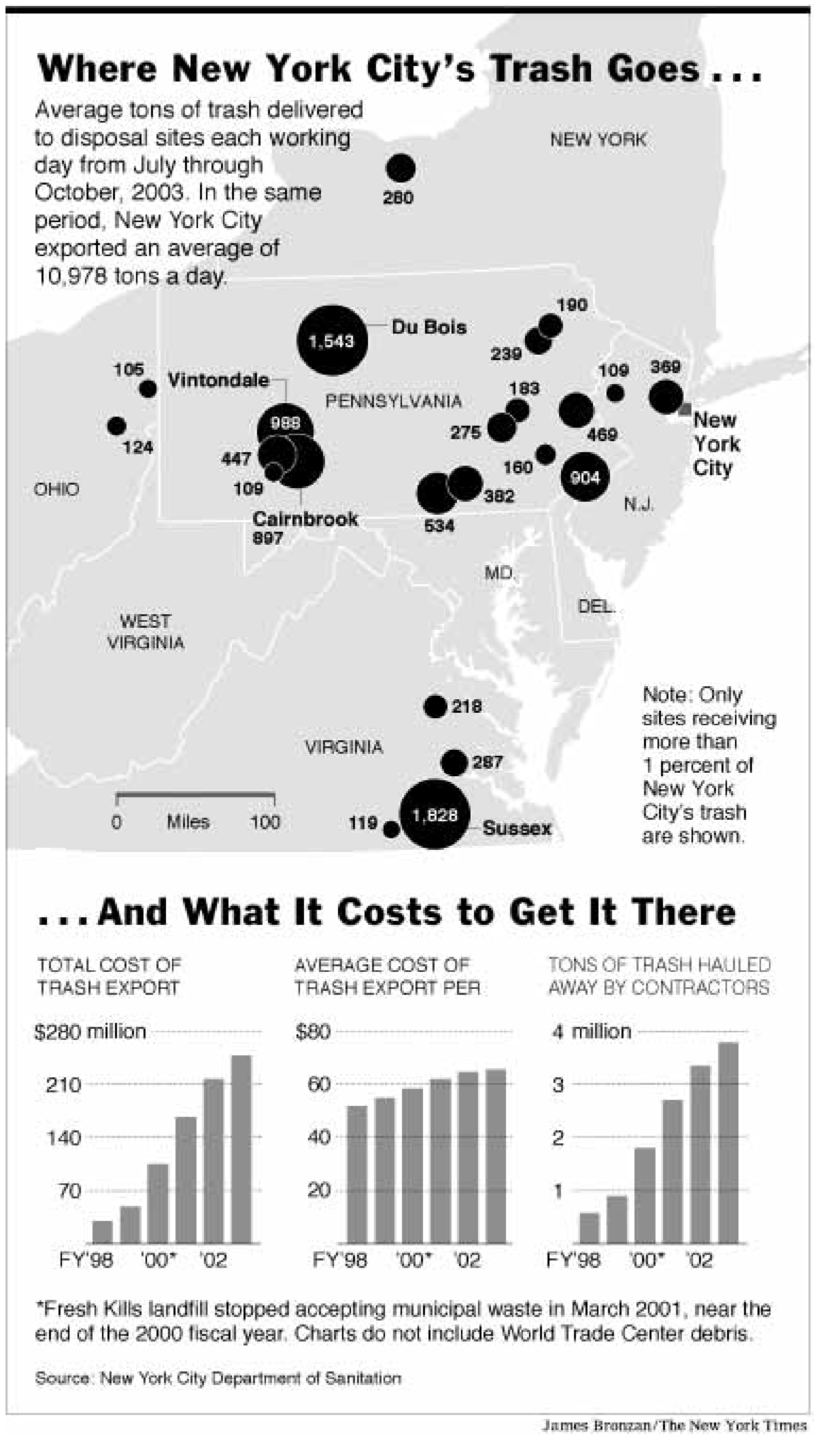

Figure 1 NY Times Graphic on Disposition of NYC Waste

The overall system of waste management in the United States is somewhat irrational. There is very little top-down planning. Most of the “oversight” is voluntary by nature, and comes from the US Environmental Protection Agency (EPA). In this office, measures such as non-binding recycling targets are set, and there are often educational (usually web-based) guidelines for achieving these targets (EPA 2005). It falls to states and municipalities to pass laws that bind entities to specific targets.

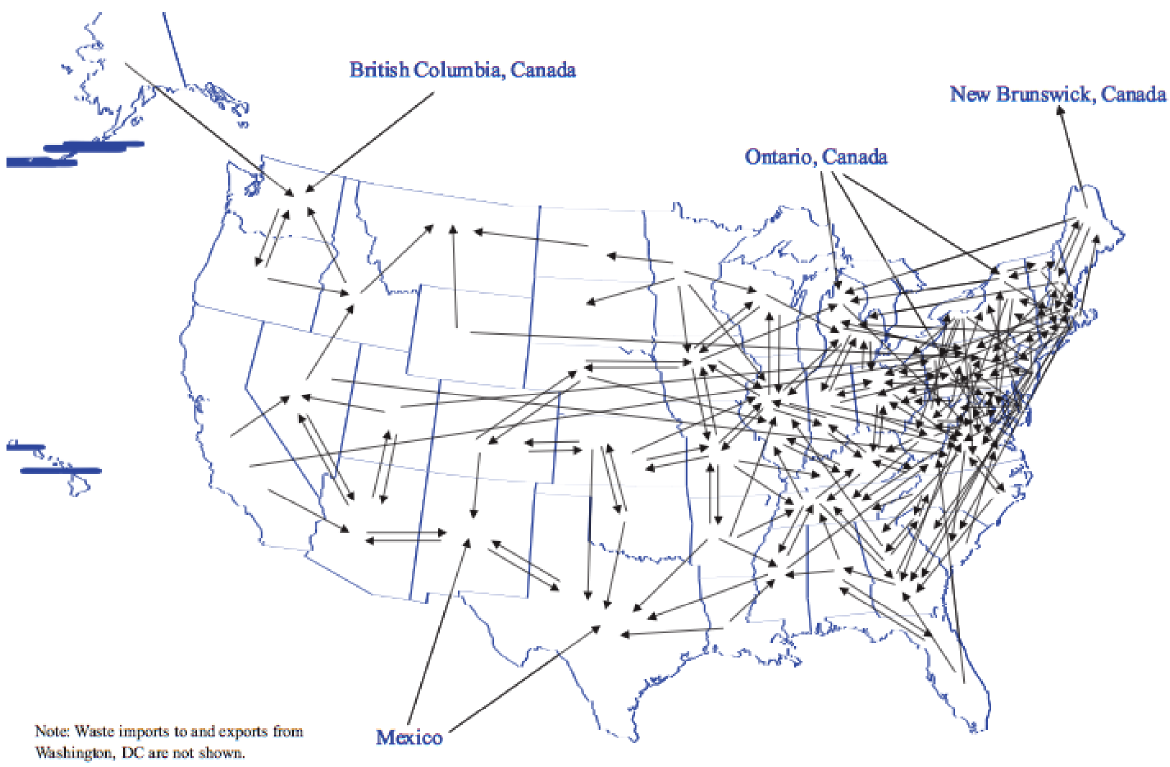

Figure 2 Map of exports of MSW between US States

This leads to the concept of goal-oriented waste management (Maystre and Viret 1995). Perhaps because of the tone of the national conversation, which as mentioned has been dominated by small cadres of passionate activists and somewhat bereft of rationality; perhaps because of the very nature of the problem – waste is dirty, smelly and not something most people want to think about; perhaps due to the shady, criminal elements of waste management in this country; whatever the reason, attitudes towards waste management have evolved haphazardly. Discussions about MSW often revolve around lofty goals such as "zero waste" on the one extreme, and thrift or expediency on the other (Tierney 1996), without a strong enough understanding of realities on the ground.

What is really missing from these discussions is a rational, objective system of measurement and analysis that levels the playing field and provides a basis of understanding that steers the conversations away from "feelings" and more towards actionable facts.

The ultimate goal of this thesis is to address two main concerns about MSW management in the US

1. The lack of readily accessible, transparent, and reliable data
2. The lack of an intuitive, objective methodology for analyzing the environmental effectiveness of waste management systems and policies.

Much of this can be accomplished by introducing the concept of "goal oriented waste management" to the national discussion.

Goal-Oriented Waste Management

Much of the information currently published about MSW – including data found in the EPA and BioCycle/EEC State of Garbage reports – simply quantifies flows of materials. Outside of vague, idealistic goals such recycling as much as possible, there is little discussion about what to actually do with all of this waste.

The EPA has for a long time published a hierarchy of suggested waste treatment options (EPA 2006). While this is a useful set of guidelines to aspire to, it does not constitute a workable set of goals. If the US is to maximize the efficiency and environmental performance of its MSW management systems, it is essential to clearly articulate specific goals. Steps towards the attainment of those goals can then be measured and evaluated.

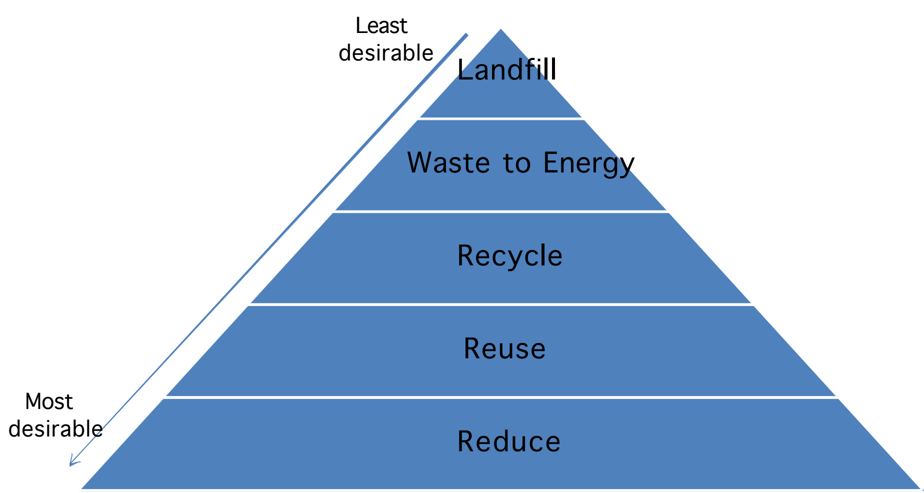

Figure 3 EPA Waste Hierarchy

The Austrian Waste Management Act serves as a good guideline for at least starting the discussion about goals for US waste management. There are three paramount goals established under the Act (and a similar, Europe-wide set of rules called the “European Waste Directive”), and they are as follows: (Brunner and Rechberger 2004)

1. Protect human health and the environment
2. Conserve resources such as materials, energy, and land

3. Treat wastes prior to disposal so they do not need aftercare when finally stored in a landfill.

No one is likely to argue with the first goal and, to a large extent, it has already been reached by the developed world (Brunner and Fellner 2007). Even though many may find it objectionable to landfill or incinerate wastes that may – under the right circumstances – have “higher uses, it is difficult to question the tremendous progress we have made as a society in the past century or so. Whereas in the late 1800’s and early 1900’s one would be likely to find refuse piled on city streets or dumped into rivers and oceans, contemporary thinking finds this almost entirely unacceptable and modern practice renders it archaic. State of the art landfills and waste to energy facilities have a much lesser effect on the environment than their predecessors, and the technologies are improving all the time.

As far as resource conservation is concerned, energy/materials recovery schemes have dramatically increased performance in that area as well. But this is where a goal-oriented approach is likely to have the greatest impact in the developed world. As can be seen from the map of exports between states (Figure 2), we are currently a long way from a rational, goal-oriented system of this nature. It is hoped that the research presented in this paper will push us in that direction.

The research is broken up into two main components. Chapter 1 presents the problem of waste data acquisition. Up until now, there have been two main methods of data acquisition in the US – the EPA/Franklin “Facts and Figures” study (EPA 2007), an annual publication of the EPA; and the *BioCycle*/EEC “State of Garbage in America,” a biannual study comprising research largely conducted by researchers at Columbia University, including the author of this thesis.

Each of these approaches have their strengths and weaknesses, and it can be argued that the solution we have come up with – an interactive, web-based database called "WasteMap" – takes the best of both approaches and combines them into a more unified whole. In any event, the methodology behind the data acquisition for WasteMap is presented in Chapter 1. This presentation comes in the form of case studies performed for the EPA Region 9, in which detailed waste data was collected from three states under their jurisdiction – California, Nevada, and Hawaii. Data from these states are analyzed to draw conclusions about the overall state of MSW data collection, and some of the lessons that can be drawn from higher-quality data once it is made available. Finally, the development and Flash® programming of the geographical, web-based database is illustrated.

The second component of the thesis is the search for – and ultimate development of – a system for rationally quantifying the effectiveness of waste management systems is detailed. This component is broken up into two chapters. The first, on statistical entropy, goes over the efforts we have made to extend an already-existing system of environmental analysis to cover a broader range of materials. More specifically, we extend the system of statistical entropy analysis – previously applied only to metals – to account for the flows of carbon through waste management systems.

Though we were pleased with the results of this extension, and are impressed by the concept of statistical entropy as originally devised by its creators (Rechberger and Brunner 2002), it ultimately did not accomplish what we were hoping to achieve – a scientific approach to MSW management systems assessment that is rigorous enough to gain the respect of various stakeholders in the industry, but intuitive enough to be easily grasped an understood by the public and political decisionmakers.

We believe we have gone a long way towards accomplishing those goals with the creation of a new metric to measure the efficacy of waste management systems – Resource Conservation Efficiency, or RCE. This new metric, based on life cycle assessment (LCA) and cumulative energy demand (CED), provides a means for making apples-to-apples comparisons between cities. It is fully explained in Chapter 3, where we also apply it to the analysis of waste management in two American cities – Honolulu and San Francisco. These cities were chosen largely because they rely upon two competing strategies for disposing of non-recycled waste – WTE in the case of Honolulu and landfilling in the case of San Francisco. The results are unexpected, and we hope to stimulate a great deal of discussion once this new metric is introduced to the larger community of waste management stakeholders.

2. Background and Literature Review

Environmental Evaluation options

There are many possible reasons to "evaluate" waste management systems for municipalities. A top concern is economic – different regions, for example, have different economic realities. For example, many western US states are restricted to landfilling as the primary means for waste management because land is so cheap and abundant, it is virtually free to bury waste there (Simmons, Goldstein et al. 2006). Other states, such as New England, face the opposite problem – land is relatively scarce and expensive, so alternatives to landfilling such as recycling and waste to energy (WTE) are more prevalent.

The other main arena for waste management analysis is that of environmental performance. Here there are numerous options and strong precedents to follow. One of the "fathers" of what may best be termed as "environmental quantification" is Robert U. Ayres, who was among the first to introduce the concept of "industrial metabolism," which is a core principle of Industrial Ecology (Ayres 1999). Among the fundamental ideas of this school of thought is that Earth systems, like the human body, have a metabolism, and that the corresponding use of materials – much like nutrients in the human body – must be accounted for.

Materials Flow Analysis

The resulting discipline of Industrial Ecology has largely relied upon Materials Flow Analysis (MFA) to do this accounting. MFA has been defined by Brunner as a method to assess flows and stocks of materials within a defined system. A description and illustration of MFA is presented in Chapter 4 of this thesis. For now it is sufficient to point out that MFA relies upon the law of conservation of mass. This implies that all materials must be accounted for in the system being examined – input must equal outputs plus stocks.

The core strength of the MFA approach is its ability to intuitively organize materials flows. MFAs can be performed for substances. For example, if one were interested in the flows of copper through an economy, as in Spatari et al, MFA provides a set of protocols resulting in a finished product illustrating sources, sinks, and usage pathways for the substance (Spatari, Bertram et al. 2002). The Spatari study produced a "comprehensive accounting of the anthropogenic mobilization and use of copper" in which the copper flows through all life cycle stages were quantified, from "mining and processing, fabrication, utilization, and end of life."

These kinds of studies, along with the underlying philosophy found in seminal works like Ayres, allow for a kind of "balance sheet" for vital resources. Graedel, another prominent figure in the field of Industrial Ecology, has embarked on a program through Yale's Center for Industrial Ecology called "The Stocks and Flows Project." In this project, the stocks of important earth resources are being catalogued, along with the current metabolism – rate of material extraction, period of use in the technosphere, and pathways for disposal and/or recycling (Johnson, Harper et al. 2007). It has been suggested that the Stocks and Flows project will attempt to catalog in this way all of the elements on the periodic table (Graedel 2008).

It is easy to see the attractiveness and utility of such a project. It has often been lamented that, at least since the Industrial Revolution, we have treated natural resources as if we have a blank check, with a bottomless reservoir of funds upon which to draw. The important step of drawing account balances – and methodically accounting for the "funds transfers" of these accounts – would go a long way towards the vital goal of arriving at some kind of state of sustainability for society. Indeed, it would seem that without such accounting procedures in place, sustainability would be a forever-elusive goal.

So, it is clear that MFA is a necessary tool for sustainable development. It is, however, a descriptive tool, not an analytical one. For the actual analysis of the environmental effects of human activity, we must turn to one of the suite of tools that have been developed for this purpose.

Daniels and Moore provide an excellent overview of the common analytical tools available for this next step in environmental quantification (Daniels and Moore 2001; Daniels 2002). In their two-part paper "Approaches for Quantifying the Metabolism of Physical Economies," they break up the set of examined tools into two major categories – "macro-scale tools" and "micro-scale" tools.

Macro-scale tools "tend to cover the totality of economic activity as the relevant container of the driving forces behind material and energy flows or requirements and look within a defined geographic region, over a given period of time." These approaches include, but are not limited to, the following:

- Total Material Requirement and Output (TMRO)
- Bulk Internal Flow MFA (BIF-MFA)
- Phyiscal Input-Output Tables (PIOT)
- Substance Flow Analysis (SFA)

- Ecological Footprint Analysis (EFA)
- Environmental Space (ES)

Some Macro-scale tools for Environmental Analysis

The first four approaches – TMRO, BIF-MFA, PIOT, and SFA, are some of the basic tools of MFA. TMRO is an approach that examines materials flows at aggregated levels, usually nations or regions. This aggregated approach is useful for comparing the ecological impact of different countries.

BIF-MFA, was described earlier above under the more general term "MFA," and is used to examine materials flows and quantify the metabolism of regions, cities, countries, etc. It is the foremost tool for materials accounting within the boundaries defined under the methodology.

PIOT are also used primarily for national-level analyses. They are similar to conventional input-output accounts, but extend upon them by incorporating "environmental resource and waste output 'sectors' to provide measures of the physical flow of materials and goods within the economic system and between the economic system and the natural environment." (Daniels and Moore 2001) This is done by tracking the movements of materials and evaluating the environmental burden of activities by quantifying the demand on studied resources by the economy.

SFA, also mentioned above, focuses on the flows of one chemically defined substance through the metabolism of a defined region. Again, as a macro-scale tool, this methodology tends to encompass all economy-wide sources acting as driving forces behind the flows of the examined substance. It can also be used on a micro scale to trace a substance through the metabolism of a given technology, but this is a rarer application.

(In the Entropy chapter of this study, we will use a form of SFA to examine carbon in waste management systems, certainly a micro approach.) SFA is often used for high impact, "indicator" substances like mercury, lead or other heavy metals, and water, carbon, etc. (Baccini and Brunner 1991)

EFA was intended as an intuitive approach to quantifying "carrying capacity" – or the ability of the Earth to supply resources. It calculates material and energy requirements of regions and converts these calculations into the "ecologically productive land area required to produce the resources used in these activities." (Daniels and Moore 2001) The resulting "footprint" can be used to compare the resource intensity of different regions. There are also many online footprint calculators that allow individuals to calculate their own footprint, such as the Australian Conservation Foundation's version (ACF 2008).

The final macro approach examined in the Daniels paper is Environmental Space (ES). ES analysis attempts to quantify sustainable development by "comparing resources demands with available 'environmental space.'" The concept of environmental space is closely related to carrying capacity, and is essentially a measure of the earth's supply of available resources to satisfy the needs of humans. The analysis is broken up into categories (e.g. energy, raw materials, waste assimilation, etc.). Ecological limits are quantified and actual consumption patterns in the examined region are compared against these limits to yield the level of sustainability achieved.

Some Micro-scale tools for Environmental Analysis

The micro level approaches, in contrast to the macro level approaches, focus primarily upon individual or small groups of products or services within the economy. The micro scale systems of analysis described by Daniels are:

- Material Intensity per Unit Service (MIPS)
- Sustainable Process Index (SPI)
- Company level MFA
- Life Cycle Assessment (LCA)

MIPS attempts to account for the primary material and energy requirements in specific products and services. MIPS is reported as the ratio of the environmental burden of a product or service to a given unit (physical or monetary) of services provided to society. By reporting environmental impact in this manner, different processes or products can be compared against other, similar processes or products and improvements can be made in the environmental burden per unit of output.

The SPI calculates the total land area required to provide a product or service, expressed in square meters of land per service/product. It compares this land area requirement with the available, natural supply of land and its associated geogenic flows and determines the sustainability of the given product or service. In this way, it is similar to the EF, though on a micro rather than macro scale.

Life Cycle Assessment

The final micro scale technique is LCA. LCA is a methodology used to assess the "cradle to grave" – or the entire life cycle, from raw materials extraction to final disposition – environmental impacts of a product or service (EPA 2008). The main steps involved in LCA include:

- The Life cycle inventory (LCI), where relevant material and energy inputs and environmental releases are compiled
- Evaluation of environmental impacts associated with inputs and releases
- Interpreting results and recommending actions.

LCA has evolved a set of internationally-accepted standards for accurate life cyle analysis. These standards, developed largely under the auspices of the Society of Environmental Toxicology and Chemistry (SETAC), have been codified into International Standards Organization (ISO) 14000-level standards (UNEP 2008). The SETAC approach involves the following steps:

- Pre-analysis
- Life Cycle Inventory
- Life Cycle Impact Assessment
- Interpretation and recommendations

In the pre-analysis stage, the motivation for performing the LCA is determined and articulated. In addition, the scope is decided upon – which, if not all, of the stages of LCA (raw material acquisition, manufacturing, use/reuse/maintenance, and recycle/waste management) are to be included in the study. For the study to qualify as a full LCA, all four stages are required, but there are some occasions when sub stages may have to be left out. This is usually the result of limited time or other resources to complete the full study (Curran 2006).

The life cycle inventory is the process of quantifying energy and raw materials requirements on the one hand and emissions (atmospheric, water, solid waste, etc.) on the other. The steps required in the inventory stage include the formulation of a data collection plan; the actual data collection; and the evaluation and reporting of the results.

The next step – life cycle impact assessment (LCIA) – is the meat of the analysis. Here, potential human health and environmental impacts of the cataloged energy/materials flows and associated emissions are identified. There are several varieties of impact assessment. The two most common are the "midpoints" approach, including the CML/SETAC method and "damage-based" methods, including the well-known and utilized EcoIndicator method.

In the midpoints approaches (also known as "problem-oriented") flows are classified into "damage themes." These usually include climate change, resource depletion, stratospheric ozone depletion, acidification, photochemical ozone creation, eutrophication , human toxicity and aquatic toxicity (UKS 2008). Inventory data is fed into these categories and impact factors are assessed. Scores are then normalized. For example, emissions of a particular pollutant from a product's life cycle are compared with the typical emissions impact of that same pollutant per person per year in the country under examination.

In the damage-based method – the most prominent of which is EcoIndicator – damage themes are also used, but they are somewhat broader, consisting of damage to human health, ecosystem health or damage to resources. The main steps in the damage-based method are

- Inventory analysis
- Damage analysis
- Normalization and weighting

In the first step, an inventory analysis is performed as described above for general LCAs. The damage analysis step converts the results of the first step in terms of changes in human health, ecosystem and resource depletion. This step consists of a fate analysis, where the transfer between environmental compartments and the fate of substances (for example, how degradable a substance is once released) are modeled. Next, the exposure is assessed – that is, in what concentrations is the substance picked up by people or the environment. These emissions are then analyzed for the likelihood of their causing diseases or other adverse effects. Finally, the damage analysis can be performed where, for example, the probability of a certain exposure causing cancer is calculated; the average age people get this cancer is factored in; and the years of life lost due to this exposure is inventoried (Goedkoop and Spriensma 2001). Many substances are calculated on a European scale. Greenhouse gases and a few other emissions are calculated based on worldwide levels.

In the weighting step, scores are normalized so that the impact of each emission is weighted according to (in the case of the EcoIndicator method) a metric called DALY – disability adjusted life years. This is essentially the life years lost due to the life cycle impact of the product, and is thus a measure of human health impact. Ecosystem quality is measured according to species lost per unit area over a given time, and damage to resources is calculated as the excess energy needed for additional extractions of minerals and fossil fuels attributable to the examined product or service.

Out of the tools described above, LCA is perhaps the most widely recognized and utilized. This is reflected by the ISO standards that have evolved around LCA, and the suite of software options (both commercially sold and freely distributed on the web) available for use. (Input-output LCA is a prominent example of web-based LCA tools, and it is free to use on the internet at www.eiolca.net. It is programmed and maintained by Carnegie Mellon University. (EIOLCA 2008))

There are, of course, some inherent problems with the LCA method. It is, as mentioned, time and resource intensive. Data availability can be a problem – in fact, the inventory stage is often the most time-consuming, as data sources can often be quite dispersed and difficult to find. Traditional LCAs do not account for the cost-effectiveness of examined processes, products, and services, and so must be combined with other tools if this information is required. Finally, when performing the impact assessment, some subjective choices must be made, especially in the weighting step – which impacts are deemed most important will often vary from practitioner to practitioner (Curran 2006).

Nonetheless, LCA is a powerful tool for assessing environmental impacts. And, as waste management systems are essentially the final step in product life cycles, it makes a great deal of sense to treat them as the sum of the lifecycles of individual products. Largely for this reason, LCA is chosen as the method for verifying the usefulness of our new metric – Resource Conservation Efficiency (RCE) – as an environmental indicator. Because it is the final stage in the life cycle of products, and because of the increasing prevalence of LCA as an environmental metric, there is strong precedence for the use of LCA to evaluate waste management systems. A review of a sampling of these studies can be found in the following section of this chapter. Before moving on, however, it is necessary to mention an important outgrowth of the life cycle inventory that will be the basis of the RCE metric discussed in this study – Cumulative Energy Demand, or CED.

Cumulative Energy Demand (CED)

CED is a cradle to grave account of the energy inputs and consumption necessary to manufacture, use, and dispose of a product. All inputs from each stage in the life cycle are accounted for in energy terms, including direct energy inputs; feedstock materials; and capital goods (Blok 2006). This total of cradle to grave energy inputs is the CED of a particular product or service.

CED is not surprisingly often used to evaluate energy systems, and we have included several examples of this in the chapter on RCE. However, it is useful in this background review to highlight a few examples of studies that have been conducted using CED to provide a better sense of the power of this methodology.

Rohrlich et al used CED to examine the efficiency of lignite extraction for use in electricity production. In examining all life cycle energy inputs – from site preparation, including clearance and disposal of vegetation to mining activities, including ore extraction and overburden disposal – it was determined that the fraction of primary life cycle energy demands for lignite mining accounts for 6.2% of the total CED for electricity from lignite production. This leaves 93.8% of lignite energy content available as useful energy (Rohrlich, Mistry et al. 2003).

CED clearly makes a great deal of sense for examination of energy systems, but it is also useful for other disciplines. Rozycki et al used CED to examine the system-wide environmental impacts of German high speed rail. They looked at resource consumption caused by the manufacture and use of the trains themselves in addition to the construction and operation of the supporting infrastructure. In so doing, they were able to determine which life cycle stages drew the highest energy demand and therefore identify those areas where improvements could be made. Their results showed that the infrastructure was responsible for 13% of the CED per 100 person kilometers, while the actual locomotion of the trains dominated the energy demand. They therefore pinpointed locomotion as a key area for study (Rozycki, Koeser et al. 2003).

CED has been shown to be a useful screening indicator for full life cycle studies. As will be discussed in some detail in Chapter 5 on RCE, CED correlates rather closely with environmental indicators. We were able to take advantage of this fact to design the RCE metric and deploy it in a way that will allow individuals and organizations that are not practitioners of LCA to evaluate waste systems in an intuitive but still scientifically rigorous manner.

Previous use of LCA and CED in Waste Management Evaluation

LCA has a significant history of use as a tool to evaluate waste management systems. It is not possible to cover all of the referenced studies on LCA and MSW, but it is useful to trace some of the history and point out notable studies. It should be pointed out, however, that "survey studies" – reports compiling the results of large numbers of waste-related LCA studies – do exist and can be consulted. One such study, by a British organization called WRAP, summarized the results of 55 separate LCAs in determining that recycling is a net environmental benefit for most materials (Georgeson 2006). In addition, their exhaustive international literature review identified over 250 reputable LCA studies on MSW.

Life Cycle Inventory studies

There are a number of papers that focus on life cycle inventories of "generic" waste technologies. Camobreco, for example, used data both from actual landfills and the literature to develop an LCI for a typical modern MSW landfill (Camobreco, Ham et al. 1999). In this study, he used the average composition of US MSW, then modeled three separate time scenarios – a short term (20 years) corresponding to the landfill's most active period of decomposition; an intermediate term (100 years), corresponding to what he refers to as a "generational" life span; and a long term (500 years), or what he refers to as an "indefinite" time period. His model inventoried total material and energy inputs over the given timeframes, in addition to all significant emissions to air, water, and land.

Harrison performs an LCI for a typical combustion facility, delineating material-specific energy generation and emissions (Harrison, Dumas et al. 2000). McDougall has an entire book on LCI for waste, in which some original research and a great deal of literature sourcing and review provides very useful LCIs for all relevant waste management materials and processes (McDougall and White 2001). Finally, Finnveden et al performed a comprehensive LCA on MSW management, for which they published inventory data in the form of SimaPro inputs (Finnveden, Johansson et al. 2000). (SimaPro is a popular LCA software package.)

As a final example of the available LCI data sets, Weitz et al performed a full LCA on the management of typical components of US MSW for the EPA, for which they also published a complete set of inventory data. Though this study focused primarily on materials suitable for curbside recycling in the US, it is an extremely thorough and well carried-out work, and is used as a basis for much of the work in this thesis (Weitz 2003).

Life Cycle Analysis Studies on MSW Management

LCA studies on waste tend to be comparisons between different strategies and technology suites for waste management. For example, Arena et al looked at various treatment options for different kinds of packaging waste. Their results indicated that, for readily recyclable plastic materials such as PET, there are significant energy and environmental savings to be achieved from recycling plastics (Arena, Mastellone et al. 2003). For example, from an energy perspective, PET recycling is said to save between 22 and 25 MJ/kg over virgin production plus disposal (landfilling) of the same material.

Often, LCAs on waste management will be in the form of comparisons of the status quo of a particular city versus some hypothetical scenario. Craighill, for example, looked at the global warming, acidification effects, and nutrification of surface water associated with recycling the MSW produced in Milton Keynes (Central England) versus disposal (in this case landfilling) plus virgin production. He found that the recycling system performs better across all the given metrics (Craighill and Powell 1996). He also went on to evaluate the net economic impact of recycling versus disposal systems and found that, for most materials (with the exception of HDPE, PVC, and PET[1]) there is a significant net economic benefit to recycling.

A well-referenced study (EDF 1995) commissioned by the Environmental Defense Fund (EDF), a prominent environmental NGO based in New York City,

[1] It should be noted that this study was performed in 1995, and the economic benefits of recycling plastics have increased since that time.

examined the environmental effects of virgin vs. recycled paper. This full lifecycle study compared three competing scenarios against one another:

1. Acquisition of virgin fiber + manufacture of virgin paper + landfilling
2. Acquisition of virgin fiber + manufacture of virgin paper + incineration
3. Manufacture of recycled paper + recycling collection/processing/transport to site of remanufacture.

The measured environmental effects were categorized in terms of solid waste output, energy use, air emissions, and waterborne wastes. The findings suggested that Scenario 3 offers the best performance for aggregated paper grades (the cumulative effects of managing newsprint, corrugated board, paperboard, and high grade office paper) across all impact categories. In other words, the recycling option generated the least waste, required the least system-wide energy, and produced the fewest water and air emissions than either virgin manufacturing scenario. The LCI for this study was gathered through both literature review and contacts with actual manufacturers. The impact assessment was performed by a panel of experts from various backgrounds – industry, academia, government and non-governmental organizations.

Finnveden et al, as mentioned earlier, produced a major LCA study focusing on MSW in Europe (Finnveden, Johansson et al. 2000). The study's reports include LCIs and detailed analysis of the LCAs performed. The LCAs included evaluations of the environmental and energy effects of different waste treatment options by material, in addition to more aggregated results for the treatment of MSW as a whole. The aggregated results showed that recycling systems perform better than virgin-based systems for MSW in general across the broad impact categories – especially with respect to total energy use and emissions of greenhouse gases.

Summary of Waste LCA Findings

As evidenced by this review, recycling tends to be the most favored option from an environmental perspective for most materials in the MSW stream. The number of studies – and the reputability of the practitioners – obviates the need to reproduce LCAs of waste management by material or system. The results of previous studies also tend to agree that the use of WTE has environmental and energy benefits that fall somewhere between those of recycling and landfilling.

Use of CED in Waste Management Evaluation Studies

Though not always invoked by name, CED is a frequently cited metric when discussing the relative merits of different waste treatment options – especially when recycling is involved in the discussion. Many of the above-mentioned studies focus a great deal of attention on the life cycle energy (or CED) of different products with respect to MSW management. In addition, there are other books and papers that either explicitly or implicitly employ CED as a central metric in their analyses.

Perhaps the best-known waste related CED based study is a book called "Energy Savings by Wastes Recycling," edited by Porter and Roberts (Porter and Roberts 2005). This book presents the results of a study commissioned by the European Commission (EC) to examine potential energy savings available to the European Union through recycling. Much like the EDF paper, it relies on a combination of literature review and industry experts to calculate life cycle energy inputs for a variety of materials, most notably paper, aluminum, and glass. It compares energy savings available from recycling to disposal-based scenarios. The scope is purely energy-based – there is no further attempt to quantify the environmental effects of management options.

A final example of the use of CED to evaluate waste management options – this time explicitly – is a paper by Truttman and Rechberger incorporating the use of CED to measure the contribution to resource consumption made by the reuse of electronic household waste. This fascinating study attempted to evaluate the merits of the increasing pressure in European nations to force the reuse of electronic equipment as a means to savings resources. The authors utilized CED to calculate the energy savings from the extensive reuse of electronics. Among their conclusions was that in the aggregate (i.e. when combining CED for all examined electronic products), there is a 12% energy savings. The authors additionally observed that emissions reductions were similar in scope to energy reductions since "energy and emissions are usually closely connected (Truttmann and Rechberger 2006)."

This last observation is crucial to the argument that will be advanced in Chapter 5 (RCE) – that CED-based metrics are useful screens to full LCAs due to their correlation with environmental impact measures.

Previous studies on Waste Data Acquisition

Chapter 3 details the work we have done on waste data acquisition and analysis. In it, the two best-known publicly available sources of national US MSW data – the *BioCycle*/Columbia Earth Engineering Center "State of Garbage in America Survey" (SOG) and the EPA/Franklin "MSW Facts & Figures" report are explained and contrasted. Before moving on to the analysis, however, it is useful to quickly review some approaches to MSW data acquisition and analysis.

Brunner explains that there are two main methods of acquiring MSW data – *direct* and *indirect*. The direct method relies upon such techniques as waste characterization studies that take statistical samples of actual MSW to determine the amounts and makeup of solid waste. The indirect method, on the other hand, uses data sources such as production data from commodity organizations, economic data, and census data to arrive at estimates of materials consumption (Brunner and Rechberger 2004).

For our US MSW Database (MSWDB or WasteMap) project (described in detail in Chapter 3), we suggest a combination of each of these approaches. As it stands now, SOG has a handle on disposal data (measured directly), EPA/Franklin has an excellent grasp of recycling tonnages (measured both directly and indirectly), and there are additional directly-measured waste characterization studies that we used to fill out the data gaps in test-case states for the WasteMap.

The proposed WasteMap methodology is to combine these already-existing data sources and perform targeted waste characterization studies to determine the actual capture rates of various materials in the waste stream. This methodology was developed independently as a result of our involvement with the SOG in addition to contracted work with EPA Regions 3 and 9. In wrapping up our work in this area, however, we came across a similar project in Australia – the Australian Waste Database (AWD).

The stated aims of the AWD are to agree upon a national classification system for solid waste; establish a protocol for sampling and characterizing waste; and to establish a national waste generation database to provide waste data information by region for benchmarking purposes (Moore, Kung et al. 1994). The data sources appear to be similar to those we have identified for the MSWDB – state EPAs and "waste authorities." We discovered the existence of this project too late in our research to incorporate it in this thesis, but it is hoped that there will be communication between the WasteMap project developers and those of the AWD to help ensure best practices in the development of both.

3. Municipal Solid Waste Data Collection and the Creation of a National Waste Management Database

Summary

For the past five years the Earth Engineering Center (EEC) at Columbia University has led the conduct and analysis of the State of Garbage in America (SOG), a survey of waste generation and management in the fifty states published bi-annually in BioCycle journal. During this time, we have found that although there are myriad sources of municipal solid waste (MSW) data, much of this data are not transparent and also extremely difficult to find. This chapter, based in large part on research sponsored by EPA Region 9, shows how a national database can be built upon the foundation provided by the SOG survey, complemented by an in-depth analysis of state data from various sources within a state. The primary goal of the MSW database (MSW-DB; www.wastemap.us) initiated during this study is to provide a central storehouse of reliable, transparent, tonnage-based and readily available MSW data for use by policymakers, MSW managers, and the general public.

California was used as the starting point due to the high volume of data available for that state, as well as the controversy surrounding its unusual method of collecting and reporting recycling rates. Also, because of California's size, its recycling tonnage has an outsize effect on overall US national figures. It is therefore important to accurately quantify MSW management there. This study showed that California – while recycling a higher percentage of waste than most other states – overstates its recycling of MSW: Despite the tremendous success in increasing recycling percentages in California, since

passage of a major law requiring 50 percent statewide diversion, per capita disposal rates have remained flat and even increased (after an initial period of decline in the immediate aftermath of the law's passage). This indicates that recycling or diversion rate targets are not comprehensive enough, by themselves, to ensure or encourage a sustainable, integrated waste management system.

Introduction

While increasingly ubiquitous around the globe, the US has yet to adopt a comprehensive, goal-oriented framework for integrated waste management (IWM) at home. The closest we have come is the well-known EPA Hierarchy of waste management options, depicted in Figure 4 (EPA 2006).

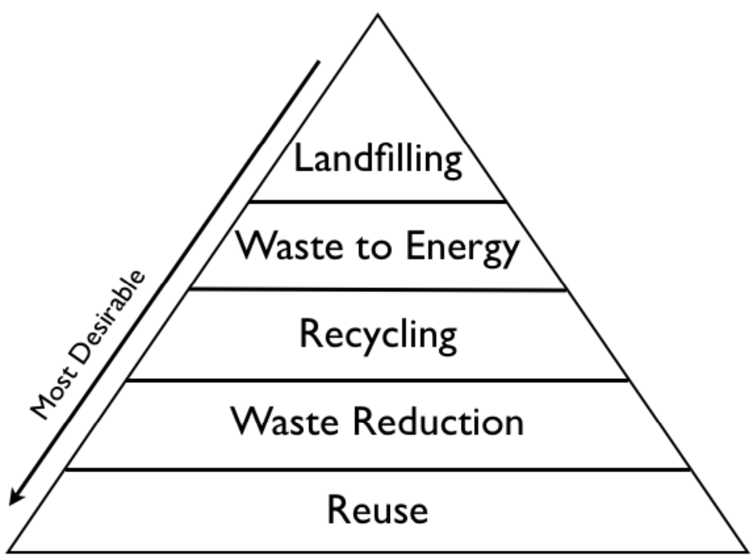

Figure 4 EPA Hierarchy of Waste Management

Though perhaps useful as a guideline, recycling rate targets by themselves are not sufficient goals. It will be demonstrated later in this chapter that, in fact, recycling rates only provide limited information as to the overall success of waste management systems.

Instead, the national aim should be to develop a system of rational, measurable goals for an integrated waste management system that:

- Conserves resources – materials, energy and land
- Protects the environment
- Treats wastes prior to disposal so they do not pose a problem for future generations.

In order to establish such a system, flows of waste must be accurately measured. This is the main focus of this chapter – the establishment of a US national interactive web database that accurately measures flows of municipal solid wastes (MSW) in the country.

Existing Sources of MSW Data

Currently, there are two major national studies of municipal solid waste (MSW) data in the US – the EPA/Franklin "Facts & Figures," an annual report commissioned by US EPA (EPA 2006); and the BioCycle/Earth Engineering Center State of Garbage in America (SOG), a bi-annual study published by *BioCycle* journal (Simmons, Goldstein et al. 2006). A summary graph of the results of the 2004 SOG is shown in Figure 5.

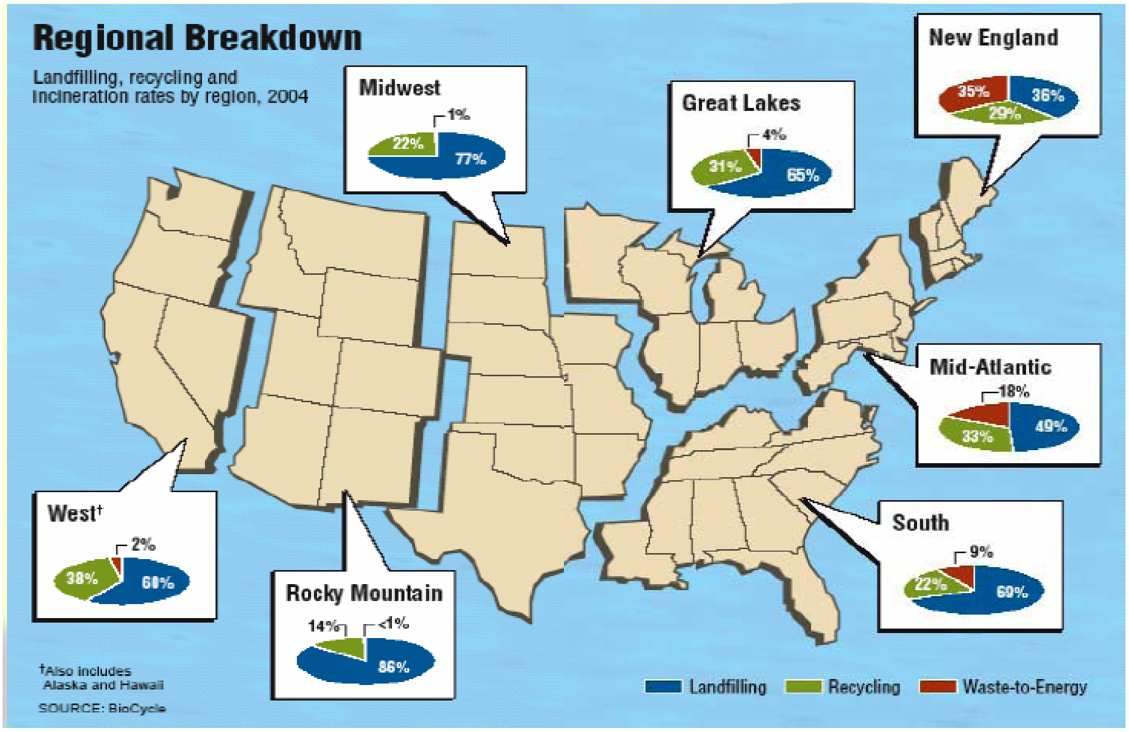

Figure 5 BioCycle Regional Breakdown Results

Franklin uses a form of Materials Flow Analysis (MFA) to perform their analysis. It is, however, quite difficult to assess the effectiveness of their approach, as the specific methodology behind their calculations is not published. A generalized (unpublished) description of the original methodology was, however, obtained for this research study(Franklin Associates 1995). While the steps taken to calculate MSW flows are not clearly delineated in this document, the underlying methodology is apparent – estimates are based on production data for materials and products that end up in the waste stream, and adjustments are made for imports/exports and for the expected lifetime of materials. The report's bibliography includes many references to the US Department of Commerce, as well as trade organizations like the US Steel Recycling Institute, Battery Council International, etc.

Because collected recyclables are used as feedstock for remanufacturing into new materials, it turns out that the Franklin methodology (which appears to rely on close contact with trade/commodity associations) does a good job of tracking the national trends of traditional curbside recyclables – i.e. metal, glass, paper, and plastic (MGPP). It falls short, however, when it comes to organics collection and, more significantly, in its estimate of tonnage of MSW landfilled. For whatever reason, industrial production data does not seem to provide adequate information for estimating disposal tons.

The SOG survey employs an entirely different strategy. Since most states have regulations requiring landfills and waste to energy (WTE) facilities to report tons received, it is possible to obtain reasonable disposal tonnage reports from the relevant regulatory authorities in each of the fifty states. Recycling tons are typically not regulated, but the same agencies tend to keep track of these figures as well – though the numbers are generally not as reliable as the reported landfilled and WTE tonnages.

The State of Garbage in America survey has been conducted since the late 1980's. EEC first became involved in 2003, and developed the current methodology, which focuses on tonnage reports rather than the previous (and more subjective) "percentage estimates" (e.g. % recycled, % landfilled, etc.) requested in other years. Detailed questionnaires are sent to representatives of the Waste Management departments of each state.

The data provided in the SOG survey are thoroughly reviewed by EEC and BioCycle researchers. The main goal of this phase of the survey is to allow for an "apples to apples" comparison of states – that is, to adjust reported values to the EPA standard definition of MSW. State officials are then re-contacted to clarify misunderstandings and fill in missing data where possible. This iterative effort results in a report that characterizes WTE and LF tons quite accurately, but which still leaves some questions regarding the accuracy of recycling tons. Figure 6 shows a comparison between the SOG and EPA results.

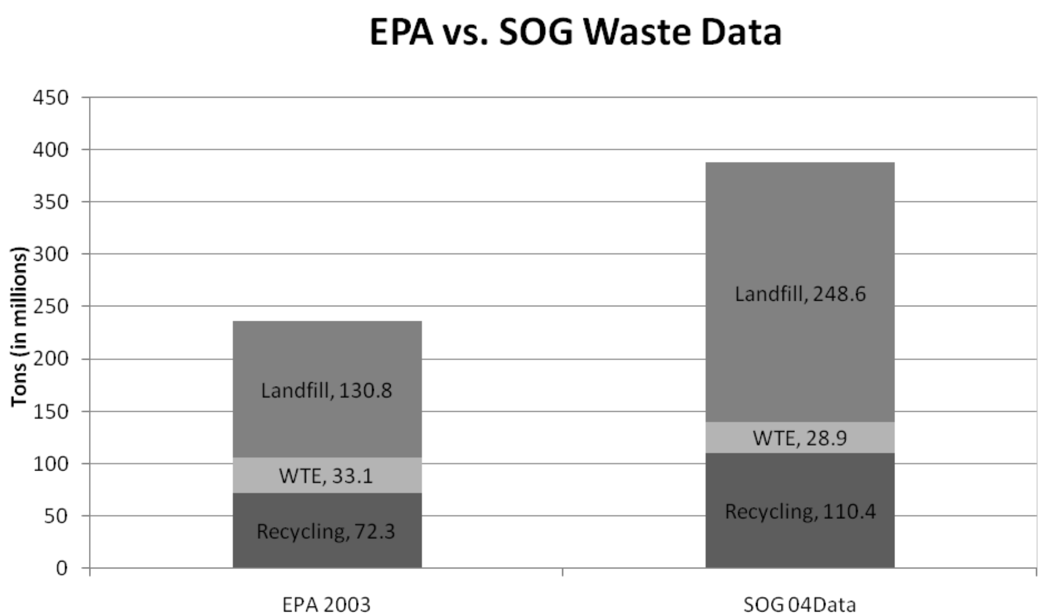

Figure 6 EPA vs. SOG Comparison

Methodology: Data Mining for EPA Region 9

As mentioned, there are several sources of municipal solid waste (MSW) data but – depending on the locality or region one wishes to assess – much of this information can be extremely difficult to find. Furthermore, once this information is located it can be very hard to decipher. This is due mostly to inconsistencies in reporting methods across regions and, often, a lack of financial resources on the part of the data collectors.

It is important to make these data – which are vital to the environmentally responsible management of resources – easier to obtain. The primary objective of the municipal solid waste database (MSW-DB) developed during this study at Columbia University was to provide a central storehouse of reliable and readily available MSW data for the general public to use.

Because so much data already exists in published form, the MSW-DB is designed to utilize, as much as possible, preexisting information. The process of aggregating and organizing this information in one location identifies data discrepancies and gaps and is likely to indicate future research needs in terms of data acquisition and reporting.

The steps taken in the creation of the MSW-DB are as follows:

1. Aggregate waste data reports published by local and state agencies to arrive at statewide estimates of waste generation, recycling, and disposal tonnages; and draw conclusions for further action (e.g., delineate needs for further data research).

2. Perform a materials flow analysis (MFA) for all four Region 9 states using existing data from a variety of sources – state reports and contacts; information (published an unpublished) provided by waste managers; and other published information – coupled with waste characterization studies to estimate waste stream makeup and flows.

3. Perform as-needed facilities level research. The focal points here are likely to be those sectors that traditionally suffer from lack of transparent data, namely material recovery facilities (MRFs) and exporters of waste scrap materials.[2]

The first step in cross-referencing municipal solid waste MSW data from various states is to eliminate – to the extent possible – apparent inconsistencies arising from different

[2] Due to time constraints, this has only been performed on a preliminary basis for this project. The results are not included in this report, but can be summarized by the authors upon request.

methods of collection and reporting. The easiest way to do this is by following the EPA definitions.

Results from this study have been published in the Columbia University new online waste management database at www.wastemap.us.

EPA Municipal Solid Waste Definitions

In 1997, the US EPA published a large document with advice for state and local governments on measuring recycling in their jurisdictions (EPA 1997). The goal of this document was to help waste management authorities across the country to publish standardized recycling data that could be compared across regions. The report was based on the expert advice of waste industry and government officials across the US and it was hoped that stakeholders would adopt these standards, thus allowing for easier top-down planning of waste systems, where appropriate. Though the standards have not been adopted universally, they are generally accepted and were certainly useful for the purposes of this study.

The EPA divided the definitions into two main parts – a "scope of materials" and a "scope of activities". (See Figure 7 and Figure 8.)

MATERIAL	WHAT IS MSW	WHAT IS NOT MSW
Food Scraps	Uneaten food and food preparation wastes from residences and commercial establishments (restaurants, supermarkets, and produce stands), institutional sources (school cafeterias), and industrial sources (employee lunchrooms).	Food processing waste from agricultural and industrial operations.
Glass Containers	Containers; packaging; and glass found in appliances, furniture, and consumer electronics.	Glass from transportation equipment (automobiles) and construction and demolition (C&D) debris (windows).
Lead-Acid Batteries	Batteries from automobiles, trucks, and motorcycles.	Batteries from aircraft, military vehicles, boats, and heavy-duty trucks and tractors.
Tin/Steel Cans and Other Ferrous Metals	Tin-coated steel cans; strapping; and ferrous metals from appliances (refrigerators), consumer electronics, and furniture.	Ferrous metals from C&D debris and transportation equipment.
Aluminum Cans and Other Nonferrous Metals	Aluminum cans; nonferrous metals from appliances, furniture, and consumer electronics; and other aluminum items (foil and lids from bimetal cans).	Nonferrous metals from industrial applications and C&D debris (aluminum siding, wiring, and piping).
Paper	Old corrugated containers; old magazines; old newspapers; office papers; telephone directories; and other paper products including books, third-class mail, commercial printing, paper towels, and paper plates and cups.	Paper manufacturing waste (mill broke) and converting scrap not recovered for recycling.
Plastic	Containers; packaging; bags and wraps; and plastics found in appliances, furniture, and sporting and recreational equipment.	Plastics from transportation equipment.
Textiles	Fiber from apparel, furniture, linens (sheets and towels), carpets and rugs, and footwear.	Textile waste generated during manufacturing processes (mill scrap) and C&D projects.
Tires	Tires from automobiles and trucks.	Tires from motorcycles4, buses, and heavy farm and construction equipment.
Wood	Pallets; crates; barrels; and wood found in furniture and consumer electronics.	Wood from C&D debris (lumber and tree stumps) and industrial process waste (shavings and sawdust).
Yard Trimmings	Grass, leaves, brush and branches, and tree stumps.	Yard trimmings from C&D debris.
Other	Household hazardous waste (HHW), oil filters, fluorescent tubes, mattresses, and consumer electronics.	Abatement debris, agricultural waste, combustion ash, C&D debris, industrial process waste, medical waste, mining waste, municipal sewage and industrial sludges, natural disaster debris, used motor oil, oil and gas waste, and preconsumer waste.

Figure 7 EPA MSW Definitions - Scope of Materials (Adapted from Measuring Recycling)

RECYCLABLE MATERIAL	WHAT COUNTS AS RECYCLING	WHAT DOES NOT COUNT AS RECYCLING
Food Scraps	Composting of food scraps from grocery stores, restaurants, cafeterias, lunchrooms, and private residences, and the use of food scraps to feed farm animals.	Backyard (onsite) composting of food scraps, and the use of food items for human consumption (food banks).
Glass	Recycling of container and packaging glass (beverage and food containers), and recycling of glass found in furniture, appliances, and consumer electronics into new glass products such as containers, packaging, construction materials (aggregate), or fiberglass (insulation).	Recycling of glass found in transportation equipment and construction and demolition (C&D) debris, recycling of preconsumer glass or glass from industrial processes, and reuse of refillable glass bottles.
Lead-Acid Batteries	Recycling of lead-acid batteries found in cars, trucks, or motorcycles into new plastic and lead products.	Recycling of lead-acid batteries used in large equipment, aircraft, military vehicles, boats, heavy-duty trucks and tractors, and industrial applications.
Metals	Recycling of aluminum and tin/steel cans, and recycling of metals found in appliances and packaging into new metal products.	Reuse of metal containers, packaging, furniture, or consumer electronics, and recycling of metals found in transportation equipment (autobodies) and C&D debris.
Paper	Recycling of paper products (old newspapers and office papers) into new paper products (tissue, paperboard, hydromulch, animal bedding, or insulation materials).	Reuse of paper products, recycling of preconsumer or manufacturing waste (trimmings, mill broke, print overruns, and overissue publications), and combustion of paper for energy recovery.
Plastic	Recycling of plastic products (containers, bags, and wraps), and recycling of plastic from furniture and consumer electronics into new plastic products (fiber fill and plastic lumber).	Reuse of plastic products (storage containers and sporting equipment), recycling of preconsumer plastic waste or industrial process waste, and combustion of plastics for energy recovery.
Textiles	Recycling of textiles into wiper rags, and recycling of apparel and carpet fiber into new products such as linen paper or carpet padding.	Reuse of apparel.
Tires	Recycling of automobile and truck tires into new products containing rubber (trash cans, storage containers, and rubberized asphalt), and use of whole tires for playground and reef construction.	Recycling of tires from motorcycles, buses, and heavy farm and construction equipment, retreading of tires, and combustion of tire chips for energy recovery.
Wood	Recycling of wood products (pallets and crates) into mulch, compost, or similar uses.	Repair and reuse of pallets, combustion of wood for energy recovery, recycling of industrial process waste (wood shavings or sawdust), and recycling of wood from C&D debris.
Yard Trimmings	Offsite recycling of grass, leaves, brush or branches, and tree stumps into compost, mulch, or similar uses; and landspreading of leaves.	Mulching of tree stumps from C&D debris, backyard (onsite) composting, grasscycling, landspreading of leaves, and combustion of yard trimmings for energy recovery.
Other	Household hazardous waste (HHW), oil filters, fluorescent tubes, mattresses, circuit boards, and consumer electronics.	Recycling of used oil, C&D debris (asphalt, concrete, and natural disaster debris), transportation equipment (autobodies), municipal sewage sludge, and agricultural, industrial, mining, and food processing waste.

Figure 8 EPA MSW Definitions - Scope of Activities (Adapted from Measuring Recycling)

Waste Characterization

The three states examined in this study perform waste characterization studies at different times, making it difficult to determine the average composition of a ton of MSW in a given year. The most recent comprehensive characterization was performed in the state of California by CIWMB in 2004. The results will be used as a reference – even for states other than California - throughout this study. Also, it has been shown elsewhere

(Kaufman In Press) that the EPA/Franklin Waste Characterization provides a reasonably accurate picture of MSW composition. It will therefore also be used in this study.

California Waste Data Analysis

Because California has the most comprehensive set of MSW reports available, it was decided to start with this state. What follows is a narrative of the journey through the vast storehouse of online MSW data and reports provided by the California Integrated Waste Management Board (CIWMB).

Aggregation and Interpretation of California Data

The CIWMB is the largest, most comprehensive and complex organization of its kind in the United States. In the Governor's 2007-08 proposed budget, over $200 million were allotted to "waste reduction and management" activities. This works out to roughly $5.50 per capita (2007). More than three-quarters of non-administrative expenditures are directly related to waste reduction and recycling. The source of this funding is provided by landfill fees that are assessed for each ton of waste sent to landfills in California (Finance 2007).

This is often a source of envy for neighboring states, which appear to be considerably less well funded. This is only partially the case. While some of the per ton fees are directly applicable to diversion activities, the $200 million funds a wide range of waste-related activities, including "the Waste Management Board for oversight of jurisdiction (city, county or region) and state agency waste management planning and diversion program implementation activities; market development activities; and oversight of local government enforcement of requirements to ensure solid waste handling and disposal facilities protect public health and safety and the environment." (Van Kekerix 2007)

While CIWMB publishes vast quantities of data – both in online databases and published reports – it is sometimes difficult to track the source of the information. This is particularly the case with recycling tonnages. CIWMB uses a complicated formula in which a base-year waste characterization is applied in combination with disposal tonnages – which are meticulously tracked by the board since their income derives from landfilling fees – to arrive at an estimated recycling tonnage. Generally speaking, then, disposal tonnages (LF + WTE) are measurement-based while recycling tonnages are estimated.

However, due to the breadth of the research conducted and funded by CIWMB, it is possible to combine the California waste information with the findings of several of the published reports to arrive at a reasonable calculation of waste flows in the state.

CIWMB reported that 42,089,545 tons of solid wastes were landfilled in the state in 2005 (2006). In the previous year, the Board released a comprehensive report that used detailed sampling procedures at disposal facilities across the state to statistically determine the composition of California's solid waste (CIWMB and Group 2004). Using that report and the EPA definitions, it was possible to estimate the non-MSW tons landfilled in the state (Figure 3). WTE tonnages[3] were derived from the EEC/BioCycle 2005 "State of Garbage in America" survey and are also listed in Table 1 (Simmons, Goldstein et al. 2006).

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[3] WTE tonnages were calculated as follows: MSW adjustments were made to California's raw disposal tons on a percentage basis – i.e. C&D, HHW, and special waste accounted for approximately 29 percent of the raw disposal tons. This percentage was applied to BioCycle's WTE tonnage to arrive at MSW WTE tonnage.

Table 1 2005 Derived Tonnages of MSW Disposal in California[4]

California 2005 MSW Data		Tons
Disposal (unadjusted)		42,090,000
	Less construction & demolition (C&D)	9,133,000
	Less household hazardous (HH)	126,000
	less special waste	2,904,000
	Total MSW Disposal	**29,926,000**
	MSW to Landfill	**29,335,000**
	MSW to WTE	**591,000**

In 2006, Cascadia Consulting Group and R.W. Beck released the results of a CIWMB-funded study (CIWMB, Beck et al. 2006) characterizing the residuals from materials recovery facilities (MRF) in California. A total of 390 samples were taken from a representative cross-section of MRFs across the state. Using the data reported from these activities, it was possible to back-calculate recycling tonnages that passed through the MRFs (CIWMB, Beck et al. 2006).

To incorporate the composting and mulching of organic wastes into this assessment of recycled tonnage, a CIWMB-funded report on California's composting infrastructure was utilized (CIWMB and Consulting 2004). This report was an attempt to quantify the amounts of organic waste being handled by compost and mulch producing facilities in California. The estimated recycling and organics processing tonnages are shown in Table 2.

Table 2 2005 Derived Tonnages of MSW Recycled in California

Recycling		Tons
	Single stream MRFs	3,547,000
	Multi-stream MRFs	598,000

[4] Note: all tonnage totals are rounded to the nearest thousand to account for likely measurement errors by states.

	Mixed waste MRFs	1,566,000
	Total (MSW) MRF Rec.	**5,712,000**
	Direct to Recycler Tons	6,719,000
	Total MSW Recycling Tons	**12,431,000**
Organics processing		
	Composters	4,730,000
	Processors	5,138,000
	less alternate daily cover (ADC)	2,100,000
	less agricultural	395,000
	less waste water treatment plant residues (WWTP)	395,000
	Total Organics Recycling	**6,979,000**
Total Recycling Tons		**19,409,000**

The methodology used to estimate direct-to-recycler tonnage is as follows: The paper recovered in the US in 2005 was 51 million tons. We contacted Governmental Advisory Associates to determine the amount of fiber going through US MRFs in 2005 (17.2 million tons) (Berenyi 2007). The difference is US "direct-to-recycler" fiber tons. We then multiplied this amount by California's share of US recycling tonnage, according to the BioCycle/Columbia State of Garbage report (20 percent) to arrive at California's share of US direct-to-recycler fibers of approximately 6.7 million tons. [5]

Putting all of this information together resulted in the estimate of California's recycling, WTE, and landfilling rates that are shown in Table 3.

Table 3 2005 Derived Tonnages of Waste Generation and Disposition in California

	Tons	Percent
Total California Recycling Tons	19,409,000	38.9%
Total MSW WTE	591,000	1.2%

[5] It is likely that other materials – particularly steel – also have "direct-to-recycler" tons, but we were unable to account for those materials in this study. These tons would be significantly less than paper, however.

Total MSW Landfilled	29,926,000	59.9%
Total California MSW Generation	**49,925,000**	
Per capita MSW Generation	**1.38**	

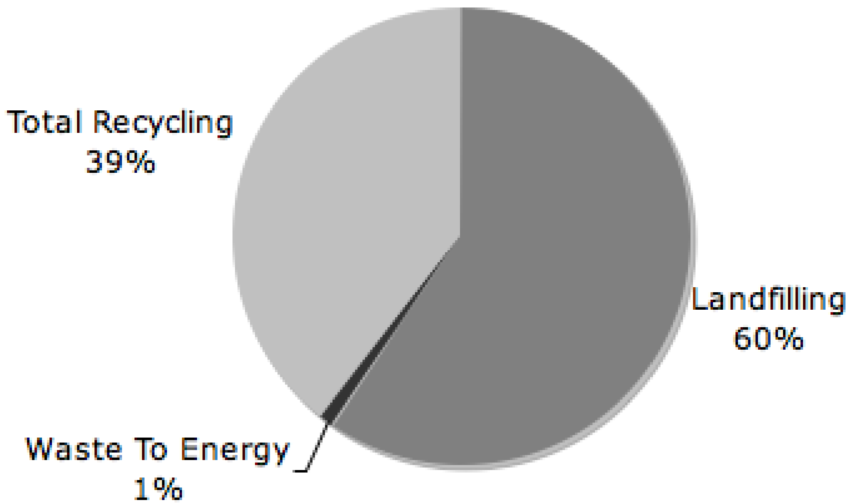


Figure 5 California MSW Management

Comments on California Data

California's Waste Management Act of 1989 (AB 939) mandated a 50 percent diversion rate by the year 2000. According to CIWMB, this was in fact achieved, and the rate has continued to increase since then – indeed, California's 2005 Diversion rate was reported by CIWMB to be 52 percent (CIWMB 2007). However, it is important to note that landfilling tonnage per capita – while decreasing dramatically in the first few years after the law went into effect – has remained flat or increased over the subsequent years. This trend is clearly demonstrated in Figure 9.

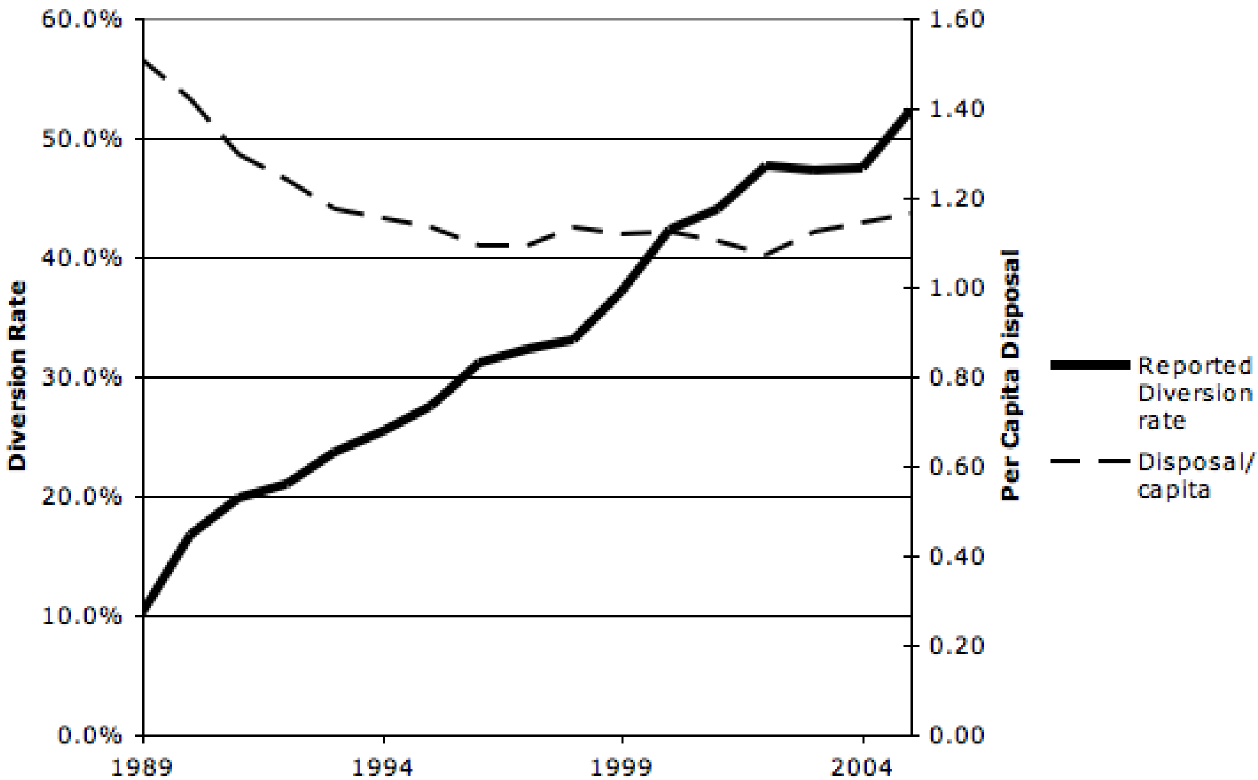


Figure 9 California reported diversion rate vs. reported disposal per capita

It should also be noted that the average U.S. generation rate in 2004, as reported in the SOG survey(Simmons, Goldstein et al. 2006), was only 1.30 tons per capita. Although California enjoys a higher standard of living than some other states, it is simply not credible that its per capita generation of MSW is 1.4 tons recycled plus 1.20 tons landfilled, i.e. nearly 100% higher than the rest of the nation (Kaufman, Millrath et al. 2004). A possible reason for the inflated generation numbers produced by California include the fact that CA counts C&D and some other non-EPA defined waste as part of their recycled stream. Additionally, a study (Themelis and Todd 2004) of recycling in New York City showed that many sorted recyclables were unmarketable and were therefore landfilled. This "double-counting" may be responsible for some of the generation inflation in CA as well. Finally, it must be assumed that there are faults in the formula CIWMB uses to estimate MSW generation. Figure 10 shows the comparison between California's reported diversion rate and its generation of waste per capita.

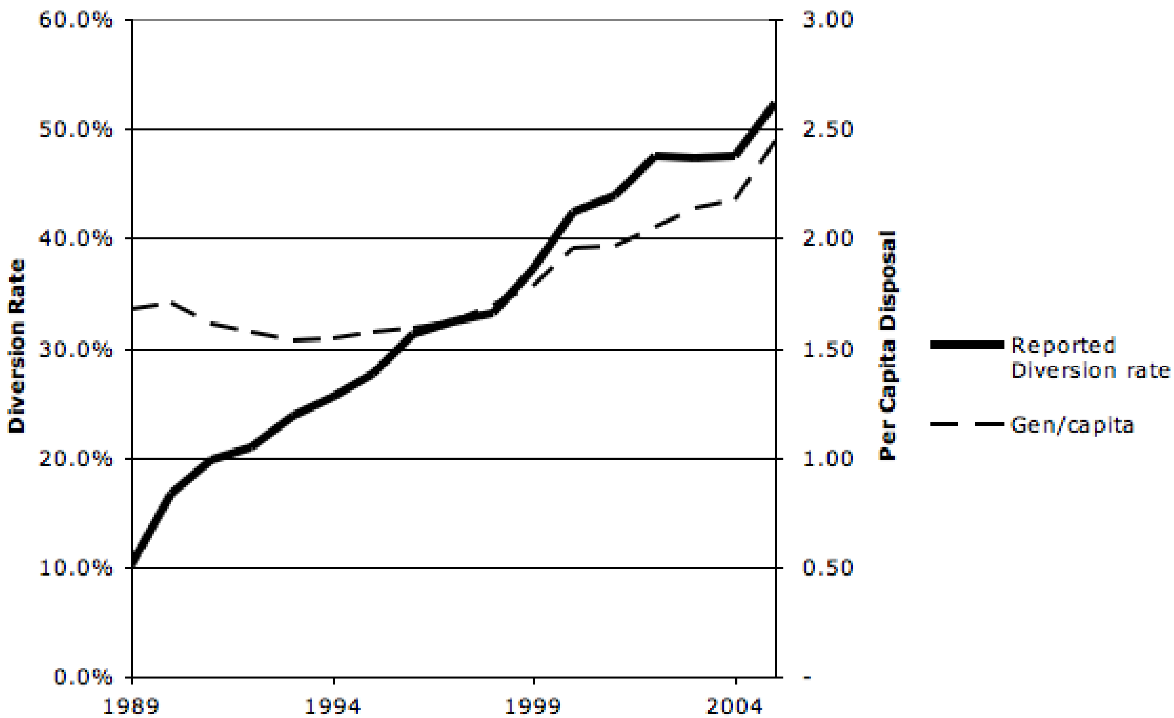


Figure 10 California reported diversion rate vs. reported generation per capita

The above is not meant to downplay the tremendous achievement of California and CIWMB in increasing recycling and composting. In fact, it can be argued that California has led the way in establishing recycling as a mainstream activity through most of the U.S. Nevertheless, the CIWMB numbers themselves indicate that, on a per capita basis, California is the nation's major recycler and, also, the largest landfiller.

The point in highlighting this paradox is to bring attention to the fact that, despite the most demanding regulations and targets for recycling, there needs to be a) integration and analysis of both generation and disposal data and, b) a viable plan for disposing the non-recycled and non-composted fractions of the MSW generated.

Nevada Waste Data Analysis

Nevada law requires the preparation and publication of a biennial report on the status of recycling in the state. This legislation more specifically calls for mandatory disposal facility tonnage reports, which appear to be accurately completed by all eligible facilities. The legislation also calls for reporting by recycling facilities, which has not been as successful, due largely to the limited resources of the municipal offices required to carry out the data reporting compilation (2003).

The following data are drawn mostly from a single, unpublished Nevada Bureau of Waste Management (BWM) report (Fergus 2006). Additionally, there was consistent communication with BWM staff to validate data and secure supplementary information. We also established contact with Republic Waste Services, which manages Las Vegas area landfills and has a great deal of data on Nevada waste management. (Gaddy 2007)

An attempt was made to follow the same procedures used in the California data analysis. The lack of parallel companion reports, such as the MRF and compost facilities publications and, most importantly, a waste characterization study, made this task more difficult. Nevertheless, the data provided were sufficient for a reasonable estimate of Nevada waste flows (Table 4).

The biosolids[6] tonnages were calculated as follows: A US EPA report (1999) on biosolids generation and disposal was used to estimate a US per capita biosolids generation rate. This was combined with census population numbers to estimate Nevada generation rates. Biosolids recycling tonnages, which are documented in the BWM report, were subtracted from this derived generation rate to arrive at biosolids tonnages disposed in landfills.

Table 4 2005 Derived Tonnages of MSW Disposed in Nevada

Nevada 2005 MSW Data		**Tons**
Landfilling		**3,566,000**
	less C&D	774,000
	less biosolids	404,000
	Total MSW landfilled	**2,388,000**
	Total disposal tons	**2,388,000**

Table 5 shows the data used to calculate recycling tons in Nevada. Automobile scrap and biosolids recycling tonnages were both detailed in the BWM report.

Table 5 2005 Derived Tonnages of MSW Recycled in Nevada

Nevada Recycling		**Tons**
Totals		968,000
	less auto scrap	243,000
	less biosolids	106,000
	Total MSW Recycled	**619,000**

[6] Biosolids are the solid materials left after treating municipal wastewater. They are often used as fertilizer, and are not counted in the EPA MSW definition.

Table 6 shows the aggregate tonnages and percentage breakdowns for MSW management in Nevada.

Table 6 2005 Derived Total Tonnages Recycled and Landfilled in Nevada

	Tons	Percent
Total Nevada Recycling tons	619,000	20.6%
Total MSW Landfilled	2,388,000	79.4%
Total Nevada MSW Generation	**3,007,000**	
Nevada per capita MSW Generation	**1.50**	

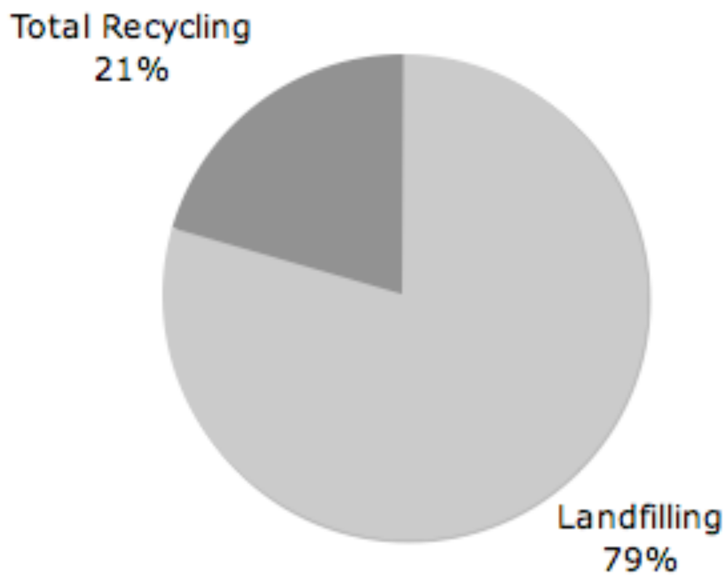


Figure 11 Nevada MSW Management

Hawaii Waste Data Analysis

Hawaii presents some unique challenges due to its geography – as a group of islands, it is apparently more difficult to track the state's waste through a central authority. This project researched data from the state as well as from counties and municipalities.

Fortunately, much of Hawaii's population is concentrated in the Honolulu area on the island of Oahu – 905,000 people in 2005, or 71 percent of the statewide population (2007). Also, Honolulu keeps excellent and detailed records of waste handling on the island of Oahu.

Most of the remainder of Hawaii's population is concentrated on two islands – Maui (11 percent) and Hawaii (13 percent), bringing these three islands' share of the state's total population to 95 percent. The quality of Maui and Hawaii data is also high and allowed us to piece together a representative picture of most of the state's waste flow. The methodology used to gather overall statewide data for Hawaii was as follows: Published reports were gathered from the major areas studied: Oahu, Hawaii, and Maui. Additionally, communications were established with representatives responsible for data collection in these jurisdictions. Finally, the data were normalized and agglomerated.

Honolulu/Oahu

As mentioned above, Honolulu comprises over 70 percent of Hawaii's population. Solid waste data in Honolulu is managed by the Department of Environmental Services (DES) of the City & County of Honolulu. They maintain an informative website (www.opala.org) with yearly updates on waste data. This was a principal source of the data used in this report. It was particularly helpful that DES released the final report of their 2006 waste characterization study in the Summer of 2007 – this allowed for a fuller accounting of waste flows in Honolulu (Beck 2007).

Because of the availability of the comprehensive waste characterization study, it was decided to use 2006 data for Honolulu; 2005 data was used for the rest of the examined states and municipalities in this study. The total MSW disposal (WTE +

Landfilling) in 2006 was 940,187 tons. MSW recycling accounted for 297,000 tons, resulting in 1,237,000 tons of MSW generated overall (2007).

A major adjustment had to be made in the category of "ferrous metals (including autos)" recycling. Scrap metal from the recovery of automobiles does not fall under the EPA definition of MSW and therefore was excluded. However, automobile scrap was not specifically reported in Hawaii, so we had to estimate this tonnage value[7]. The original amount of ferrous metals (including autos) diverted was reported to be 131,591 tons. The adjusted amount was estimated at only 13,146 tons.

Recycling adjustments are shown in Table 7. MSW disposal tonnages for Honolulu are shown in Table 8, while overall Honolulu totals are shown in Table 9.

Table 7 Derived Tonnages of MSW Recycled in Honolulu

		Tons
Recycling (unadjusted)		**543,000**
	less auto scrap	118,000
	less C&D	122,000
	less biosolids	6,000
	MSW Recycled	**297,000**

Table 8 Derived Tonnages of MSW Disposal in Honolulu

		Tons
Disposal		868,000
	MSW to WTE	**756,000**
	MSW Landfilling	**113,000**

[7] The methodology used was as follows: We estimate that the typical ratio of diverted Paper to metal/glass/plastic (MGP) is approximately 60/40. We used this ratio in combination with actual tonnages of non-ferrous diverted materials in Honolulu to arrive at an estimate of ferrous metals diverted.

Table 9 Derived Tonnages of Waste Generation and Disposition in Honolulu

	Tons	Percent
Total MSW Recycling Tons	297,000	25.5%
Total MSW WTE	756,000	64.9%
Total Honolulu Landfilling Tons	113,000	9.7%
Total Honolulu MSW Generation	**1,116,000**	
Per capita MSW Generation	**1.30**	

Maui

Maui's waste disposal numbers were provided by the Maui County Recycling Section of the Department of Public Works and Environmental Management (DPWEM). Biosolids were removed from MSW disposal tonnages as per the method developed and used for Nevada in this study. Maui's recycling tons were adjusted for biosolids as shown in Table 10.

Table 10 Derived Tonnages of MSW Recycled in Maui

		Tons
Recycling (unadjusted)		**73,000**
	less biosolids	24,000
	MSW Recycled	**49,000**

Table 11 Derived Tonnages of MSW Disposed in Maui

		Tons
Disposal (unadjusted)		**163,000**
	less biosolids	36,000
	MSW landfilling	**127,000**

Finally, Table 12 shows a summary of MSW tonnages generated and disposed in Maui.

Table 12 Derived Tonnages of Waste Generation and Disposition in Maui

	Tons	Percent
Total MSW Landfilled	**127,000**	**72.2%**
Total Maui Recycling Tons	**49,000**	**27.8%**
Total Maui MSW Generation	**176,000**	
Per capita MSW Generation	**1.26**	

Hawaii County

The Recycling Section of the County of Hawaii Dept. of Environmental Management (DEM) provided Hawaii County data. They made available to Columbia their FY01-02 to FY05-06 Solid Waste Disposal Summary. An additional landfilling characterization report was provided by the same department (Chin-Chance 2007). This quantifies landfilling tonnages and details recycling tonnages as well. Biosolids were accounted for as per the method introduced in the Nevada section of this report, and C&D tons disposed were estimated using the CIWMB characterization report. The results are shown in Table 14. Household hazardous waste and auto scrap were both included in Hawaii County's reported recycling tons. They had to be removed from MSW calculations. This step is shown in Table 13.

Table 13 Derived Tonnages of MSW Recycled in Hawaii (County)

Recycling (unadjusted)		**78,000**
	less household hazardous waste	Negligible
	less auto scrap	13,000
	MSW Recycled	**64,000**

Table 14 Derived Tonnages of MSW Disposal in Hawaii (County)

		Tons
Disposal (unadjusted landfilling)		**223,000**
	less biosolids	2,000
	less C&D	48,000
	MSW disposal	**172,000**

Totals and rates for Hawaii County are shown in Table 15.

Table 15 Derived Tonnages of Waste Generation and Disposition in Hawaii (County)

	Tons	Percent
Total MSW Landfilled	**172,000**	**72.9%**
Total Hawaii (County) Recycling Tons	**64,000**	**27.1%**
Total Hawaii (County) MSW Generation	**236,000**	
Per Capita MSW Generation	**1.41**	

With 95 percent of Hawaii's population accounted for, it is now possible to combine the results from the islands and produce a set of tonnages and rates for the state. The results are shown in Table 16.

Table 16 Derived Tonnages of Waste Generation and Disposition in Hawaii (Overall State)

	Tons	Percent
Total MSW Recycling Tons	410,000	**24.9%**
Total MSW WTE	756,000	**45.8%**
Total Hawaii Landfilled Tons	483,000	**29.8%**
Total Hawaii MSW Generation	**1,650,000**	
Per capita MSW Generation	**1.39**	

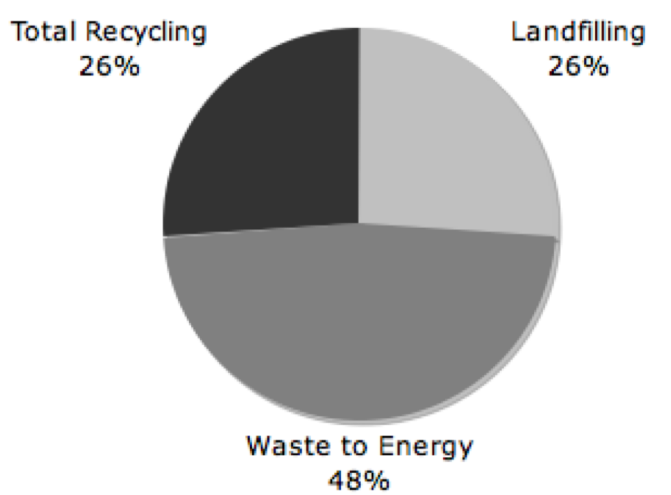


Figure 12 Hawaii (Overall State) MSW Management

Effect of Arizona Data on Overall Region 9 Management

Though Arizona was not able to participate in this study (there was staff turnover during the active research phase), a simple sensitivity analysis shows the possible effects their data would have on overall Region 9 numbers. As shown in Figure 8, Arizona would need to be recycling a minimum of 16 percent of their MSW in order to maintain a Region 9 recycling rate of 35%. Arizona's likely recycling rate of greater than 20 percent (based upon Phoenix's success in reaching that target) would maintain a Region 9 overall rate of greater than 35.5 percent, easily surpassing the national target.

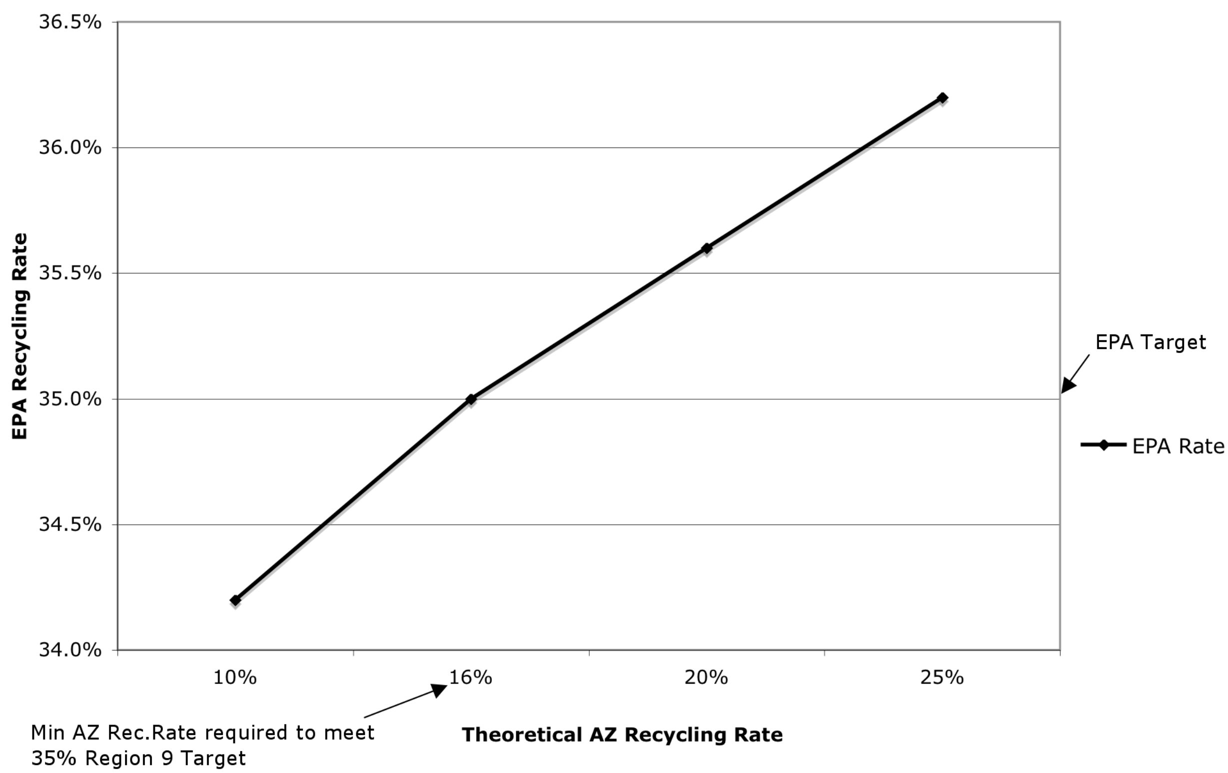


Figure 13 Arizona's Potential Effect on Overall Region 9 Recycling Rates

Overall Region 9 MSW Management Totals & Rates

Table 17 shows the results of the three states of Region 9 that were examined in this study (California, Nevada, and Hawaii).

Table 17 Overall Region 9 (Examined) MSW Management Totals & Rates

	Tons	**Percent**
Total Region 9 Recycling Tons	20,438,000	**37.9%**
Total MSW WTE Tons	1,347,000	**2.5%**
Total Region 9 Landfilling Tons	32,135,000	**59.6%**
Total Region 9 MSW Generation	**53,920,000**	
Per capita MSW Generation	**1.39**	

Opportunities for Increased Diversion in Region 9

In this study, we were mostly confined to bulk metal-glass-paper-plastics (MGPP) data, with little information on the recycled tons of each material. Without detailed data on specific types of materials being recycled it is difficult to know the tonnages of individual materials that are recycled in Region 9. However, when certain jurisdictions perform waste characterization studies – and when these are accompanied by high quality recycling data – we are able to open a window into opportunities for higher diversion. California and Honolulu are jurisdictions with recent enough characterization studies that allow us to look more closely at recycling opportunities.

We start with Honolulu, which, as mentioned, accounts for over 70 percent of Hawaii's population and is thus an important barometer of statewide recycling activities. An analysis of the characterization study in combination with county-reported recycling data showed that there are several key materials that can be targeted for increased diversion.

Only 2.9 percent of plastics are currently captured for recycling in Honolulu. The next-lowest examined commodity is paper, with a 19 percent capture rate. If plastics were to be brought up to the level of paper recycling, the overall Honolulu diversion rate would jump from 25.5 percent to 27.4 percent, an increase of 1.9 percent. Paper is a similar "low-hanging fruit" – if Honolulu were able to increase the diversion of paper from the current level of 19 percent to 25 percent (a modest goal), the overall recycling rate would increase to 27.7 percent, a 2.2 percent improvement. Table 18 summarizes the above and two other hypothetical scenarios for food and yard wastes.

Table 18 Material-Specific Recycling Tonnages in Honolulu

Honolulu Tons	Food	Yard	Paper	Plastic
Disposed	120,000	82,000	345,000	132,000
Recycled	37,000	77,000	81,000	4,000
Total Commodity Generated	157,000	159,000	426,000	136,000
Commodity Recycling Rate	23.6%	48.4%	19.0%	2.9%
Contribution to Overall Diversion	12.5%	25.9%	27.3%	1.3%
Next Target %	30%	55%	25%	19.0%
Next target tons	47,000	87,000	107,000	26,000
Tons to next target %	10,000	10,000	26,000	22,000
Percent overall increase if target attained	0.9%	0.9%	2.2%	1.9%
Overall Recycling Rate if target attained (baseline = 25.5%)	26.4%	26.4%	27.7%	27.4%

The picture in California is not as easy to analyze, as detailed commodity-by-commodity recycling information is not available. The only reliable broad-based commodity category that could be analyzed was organic waste, including food and yard scraps. As Table 19 shows, California recycles 44.8 percent of the organic waste it generates. There is still a great opportunity to divert more organics, as food waste is still disposed of in large quantities.

Table 19 Diversion of Organics in California

	Organics
Recycled (composting)	6,979,000
Disposed (landfilling + WTE)	5,854,000
Food scraps disposal	*3,118,000*
Yard waste disposal	*2,736,000*
Total Generated	12,833,000
Organics Recycling Rate	44.8%

Comments and Analysis

The results show that the combination of the three EPA Region 9 States examined in this study are recycling more than 37 percent of their generated MSW, surpassing the 35 percent national MSW recycling goal set by EPA. (EPA 2005) Though Arizona's tonnages are not included in this analysis, we have calculated that AZ would only have to be recycling roughly 16 percent of its waste at current levels of Region 9 MSW generation to maintain a 35 percent region wide recycling rate. It is reasonable to expect that this minimum is being met, as Phoenix – by far the state's largest city – is currently recycling more than 20 percent of the waste it generates. (Phoenix 2007)

Though recycling is a clear success in the region, landfilling remains the predominant means of dealing with MSW in all areas (with the exception of Honolulu, which relies primarily on WTE for all non-diverted waste). Food waste is a prime candidate for increased diversion, as it is present in large volumes and is responsible for much of the negative environmental effects associated with landfilling. States that figure out means for economically and efficiently dealing with this fraction of the waste stream will go a long way towards achieving a successful integrated waste management system.

Finally, one of the most important conclusions we have drawn from this study is that ***there is a tremendous opportunity for convergence between the U.S. EPA and the BioCycle/Columbia studies of waste management in the U.S.*** EPA has excellent data on recycling of MSW, due to strong partnerships with industry organizations. The BioCycle/Columbia team has developed good relations with a robust network of state waste managers who have direct access to MSW generation and disposal data. In addition Columbia has collected data directly from MRFs and compost facilities that are sometimes unwilling to share information with government agencies due to privacy and competitiveness concerns. The strengths of both EPA and Biocycle/Columbia data could be combined to produce a more reliable overall set of MSW management figures. Improved MSW measurement data would support the prioritization and implementation of cost-effective waste reduction, recycling, and compost program development.

Building and Programming of the Interactive Database

After the basic data has been collected, the remaining question is how best to present and disseminate information. The solution reached by the Earth Engineering Center of Columbia University is the MSW Database (MSW-DB), a geographic, interactive map, programmed using Adobe Flash, that is now published on the web (www.wastemap.us). Figure 14 shows the home page interface of the MSW-DB.

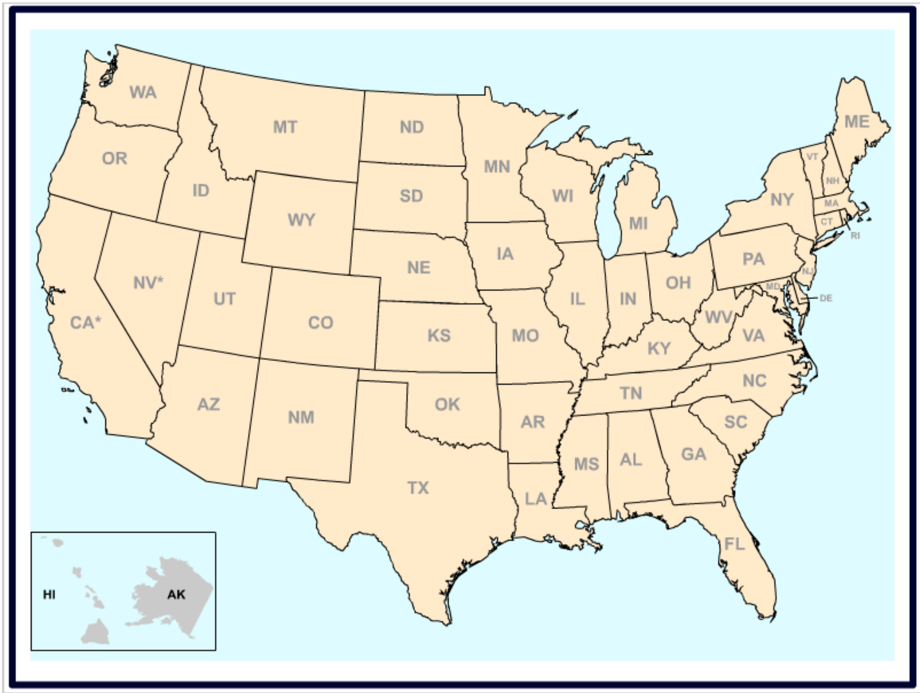
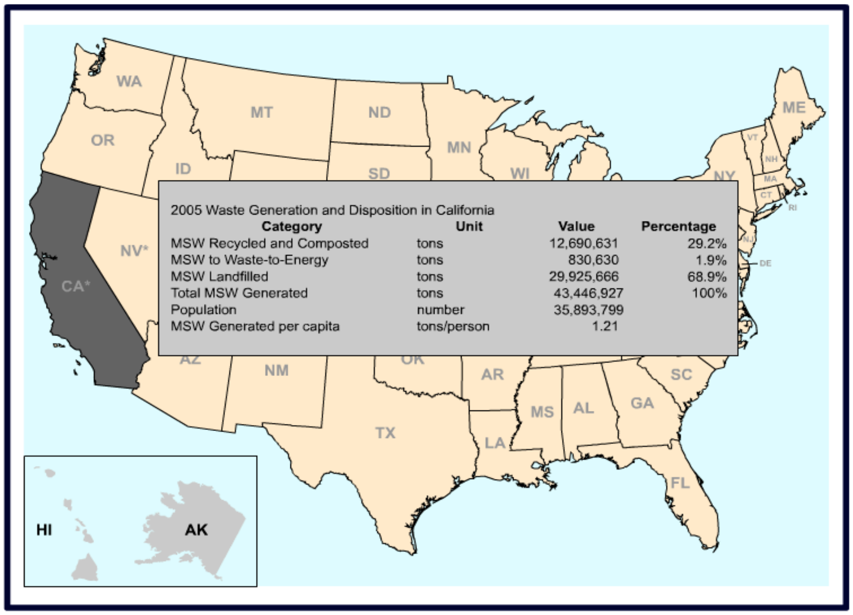


Figure 14 Home page of MSW Database

When the user "hovers" over a state on the home page map, a window pops up showing basic information on MSW flows. For example, Figure 15 shows the result when a user hovers over the state of California on the interactive map.



*Denotes data from EPA Region 9 Grant Activities (see Help for more information)

Figure 15 California hover screen

The same level of information is now available for all states of the Union, on the basis of the data collected during the 2004 SOG Survey (ref BioCycle 2005).

When the user clicks on the state, the map links to a page showing more detailed information on the methodology used to calculate that state's data, and also the sources of the information.. For example, Figure 16 shows the waste flows in California. In addition, the State Detail Pages feature "In Perspective" analyses, where the data from the state are compared with both the region (in California's case, the EPA Region 9 states of Hawaii and Nevada) and the nation, as a whole. California's In Perspective analysis is shown in Figure 17.

2005 California Waste Flows **Columbia University MSW Database**

Click the buttons on the State map for MSW Flows of Different CA Cities

California 2005 MSW Data		**Tons**
Landfilling		42,089,545
	less C&D	9,133,431
	less HH	126,269
	less special waste	2,904,179
	Total MSW Landfilled	**29,925,666**
Waste to Energy		**830,630**
Total Disposal Tons		**30,756,296**
Recycling		
	Single Stream MRFs	3,547,143
	Multi-Stream MRFs	598,333
	Mixed Waste MRFs	1,566,491
	C&D MRFs	703,043
	Total (MSW) MRF Rec.	**5,711,968**
Organics		
	Composters	**4,730,081**
	Processors	**5,138,031**
	less ADC	2,100,000
	less agric	394,724
	less WWTP	394,724
	Total Organics Rec.	**6,978,663**
Total Recycling Tons		**12,690,631**

Back to US Map

Figure 16 Detailed information on State of California

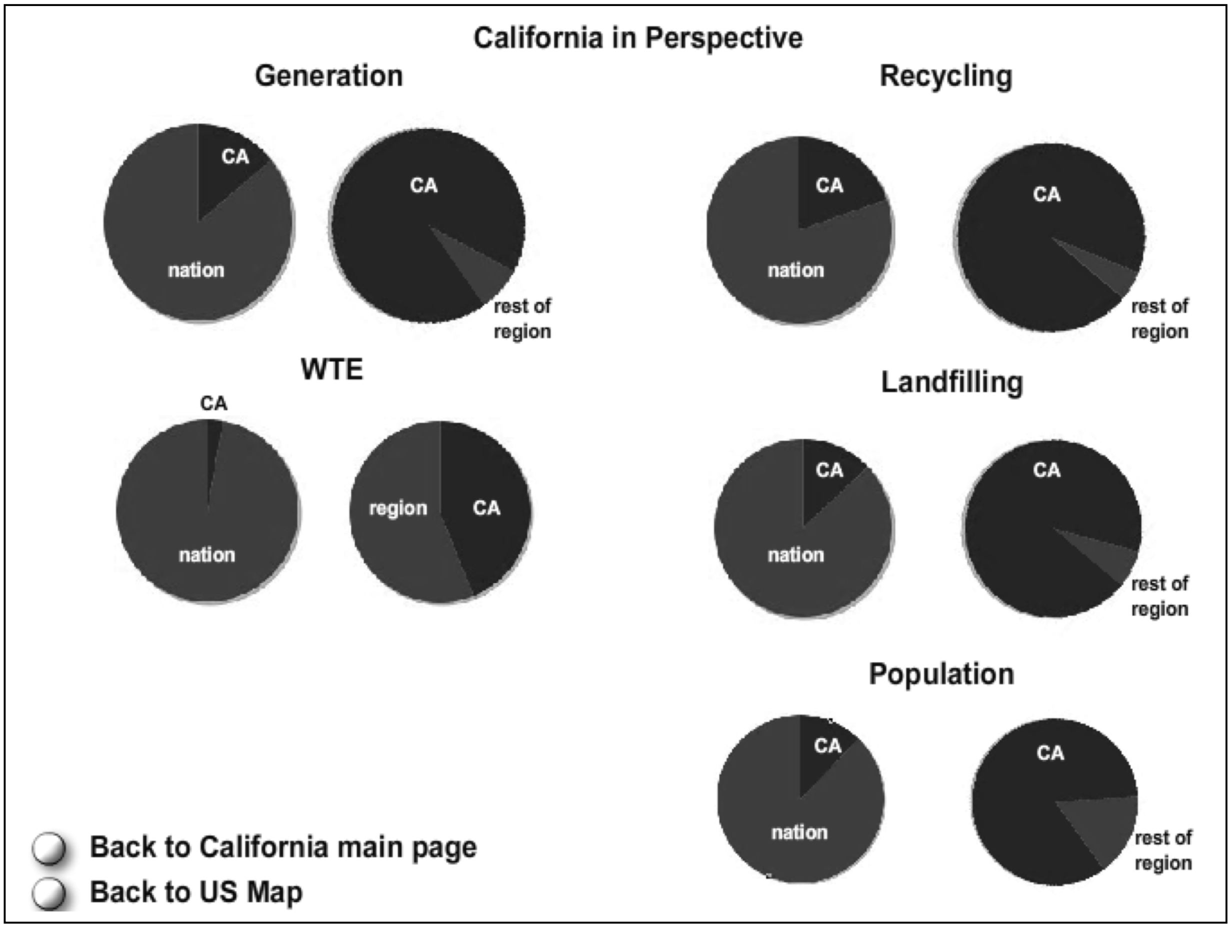


Figure 17 California comparative ("In Perspective") charts

At this time, such detailed information is provided only for three of the states of EPA Region 9 (CA, HI, and NV) but it is expected that with time the WasteMap of Columbia University will encompass all fifty states.

4. Examination of the Fate of Carbon in Waste Management Systems through Statistical Entropy & Life Cycle Analysis (LCA)

Introduction

In examining possible means for the development or adoption of a more streamlined and publicly accessible method for measuring the impacts of waste management, one of the early options was statistical entropy.

The statistical entropy (SE) function – a method adapted from the field of information theory (Shannon 1948) – was applied by Rechberger and Brunner to waste treatment to account for the tendency for processes to either concentrate or dilute substances (Rechberger and Brunner 2002; Rechberger and Graedel 2002). It is a logarithmic function that requires input and output concentrations as well as mass flows to be accurately calculated.

In the original paper by Rechberger and Brunner, the partitioning of four heavy metals – cadmium, mercury, lead, and zinc – were analyzed by incinerators with different pollution control systems. Their analysis showed that with increasingly sophisticated environmental controls (the simplest being an incinerator with no flue gas treatment, the most sophisticated being a "best possible" treatment scenario where all solid residues are either recycled or placed in a "final storage" landfill with no need for aftercare), the ability of a treatment system to concentrate the examined substances increased as well. This in essence proved the theory that the authors set out to demonstrate, namely that increasing substance concentration in waste systems equates to increasing environmental performance (Rechberger and Brunner 2002).

The extension of the methodology to carbon flows is complicated by several factors. For example, input concentrations are relatively straightforward, as they are usually known – the concentration of carbon in municipal solid waste, for instance, is well-referenced in the literature (Tchobanoglous and Keith 2002). The output concentrations are more difficult to compute, for a variety of reasons – such as transformation reactions, mixing and, in the case of carbon, sheer number of output species (Castaldi and Kwon 2007). These complexities will be examined in the Experimental and Modeling sections.

SE is an offshoot of materials flow analysis (MFA) that has been shown to be a valuable complement to other evaluation methods such as LCA – when different incineration technologies are compared against each other, those facilities with greater environmental controls perform better in SE terms (Rechberger and Brunner 2002). In light of recent environmental focus related to energy use and carbon emissions, there is a need to expand the SE methodology to more comprehensively account for substance flows of carbon through systems. However, due to the large number of species present when carbon outputs are considered, we need a method that will account for carbonaceous inputs and outputs in a tractable manner.

This chapter outlines a methodology to extend the SE analysis to carbon in two parts. First, in order to establish the ability of this methodology to deal with large numbers of carbon containing output species, we incorporate our own laboratory experimental data on the gasification and combustion of waste tires. By doing these calculations with our own experimental data, we can comprehensively understand the complete set of species and feel confident that everything has been accounted for. Also, this approach demonstrates that even a relatively simple process like the combustion of natural rubber produces very complex and quite numerous outputs.

To demonstrate its practical usefulness in a real world application, we then use the SE approach to assess alternative options to manage municipal solid waste (MSW). This is first accomplished in a relatively straightforward analysis of landfills versus incinerators. We then expand the analysis to include energy effects (i.e. the carbon emissions that result from energy that has to be produced from primary sources that could have been offset by waste to energy - WTE - plants); and radiative forcing (i.e. the relative potency of methane-heavy emissions from landfills versus the carbon dioxide emissions of WTE's.)

Background methodology

Materials Flow Analysis

Statistical entropy is a subset of the materials flows analysis (MFA) method of accounting for flows and stocks of materials in a given system. It is based on the principle of mass conservation - any mass entering a system must either exit the system of remain as "stock." An MFA, therefore, comprises a mass balance of all inputs, outputs, internal flows, and stocks of a system.

A system consists of one or more processes and materials flows. A process denotes the transformation, transport, or storage of materials. Processes are linked by flows of goods. A good consists of substances, which are either elements (e.g. carbon, hydrogen) or compounds (e.g. carbon dioxide, water). These differences are clearly delineated because economic decisions are typically made based on flows of goods, while environmental management decisions are better understood on the basis of substances.

Figure 18 shows a simplified MFA diagram incorporating all of the key terms introduced above. The system depicted is a highly simplified waste management scenario. The input is waste tires, and the processes within the system boundary are waste-to-energy (WTE) and landfilling (LF). Tires are the goods, and they carry any number of substances, such as carbon, hydrogen, zinc, etc. The arrows represent flows of goods.

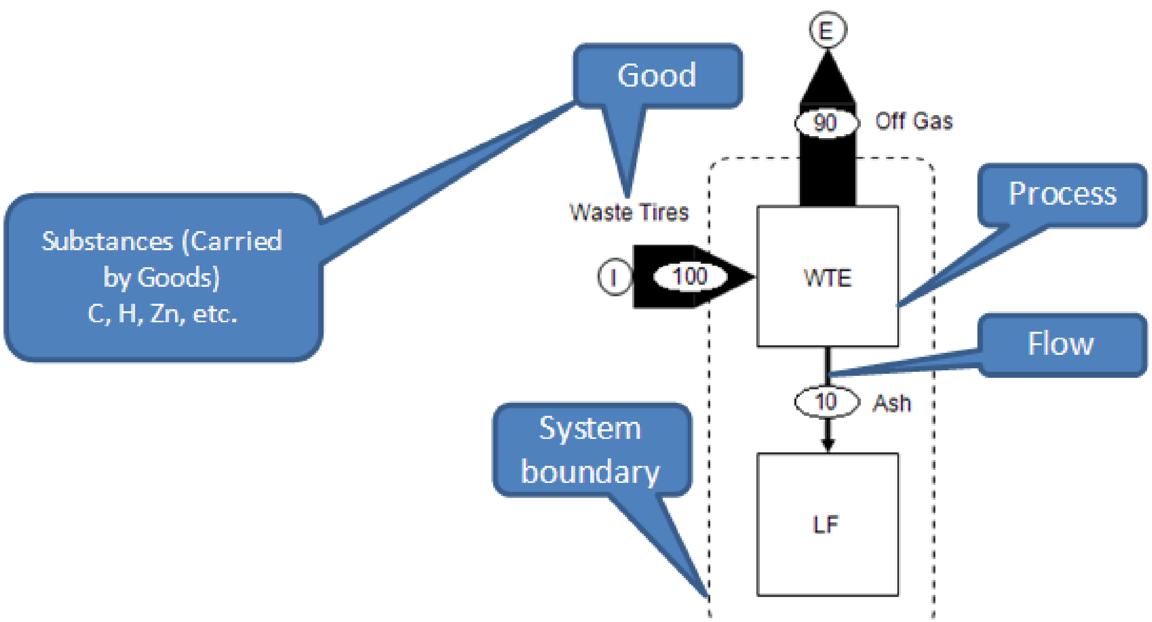


Figure 18 Material Flow Analysis (MFA) Terminology

An example of an MFA for a real-life scenario is presented in Figure 19, where California waste management data collected for the EPA Region 9 grant is utilized.

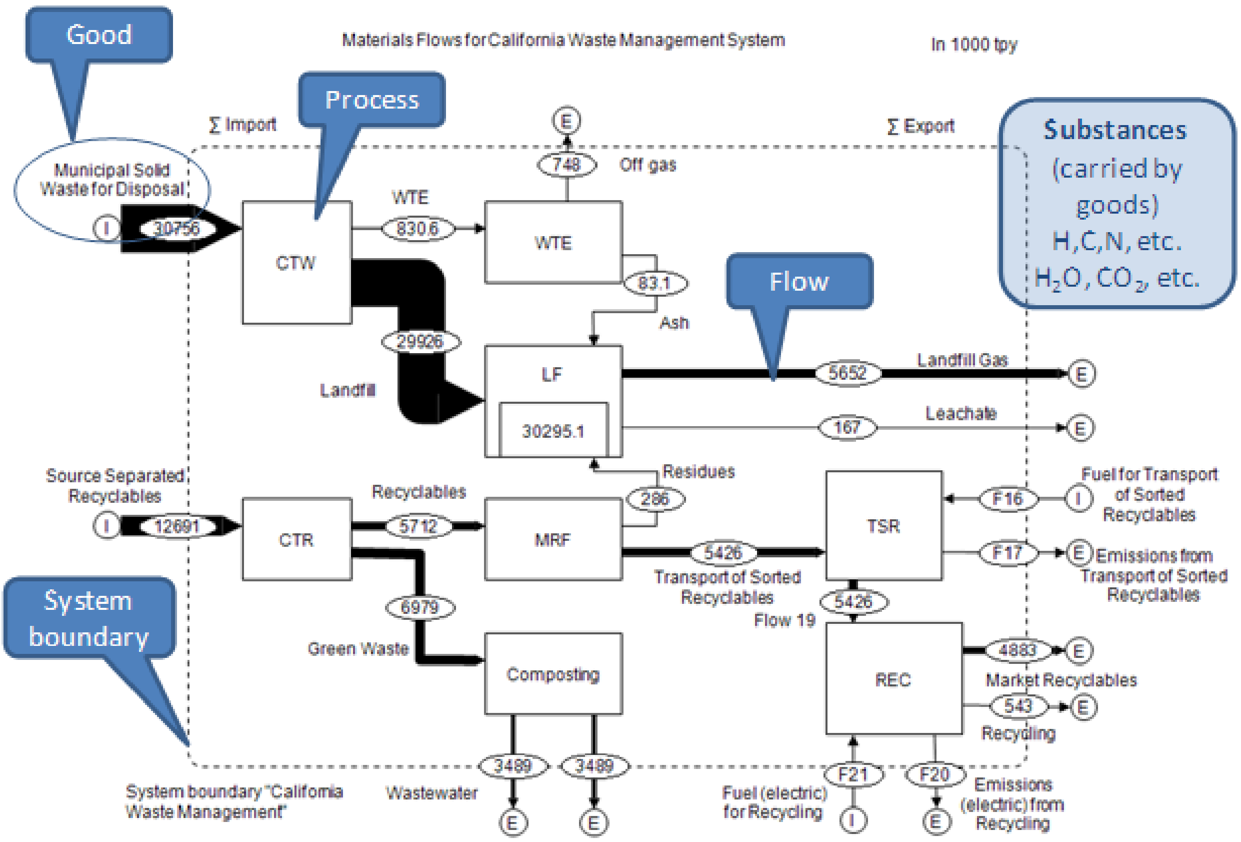
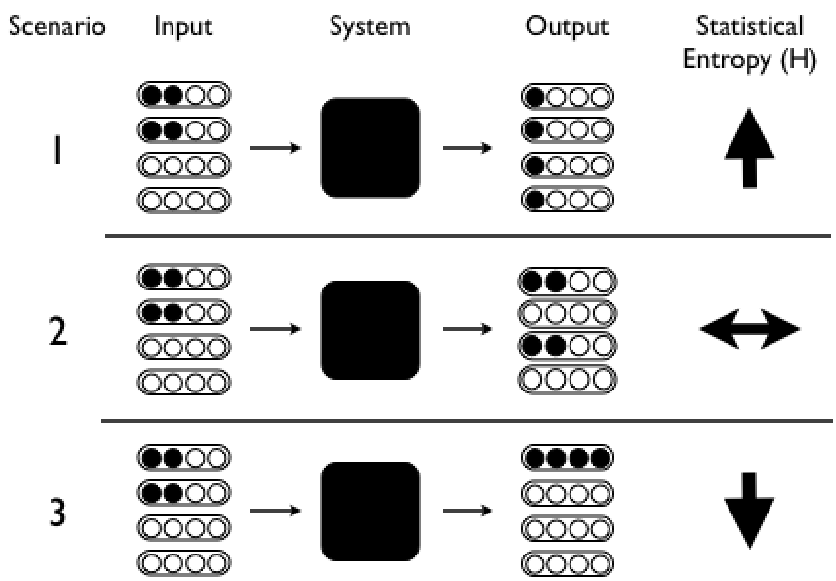


Figure 19 MFA of California Waste Management

MFA is a proven method of resource accounting. It is, however, a descriptive exercise – it does not in itself provide the analytical tools necessary to determine, for example, the environmental effectiveness of a given waste management strategy. For that, it is necessary to introduce another set of tools – statistical entropy analysis and modified life cycle assessment.

To perform SE analysis, input and output entropies are calculated and then compared to determine whether the system concentrates or dilutes the examined substance. A schematic representation is presented in Figure 20.



Theoretical example of statistical entropy calculations. Each scenario (1,2 and 3) share the same input mass and concentrations, and are transformed when passed through a black box system. Scenario 1 dilutes the substance (black balls) so that it is spread out more in the goods, thus the substance concentrating efficiency (SCE) is negative. Scenario 2 produces the same output distribution as the input, so the entropy remains the same (no concentration or dilution). Scenario 3 concentrates the all of the black balls into one output good, resulting in a positive SCE.

Figure 20 Schematic of theoretical statistical entropy calculation

Actual SE calculations require computing the input and output entropies and comparing them to a theoretical maximum. The same basic form of the statistical entropy equation is used for inputs and outputs (eq. 1).

$$H(c_{ij}, m_j) = ld(\dot{X}_j) - \frac{1}{\dot{X}_j} \cdot \sum_{i=1}^{k} \dot{m}_i \cdot c_{ij} \cdot ld(c_{ij}) \qquad (1)$$

where H is the statistical entropy (measured in bits); c and m are the concentration and mass flows, respectively; ld is the logarithm to the base 2 (allowing for conversion to binary units); and $\dot{X}_j$ is the total substance flow induced by the set of goods. The subscripts i and j are indexes for goods and substances, respectively.

For the output entropies, c_{ij} is defined as

$$c_{ij} = \begin{cases} c_{j.geog.g}/100 & \text{gaseous outputs} \\ c_{j.geog.a}/100 & \text{aqueous outputs} \\ c_{ij} & \text{solid outputs} \end{cases} \qquad (2)$$

where *geog* signifies the geogenic concentration of the examined substance, and *g* stands for gaseous and *a* stands for aqueous. For output entropy calculations, m_i is defined as

$$m_i = \begin{cases} \dfrac{\dot{X}_{ij}}{c_{j.geog.g}} \bullet 100 & \text{gaseous outputs} \\ \dfrac{\dot{X}_{ij}}{c_{j.geog.a}} \bullet 100 & \text{aqueous outputs} \\ \dot{m}_i & \text{solid outputs} \end{cases} \qquad (3)$$

The maximum entropy is calculated as per (eq. 5). A more detailed derivation of the maximum entropy is available in the literature(Rechberger and Brunner 2002).

$$H_{\max,j} = ld\left(\frac{\dot{X}_j}{c_{j,geog,\min}} \bullet 100\right) \qquad (5)$$

Once the raw input and output entropies have been quantified, the *relative statistical entropies* (RSE) must be quantified for each as well (eq. 6). The RSE relates the input and output entropies to a theoretical maximum entropy, enabling one to determine the degree of concentration or dilution. The difference between the input and output RSE of a system is defined as the *substance concentrating efficiency* (SCE) of the system (7). It is this SCE that provides a quantitative measure of how the chosen system digests or transforms the input material.

$$RSE_j = \frac{H_{actual}}{H_{\max}} \qquad (6)$$

$$SCE_j \equiv \frac{RSE_{j,I} - RSE_{j,O}}{RSE_{j,I}} \bullet 100 \qquad (7)$$

The Statistical Entropy method was developed to deal with flows of conservative substances through systems – that is, the substances had a small, finite number of stable chemical species that they could yield and enter into the environment. In order to use the methodology to fully account for substance flows, however, it is desirable to look at more complicated materials flows as well. Perhaps the most complex substance flow is carbon, due to its prevalence in the environment and waste management systems in different forms, and because reactions involving carbon are so complex and their products so diverse and numerous. This complexity is apparent in the series of experiments on the thermal degradation of tires performed and referenced in this chapter (Castaldi and Kwon 2007).

Adaptation of the SE Method to Carbon Flows

Experimental

To identify output species of gasification or combustion reactions, intensive Gas Chromatography/Mass Spectroscopy (GC/MS) coupled to Thermo-Gravimetric Analysis (TGA) measurements are used. Experimental work was carried out on the main constituents of tires, Styrene-Butadiene Rubber (SBR) and Natural Rubber (Polyisoprene or IR), as well as on pieces of tires themselves, to better understand the mechanistic behavior of waste tire, and thus better characterize organic outputs. Significant data have been established from these experiments that enable an understanding of the thermal degradation mechanism and main gaseous outputs from waste tires during the combustion and pyrolysis/gasification processes (Castaldi and Kwon 2007; Castaldi, Kwon et al. 2007). These data are used as the basis set in the SE analysis for quantifying the flows of carbon in the investigated systems[8].

All experiments were performed using a Netzsch STA 409 PC/4/H TGA unit capable of simultaneous TGA and Differential Thermal Analysis (DTA) measurement. The heating rate was 10˚C/min. All data were digitally recorded and S-type thermocouple readings were compared simultaneously. The flow rate for purge and protective gases were set using Aalborg thermal mass flow controllers (GFCS-010378; Aalborg® Inc.) and the total flow rate was 100ml/min. The initial test sample weights were typically about 10 mg and all samples came from the same chemical batch. The effluent of the TGA was sent to either a µ-GC (Agilent 3000) or GC/MS (Agilent 9890/5973) and the sampling system, that includes transferred lines coupled to a vacuum pump, was

[8] All described experiments were carried out by Eilhann Kwon, a PhD student in the Earth & Environmental Engineering Department at Columbia University, under the direction of Professor Marco J. Castaldi.

maintained over 300˚C using Omega heat tape (SRT 101 Series) to mitigate the condensation and/or adsorption of hydrocarbon onto its surface. The concentrations of standard (Lot#A03448), a Sigma Aldrich aromatic standard (PIANO Aromatic Lot #2102) and a Japanese indoor air standards mixture (Lot#4M7537-U).

The experiments showed that there were more than 50 carbon-based output species. This was the case for all three examined input samples – waste tires, SBR, and IR. These species were detected on a parts-per-million-volume (PPMV) basis). To directly use the RSE methodology, the volumetric concentrations were converted to mass concentrations.

The next step involved analyzing whether the PPMV contributions of the major output species to the total were consistent across the entire temperature range. We found that the PPMV share was highly consistent across all recorded temperatures. Table 1 identifies the major gaseous product species from the experiments and their percent contribution to the total. Table 20 shows the PPMV share of the five most prevalent gaseous species in the thermal treatment of SBR. These five species accounted for just over 97 percent of total output PPMV. Their share of PPMV across all temperatures had a median value of 96.5 percent.

Table 20 Major gaseous output species share of total PPMV in thermal treatment of SBR

Substance	Total PPMV	% of Total
Butane (C4)	2575.2	71.6%
Styrene	733.1	20.4%
Toluene	105.1	2.9%
1.2.4-Trimethyl-Benzene	44.1	1.2%
Acenaphthene	37.0	1.0%

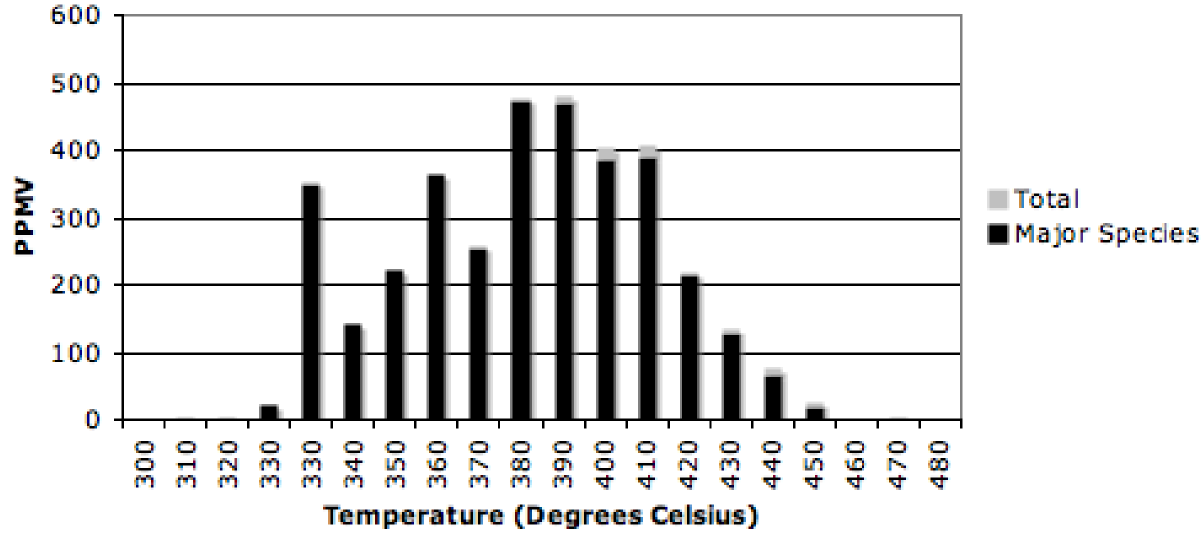


Figure 21 Major species PPMV share in thermal treatment of SBR

A similar graph for inputs and outputs in terms of PPMV for tires is presented in Figure 22.

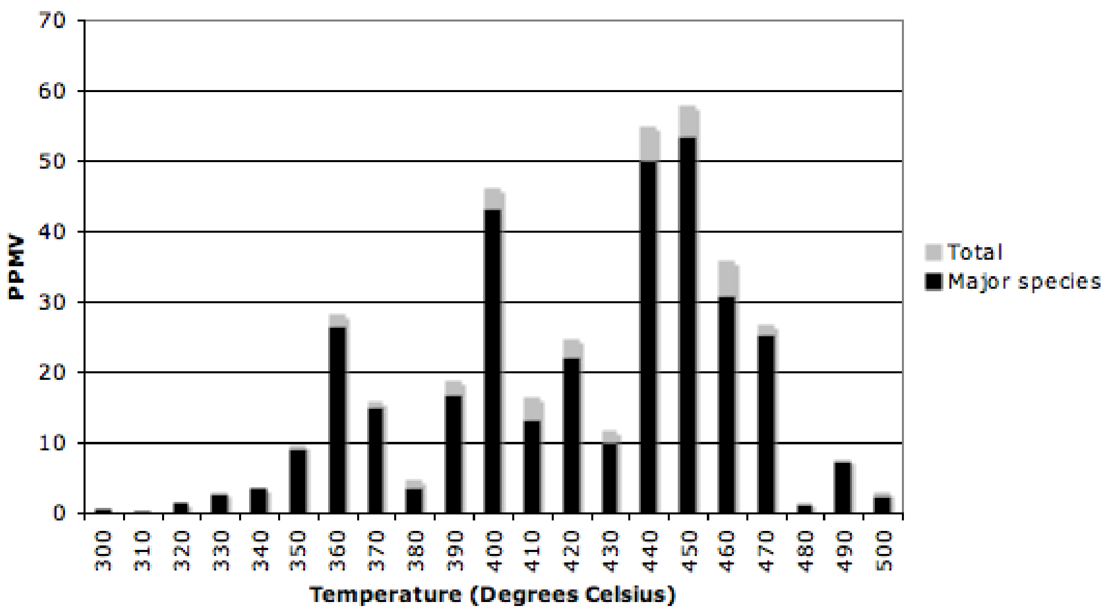


Figure 22 Major species PPMV share in thermal treatment of Tires

As shown in Figure 22, the "major species" contribution to the overall output flows of materials is dominant. The same trend can be seen for the other experimentally examined material, Natural Rubber (Polyisoprene, IR), the results for which are shown in Figure 23.

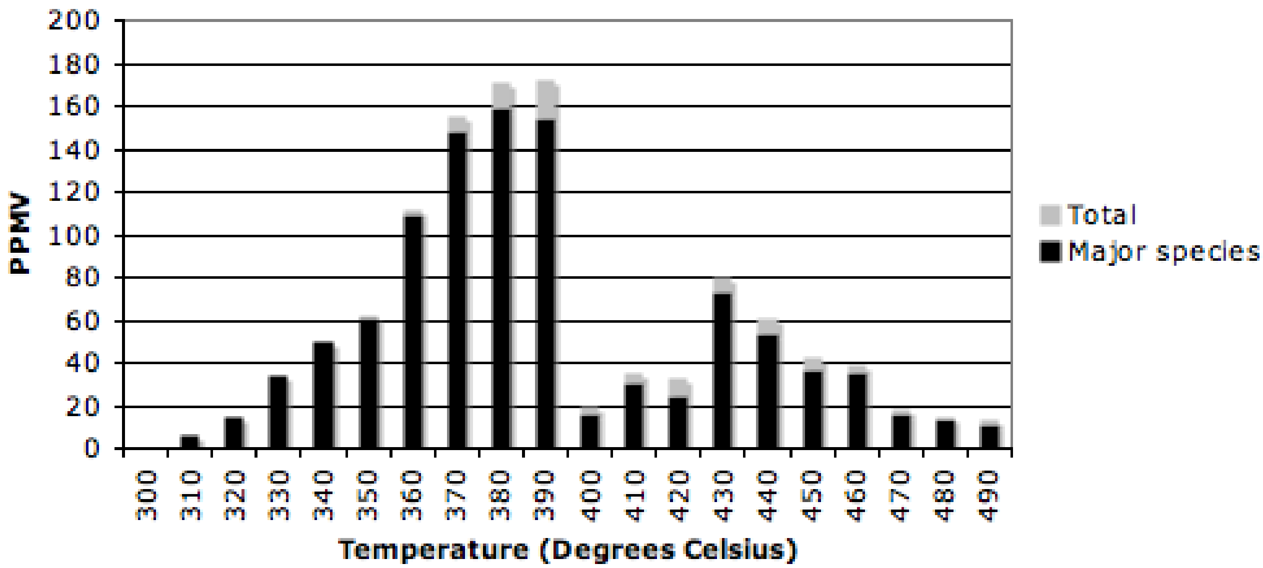


Figure 23 Major species PPMV shares in thermal treatment of IR

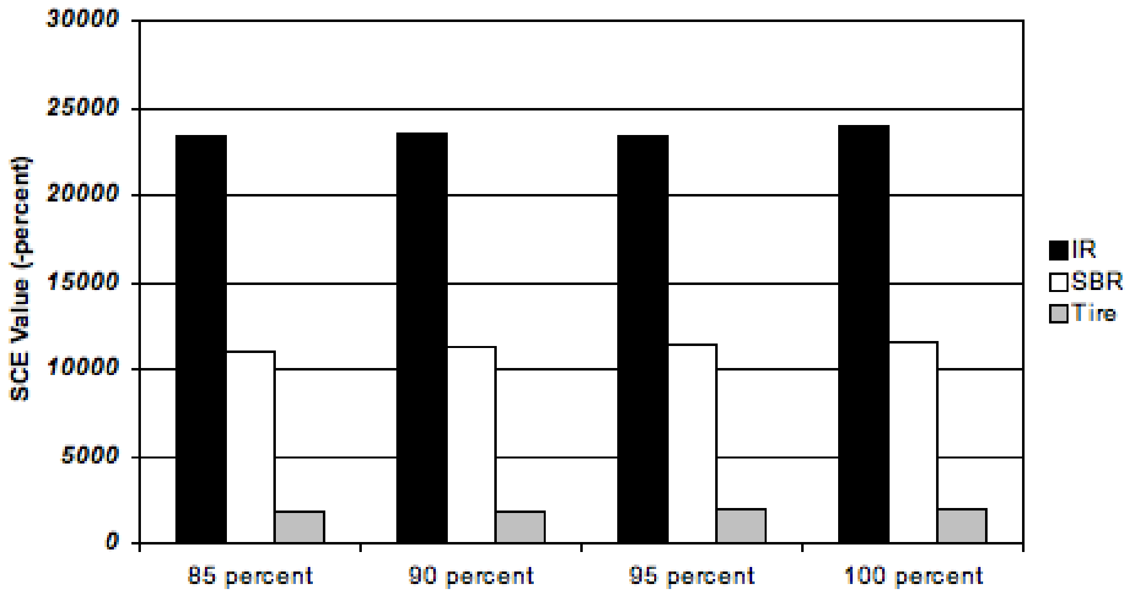
These results were consistent across all experimental data ranges. We were therefore able to conclude that, in all examined cases, we can sum the total measured PPMV values of individual measured species; and then compute the individual species share of the total output PPMV.

In the case of SBR gasification, the input, once thermally treated, fractionates into two main output products: off gas (with more than 50 carbon-containing product chemical species) and tar. The off gas accounts for 90 percent of the total output mass, while the tar accounts for the remaining 10 percent. The specific output product flows in the off gas are determined by multiplying the "% of totals" figures from by the output mass of off gas.

To calculate entropies for gaseous and aqueous species it is necessary to reference the geogenic concentrations of the substances being examined. We used two primary sources to find geogenic concentrations for all substances (1999; 2007). Some substances are industrial byproducts (i.e. not found in nature); in these cases we had to establish natural concentrations based on government-mandated emissions limits. These "natural" concentrations were approximated by taking emissions or workplace standards (such as EPA emission limits or OSHA air concentration standards) and using them in place of geogenic values. (Note that calculations were performed using fractions of these values and that SCE results were not sensitive towards these changes.) Many of these data are published in TOXNET, a National Institutes of Health online database.

Once all input and output species are quantified in this manner (i.e. represented on a mass basis), it is possible to calculate the entropy changes in the system. Mass balances were performed for SBR gasification, IR gasification, and tire gasification, accounting for 100 percent of output species. We then adjusted inputs and outputs so that 85 percent, 90 percent, and 95 percent of gaseous output masses were accounted for. (As the methodology is mass-weighted, the results are sensitive to output mass and much less so to concentrations.) The purpose of this sensitivity analysis was to determine the minimum number of gaseous output species required to perform a reliable SE analysis for carbon. The analysis shows that differences in SE for gas phase carbon outputs are minimal, and that an acceptable determination is 90 percent as a threshold for reliable SE calculations (Figure 24). The following tables show the inputs used for the SBR entropy model.

Figure 24 SCE Values for thermal treatment of IR, SBR, and Tires at different levels of output mass (values in minus percent)



When the SE method was developed and the substance concentrating efficiency metric was devised, it was expected that the analysis would be applied to conservative substances such as metals, yielding SCE values between 0 and 100 (Rechberger 2008). With SCE values between 5000 and almost 25,000, it becomes attractive to limit the analysis to the use of straight entropy values (H) – a more intuitive approach. The graphs for entropy values and SCE values look nearly identical. Both will be used in different situations in further analysis in this chapter. The H-values of the three compounds tested for different levels of output mass are shown in Figure 25.

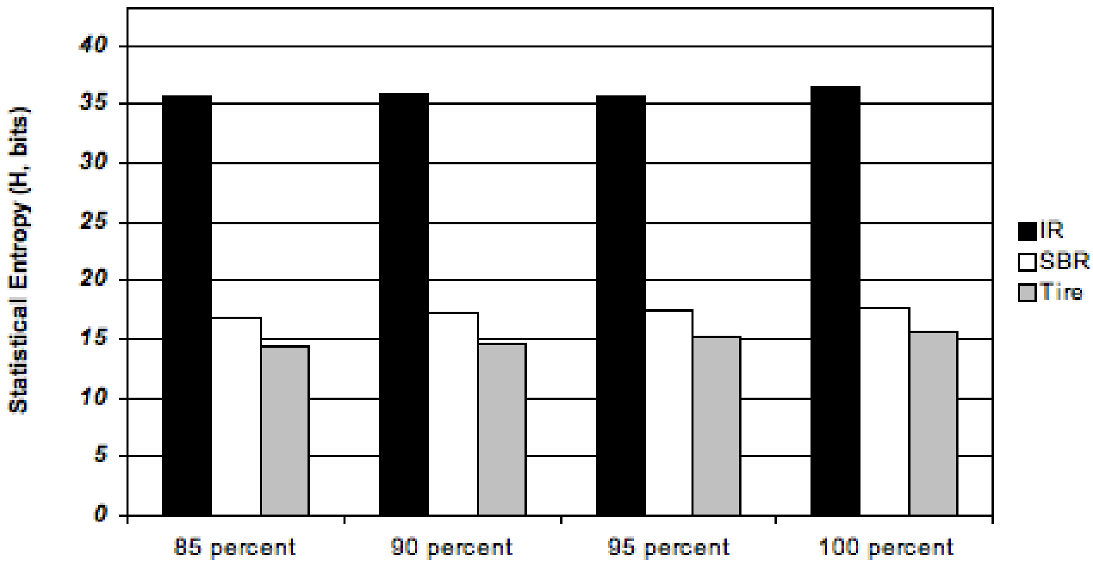


Figure 25 H-Values for thermal treatment of IR, SBR, and Tires at different levels of output mass

Figure 25 shows that whether one chooses to use SCE or H-values as the metric of choice to examine the fate of carbon in waste treatment systems, the illustrative results are the same – the concentrating or dilution effects of the examined system are well-described. The important conclusion here is that one arrives at nearly the same result irrespective of the chosen mass fraction of output species.

As a final test, models were constructed to determine the number of output species necessary to significantly effect the hypothesis that 90 percent of mass is sufficient to reliably calculate entropy emissions. The model was constructed so that there was one output species that accounted for 90 percent of mass, and the remainder was distributed evenly among X number of output products. This test demonstrated that the entropy results only began to be affected after X reached 250 species.

Energy Considerations

Later in this chapter we will present a case study comparing the entropy effects of two different treatments for municipal solid waste (MSW) – landfills and waste to energy (WTE) facilities. For a complete analysis, this will require the inclusion of the entropy effects of the difference in net energy utilized by the two respective treatments. For an average ton of MSW, a landfill can be expected to extract 3,700 MJ of net energy (considering a conversion efficiency of 32 percent) through the capture of landfill gas (Themelis and Ulloa 2007). WTE facilities, on the other hand, utilize the heating value of MSW, amounting to approximately 13,000 MJ of net energy per ton (Themelis 2001). This difference must be made up by production of energy from the grid – a significant portion of which will be generated from carbon-rich sources of energy (i.e. coal, natural gas, and oil) – resulting in increased emissions of carbon (and an associated increase in entropy). This has been accounted for in our methodology extension and is discussed in the "entropy calculation" section below.

Forcing Factor

The last piece of the extension of the SEA methodology to carbon involves the addition of a forcing factor to account for the environmental impacts related to the "quality" of different emissions. One such consideration is the global warming potential of greenhouse gases. For instance, it is generally accepted that the radiative forcing effect of methane emissions amounts to 21 times that of carbon dioxide (Harvey, Gregorgy et al. 1997). As the statistical entropy model is built upon the foundation of mass flows and the concept of a closed mass balance (that is, total output mass must equal total input mass), it is necessary to modify the methodology to account for environmental impacts related to inputs and emissions (such as global warming potential of greenhouse gas emissions).

To account for the potency of methane as a greenhouse gas, a forcing factor needs to be introduced to the entropy calculations. Earlier, the total substance flow $\dot{X}_j$ was introduced in equation (1). This can be more precisely defined as the product of the total mass flow of the good (m_i) and the concentration of the substance in that good (c_i). The forcing factor is added here to account for the effect of methane:

$$\dot{X}_j = \dot{m}_i \cdot f_k \cdot c_i \qquad (8)$$

where f_k is the forcing factor.

This forcing factor can also be used for other adjustments, both positive (e.g. "value-added" goods that are produced, such as ethanol) and negative (e.g. toxicity of produced goods).

Summary of carbon extension

In the preceding section we showed examples of how the statistical entropy method can be extended to carbon first through the analysis of simple chemicals (IR and SBR, which are "pure materials") and then through a more complex material (tires). It was shown that 90 percent of gas phase outputs could be used to make reliable entropy calculations, even though carbon fractionates to numerous different chemical species – especially in thermal systems. This selection makes sense, since typically thermal systems achieve equilibrium product concentrations. However, it is unlikely that this can be used for chemical synthesis or other processes where the exact kinetics needs to be known to determine the potential by-product formation. It is effectively a way to analyze a multitude of processes without the need for actual experimental or field data. The next step is to apply the analysis to a real world scenario involving MSW, namely the comparison of the SCE (or delta H) of landfills versus waste to energy facilities.

Application of the Extended Statistical Entropy Methodology

The two most common methods for final disposal of municipal solid waste (MSW) are landfilling (LF) and combustion with energy recovery or waste to energy (WTE). Studies have been performed which provide a lifecycle inventory (LCI) of waste processes (Harrison, Dumas et al. 2000). In addition, the literature includes many analyses of the overall environmental effects of landfilling from a lifecycle perspective (McDougall and White 2001). Furthermore, well-known databases such as EcoInvent contain detailed inputs and output emissions factors for a range of substances. All of these types of resources can be used to help construct the input data necessary to perform the SCE calculations for landfilling and combustion of MSW.

For this study, choices must be made as to which transfer coefficients will be used for carbon in landfills and WTE facilities. Foremost among the determining factors is which timescale to use when it comes to landfills. While WTE plants employ combustion reactions that yield instantaneous (and quantifiable) emissions, landfills are essentially giant bioreactors. Many of the biological and chemical reactions take place over centuries, making direct comparisons with WTE plants more difficult.

Many attempts have been made to estimate transfer coefficients for different substances entering a MSW landfill (Baccini, Henseler et al. 1987; Belevi and Baccini

1989). Some primarily use water balance models to predict the production of leachate while using sampling methods and kinetic models to determine the production and composition of landfill gas. Others have used these models to determine specific flows of substances based on inputs of MSW to landfills in temperate climates (Doberl, Huber et al. 2002; Themelis and Ulloa 2007).

For our baseline comparisons, we have chosen 100 years as the timeframe by which to measure landfills. To some extent this is standard practice in LCA studies; on the other hand, technological development is rapid and the way we manage landfills (even "inactive" ones) may be drastically different in a century's time. However, we have also included transfer coefficients for landfills on the 1000 year and 10,000 year timescales. It is clear from the results of these additional calculations that the choice of timescale has a significant effect on the entropy of carbon emissions (see "entropy calculations" and "discussion" below).

Brunner et al calculated the transfer coefficients for carbon in a moderate climate landfill at the 100, 1000, and 10000 year timescales (Table 21) (Bruner, Doberl et al. 2001). Also, Snilsberg et al calculated transfer coefficients for the combustion of MSW in waste to energy facilities (Snilsberg, Jonasson et al. 2004). These transfer coefficients allow us to calculate the input and output mass balances and entropies for landfills at the different timescales.

Table 21 Transfer Coefficients for waste facilities at different timescales [19]

C Output (emission)	Transfer Coefficient (100 yr)	TC (1000 yr)	TC (10000 yr)	TC WTE
Biogas	.54	.70	.92	.99
Buried in LF	.46	.29	.5	n/a
Leachate	Neg.	Neg.	Neg.	n/a

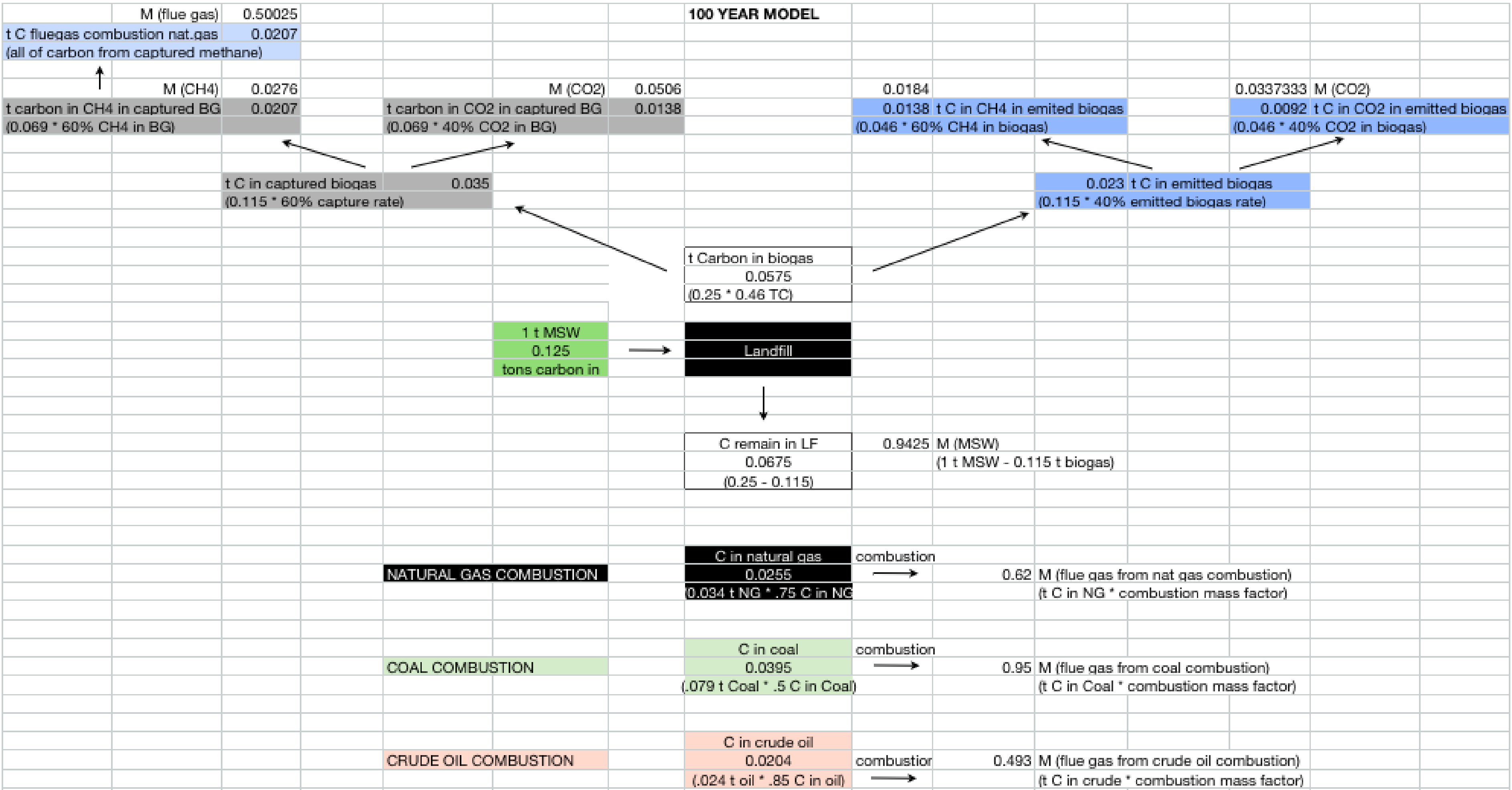
Entropy calculations

Like the SBR, IR, and waste tire models detailed in previously, geogenic concentrations of emissions are necessary in order to perform SCE and/or delta H calculations for landfills. Atmospheric concentrations of methane and carbon dioxide are well documented – we used recent values of 380 ppmv CO_2 (2007) and 1750 ppbv for CH_4 (2007). As other carbon emissions are negligible from the point of view of entropy calculations it was unnecessary to reference their associated geogenic concentrations.

Once all input and output flows for each scenario are quantified, it is possible to calculate the substance concentrating efficiencies as per the method introduced (above). The first step in our analysis was to perform a "direct comparison" between a generic landfill (100 year timescale) and a generic WTE facility. From an initial inspection of the data, landfills perform significantly better with respect to carbon than WTE facilities. This is expected (and makes intuitive sense) because most of the carbon remains "undigested" in the landfill over relatively small timescales.

Figure 26 shows the 100 year carbon flow model we constructed to allow for the accurate calculation of statistical entropy for landfills.

Figure 26 100 Landfill Carbon Flow Model



Energy considerations

While the results of the entropy analysis clearly show landfilling on short time scales has a lower SCE, the analysis does not consider the effect of "co-products" produced as part of the waste treatment process. Energy (or electricity) is a principle co-product of waste treatment processes (both landfills and WTE plants). One of the key advantages of waste to energy as an MSW treatment process is that it takes fuller advantage of the inherent energy in waste materials. From an entropy perspective, energy credits are important because we are able to "offset" emissions related to electricity produced from conventional fossil sources.

In our analysis we therefore account for the reduction in statistical entropy associated with alternative routes of waste management as follows. The energy extracted from a ton of landfilled MSW is subtracted from the energy extracted from a ton of MSW sent to a WTE facility. This difference is assumed to be energy needed to be produced by the grid. Thus, WTE can be expected to achieve a resource efficiency savings of roughly a factor of four over landfilling.

To better compare the energy factors on a common scale we next converted the energy factors into "carbon equivalents" as has been done elsewhere (Harvey, Gregorgy et al. 1997). It was assumed that the "lost electricity" attributable to landfilling had to be compensated by the production of an equivalent amount of electricity from the grid. Thus, .034 tons of natural gas; .079 tons of coal; and .024 tons of crude oil (roughly 4,000 MJ) were added to the carbon entropy input model. At this point, it is clear the difference between WTE and landfilling is about 30% in terms of SCE (Figure 27). Note that the ratio of CH_4 to CO_2 to is 3:2. Tables with input data for the 100-year landfill and the WTE follow.

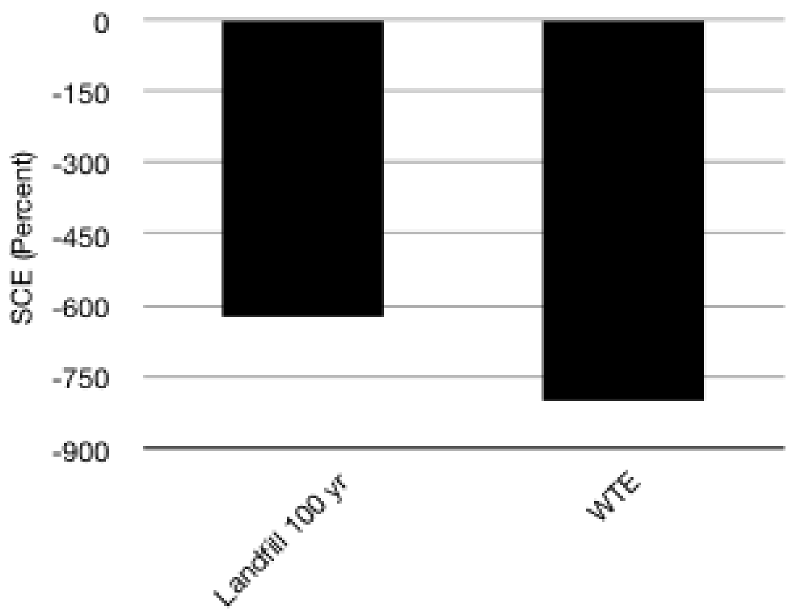


Figure 27 SCE of LF vs. WTE with Energy Offsets

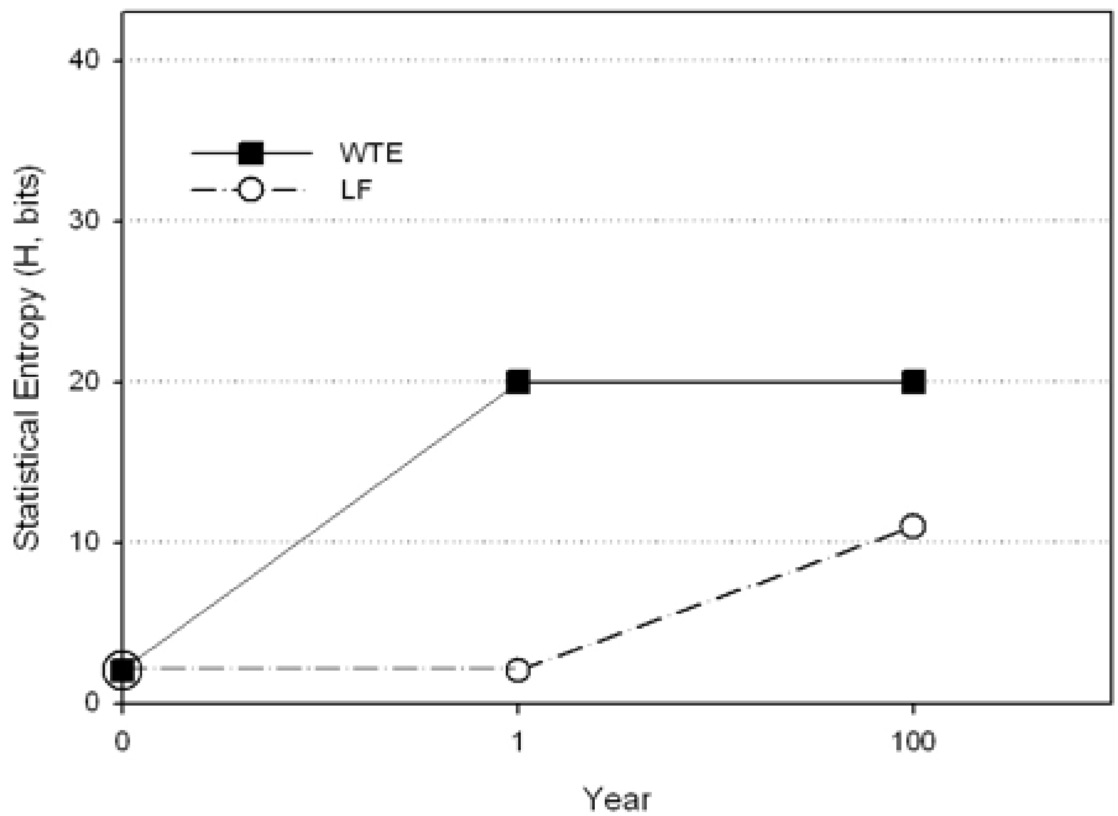


Figure 28 Entropy changes in MSW WTE and 100 year LF

As Figure 28 shows, landfills perform favorably versus a WTE facility in treating MSW with respect to carbon over a 100-year time span, even when factoring in energy offsets (energy that has to be produced from the grid that could have been harnessed in a WTE facility). To get a true sense of the impacts of the two treatment methods, however, there are additional factors that have to be analyzed. The first is radiative forcing.

Geogenic concentrations in air		
CO2	0.0001	
CO	0.00001	
TOC	0.000001	
PCDD/F	1.00E-11	
Geogenic concentrations in water		
TOC	0.000001	
Input	**M**	**c**
MSW	1	0.075
Air	5.1	0
Water	0.7	0
Aux. agents	0	0
Output	**M**	**c**
Flue gas	6	0.01
Bottom ash	0.28	0.01
Fly ash	0.025	0.005
Filtercake	0.003	0.005
Pur. waste water	0.5	0.000001
Iron scrap	0.003	0.01

Table 22 WTE basic model inputs and outputs

Entropy Calculation of the C-Balance of MSW Landfilling

Required Input Data

Geogenic Concentrations in Air	(mg/mg air)	
CO_2	0.0004	
CH_4	0.0002	
Input	**M**	**c**
MSW	1.00	0.25
Natural gas	0.03	0.75
Coal	0.08	0.50
Crude Oil	0.02	0.85
Air	3.05	0.00
Methane source	1.38	0.80
Outputs	**M**	**c**
Flue gas (BG combustion)	1.00	0.04
CO_2 (captured BG)	0.10	0.30
Emitted biogas CO_2	0.07	0.30
Emitted biogas CH_4	1.41	0.80
Flue gas (NG combustion)	0.62	0.04
Flue gas (coal combustion)	0.95	0.04
Flue gas (crude combustion)	0.49	0.04
MSW remaining LF	0.89	0.15

Table 23 Landfill 100 year basic model inputs and outputs

Effects of radiative forcing

The default assumption under the entropy model is that input molecules of carbon go to carbon dioxide. Thus, when the forcing factor is applied, and additional carbon appears as a result, the input must be adjusted to account for this difference. In the case of methane in our examined scenario, a "methane source" is added to the model to close the mass balance. With greenhouse gas forcing considered, the effectiveness of landfills with respect to carbon concentration is greatly diminished (Figure 30).

Effects of time and carbon sequestration and storage (CCS)

The addition of energy offsets and climate forcing changes the results of the analysis significantly, as the SCE of landfills decreased by a factor of 4. After 100 years, however, there is still a significant amount of carbon remaining in landfill deposits. When one examines the fate of carbon for much longer timescales the comparison with WTE becomes even more dramatic. Using transfer coefficients on 1000 and 10,000-year timescales and including the effects of energy offsets and forcing, we re-calculated the entropy models (Figure 29).

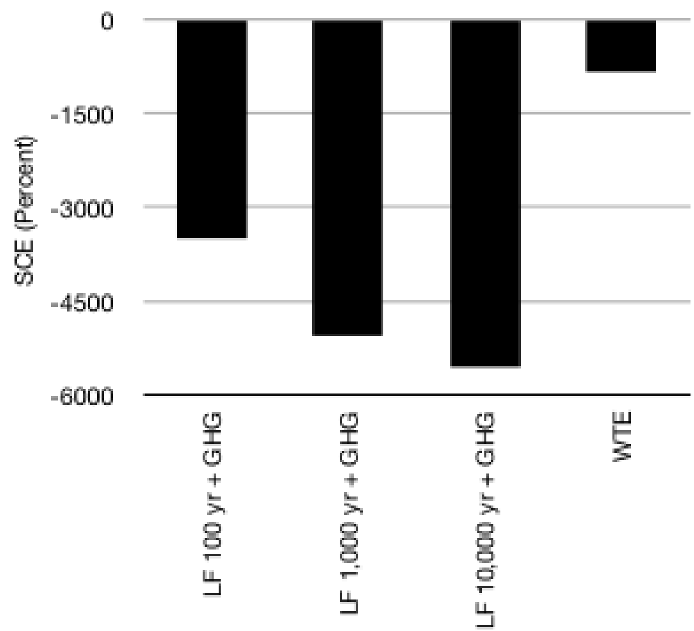


Figure 29 SCE comparison with energy offsets and different timescales

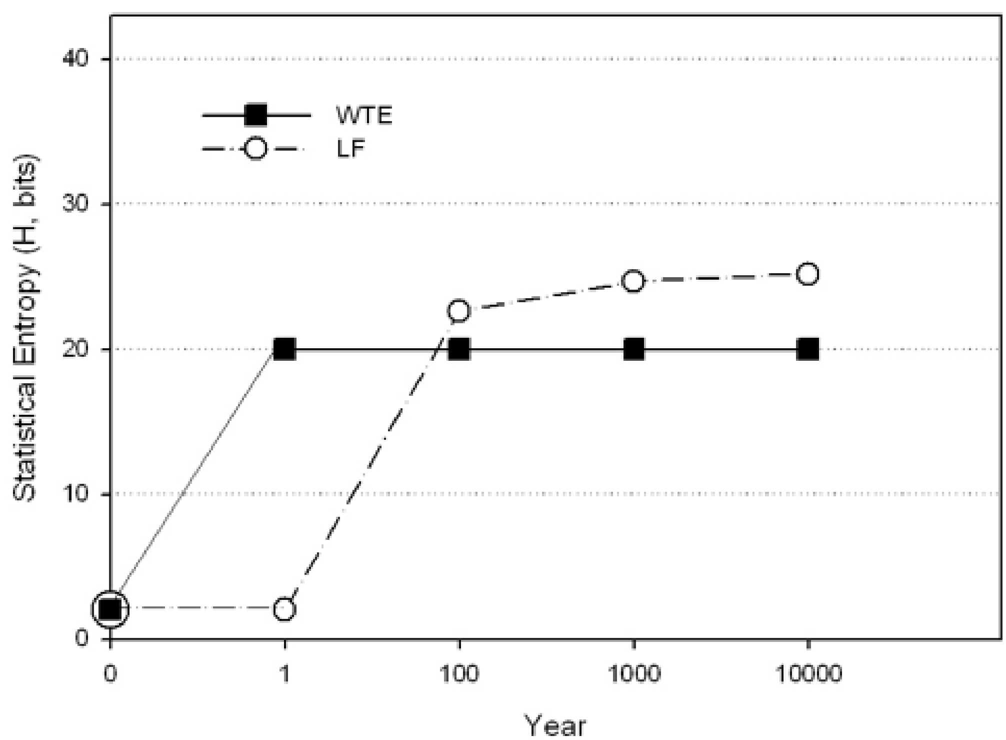


Figure 30 Landfill GHG Model vs. WTE Delta H Graph

It is clear from these results that, when including longer timescales and energy offsets, landfills become less attractive with respect to carbon management. The final step in this analysis is to consider the likely advancement of carbon capture and storage (CCS) technologies.

As global warming concerns grow, CCS is likely to play a larger role in the package of solutions applied to reducing the greenhouse gases we emit into the atmosphere. For this study, we incorporated into our model some basic assumptions on how a CCS system would perform in both a landfill gas-to-energy and WTE operating environment. Assuming the CCS system of a WTE facility would operate with the efficiency of a coal-fired power plant (85%); and that a landfill CCS system would perform similarly to a natural gas power plant (86%); we recalibrated the model to account for CCS (Figure 31). (Metz, Davidson et al. 2005)

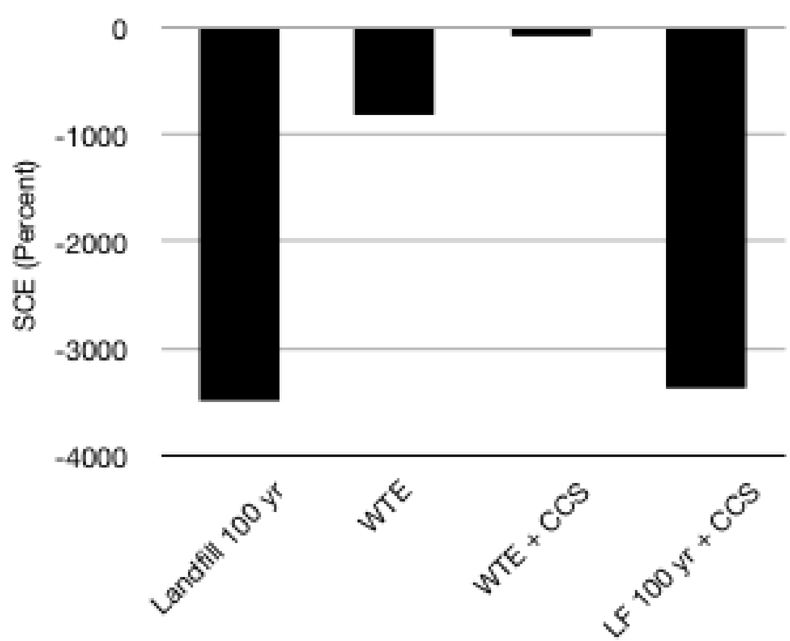


Figure 31 SCE comparison with energy offsets, timescale adjustments, GHG forcing and carbon capture and storage

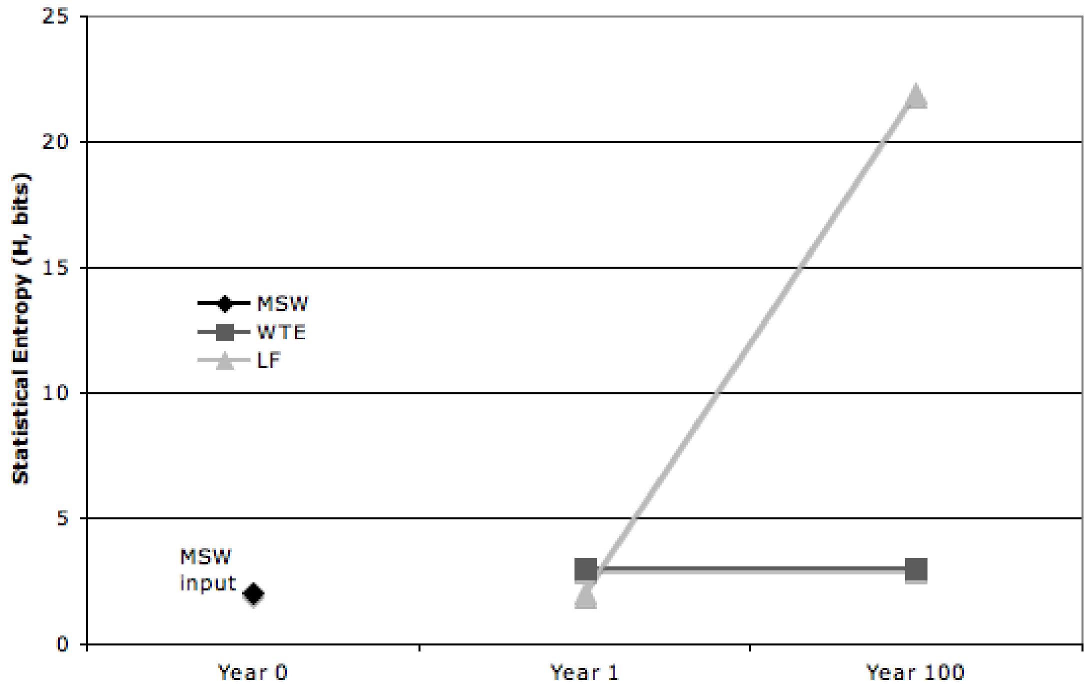


Figure 32 Delta H including CCS

The inclusion of CCS demonstrates the increased complexity of managing carbon through landfilling. With the flows of carbon concentrated in the flue gas of a WTE plant vs. the dispersion of carbon flows in a landfill (both over time and space), it is much easier to capture significant amounts of carbon in the combustion scenario. (In the landfill model, it was assumed that – as in other cases – 60 percent of the landfill gas production for the first hundred years was captured, and that the associated carbon emissions for this fraction were sequestered. The remaining emissions of biogas (methane + carbon dioxide) over the modeled lifespan were assumed to have been lost to the atmosphere.)

Discussion

This chapter discusses the extension of the statistical entropy method to carbon through the use of data produced from laboratory experiments. It should be noted, however that the method introduced here would also work in situations where no real data were available. For example, if one wished to compare the SCEs of two thermal systems, it would be possible to use equilibrium calculations to estimate the generation of major gas phase carbon species and use those values to calculate the entropies.

As climate concerns increase, it becomes more and more important to account for the flows of carbon through the technosphere. Statistical entropy analysis is a quantitative measure of the carbon lifecycle. The results of this chapter show that certain intuitive assumptions about the fate of carbon in waste management systems are erroneous upon further inspection.

First of all, landfills appear to be more efficient at concentrating carbon (or at least minimizing its dilution) in moderate timescales. However, when additional factors are considered (most notably the active management of carbon emissions, the inclusion of lifecycle energy metrics, and the greenhouse gas forcing effect) they are a less attractive option.

With these results in mind, it is important to consider the relative strengths and weaknesses of this approach to waste management systems evaluation. The SE method is excellent for decisions based upon the analysis of "pure" materials flows (such as metals). It also has a great deal of usefulness when the appropriate analysis boundaries are considered, and the appropriate environmental metrics (e.g. radiative forcing) applied. Finally, it is a relatively quick method to employ – a key advantage for policymakers.

On the other hand, a key shortcoming of this method is that it is a substance specific approach. Using an SE methodology, it may be possible to assess the extent to which waste management systems dilute or concentrate specific substances. Waste management systems are essentially simultaneously managing multiple materials. The method might be difficult to use to benchmark the overall performance of waste management systems, in which some substances are concentrated while others are made more dilute.

This leads to a question of whether there is a method closer to LCA that provides a fuller (though still "quick and quantifiable") picture of the effectiveness of waste management systems. Lifecycle embodied energy has been used in other applications (Scheuer, Keoleian et al. 2003) and can perhaps also be employed for waste management system comparisons. We have identified the development of standardized metrics and approaches to assess life cycle embedded energy of waste management systems as an important area for future research. This is what we attempt to accomplish in the next chapter, on Resource Conservation Efficiency.

4. Resource Conservation Efficiency

Introduction

Life Cycle Assessment (LCA) has long been used to model and evaluate the environmental impacts of various products and services (cite references). Among the sectors of the economy that have received attention from LCA practitioners is municipal solid waste (MSW) treatment and disposal. LCA is often used by waste stakeholders to compare technologies and/or treatment regimes (Doka and Hischier 2005) (examples are also presented in the Literature Review chapter of this thesis). The problem with conducting full-scale LCA for these purposes is that it is expensive, time consuming, and requires bringing in specialists to complete.

Many “workarounds” have been suggested and used to substitute for full LCAs. One such method is the use of cumulative energy demand (CED, (Blok 2006)), also known as lifecycle embodied energy (LEE). CED, as described in the Literature Review chapter, is a cradle to grave account of the energy inputs and consumption necessary to manufacture, use, and dispose of a product. All inputs from each stage in the life cycle are accounted for in energy terms, including direct energy inputs; feedstock materials; and capital goods

In this chapter, we propose a variation of the CED method – a new metric called Resource Conservation Efficiency (RCE) – to evaluate the effectiveness of MSW management systems with respect to environmental conservation and resources protection. As will be explained in the methodology and discussion sections of this Chapter, this metric allows for an apples-to-apples comparison of different cities’ waste management system performance, even where different technology suites and strategies are employed.

This method grew out of the common credo that recycling of materials "saves energy" over manufacturing from virgin sources plus landfilling and/or incineration of waste materials that cannot be recycled. This is a familiar claim and has been extensively studied in the literature (Porter and Roberts 2005). Our research has shown that this is, in fact, the case for many familiar materials. However, it is also clear that not all MSW is currently recyclable and those materials that are not recyclable are often high in calorific value and, thus, excellent (or at least passable) fuels. Furthermore, even recyclable materials are not fully captured by any city. Any system of waste management analysis must account for the studied systems' tendency to capture these energy potentials. Additionally, it is hoped that any such evaluation system also factors in the environmental benefits or hazards associated with waste management choices. A system that does this rationally, objectively and – perhaps most importantly – intuitively (for the general public's understanding) would be an invaluable addition to the world of waste management decision-making.

It is believed that RCE is just such a tool, and this paper outlines the development, rationale, and applicability of this new evaluation process. First, the quantifiable underpinnings of the RCE system are developed and examined. This is primarily accomplished through the use of SimaPro LCA software (Goedkoop 2007), with its many included databases. Materials typically found in MSW are input to the database, and the relevant data are recorded – namely, CED and the environmental effects of manufacture and waste treatment of the examined materials, measured by means of EcoIndicator 99, an LCA industry standard method for assessing and weighting human health, resource depletion, and environmental impacts of products and services. Once these factors are tabulated, a correlation is drawn between the CED and EcoIndicator scores, in effect vetting a CED-based evaluation score for use as a proxy for overall waste management system performance measurement. Next, the CED values are used to calculate material-specific RCE values, which are also vetted by means of regression plots against EcoIndicator scores.

As a practical means for determining the suitability of this new metric to evaluate the MSW management systems of different localities or regions, RCE is applied to the actual treatment regimes of two American cities, Honolulu, HI and San Francisco, CA. These cities are chosen for two important reasons. First, they rely on different and opposing methods of final disposal for non-recycled wastes – landfilling in the case of San Francisco, and waste to energy (WTE) in the case of Honolulu. Second, in 2007-8, we received a grant from the US EPA Region 9 to examine the MSW data of Region 9 states, which gave us a unique opportunity to examine in depth the waste management data of these two interesting cities.

Background

Different methods of Environmental Evaluation

There are many methods available for quantifying the environmental impact of human activities, each with their own strengths and weaknesses (Daniels and Moore 2001; Daniels 2002). Life cycle assessment is attractive in the case of waste management, for reasons discussed earlier in this thesis. As garbage is an everyday part of all citizens' lives, it is often a hot-button issue for environmentalists. Perhaps for this reason, there have been many LCA studies on MSW management, even though its actual environmental impacts might not measure up to the incredible attention paid to it.

Many cities hire experts to perform detailed lifecycle impact assessments for their current and future waste management planning and infrastructure (Denison 1996). These studies are often quite useful and informative but hardly ever conclusive – the way LCAs are typically performed and reported, it takes a team of experts to interpret them in addition to performing them. So-called streamlined LCAs are performed at least in part as a response to this (1999) but, again, the results are usually open to interpretation and difficult to use as benchmarks against other cities.

Use of CED

As discussed earlier in this thesis, cumulative energy demand (CED) is a useful survey-level indicator for the environmental performance of products and services (Klopffer 1997; Huijbregts, Rombouts et al. 2006). A previous concern about the usefulness of CED in waste management – the relatively high uncertainties related to waste treatment – have largely been addressed by the latest version of the EcoInvent database and by the present work described in this report (Doka and Hischier 2005).

CED has been used to evaluate the environmental and energy impacts of several different sectors. It is frequently employed to determine energy payback periods for alternative generation technologies such as solar (Knapp and Jester 2001) and wind (Wagner and Pick 2004); and is also commonly used to evaluate the efficacy of efforts to produce energy from biomass (Kim and Dale 2004). It is also commonly used to assess the life cycle environmental impact of buildings (Thormark 2002; Scheuer, Keoleian et al. 2003). Though these studies and others like it include waste management as a phase of the overall lifecycle of the systems they are exploring, there has not yet been a CED/LCA-based metric that focuses specifically on evaluating MSW management systems. The RCE method described in this Chapter has been developed to address these issues.

It is hoped that the development of this metric will allow for a more rational, dispassionate approach to strategizing about municipal waste management decisions. Too often, opinions are formed based on noble but immeasurable goals such as "zero waste." While attractive as theoretical ideals, these kinds of targets have proven impossible in practice (to this point). In the meantime, cities continue to generate more and more waste that has to be dealt with somehow. This new metric provides a benchmark to more objectively judge how cities are doing in the present time with respect to one another, and a more objectively determined theoretical maximum performance.

Methodology used in Development of SimaPro "Test Model"

Though CED is a useful proxy to evaluate environmental performance, as already mentioned, it has not yet been applied to waste management systems evaluation. To verify the appropriateness of the use of CED as an LCA proxy and to determine how well it does work for waste management systems evaluation, it was decided to build our own database test models before using CED literature values to perform RCE calculations for actual cities.

The test models were built by choosing representative materials in SimaPro databases meant to represent the virgin and recycled content materials that are actually managed in real cities and covered in the literature on LCA and waste management. The primary purpose of this choice in vetting methodology is to take advantage of the many different materials present in SimaPro databases to get as wide a range of materials and comparisons as possible. The goal is to show a correlation between CED and environmental impact indicators for evaluated product/waste systems.

The materials chosen for the test model are listed in Table 24.

Table 24 Materials used in analysis

Plastics
Polypropylene
HDPE
PET
Papers
Kraft paper
Mixed low grade paper
Newspaper
Glass
Recyclable glass
Metals
Aluminum

These materials were each individually plotted using LCA database values for their corresponding production and end of life treatment regimes (Figure 37). Additionally the model is set up so that the following three scenarios are depicted:

- **A "maximum RCE" scenario**. This scenario is defined as follows
    - **for recyclable materials:** 100% recycling of the used material with zero disposal (i.e. 100% of the material is recycled into a new product). This is clearly a *theoretical maximum* scenario, representing an unattainable upper boundary of system performance.

- **For non-recyclable but combustible materials.** 100% virgin production plus 100% combustion with energy recovery of equal amounts of the waste product. In other words, for non-recyclable plastics, if one ton of material is produced, one ton of that material is also assumed to be collected and combusted in a WTE with energy recovery.

- **"Honolulu RCE" scenario.** Honolulu RCE scenarios are defined as follows:
  - A product assembly is defined such that it matches the actual waste management scenario as experienced for that product in Honolulu. For example, assuming that Honolulu manages one ton of PET in its waste stream, recycles 50%, combusts 40% with energy recovery, and landfills the remainder of 10% , it is modeled as follows:
    - Virgin production of 0.5 tons of PET to account for the replacement of the 0.5 tons of PET (0.4 tons combusted and 0.1 tons landfilled) that are not recovered from the waste stream;
    - Recycled production of 0.5 tons of PET, assumed to be remanufactured from the 0.5 tons collected from the Honolulu waste stream
    - Combustion of 0.4 tons of PET
    - Landfilling of 0.1 tons of PET.
- **"San Francisco RCE" scenario.** This is defined in the same way as the "Honolulu RCE scenario" setup described above, but using San Francisco waste management data used to determine ratios.

Once the assemblies are defined as described above, models are run in SimaPro using two evaluation techniques:

- EcoIndicator 99, evaluated for "Single Score" points (Goedkoop and Spriensma 2001). Single score points take the major environmental impact category scores and aggregate/weight them to approximate an overall score for total environmental/human health impact.
- Cumulative Energy Demand (CED)

As with the other impact assessment methods, choices have to be made as to which evaluation strategy one wants to employ. EcoIndicator provides three choices for weighting and normalization(Baayen 2000):

- Hierarchist: Represents a balance between short and long term time perspectives, and holds philosophy that proper policy can avoid many problems
- Individualist: Represents a bias towards shorter time frame of assessment, and holds philosophy that technology can avoid many problems
- Egalitarian: Strictest "environmentalist" approach, with a very long term view and the philosophy that problems are potentially catastrophic.

For this test model set up and for all subsequent models we have chosen the "hierarchist" approach, as it represents the "happy medium" (strongest balance) between the two extremes.

Test Model Results (Real-world scenarios)

There is a strong correlation between CED and EcoPoints scores for most materials in the test model. Each individual material listed in Table 24 is analyzed, and graphs are plotted depicting CED vs. EcoIndicator Single Scores. Figure 33 shows the plot for PET plastic. The results for PET show a strong correlation between CED and EcoPoints scores. Included in this model are the energy offsets from the production of electricity from the US Grid (for PP waste combusted in Honolulu). Landfilled polypropylene is assumed not to biodegrade and/or produce energy. Similar graphs from selected materials follow.

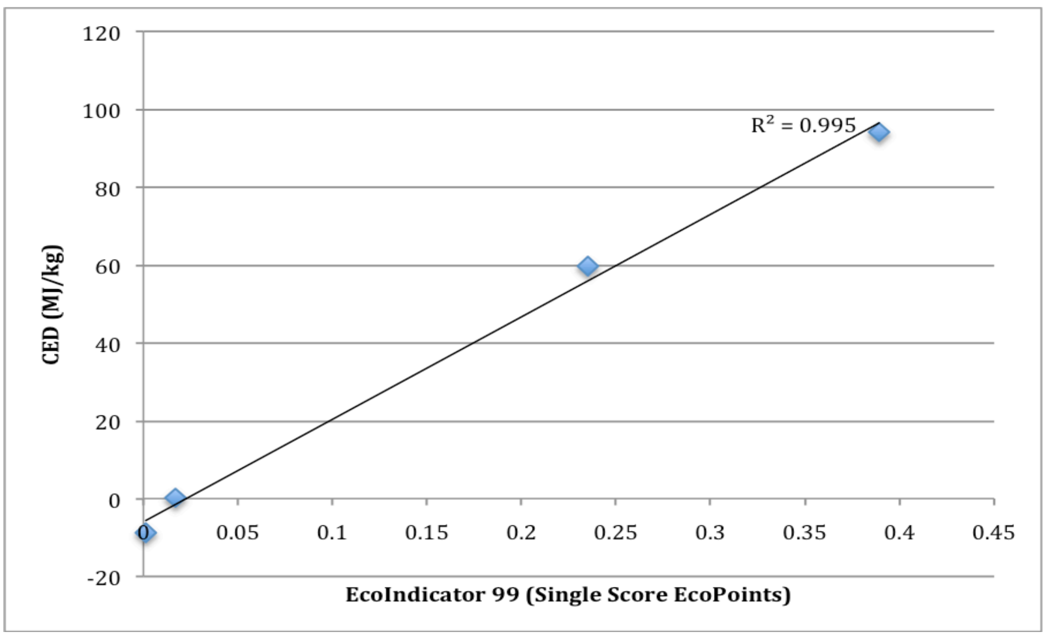


Figure 33 PET Plastic Test Model CED vs. EcoPoints

PET

When discussing PET in terms of municipal solid waste management, we are most often referring to post-consumer beverage bottles. While PET is a very versatile product, it is most often found in this form in waste management systems. It is also one of the most successful plastics in terms of recovery efforts – while many other plastics are sent to landfills even after separation in recovery schemes (Themelis and Todd 2004), there is a robust market for sorted PET.

The results for HDPE also show a strong correlation between CED and EcoPoints scores. Included in the model are energy offsets from the production of electricity from the US Grid. As in the case of PP, the landfilled waste is assumed not to biodegrade and/or produce energy.

Mixed low-grade paper also shows a correlation between CED and EcoIndicator scores. The model for mixed low-grade paper includes energy production, both for paper combusted with energy recovery in WTEs in Honolulu as well as for paper landfilled in San Francisco and Honolulu. (It is assumed that the landfill collects and combusts methane gas produced as a result of landfilling the given amount of paper, producing electricity that offsets grid production.) The correlation between CED and EcoIndicator 99 for newspaper is also present. The model once again incorporates energy production from landfilled and combusted materials, offsetting US grid production. Figure 34 shows the test model results.

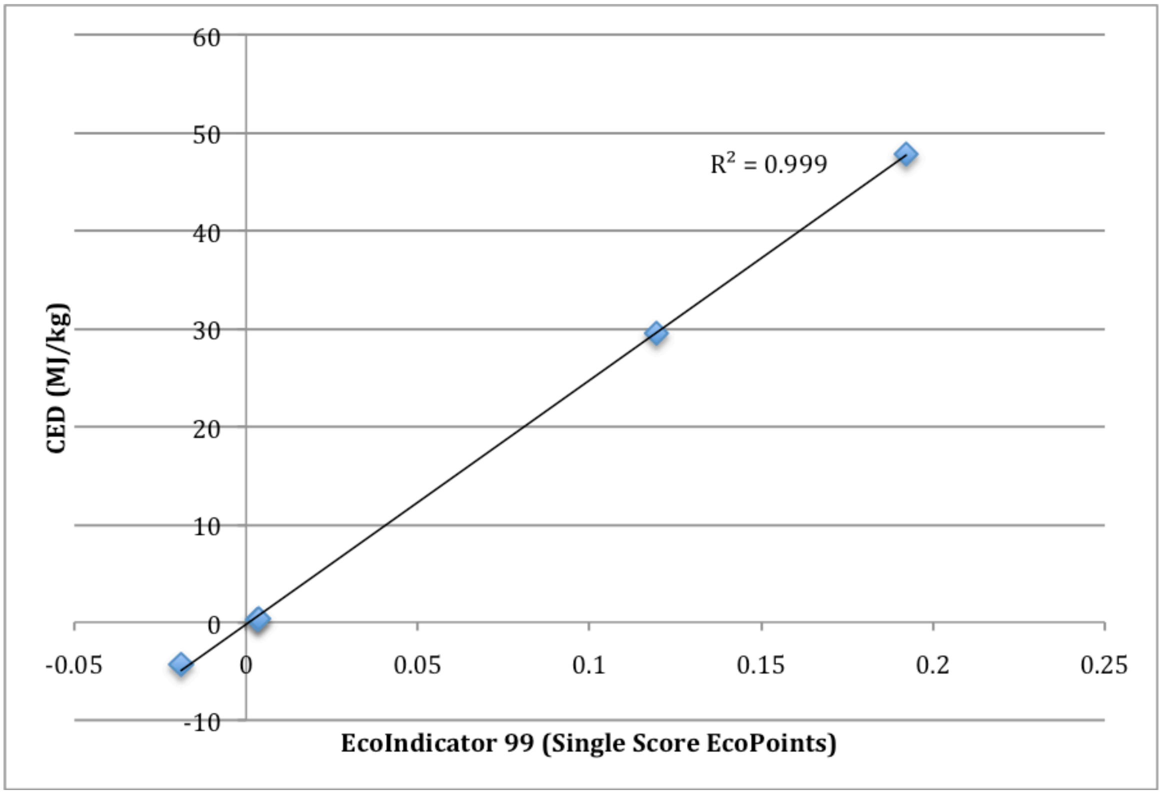


Figure 34 Newsprint Test Model Results

Glass produces no energy when combusted or landfilled. Any energy savings from glass must therefore be realized through recycling. Figure 35 shows the modeled performance of glass in the test model. The correlation is very strong between CED and EcoIndicator 99 scores.

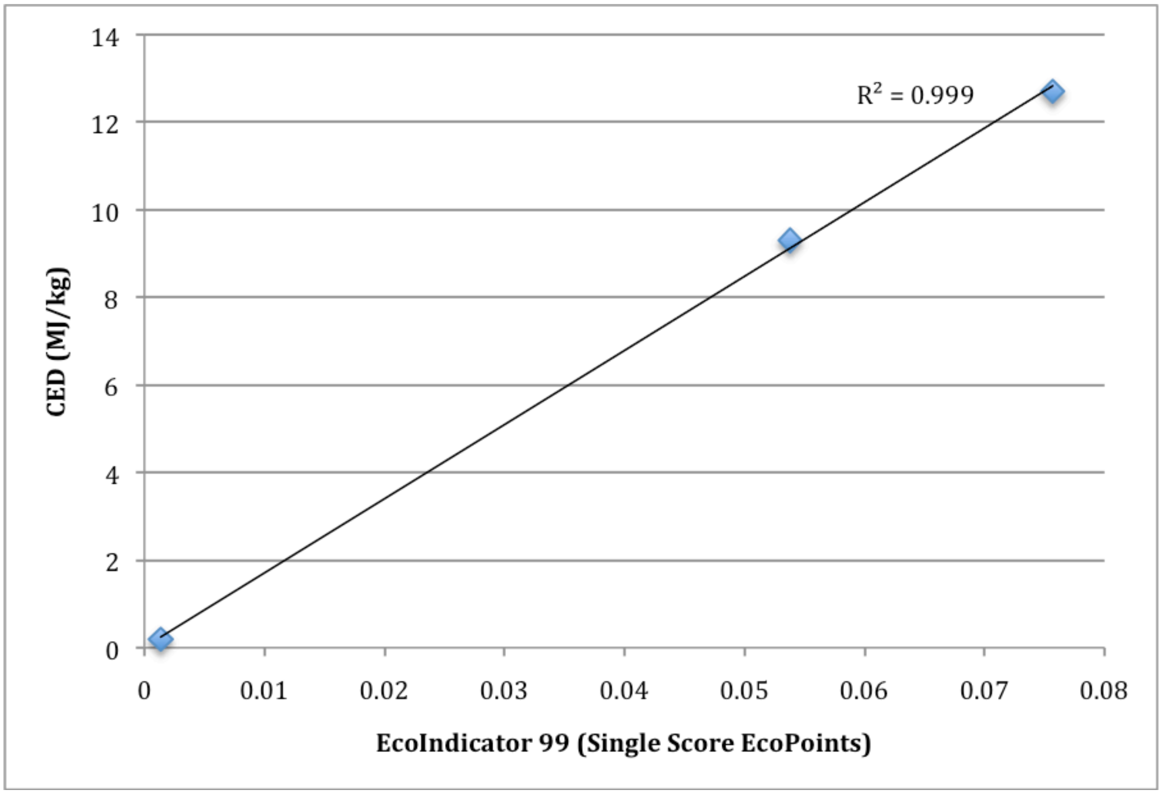


Figure 35 Glass Test Model Results

For our test model analysis, metals are represented by aluminum, as there are robust LCI data for this material and the discrepancy between recycled and virgin material CED is so high. Aluminum is represented in several databases in the SimaPro program. EcoInvent was chosen as test model database for aluminum.

There is a very high correlation between aluminum CED and EcoIndicator scores. It should be noted that the Honolulu's performance data includes the assumption (2007) that "nearly 100 percent of ferrous and nonferrous metals are recovered for recycling" in Honolulu's waste to energy facility. Figure 36 shows the test model results.

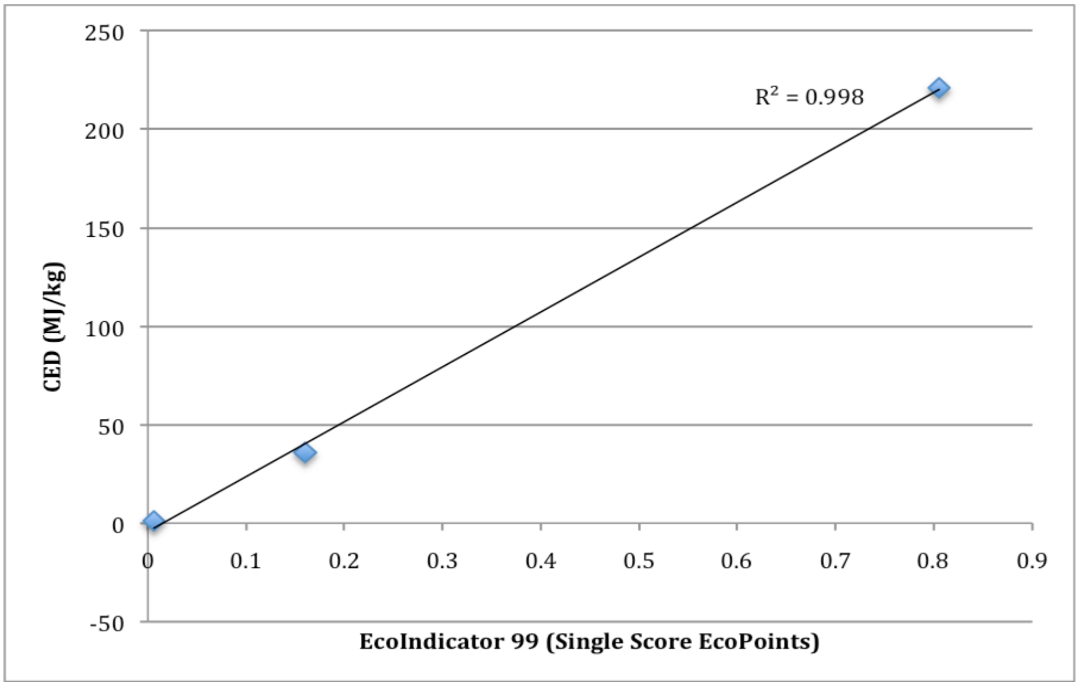


Figure 36 Aluminum Test Model Results

Aggregated CED vs. EcoPoints Regression Plot

To test the correlation between CED and EcoIndicator single scores for all materials examined in the test model, regression plots were constructed (Figures X and X). The first is the generic model for all of the materials without incorporating the "real-world" scenarios, and the second is the aggregated "real world" models. The results show a strong correlation, leading to the conclusion that CED is in fact a reasonable metric to screen for environmental impacts in waste treatment systems.

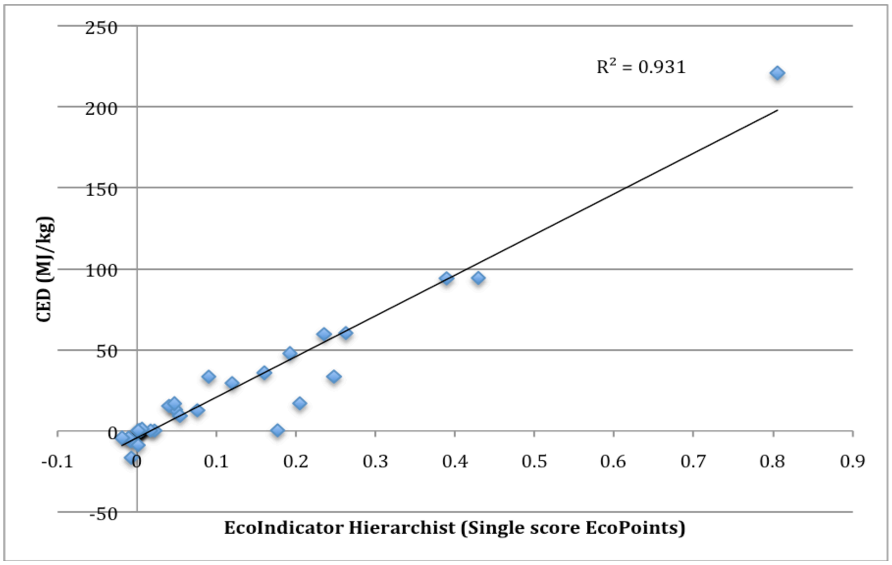


Figure 37 Aggregated generic model CED vs. EcoIndicator 99

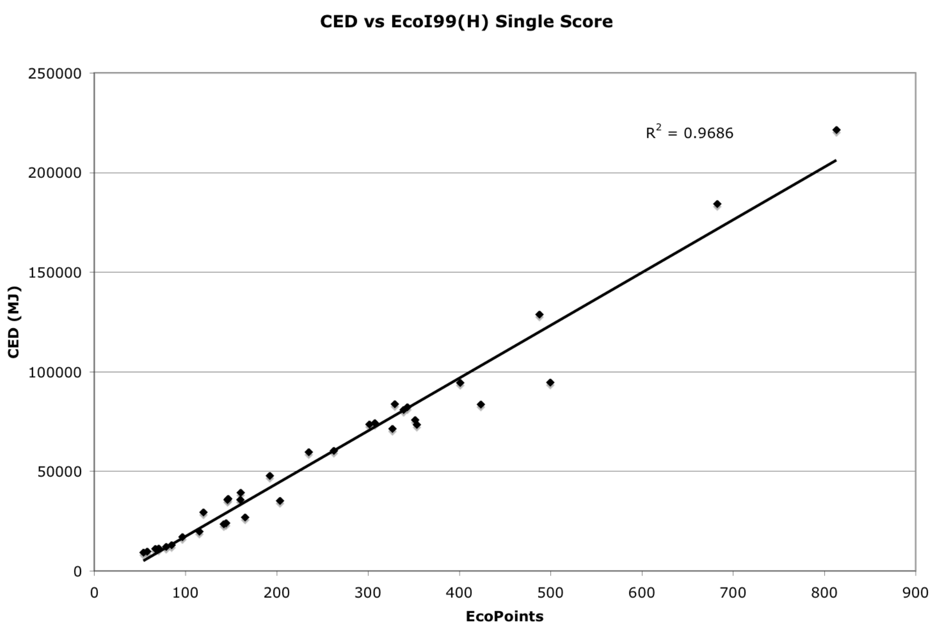


Figure 38 Aggregated Test Model Regression Plot

As a further check, it is useful to plot CED scores for the different scenarios against other environmental indicators, including the "Egalitarian" and "Individualist" strains of the EcoIndicator99 method. Figure 39 shows the plot for the Egalitarian analysis, while Figure 40 shows the plot for the Individualist version. Both show tight fits with the CED values.

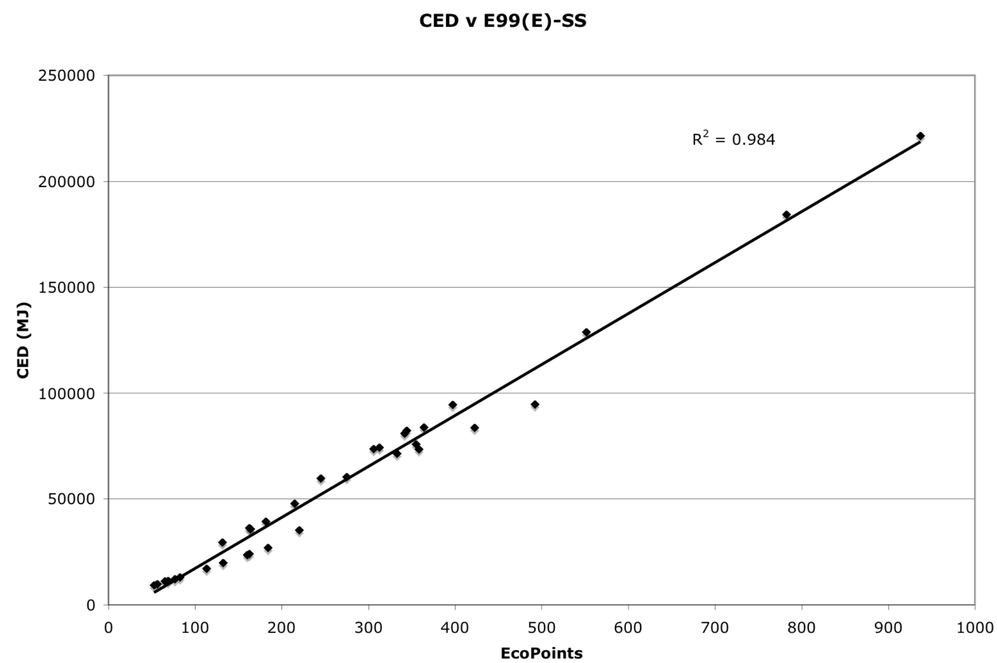


Figure 39 CED vs. EcoIndicator99 for Test Model (Egalitarian)

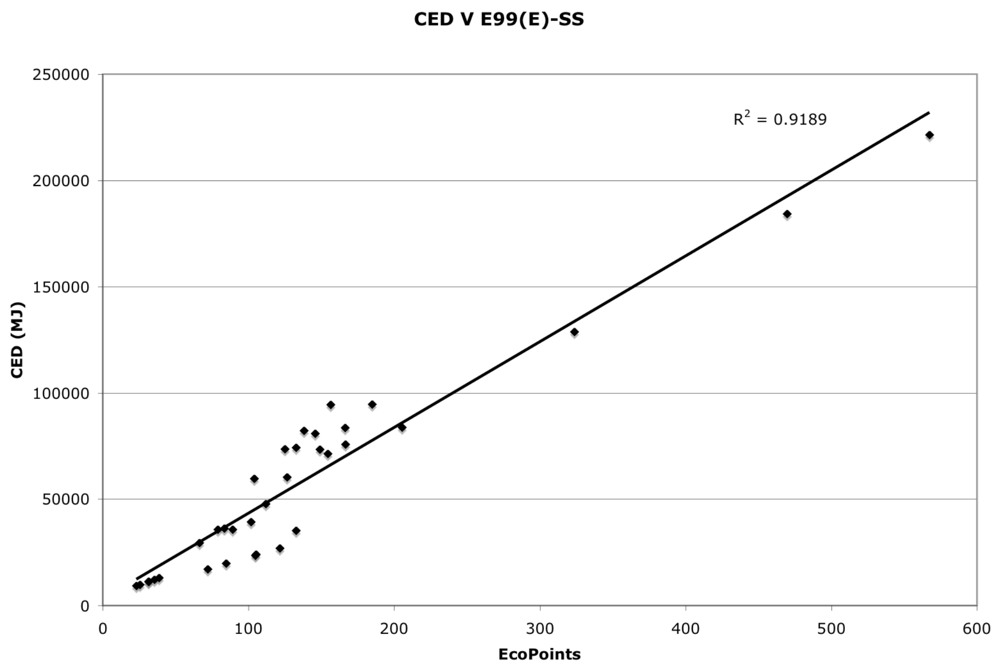


Figure 40 CED vs. EcoIndicator99 for Test Model (Individualist)

Two other impact models were used as well – the CML 2 baseline 2000, developed by the University of Leiden's Center for Environmental Studies (CML 2000). The CML analysis is for human toxicity impacts, which are determined from toxicological experiments meant to demonstrate "acceptable levels" of human exposure to various substances. These results shown in Figure 41. The other tool is IMPACT 2002+, developed by the Risk Science Center at the University of Michigan (Jolliet, Margni et al. 2003). The values presented in these results (Figure 42) are single scores for human health, weighted for the impacts of different LCI emissions.

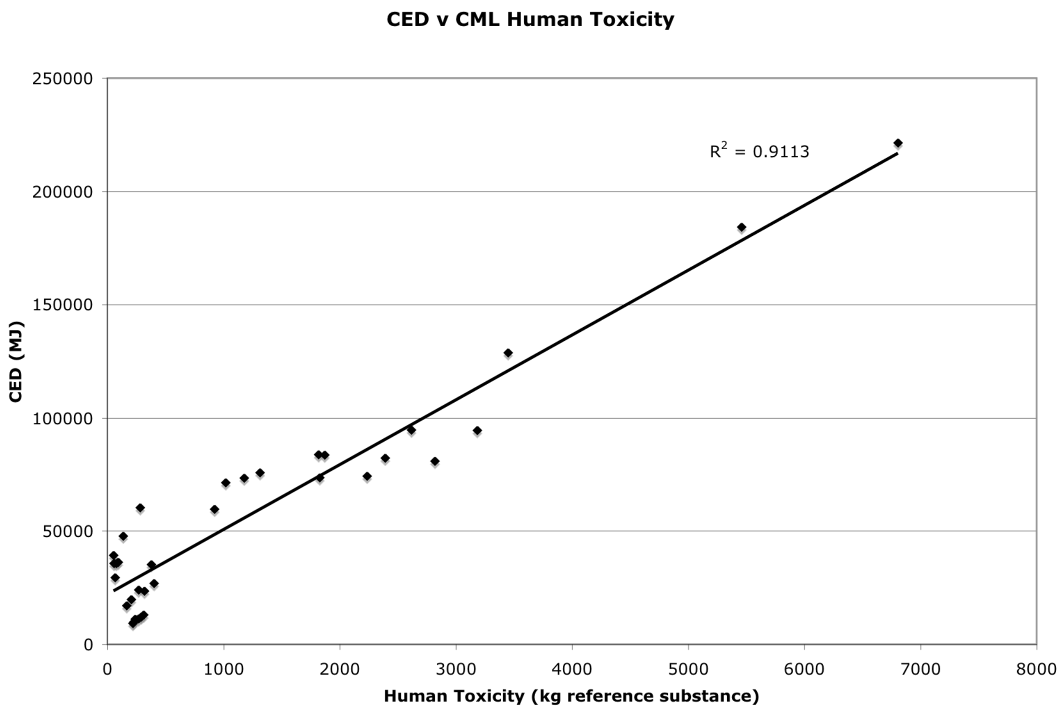


Figure 41 CED vs. CML 2 for Human Toxicity for Test Model

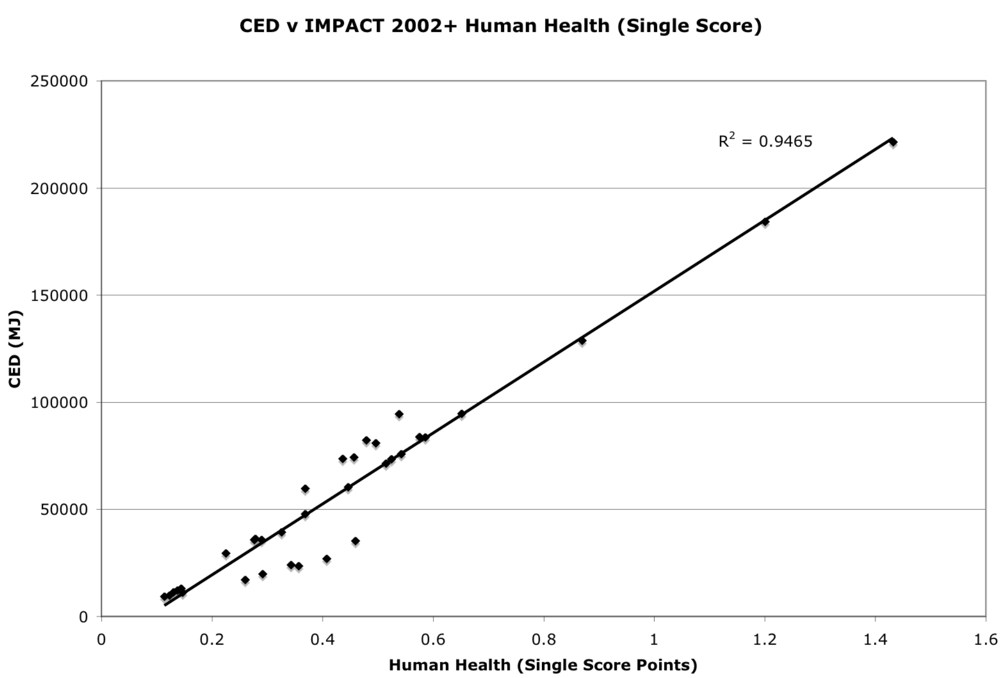


Figure 42 CED vs. IMPACT 2002+ Human Health for Test Model

Resource depletion

Though energy accounts for a large fraction of the resource consumption associated with the production and end of life management of a product or service, there are other measures that can be used to validate RCE as a resource conservation metric. One such measure is "resource depletion" in EcoIndicator 99. This is a measure of the surplus energy required to extract non-renewable materials (i.e. fossil fuels and minerals) based on the depletion of readily available stocks. This was also plotted against CED, and a decent correlation was observed (Figure 43).

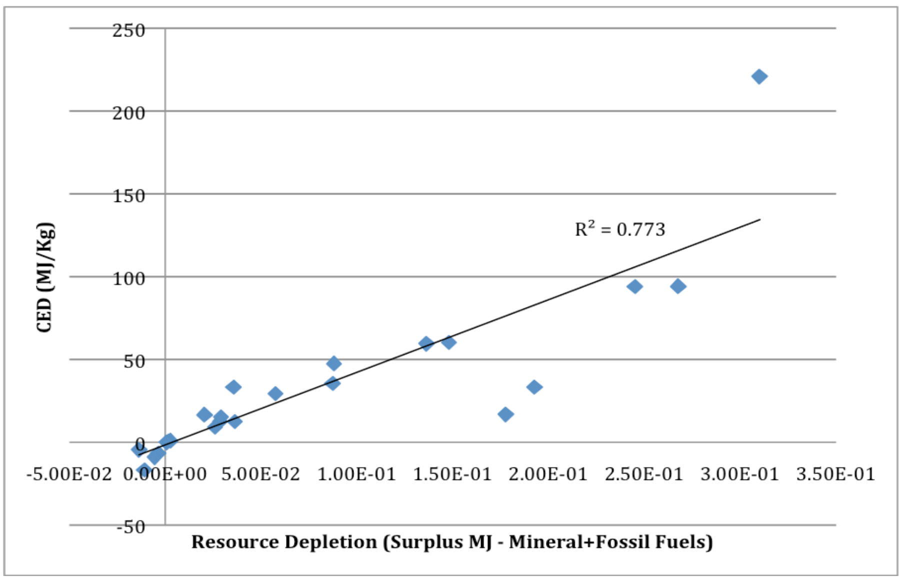


Figure 43 CED vs. Resource depletion Generic Model

Test Model Discussion

To reiterate, the purpose of the proposed RCE metric is to allow for a quick and replicable method of evaluating the effectiveness of waste management systems. The purpose of designing and running the test model explained in the above sections is to establish a correlation between CED values as calculated to represent waste management systems and the associated EcoIndicator scores. A fair correlation demonstrates that a

CED-based metric is a reasonable means to accomplish the stated objective. As the model results show, our method of calculating CED does indeed show a strong correlative relationship with EcoIndicator 99 single score, CML 2, and IMPACT 2000+ values.

Material Life Cycle Inventories and Recycling Offsets

In order to aid in the "intuitive" understanding of the benefits of proper management of different materials in the waste stream, it is useful to have a stronger sense of the processes behind the numbers. It is one thing to say that recycling "saves energy," but it is another to have a clearer picture of the processes that are avoided in recycling systems that allow energy to be saved.

It can be assumed for most materials that there is a process flow that is common to recycled virgin materials after a certain point in virgin production, as depicted by Figure 44 below. This is a simplified flow diagram for the production of aluminum, showing processes offset by recycling, but all virgin materials share the basic steps – raw material extraction and processing; fuel extraction and processing; and primary materials production. Similarly, recycling requires collection and processing before re-entering the production change and offsetting virgin materials steps.

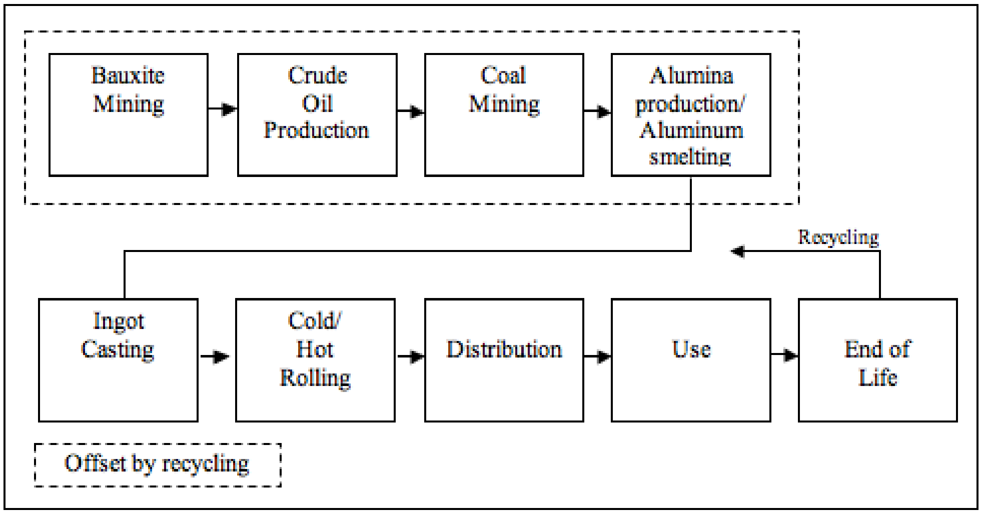


Figure 44 Simplified Aluminum Production Flow Diagram

In the case of aluminum, the large percentage of energy use and environmental impacts in virgin production come from the mining and processing of bauxite into alumina for later processing into aluminum products (McDougall and White 2001). All sources agree that the savings when recycling aluminum from scrap instead of manufacturing from virgin sources are vast (Choate and Green 2003). Indeed, aluminum is the most ideal product for recycling – not only there are significant energy and resource savings, but it can be recycled nearly infinitely as process losses are minimal. Table 25 summarizes the energy savings collected from different sources in the literature. (Note: some of this work can be found in (Weitz 2003), a report produced for the US EPA in 2003, a major piece of which was the collection of literature values for life cycle inventories for the different products in MSW.)

Table 25 Aluminum Energy Savings Literature Summary

Aluminum

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
RTI	176.78	8.99	167.79
McDougall	182.8	8.24	174.56
Porter	235	13	222
Average			**188.1**

The key difference between the production of steel from virgin vs. scrap sources (aside from the mining of raw materials) is the use of basic oxygen furnaces (BOF) in the virgin production. These BOFs are more energy intensive than their electric arc furnace (EAF) cousins, which are used in the production of steel from scrap metals (Margolis and Brindle 2000). The Summary of literature source values for virgin and recycled steel production is shown in Table 26.

Table 26 Steel recycling energy savings literature summary

Steel

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
RTI	22.9	8.6	14.3
McDougall	35.8	17.2	18.6
Average			**16.4**

The most significant energy expenditure that is avoided in the recycled production versus the virgin production of glass is the mixing and melting of primary materials –

mostly (and most commonly, as different kinds of glass can include some different components) Limestone, silica, and soda ash (Pellegrino 2002). The energy savings estimates are listed in Table 27.

Table 27 Glass recycling energy literature summary

Glass

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
RTI	7.6	5.0	2.6
McDougall	14.5	11.0	3.5
Porter	23.0	17.0	6.0
Average			**4.0**

The LCI literature for plastics manufacturing (at the very least from a recycling point of view) contains less information on recycling processes than the other materials in the MSW stream. This is most likely due to the much greater complexity of producing plastic products, and to the sheer variety of finished plastics that can be achieved. Due to this complexity, plastics are often "downcycled" into alternative, alloyed plastic products (such as plastic lumber) (McDougall and White 2001). It is very difficult (perhaps impossible) to determine from the literature and from municipal waste tonnage reports which plastics go to which final products. For these reasons, it is assumed (for simplicity) that HDPE plastics are recycled to HDPE flakes to replace virgin HDPE.

It is clear, however, that when plastic is successfully recycled it does yield significant energy savings over virgin production (Arena, Mastellone et al. 2003). The plastics included in this study are high-density polyethylene (HDPE) – assumed to include most post consumer plastic bottles; polyethylene terephthalate (PET) – assumed to include "other" post consumer plastic bottles such as laundry detergent containers; and polypropylene (PP) – assumed (again for simplicity) to include plastic tubs and silverware, etc.

HDPE is one of the possible end products produced from ethylene, which is itself a common and major product from petroleum cracking (Weitz 2003). Once ethylene is available as a feedstock, the type of finished product depends on the reactors and catalysts chosen and the desired result. HDPE resins are linear polymers with densities approximately 5 percent greater than LDPE resins. In MSW they can be found in plastic containers, etc.

Table 28 HDPE recycling energy literature summary

HDPE

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
McDougall	33.26	7.62	25.6
RTI (EPA)	64.0	7.6	56.4
RTI (APME)	69.63	n/a	n/a
RTI (SFAEFL)	69.6	n/a	n/a
Franklin	94.4	60.4	34.0
Average HDPE			**38.7**

Table 29 PET recycling energy literature summary

PET

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
RTI (EPA)	66.5	n/a	n/a
RTI (APME)	72.05	n/a	n/a
RTI (SFAEFL)	70.2	n/a	n/a
Arena	77	42	35.0
Franklin	85.4	54.2	31.2
Average			**33.1**

The only literature source of information found for this study was the Franklin 98 US database entry for virgin vs. recycled content polypropylene production. Franklin explicitly mentions that most of the recycled PP products in the US at the time of their study (the late 1990s) were from a relatively few curbside programs that collected food tubs and cups.

Table 30 PP recycling energy literature summary

PP

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
Franklin	99.4	65.2	34.2

Our analysis of paper is mostly confined to the few major grades of paper that are collected for recycling in Honolulu and San Francisco – linerboard, newsprint, office paper, and mixed low grade paper. Paper materials showed the lowest correlation between CED and EcoPoints in our test model analysis. This is most likely due to the fact that virgin paper manufacturers produce all of their own process energy from rejects in the forestry operations that supply their feedstock (Miller, Justiniano et al. 2005).

Linerboard is the paperboard material used to manufacture materials like shoe and cereal boxes. It is frequently produced from mixed, low-grade paper waste collected from municipal recycling collection schemes (Kaufman 2004).

Table 31 Linerboard recycling energy literature summary

Linerboard

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
Porter	35.2	21.3	13.9
RTI	28.4	17.2	11.2
EcoInvent	31.8	17.0	14.8
Average			**13.3**

Like other forms of paper manufacturing, secondary newsprint's primary advantage over newsprint manufactured from virgin resources is that it avoids the necessity of having to harvest and process trees to make pulp. Some of the benefits of recycled production of newsprint are offset by the chemical- and energy- intensive deinking step (EDF 1995). Deinking is not necessary when producing linerboard, accounting largely for its greater energy savings over newsprint production.

Table 32 Newsprint recycling energy literature summary

Newsprint

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
Porter	32.5	27.0	5.5
RTI	44.1	24.2	19.9
Average			**12.7**

High-grade office paper, like other grades of paper examined in this study, offers significant energy savings when producing from recovered fibers. As Hershkowitz points out, there are also non-energy related savings to be realized when using recycled fiber as a feedstock – virgin production produces significantly more wastewater per ton produced, and is significantly more chemical-intensive and therefore environmentally-burdensome (Hershkowitz 1997). The average energy savings per ton recycled production is 21.8 GJ.

Table 33 High grade office paper recycling energy literature summary

High Grade Office

Source:	Virgin Energy (GJ/ton)	Recycled Energy (GJ/ton)	Energy Savings (GJ/ton)
Porter	65.0	37.1	27.9
EDF	42.2	22.9	19.3
Hershkowitz	n/a	n/a	22.0
RTI	39.1	21.2	17.9
Average			**21.8**

Combustion with Energy Recovery (WTE) and Landfilling with Methane Recovery (LFGE)

Net energy values for materials combusted in WTE facilities are also necessary for RCE calculations. Net energy/heating values for materials referenced in this study come from the Columbia University report on municipal solid waste management in New York City, "Life After Fresh Kills." Landfilling net energy values are adapted from a paper on landfill gas generation in the US (Themelis and Ulloa 2007). All net energy calculations account for conversion efficiencies from thermal energy to electricity production.

Table 34 WTE & Landfilling Energy Values

Material	WTE Net Energy (GJ/ton)	LF Net Energy (GJ/ton)
Paper	5.22	1.37
Plastics	10.87	
Organics	1.8	1.9

The Resource Conservation Efficiency Metric

With the reasonableness of a CED-based metric as a proxy for larger life cycle studies of waste management systems established, and with material-specific energy savings established, it is now necessary to detail the development and structure of the one we have developed – the Resource Conservation Efficiency (RCE). RCE can be used to quickly assess options for waste treatment of different materials; to compare these options against one another; and to assess and compare the overall waste treatment systems of different municipalities or regions. The equation for RCE is shown below:

$$RCE_i = \left[\frac{(E_{LF} * f_{LF,i}) + (E_{WTE} * f_{WTE,i}) + (E_{REC} * f_{REC,i})}{\max_i\{E_i\}}\right]$$ **Equation 1**

where RCE_i = material-specific RCE

i	=	index for materials
f_{MAT}	=	fraction of material recovered
E_{LF}	=	energy recovered from LF gas
E_{WTE}	=	energy recovered from WTE
E_{REC}	=	energy saved by recycling ($E_{VP} – E_{RP}$)
(E_{VP}	=	energy used for virgin production)
(E_{RP}	=	energy used for recycled production)
$\max_i\{E_i\}$	=	energy saved from best practice

The $\max_i\{E_i\}$ term can be explained as follows. Different materials have different recycling infrastructures associated with them, i.e. not all materials are recyclable. Any material that is recyclable and for which the energy saved from recycling is greater than the energy realized from combustion with energy recovery (WTE), or landfilling with methane recovery (LFGE) will have an $\max_i\{E_i\}$ equal to E_{REC}. For any non-recyclable material – or for those materials that have E_{WTE} or E_{LF} values higher than their E_{REC} value, $\max_i\{E_i\}$ is equal to whichever of these values is highest.

For example, let us assume that the energy saved from recycling one ton of newspaper instead of disposing of that ton and producing a new one from virgin resources is 20 GJ/ton; that the energy gained from combusting one ton of newspaper is 5 GJ/ton; and the energy recovered from landfilling and collecting/combusting the gas for one ton of newspaper is 1.3 GJ/ton. In this case, $E_{MAX,sav}$ is clearly 20 GJ, or the equivalent of E_{REC}. Had either E_{WTE} or E_{LF} been higher than 20 GJ, that value would have been chosen for $E_{MAX,sav}$. The theoretical upper limit for RCE then becomes 100 percent recovery of the paper. Calculating the energy recovered from WTE and landfilling shows that the lower limit is a 100 percent landfilling scenario. Results from different hypothetical scenarios of paper waste management are shown in Figure 45.

In Figure 45, four scenarios are presented, each representing the hypothetical management of 100 tons of wastepaper. The first is maximum recovery, i.e. 100 percent recycling of paper. The second assumes 50 percent recovery and a 25/25 split of the remaining paper between combustion and landfilling. The third scenario is for 100 percent combustion with energy recovery and the fourth 100 percent landfilling of paper.

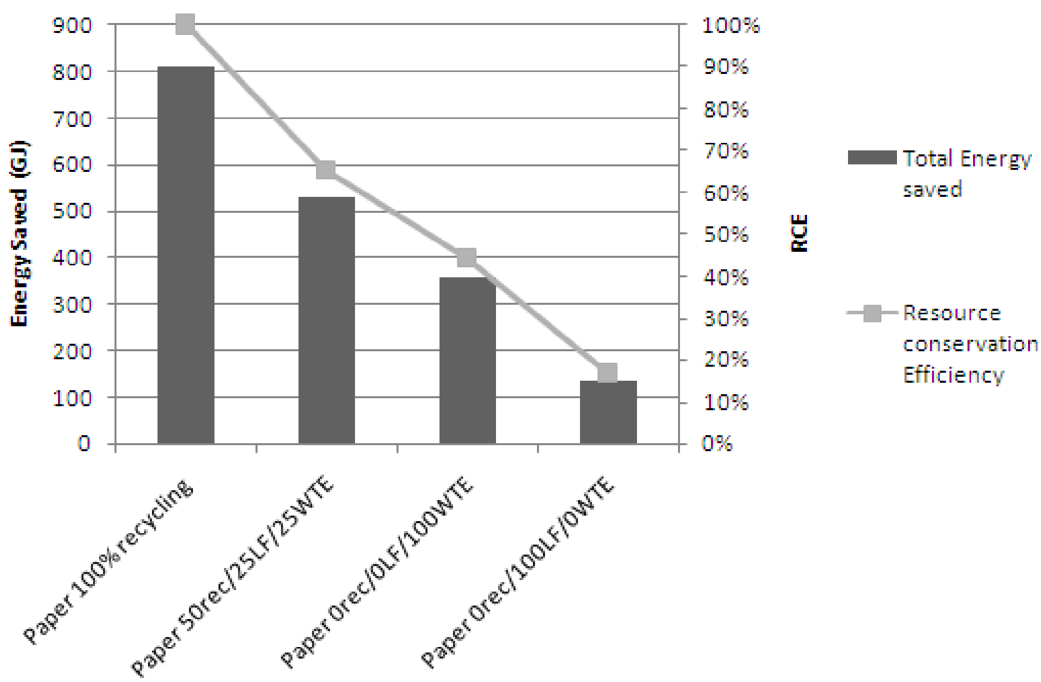


Figure 45 RCE Calculations Paper (Hypothetical)

Because it is a CED-based metric – and as CED has been shown to be a useful indicator of overall lifecycle performance metrics for waste management systems – it is reasonable to expect that RCE to correlate with chosen life cycle impact assessment indicators. Figure 46 is a plot showing the correlation between aluminum RCE and

EcoIndicator99 scores for the three modeled scenarios – the maximum possible RCE scenario (i.e. 100% recycling), and for the Honolulu and San Francisco scenarios (actual management percentages). The graph clearly shows that with the higher the RCE score, the lower the EcoIndicator99 (i.e. environmental impact) score. This is true for all examined materials.

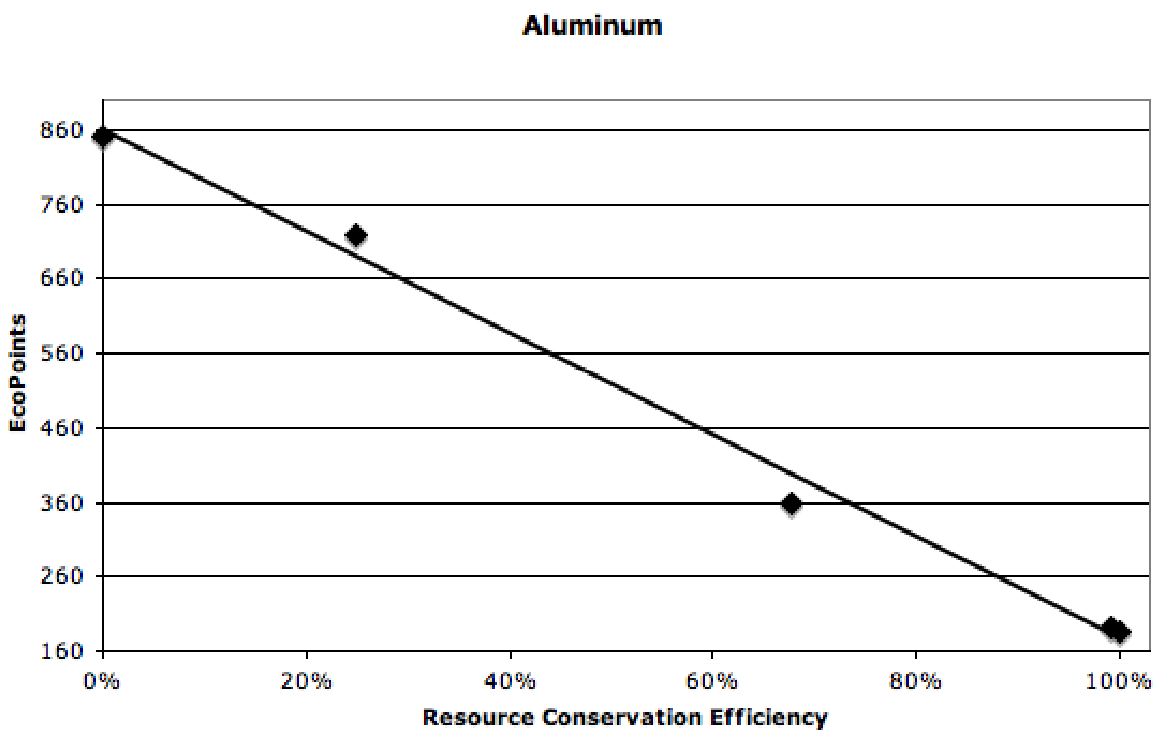


Figure 46 RCE vs. EcoIndicator99 Scores for Aluminum

With the applicability of RCE as an indicator on the materials-level scale, it is now necessary to establish an equation for the use of the metric to model the efficiency of the waste management systems of entire cities or regions.

$$RCE_{TOTAL} = \frac{\sum_{i=1}^{p}\sum_{j=1}^{m} E_{ij} f_{ij}}{\sum_{i=1}^{p} \max_{j}\{E_{ij}\}} \qquad \textbf{Equation 2}$$

where RCE_{TOTAL} = RCE for city/region
i = index for materials
j = index for treatments

The "materials" are the individual products going to waste treatment – newspaper, glass, PET plastic, etc. The "treatments" are the waste treatment options being used, i.e. landfilling, WTE, and recycling.

Modeling the Cities – Honolulu and San Francisco

LCI for Waste (EcoInvent) – Systems boundaries, offsets, etc.

With the concept of RCE established, it is now possible to model the waste management practices of the two cities – San Francisco and Honolulu.

Waste data acquisition

The San Francisco Department of the Environment, in a waste characterization report published in 2006 (2006) , delineated the materials considered recyclable under their citywide program. This materials list has been adapted for entry in SimaPro and the included materials are listed in Table 35. These will be the "standardized" materials used to evaluate our test cities. This standardized list agrees well with those materials chosen for other LCA studies of MSW management (Finnveden, Johansson et al. 2000).

Table 35 Adapted Materials List Used in RCE Analysis

Paper	**Metals**
Newspaper	Aluminum cans
Plain OCC/Kraft	Aluminum foil/containers
High Grade Paper	Other aluminum
Mixed Low Grade Paper	**Other nonferrous**
Plastic	Tin/steel cans
PET Bottles	**Food**
HDPE Natural Bottles	Other (yard waste)
HDPE Colored Bottles	Organics
Other Plastic containers	
Glass	
Glass Beverage Bottles	
Container Glass	

When cities do report recycling tonnages, the materials actually collected for recycling are not often broken down by commodity. In the cases of the two cities examined in this case study – Honolulu and San Francisco – Honolulu does a more thorough job of breaking down collected recyclables by category. Of the total tons of MGPP collected in 2005, 59% were various forms of paper, 23% were metals (split evenly between aluminum and steel), 15% comprised glass, and 3% were plastics (2007). The diverted plastics are assumed to be split evenly between PET, HDPE, PVC, and LDPE as per EPA study on MSW in the United States (EPA 2007). The 60:40 split between Paper and Metal/Glass/Plastic is typical across the country, at locations where all materials are collected (Berenyi 2002; Northup 2006).

After raw tonnage data were collected for each city they were standardized to allow for straight comparisons between the cities. This sometimes involved assumptions such as lumping “other plastic bottles” as reported in San Francisco’s waste characterization into “HDPE bottles” in the SimaPro models.

RCE Calculations

As mentioned above, energy values were obtained from several sources, including the Franklin US 98 and EcoInvent databases in SimaPro. Energy values for combustion products were applied (Themelis 2001). Additionally, we include in the analysis only those materials that are listed as "recyclable" under San Francisco's system, and/or those materials that are combustible or biodegradable. "Composite/other glass," for example – which are not recyclable under Honolulu or San Francisco's current schemes – can not yield energy savings under any current waste management system and are therefore not considered in the RCE calculations. Calculations are made as follows: First, it is necessary to determine the value of $E_{MAX,sav}$. This value can be chosen by referencing the RCE Guide Table shown in Table 36. This table is a sample set of reference data from the literature.

Table 36 RCE Reference table

Material	Rec energy (GJ/ton)	WTE energy (GJ/ton)	LF energy (GJ/ton)	$E_{MAX,sav}$ Process
Newspaper	12.7	5.45	1.37	Recycling
Kraft paper	13.3	5.45	1.37	Recycling
Waxed OCC		5.45	1.37	WTE
High Grade Paper	21.8	5.45	1.37	Recycling
Mixed low grade paper	13.3	5.45	1.37	Recycling
Polycoated paper		5.45	1.37	WTE
Soiled paper	2	1.8	1.37	Recycling
Other paper		5.22	1.37	WTE
Glass	4			Recycling
PET	33.1	8.85		Recycling
HDPE	38.7	16.38		Recycling
PP	34.2	12.64		Recycling
Aluminum	188.1	170.1		Recycling
Steel	16.4	14.76		Recycling
Yard waste	2	1.8	1.37	Recycling
Food waste	2	1.8	1.37	Recycling

There are several assumptions made in determining the values shown in the RCE Reference table. First, any materials without values in the “recycling energy” column are explicitly *not recyclable* under the rules of the San Francisco Fantastic Three program. As this is one of the more inclusive curbside recycling programs in the nation, these rules are assumed to apply to all curbside programs. Where this is not the case, values can be adjusted accordingly.

The values for “kraft paper” and “mixed low grade paper” are equal because mixed low-grade paper is assumed to be used as feedstock for recycled linerboard production. The energy savings from recycling are therefore equal to GJ saved from producing a ton of recycled linerboard instead of virgin linerboard.

The recycled energy value for “soiled paper” (known as “compostable/soiled paper” in San Francisco’s characterization study) is chosen by assuming that this material is composted in a windrow based system. The energy savings from composting – the same values assigned to “yard waste” and “food waste” – are determined by offsetting the production of the equivalent amount of chemical nitrogen, phosphorous, and potassium that would be found in one ton’s worth of composted organic waste (EcoInvent 2004).

Finally, the WTE energy values for “aluminum” and “steel” are both 10 percent lower than the recycling energy values for the same materials. H-Power, the WTE facility operating in Honolulu, claims to recover “virtually all” of the recyclable ferrous and non-ferrous metals entering the facility (2007). We assumed 10 percent losses with the rest of the incoming metals waste being diverted to recycling. These values can easily be adjusted for other cities and facilities where metals are not recovered from WTE processes.

Honolulu and San Francisco RCE Results

With the energy factors established, it is now possible to calculate RCE values for each material and for the overall performance of the two cities. Waste data was acquired as per the "Waste data acquisition" section above. Results for Honolulu San Francisco are shown in Table 37.

Table 37 Honolulu & San Francisco RCE Tabular Results

	Honolulu				San Francisco		
Material	**Recycle Rate (%)**	**WTE Rate (%)**	**Landfill Rate (%)**	**RCE (%)**	**Recycle Rate (%)**	**Landfill Rate (%)**	**RCE (%)**
Newspaper	29.9	69.3	0.9	55.8	62.5	37.5	66.6
Kraft paper	50.6	46.5	2.9	69.9	80.9	19.1	82.9
Waxed OCC	0.0	0.0	0.0	0.0	0.0	100.0	25.1
High Grade Paper	24.4	75.1	0.5	43.2	66.7	33.3	68.8
Low Grade Paper	8.9	87.5	3.6	45.1	6.2	93.8	15.9
Polycoated paper	0.0	0.0	0.0	0.0	0.0	100.0	25.1
Soiled paper	0.0	98.8	1.2	99.6	0.0	100.0	68.5
Other paper	0.0	90.8	9.2	2.3	0.0	100.0	25.1
Total paper	**22.0**	**75.8**	**2.1**	**58.0**	**49.5**	**50.5**	**63.3**
PET	20.1	76.1	3.8	40.4	34.9	65.1	34.9
HDPE	12.7	83.2	4.1	47.9	61.7	38.3	61.7
PP	11.6	87.1	1.3	43.8	44.6	55.4	44.6
Other plastics	0.0	91.5	8.5	91.5	0.0	100.0	0.0
Total plastics	**3.4**	**89.5**	**7.2**	**67.5**	**10.5**	**89.5**	**23.2**
Recyclable glass	85.0	13.5	1.5	85.5	45.7	54.3	45.7
Other glass	0.0	95.5	4.5	0.0	0.0	100.0	0.0
Total glass	**59.4**	**38.2**	**2.4**	**85.0**	**42.6**	**57.4**	**45.7**
Aluminum	70.9	28.6	0.5	99.5	74.1	25.9	74.1
Steel	39.2	34.6	26.2	73.8	39.7	60.3	39.7
Other metals	0.0	50.1	49.9	0.0	0.0	100.0	0.0
Total metals	**36.2**	**37.5**	**26.4**	**96.0**	**40.5**	**59.5**	**67.3**
Yard waste	48.5	47.6	3.9	51.2	80.5	19.5	93.9
Food waste	23.4	75.3	1.3	24.3	5.7	94.3	70.3
Total organics	**36.0**	**61.3**	**2.6**	**37.8**	**25.1**	**74.9**	**76.4**
Grand Total	**27.2**	**67.9**	**4.9**	**73.1**	**34.4**	**65.6**	**60.4**

As the table shows, the RCE method shows that recycling percentages, by themselves, are not adequate means for determining the effectiveness of cities' waste management systems. While San Francisco's recycling rate is significantly higher than Honolulu's (34.4 percent to Honolulu's 27.2 percent), Honolulu's resource conservation efficiency is higher than San Francisco's by quite a wide margin (73.1 percent to San Francisco's 60.4 percent). Figure 48 compares the results of the two cities.

Discussion

As Figure 48 demonstrates, the environmental performance of a city's waste management system depends on more than just the traditional recycling rate. While San Francisco enjoys a very high recycling rate (34.4%, which places it above the US national recycling rate according to *BioCycle* (Simmons, Goldstein et al. 2006) as well as among the top five major cities in the US (Anonymous 2006)), its RCE score is significantly lower than that of Honolulu's.

While RCE is particularly useful as a descriptive metric, it is also attractive to apply it as a *prescriptive* tool. Since these two cities have highly contrasting approaches to waste management, it is interesting to see what they might be able to do from an RCE perspective to improve their scores.

If San Francisco were to try to improve its RCE scores, there would be two obvious possible solutions. The first would be boost the recycling rate of recyclable materials with low capture rates. The second would be to combust non-recycled MSW (or a fraction of it) instead of landfilling it. The most attractive materials to target for increased recycling are mixed low-grade paper (currently recycled at a rate of 6.2%) and food waste (currently recycled at a rate of 5.7%). The alternative would be to build a WTE facility and combust at least a portion of the remaining waste after collection for recycling. The average capacity of a US WTE facility is 1300 tons per day, so we assume that San Francisco would build a facility at this capacity, resulting in approximately 500,000 tons of MSW.

RCE works very well for the "clean" stream of MSW materials. There are, however, more environmentally burdensome materials in most MSW streams (such as electronic waste and some types of construction & demolition waste) that do not have such a tight correlation between CED and environmental indicators. These materials would therefore have to be excluded from RCE studies where environmental screening analysis is an intended outcome. It would still be valid, however, from an energy savings perspective. An extreme scenario (Figure 47) is presented where the plotted landfilled or incinerated waste materials are highly toxic (different industrial sludges, drilling wastes, etc.). These materials are banned from MSW facilities, but it does show that there are exceptions to the CED vs. environmental impact rule that have to be examined further.

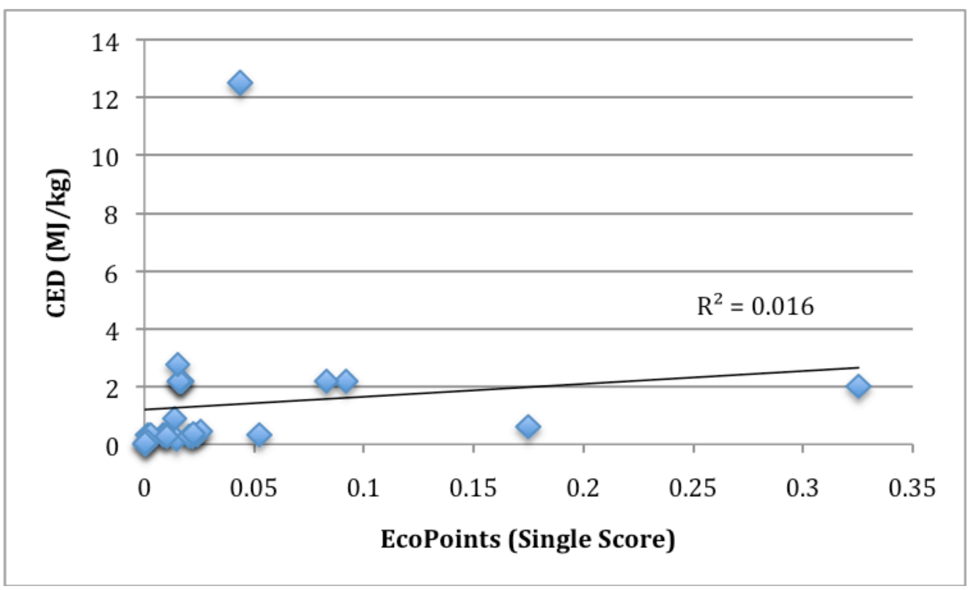
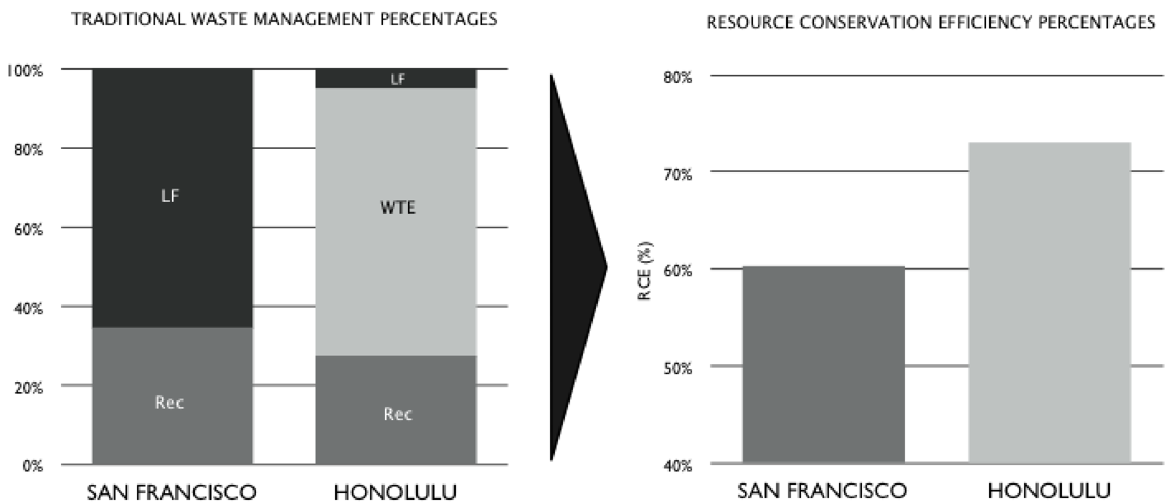


Figure 47 "Dirty Materials" CED vs. EcoIndicator 99

Figure 48 San Francisco vs. Honolulu Waste Management & RCE Percentages

Figure 45a

Figure 45b



In Figure 45a, waste treatment percentages for San Francisco and Honolulu are presented while in Figure 45b, the corresponding RCE scores are shown. As the graphs clearly display, Honolulu's RCE is higher than San Francisco's, despite the fact that San Francisco recycles more. This difference is primarily due to the fact that Honolulu sends the vast majority of its combustible non-recyclable waste to a WTE facility, while San Francisco landfills all of this material. Figure 49 shows how alternative waste treatment scenarios would affect each city's RCE.

Thus, for San Francisco, two alternative scenarios are presented. (Results for all alternative scenarios are depicted in Figure 49.) The first captures mixed low-grade paper at a rate of 65 percent, similar to the recycling rates of newspaper and high-grade paper in San Francisco, while additionally increasing the recycling of food waste from composting (from 5.7 percent in the status quo scenario to 30 percent in the first alternative scenario). This raises San Francisco's recycling rate from 34.4 percent in the status quo scenario to 44.4 percent, but raises the RCE from 61% in the status quo scenario to 64.2 percent – a modest increase.

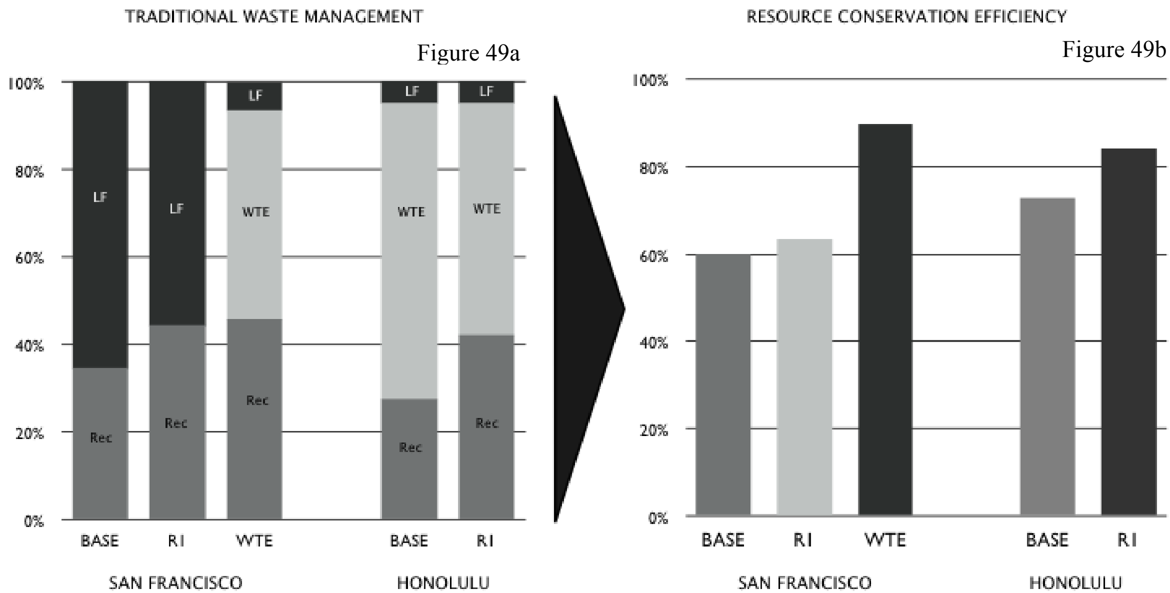
If a WTE plant were added to treat non-recycled San Francisco waste and recycling is increased as in SF R1, the RCE climbs to over 90 percent.

In the case of Honolulu, there is already quite a small percentage of waste sent to landfills, and the RCE value is already relatively high. It would seem that the best course of action to increase the value even more would be to recover more materials than in the status quo scenario. There are several materials with recycling rates that could be improved upon. Paper recycling is low relative to San Francisco – it was therefore assumed that recovery rates for all recyclable paper in Honolulu could be improved to 65 percent (similar to San Francisco's recovery rates). Similarly, San Francisco captures recyclable plastics at a 35 percent rate, so Honolulu's alternative scenario is adjusted to these rates. Finally, 85 percent of San Francisco's yard waste is captured, so Honolulu's capture rate of this material is also adjusted upwards. With these adjustments in place, Honolulu's recycling rate goes from 27.2 percent in the status quo scenario to 42.0

percent in the adjusted scenario. The RCE jumps from 74.7 percent in the status quo scenario to 84.2 percent in the alternative-recycling scenario.

It is clear from this discussion that different environments require different waste management solutions. RCE can be used as a tool to help understand what is happening in relation to other cities, and can help pinpoint the best ways (from a lifecycle perspective) to improve on the status quo.

Figure 49 SF & Honolulu alternative Waste Management & RCE Percentages



In Figure 49a, SF R1 is the first alternative recycling scenario for San Francisco, with mixed paper recycling increased from 6.2% to 65% and food waste recycling increased from 5.7% to 30%. SF WTE is the second San Francisco alternative scenario, where the recycling is increased as in SF R1, landfilling rates remain unchanged, and the WTE rate goes from zero to 56.4%. HL R1 is the alternative recycling scenario for Honolulu, where the recycling rates for several recyclables are increased, and Honolulu's overall recycling rate jumps from 27.2% to 42.0%. Figure 49b shows the commensurate RCE percentages. San Francisco's alternative recycling scenarios lead to modest RCE increases, while the addition of a WTE facility improves the RCE significantly. Honolulu's RCE is improved with the increased recycling rate.

6. Conclusions

This study has shown that municipal solid waste (MSW), when viewed objectively and quantitatively – and when employing the right analytical tools – is a relatively complex and dynamic problem with many possible solutions. The problem of collecting waste data can be addressed by using targeted characterization studies and by organizing, filtering, and interpreting the available information.

The development and application of the waste data algorithm in Chapter 3 showed that the EPA vastly underestimates the amounts of waste generated and disposed in the US. The work also showed that the BioCycle/EEC method does a much better job of accounting for these generation and disposal tonnages. The EPA, however – due primarily to their relationship with commodity organizations – accurately accounts for national recycling rates. It is hoped that the future will bring a greater convergence of these methodologies utilizing the strengths of each approach.

Improved access to waste data – in addition to aiding in the crafting of appropriate MSW management strategies – will help the overall movement towards “sustainable waste management” as well. As countries continue to adopt caps on carbon emissions, for example, there will be a greater need for improved waste data to adequately account for the waste industry’s effect on global warming. Additionally, companies undertaking carbon reduction projects will need to know the impacts of recycling, combustion, and disposal of their products. This kind of information can be obtained using the methods introduced in Chapter 3.

The entropy model described in Chapter 4 allows analysts to track the flows of various substances through waste processing systems, and to determine whether they are concentrated or diluted by the system. The SE method had previously been developed for metals, and was extended in this study to account for carbon. The analysis showed that WTE plants perform better than landfills from a global warming perspective.

While the statistical entropy method is useful in terms of tracking substance flows, the new metric introduced in Chapter 5 – Resource Conservation Efficiency (RCE) – allows users to quickly evaluate the environmental performance of alternative MSW management systems. The results are intuitive and accessible to the general population, in addition to being quantitatively rigorous.

To demonstrate the usefulness of the RCE method, a case study was performed comparing two major US cities with similar populations – Honolulu and San Francisco – but with different approaches to waste management. San Francisco relies upon landfilling for nearly all of its non-recycled waste, while Honolulu relies upon WTE. The results of the development of RCE and application to the waste management systems of these two cities show that a combination of recycling and WTE is the best way to conserve resources and optimize the environmental performance of waste systems. More specifically, San Francisco would be much better served by adding a WTE facility to their waste management mix before attempting to expand recycling. Honolulu, on the other hand, should focus on increasing recycling.

Future work

While all of these are useful tools and, we believe, valuable contributions to the fields of waste management and life cycle assessment, there is some additional work that would be attractive to undertake. First of all, it would be useful to add an economic analysis to the RCE metric that allows planners to determine, for example, the most cost-effective way to increase their environmental efficiency. While it is extremely helpful to know, for instance, that adding WTE to the San Francisco waste management mix will greatly increase the overall environmental performance of this city, it would also be beneficial to know the comparative incremental economic costs of such a change. Unfortunately, in contrast to waste-to-energy and landfilling, the economics of recycling of various materials are still rather difficult to determine and such work was therefore outside the scope of this study [1].

Waste management systems depend greatly on the feedstock and the means used for sorting wastes at the source. As collection and design for environment (DFE) practices grow, perhaps we will one day get to the point where most consumer goods at their end of their life are readily recyclable and we will no longer need waste-to-energy facilities or landfills to deal with MSW. But we are a long way from that goal. Until we get there, we need a way to structure our discussions so that we do the most beneficial thing from a resource and health perspective. It is hoped that RCE will help facilitate that kind of dialogue by creating a common language and framework that all stakeholders in the world of waste management – from environmentalists to politicians all the way to the corporate executives of management companies – can work from.

Bibliography

(1999). Agency for Toxic Substances and Disease Registry (ATSDR). U. D. o. H. a. H. Services.

(1999). Biosolids Generation, Use, and Disposal in the United States. U. S. E. P. Agency.

(1999). Streamlined Life-Cycle Assessment: A Final Report from the SETAC North America Streamlined LCA Workgroup. J. A. Todd and M. A. Curran. Pensacola, Fl, SETAC**:** 31.

(2003). 2003 Recycling Status and Market Development Report. B. o. W. M. Division of Environmental Protection, Nevada Department of Conservation and Natural Resources.

(2006). "2005 Landfill Summary Tonnage Report." Landfill Tonnage Reports, 2006, from http://www.ciwmb.ca.gov/landfill.

(2006). Waste Characterization Study Final Report. San Francisco, Department of the Environment**:** 95.

(2007). California Quickfacts from the US Census Bureau, US Census Bureau.

(2007). "Climate Change - Science." 2007, from http://epa.gov/climatechange/science/recentac.html.

(2007). Hawaii Quickfacts from the US Census Bureau, US Census Bureau.

(2007). "Methane: Science." 2007, from http://www.epa.gov/methane/scientific.html.

(2007). "Opala Solid Waste." 2007, from www.opala.org.

(2007). TOXNET Toxicology Data Network. U. S. L. o. Medicine, National Institutes of Health.

ACF. (2008). "Eco-Calculator." 2008, from http://www.acfonline.org.au/custom_greenhome/calculator.asp?section_id=86.

Anonymous (2006). "Municipal Recycling Survey." Waste News.

Arena, U., M. L. Mastellone, et al. (2003). "Life Cycle Assessment of a Plastic Packaging Recycling System." International Journal of Life Cycle Assessment **8**(2): 92-98.

Ayres, R. U. (1999). Accounting for Resources, 2: The Life Cycle of Materials. Northampton, MA, Edward Elgar.

Baayen, H. (2000). Eco-indicator 99 Manual for Designers: A damage oriented method for Life Cycle Impact Assessment. P. Consultants. Amserfoort, The Netherlands, Pre Consultants**:** 49.

Baccini, P. and P. H. Brunner (1991). Metabolism of the Anthroposphere. New York, Springer-Verlag.

Baccini, P., G. Henseler, et al. (1987). "Water and element balances of municipal solid waste landfills." Waste Management & Research **5**(4): 483-499.

Beck, R. W. I. (2007). 2006 Waste Characterization Study: City and County of Honolulu. Honolulu, HI.

Belevi, H. and P. Baccini (1989). "Long-term behavior of municipal solid waste landfills." Waste Management & Research **7**(1): 43-56.

Berenyi, E. (2002). Materials Recycling and Processing in the United States. Westport, CT, Governmental Advisory Associates.

Berenyi, E. (2007). US MRF Tons 2005. New York.

Blok, K. (2006). Introduction to Energy Analysis. Amsterdam, Techne Press.

Bruner, P. H., G. Doberl, et al. (2001). Bewertung abfallwirtschaftlicher Maßnahmen mit dem Ziel der nachsorgefreien Deponie (Projekt BEWEND). Monographien des UBA Band 149. Wien, Umweltbundesamt GmbH.

Brunner, P. H. and J. Fellner (2007). "Setting priorities for waste management strategies in developing countries." Waste Management & Research **25**: 234-240.

Brunner, P. H. and H. Rechberger (2004). Practical Handbook of Material Flow Analysis. New York, Lewis.

Camobreco, V., R. Ham, et al. (1999). "Life-cycle inventory of a modern municipal solid waste landfill." Waste Management Research **17**(6): 394-408.

Castaldi, M. J. and E. Kwon (2007). "An Investigation into the Mechanisms for Styrene-Butadiene Copolymer (SBR) Conversion in Combustion and Gasification Environments." International Journal of Green Energy **4**(1): 45 - 63.

Castaldi, M. J., E. Kwon, et al. (2007). "Beneficial Use of Waste Tires: An Integrated Gasification and Combustion Process Design via Thermo-Gravimetric Analysis (TGA) of Styrene-Butadiene Rubber (SBR) and Poly-Isoprene (IR)." Environmental Engineering Science **24**(8): 1160-1178.

Chin-Chance, C. (2007). Hawai County Landfill and Tonnage Report. R. S. Dept. of Environmental Management.

Choate, W. T. and J. A. S. Green (2003). US Energy Requirements for Aluminum Production: Historical Perspective, Theoretical Limits and New Opportunities. E. E. a. R. Energy. Columbia, MD, US Department of Energy**:** 125.

CIWMB. (2007). "California's 2005 Statewide Diversion Rate Estimate." Retrieved December 10, 2007, from http://www.ciwmb.ca.gov/LGCentral/Rates/Diversion/2005/Default.htm.

CIWMB, R. W. I. Beck, et al. (2006). Characterization and Quantification of Residuals from Materials Recovery Facilities, California Integrated Waste Management Board.

CIWMB and I. W. M. Consulting (2004). Second Assessment of California's Compost- and Mulch-Producing Infrastructure. Sacramento, CA, California Integrated Waste Management Board.

CIWMB and C. C. Group (2004). Statewide Waste Characterization Study. Sacramento, CA, California Integrated Waste Management Board.

CML (2000). CML 2 baselinee 2000. Leiden, Centre for Environmental Studies.

Craighill, A. L. and J. C. Powell (1996). "Lifecycle assessment and economic evaluation of recycling: A case study." Resources, Conservation and Recycling **17**(2): 75-96.

Curran, M. A. (2006). Life Cycle Assessment: Principles and Practice. EPA. Reston, VA, EPA**:** 88.

Daniels, P. L. (2002). "Approaches for Quantifying the Metabolism of Physical Economies: A Comparative Survey: Part II: Review of Individual Approaches." Journal of Industrial Ecology **6**(1): 65-88.

Daniels, P. L. and S. Moore (2001). "Approaches for Quantifying the Metabolism of Physical Economies: Part I: Methodological Overview." Journal of Industrial Ecology **5**(4): 69-93.

Denison, R. A. (1996). "Environmental Life-Cycle Comparisons of Recycling, Landfilling, and Incineration: A Review of Recent Studies." Annual Review of Energy and the Environment **21**: 191-238.

Doberl, G., R. Huber, et al. (2002). "Long-term assessment of waste management options a new, integrated and goal-oriented approach." Waste Management Research **20**(4): 311-327.

Doka, G. and R. Hischier (2005). "Waste Treatment and Assessment of Long Term Emissions." International Journal of Life Cycle Assessment **10**(1): 77-84.

EcoInvent (2004). EcoInvent data v1.1 Final reports EcoInvent 2000 No. 1-15. Dubendorf, Switzerland, Swiss Center for Life Cycle Inventories.

EDF (1995). Lifecycle Environmental Comparison: Virgin Paper and Recycled Paper-Based Systems. P. T. Force. New York, NY, Environmental Defense Fund**:** 165.

EIOLCA. (2008). "Economic Input-Output Life Cycle Assessment." 2008, from http://www.eiolca.net.

EPA (1997). Measuring Recycling: A Guide for State and Local Governments. S. W. a. E. Response, United States Environmental Protection Agency**:** 164.

EPA. (2005). "Resource Conservation Challenge." 2007, from http://www.epa.gov/epaoswer/osw/conserve/action-plan/act-p1.htm.

EPA (2006). Municipal Solid Waste in the United States: 2005 Facts & Figures. O. o. S. Waste, United States Environmental Protection Agency.

EPA (2007). Municipal Solid Waste in the United States: 2006 Facts & Figures Data Tables. O. o. S. Waste, United States Environmental Protection Agency**:** 56.

EPA. (2008). "Life Cycle Assessment Research." 2008, from http://www.epa.gov/nrmrl/lcaccess/.

Fergus, K. (2006). Nevada 2005 Waste Data Compilation, Nevada Division of Environmental Protection - Bureau of Waste Management & Recycling.

Finance, C. D. o. (2007). "Governer's Budget 2007-08." 2007, from http://govbud.dof.ca.gov/StateAgencyBudgets/3890/3910/department.html.

Finnveden, G., J. Johansson, et al. (2000). Life Cycle Assessments of Energy from Solid Waste. Stockholm, Sweden, Stockholms Universitet.

Finnveden, G., J. Johansson, et al. (2000). Life Cycle Assessments of Energy from Solid Waste - Appendix 5, Process Data Records in Sima Pro. Stockholm, Stockholm University**:** 590.

Franklin Associates, L. (1995). Methodology for Characterization of Municipal Solid Waste in the United States 1994 Update. U. E. P. Agency, Franklin Associates, Ltd.**:** 136.

Gaddy, A. (2007). Landfill Questions. S. M. Kaufman. Las Vegas, NV.

Georgeson, R. (2006). Environmental Benefits of Recycling - An International Review of Life Cycle Comparisons for Key Materials in the UK Recycling Sector. Banbury, UK, Waste & Resources Action Programme (WRAP).

Goedkoop, M. (2007). SimaPro. Amserfoort, Pre Consultants.

Goedkoop, M. and R. Spriensma (2001). The Eco-indicator 99 A damage oriented method for Life Cycle Impact Assessment, Methodology Report. P. Consultants. Amserfoort, The Netherlands, Pre Consultants**:** 144.

Graedel, T. E. (2008). "Center for Industrial Ecology." 2008, from http://www.yale.edu/cie/research.html.

Harrison, K. W., R. D. Dumas, et al. (2000). "A Life-Cycle Inventory Model of Municipal Solid Waste Combustion." Journal of the Air & Waste Management Association (1995) **50**(6): 993-1003.

Harvey, D., J. Gregorgy, et al. (1997). An Introduction to Simple Climate Models Used in the IPCC Second Assessment Report. J. T. Houghton, L. G. M. Filho, D. J. Griggs and K. Maskell. New York, NY, IPCC.

Hershkowitz, A. (1997). Too Good to Throw Away: Recycling's Proven Record. New York, NY, NRDC**:** 86.

Huijbregts, M. A. J., L. J. A. Rombouts, et al. (2006). "Is Cumulative Fossil Energy Demand a Useful Indicator for the Environmental Performance of Products?" Environ. Sci. Technol. **40**(3): 641-648.

Johnson, J., E. M. Harper, et al. (2007). "Dining at the Periodic Table: Metals Concentrations as They Relate to Recycling." Environ. Sci. Technol. **41**(5): 1759-1765.

Jolliet, O., M. Margni, et al. (2003). "IMPACT 2002+: A New Life Cycle Assessment Methodology." International Journal of Life Cycle Assessment **8**(6): 324-330.

Kaufman, S. M. (2004). An Analysis of the Technology and Infrastructure of the Paper Recycling Industry in New York City Earth and Environmental Engineering. New York, NY, Columbia University. **Master of Science:** 48.

Kaufman, S. M. (In Press). Using Statistical Entropy and Life Cycle Assessment to Evaluate the Effectiveness of Waste Management Systems. Earth and Environmental Engineering. New York, Columbia University. **PhD**.

Kaufman, S. M., K. Millrath, et al. (2004). "State of Garbage in America - Data and Methodology Assessment." BioCycle **45**(4): 22-26.

Kim, S. and B. E. Dale (2004). "Cumulative Energy and GlobalWarming Impact from the Production of Biomass for Biobased Products." Journal of Industrial Ecology **7**(3): 147-162.

Klopffer, W. (1997). "In defense of the cumulative energy demand." International Journal of Life Cycle Assessment **2**(2).

Knapp, K. and T. Jester (2001). "Empirical investigation of the energy payback time for photovoltaic modules." Solar Energy **71**(3): 165-172.

Lipton, E. (2003). City Seeks Ideas as Trash Costs Dwarf Estimate. The New York Times. New York**:** 2.

Margolis, N. and R. Brindle (2000). Energy and Environmental Profile of the US Iron and Steel Industry. I. Energetics. Washington, DC, US Department of Energy**:** 106.

Maystre, L. Y. and F. Viret (1995). "A goal-oriented characterization of urban waste." Waste Management & Research **13**(3): 207-218.

McDougall, F. R. and P. White (2001). Integrated solid waste management : a life cycle inventory. Oxford, UK ; Malden, MA, Blackwell Science.

Melosi, M. V. (2005). Garbage in the Cities: Refuse, Reform, and the Environment. Pittsburgh, University of Pittsburgh Press.

Metz, B., O. Davidson, et al. (2005). IPCC Special Report on Carbon Dioxide Capture and Storage. New York, NY, Cambridge University Press.

Miller, B. (2000). Fat of the Land - Garbage in New York the last Two Hundred Years. New York, Four Walls.

Miller, M., M. Justiniano, et al. (2005). Energy and Environmental Profile of the U.S. Pulp and Paper Industry. U. D. o. E. O. o. E. E. a. R. E. I. T. Program. Columbia, MD, Energetics, Incorporated**:** 99.

Moore, S., B. Kung, et al. (1994). Progress Towards the Establishment of a National Waste Database. Sydney, Australia, University of New South Wales.

Northup, H. (2006). Mix of Materials at MSW materials recovery facilities. S. M. Kaufman. Philadelphia, PA.

Pellegrino, J. L. (2002). Energy and Environmental Profile of the US Glass Industry. U. D. o. E. O. o. I. Technologies. Columbia, MD, Energetics, Incorporated**:** 108.

Phoenix, C. o. (2007). "Phoenix Recycles." 2007, from http://phoenix.gov/GARBAGE/faq0505.html#divrtfaq.

Porter, R. and T. Roberts (2005). Energy Savings by Wastes Recycling. Abingdon, Oxon, England, Taylor & Francis.

Rather, J. (2005). A Long, Long Haul from the Curb. The New York Times. New York**:** 2.

Rechberger, H. (2008). Statistical entropy. S. M. Kaufman. Vienna, Austria, personal communication - email.

Rechberger, H. and P. H. Brunner (2002). "A New, Entropy Based Method To Support Waste and Resource Management Decisions." Environ. Sci. Technol. **36**(4): 809-816.

Rechberger, H. and T. E. Graedel (2002). "The contemporary European copper cycle: statistical entropy analysis." Ecological Economics **42**(1-2): 59-72.

Rohrlich, M., M. Mistry, et al. (2003). "A Method to Calculate the Cumulative Energy Demand (CED) of Lignite Extraction." International Journal of Life Cycle Assessment **5**(6): 369-373.

Rozycki, C., H. Koeser, et al. (2003). "Ecology Profile of the Germany High Speed Rail Passenger Transport System, ICE." International Journal of Life Cycle Assessment **8**(2): 83-91.

Scheuer, C., G. A. Keoleian, et al. (2003). "Life cycle energy and environmental performance of a new university building: modeling challenges and design implications " Energy and Buildings **35**(10): 15.

Sexton, J. (1994). Brooklyn Incinerator Opponents See Victory in High Court Ruling. The New York Times. New York**:** 1.

Shannon, C. E. (1948). "A Mathematical Theory of Communication." Bell System Technical **27**: 379-423.

Simmons, P., N. Goldstein, et al. (2006). "The State of Garbage in America." BioCycle **47**(4): 26-43.

Snilsberg, B., L. Jonasson, et al. (2004). Thermal Treatment Data. AWAST (Aid in the Management and Comparison of MSW Treatment Methods for a Global and Sustainable Approach). J. Villeneuve. Brussels, European Comission**:** 18.

Spatari, S., M. Bertram, et al. (2002). "The contemporary European copper cycle: 1 year stocks and flows." Ecological Economics **42**(1-2): 27-42.

Tchobanoglous, G. and F. Keith (2002). Handbook of Solid Waste Management. New York, McGraw-Hill.

Themelis, N. J. (2001). Life After Fresh Kills: Moving Beyond New York City's Current Waste Management Plan. New York, Columbia University.

Themelis, N. J. and C. Todd (2004). "Recycling in a Megacity." Journal of Air and Waste Management Association **54**(4): 389-395.

Themelis, N. J. and P. A. Ulloa (2007). "Methane Generation in Landfills." Renewable Energy **32**(7): 1243-1257.

Thormark, C. (2002). "A low energy building in a life cycle--its embodied energy, energy need for operation and recycling potential." Building and Environment **37**(4): 429-435.

Tierney, J. (1996). Recycling is Garbage. The New York Times. New York**:** 13.

Truttmann, N. and H. Rechberger (2006). "Contribution to resource conservation by reuse of electrical and electronic household appliances." Resources, Conservation and Recycling **48**(3): 249-262.

UKS. (2008). "Life Cycle Impact Assessment." Science in the Box, 2008, from http://www.scienceinthebox.com/en_UK/sustainability/lcia_en.html.

UNEP. (2008). "Life Cycle Assessment." 2008, from http://www.uneptie.org/pc/pc/tools/lca.htm.

Van Kekerix, L. (2007). Email. S. M. Kaufman. Sacramento, CA.

Wagner, H. J. and E. Pick (2004). "Energy yield ratio and cumulative energy demand for wind energy converters." Energy **29**(12-15): 2289-95.

Weitz, K. (2003). Life Cycle Inventory Data Sets for Material Production of Aluminum, Glass, Paper, Plastic, and Steel In North America. U. E. P. A. O. o. R. a. Development. Research Triangle Park, NC, RTI International**:** 246.