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ORIGINAL PAPER

Aerobic in-vessel composting versus bioreactor landfilling using life cycle inventory models

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Abstract Potential environmental impacts associated with aerobic in-vessel composting and bioreactor landfilling were assessed using life cycle inventory (LCI) tool. LCI models for solid waste management (SWM) were also developed and used to compare environmental burdens of alternative SWM scenarios. Results from the LCI models showed that the estimated energy recovery from bioreactor landfilling was about 9.6 megajoules (MJ) per kilogram (kg) of waste. Air emissions from in-vessel composting contributed to a global warming potential (GWP) of 0.86 kg of CO_2-equivalent per kg of waste, compared to 1.54 kg of CO_2-equivalent from bioreactor landfill. Waterborne emissions contributing to aquatic toxicity is less coming from in-vessel composting than from bioreactor landfilling. However, emissions to air and water that contribute to human toxicity are greater for the composting option than for the landfill option. Full costs for in-vessel composting is about 6 times greater than for the landfilling alternative. Integration of individually collected commingled recyclables, yard wastes, and residual wastes with windrow composting and bioreactor landfilling produces airborne and waterborne emissions with the least environmental effects among the alternatives considered. It also yields greater energy savings due to the conversion of the landfill gas (LFG) to electrical energy than the option that diverts yard waste, food waste and soiled paper for aerobic in-vessel composting. However, this scenario costs 68% more than that where the commingled collection of wastes is integrated with in-vessel composting and conventional landfilling, owing to increased collection costs.

Keywords Bioreactor landfill · Life cycle inventory · In-vessel composting · Municipal solid waste management

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Introduction

Integrated Solid Waste Management (ISWM) is defined as "the selection and application of suitable techniques, technologies, and management programs to achieve specific waste management objectives and goals" (Tchobanoglous et al. 1993). An integrated system includes waste collection and sorting, followed by one or more of the following options: recycling, biological treatment of organic materials, thermal treatment, and landfilling. Composition of solid waste varies among communities or regions; as such, there can be no single acceptable solution to solid waste management (SWM). The alternatives, however, must both be environmentally sustainable and economically viable for all sectors of the community.

The use of the waste management hierarchy which consists of reduce, reuse, recycle, and dispose does not allow for a comparison of various SWM options in terms of cost and overall environmental impact. Environmental life cycle analysis is a tool used to assess the environmental burdens associated with a specific product or service. This is accomplished using: (1) an inventory of relevant inputs and outputs of a system; (2) an evaluation of the potential impacts of those inputs and outputs; and (3) an interpretation of the results in the context of the objectives of the analysis. The life cycle inventory (LCI) tool, which has been used to compare a process with another equivalent process, can provide an "educated guess" as to the potential impacts of various SWM schemes. This methodology, which had been applied to SWM systems (Komilis and Ham 2004; Barlaz et al. 2003; Wilson 2002; Camobreco et al. 1999; Denison 1996), allows SWM planners to model diverse scenarios for handling solid waste and calculate the environmental burdens of each. However, because site-specific information for every aspect of the municipal solid waste (MSW) system is time-consuming and financially not feasible, generic data is commonly used for analysis and making decisions.

This study is only concerned with the development of LCI models to describe, quantify, and compare estimates of the environmental performance of two SWM options: (1) aerobic composting of MSW; and (2) bioreactor landfills. Three SWM scenarios that feature these options into an integrated system that include waste collection, and other options such as windrow composting of yard waste and conventional landfilling are presented. An inventory of energy requirements and selected environmental emissions is performed, and cost comparisons are presented. Primarily, the goal is to analyze the effects of different SWM choices. Although the focus is on energy use and climate change, other impact categories such as acidification, eutrophication, photo-oxidant formation and human and ecotoxicological impacts are also considered. The results can be used by decision-makers as one basis for judgments on strategies and policies for SWM and investments for new solid waste treatment facilities.

The life cycle analysis is conducted strictly for comparative reasons. Site-specific analyses are needed to extrapolate the conclusions of this study to a specific project. Nevertheless, the life cycle analysis presented herein provides an analytical framework for comparing environmental burdens and costs associated with various waste management strategies and to come up with the most appropriate combination for a community.

Materials and methods

In this paper, LCI models were developed using spreadsheet program and used to compare environmental burdens and costs of aerobic in-vessel composting and bioreactor landfill. For purposes of comparison, the amount of organic materials allowed into both treatments was maintained the same (around 28,000 metric tons); non-organic materials were, however, included in the bioreactor landfill and none in aerobic in-vessel composting. Similar models for waste collection, windrow composting of yard waste, and conventional landfill were developed and used to compare three SWM strategies. Comparison among these strategies using the LCI models was conducted for roughly 91,000 metric tons of collected waste.

Boundaries of the systems considered

Figures 1 and 2, respectively, show the LCI system boundaries for the aerobic in-vessel composting and bioreactor landfill considered in the study. As shown in these figures, the LCI boundaries include all aspects of operation once the wastes are delivered to the facility. Factors contributing to the life-cycle emissions are (1) fuel used for waste placement, preparation, and other equipment use (e.g., mechanical rotation of digester, screening), and (2) decomposition of wastes. Transport of produced compost to its end use and compost land application were not included in the LCI.

Aerobic in-vessel composting process model

The Aerobic In-Vessel Composting Process Model uses an in-vessel system that features the enclosed composting

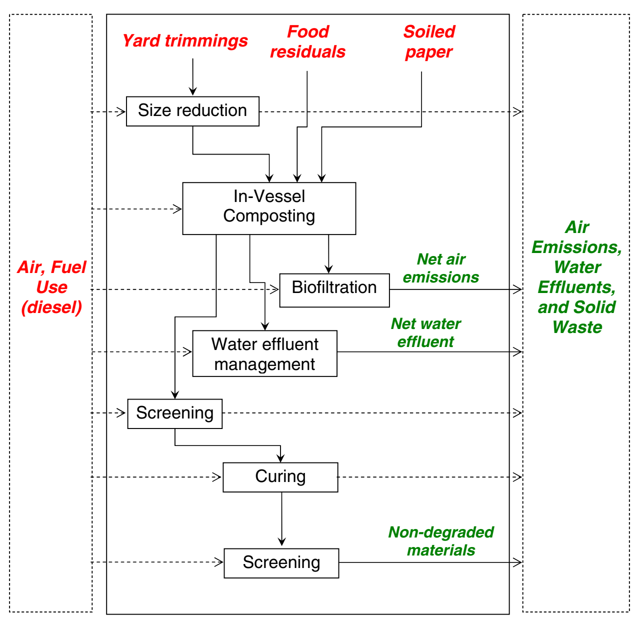

Fig. 1 Life-cycle inventory system boundaries of an aerobic in-vessel composting facility

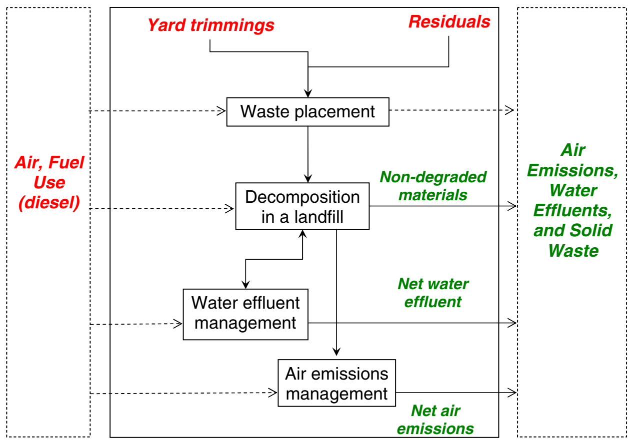

Fig. 2 Life-cycle inventory system boundaries of a bioreactor landfill

of organic wastes in a rotary vessel, therefore allowing a higher degree of process control than is possible with windrow composting. This system is excellent for composting large amounts of yard waste with food waste and soiled paper, and does not take as much space as windrow composting. Like windrow composting, carbon to nitrogen ratios and moisture contents are also considered. However, the composition of feed materials is less critical for in-vessel systems. The process is aerobic, and hence the by-products are primarily CO_2 and water. The assumptions for aerobic in-vessel composting operations considered in this study are adapted from O'Hern and O'Neill (2005) and Zimms (2005). The organic feedstock consisting of yard trimmings, food residuals, and soiled paper from materials recovery facility (MRF) are conveyed to an enclosed, rotary digester, where natural decomposition occurs for a minimum of 14 days with the addition of air and moisture. Leachate is also recirculated to provide the necessary moistening of the compost. The digester is continuously rotated by mechanical means so that materials inside the unit are aerated and broken down into smaller pieces. Additional screening of the partially stabilized compost occurs after discharge from the digester. After the active composting period of 14 days, the material is cured in windrows for an additional 6–10 weeks. Moisture and temperature are also monitored, and moisture mixing and conditioning is achieved using a turning machine, or by adding water if needed. The resulting material is screened and residuals are either disposed off or put back into the digester, depending upon the composition. Facilities are designed with containment and provisions for treatment for all leachate and runoff. Treating gases using a biofilter before emission to the atmosphere mitigates the release of nuisance odor primarily accounted to ammonia (NH_3) by around 60%.

Bioreactor landfill process model

As in sanitary landfills, basic procedures carried out in bioreactor landfills are spreading and compacting the solid waste materials in layers, and covering the material with soil at the end of each day. Bioreactor landfill systems include liquid, usually leachate, and/or air circulation systems, with leachate and gas collection. In the Bioreactor Landfill Process Model, accelerated transformation and microbial degradation of organic matter is accomplished through the controlled recirculation of leachate or other sources of moisture. In this method, leachate quality is also potentially improved, leading to reduced leachate disposal costs. LFGs are emitted earlier in the process and at a higher rate than the conventional "dry-tomb" landfill but for a total shorter duration, typically within 5–10 years of implementation.

Waste management scenarios considered

The alternatives developed represent the management of MSW from its setout at curbside through its final disposition. Three collection and waste flow combinations are defined as follows:

1. The base scenario, which includes the curbside collection of un-separated (commingled) refuse (including yard waste) in a single compartment garbage truck. These wastes are disposed of in a conventional landfill without recovery of materials.
2. The second scenario includes a curbside collection program as described in Scenario 1. The commingled refuse is brought to a mixed refuse MRF, where polyethylene (PET) beverage containers, plastic soda tubs and cups, aluminum and steel cans, paper, and glass are separated and sent to appropriate remanufacturing facilities. Food waste, yard waste, and soiled paper are sent to an aerobic composting facility using an in-vessel digester tube; all other waste is disposed of in a conventional landfill.
3. The third scenario includes a curbside collection program of commingled recyclables, where all paper components of the recyclables are placed in one compartment separate from all non-paper components. Yard waste is collected separately at the curb and transported to a windrow composting facility; all other waste is disposed of in a bioreactor landfill. This scenario requires stakeholders' participation in a commingled recyclables collection and a yard waste collection programs.

Participation rate is defined as the fraction of the community that participates in the collection option. For

Table 1 Composition and recovery of materials in MSW (percent of generation of each material)

Material	% In MSW	% Recovery
Paper and paperboard	35.2	48.1
Glass	5.3	18.8
Metals		36.3
Steel	5.9	26.4
Aluminum	1.4	31.4
Other non-ferrous metals	0.7	66.7
Plastics	11.3	5.2
Rubber, leather, and textiles	7.2	8.8
Wood	5.8	9.4
Other materials	3.4	22.7
Food, other	11.7	2.7
Yard trimmings	12.1	56.3

Source: (EPA 2005, Municipal Solid Waste in the United States: 2003 Facts and Figures)

purposes of calculations, participation rate for yard waste collection was set at 80%, while for commingled recyclables collection was 65%. Capture rate, which is the fraction of the total recyclable generation that a participating recycler is expected to be able to segregate from MSW, was assumed equal to the recovery values presented in EPA (2005). Waste compositions used in the models were also adopted from EPA (2005), shown in Table 1.

Table 2 lists the activities that contribute to energy use, air emissions, waterborne releases, and solid waste output for each of the options described above. All emissions are considered after any treatment.

Calculations for emissions and energy consumption from each solid waste unit operation were based on the quantity and composition of waste processed. Where parameters are required to represent a site-specific situation, typical values and national averages were used. Values presented are point estimates, and no data variability is included.

A total of 19 airborne and waterborne environmental pollutants or indicators were considered in the inventories. Airborne pollutants included: CH_4, $CO_{2\ biomass}$, $CO_{2\ fuel}$, SO_x, NO_x, N_2O, PM, CO, NH_3, benzene, ethene, toluene, propene, and some unnamed unburned hydrocarbons. $CO_{2\ biomass}$ refers to emissions associated with the biodegradation (aerobic or anaerobic) of organic materials during composting or landfilling. $CO_{2\ fuel}$, on the other hand, refers to emissions associated with fuel combustion during waste collection and transport, and electricity generation. CH_4, a greenhouse gas that contributes to ozone loss, is a byproduct of the anaerobic breakdown of the organic material in landfill.

NO_x, formed when fuel is burned at high temperatures, contribute to nutrient overload that deteriorates water quality (known as eutrophication). In moist environments, the oxides of sulfur (SO_x), like NO_x, may be transformed into sulfuric acid, which makes soils, lakes, and streams acidic, and damages trees, crops, historic buildings, and monuments. Ground level ozone (O_3), created by chemical reaction between NO_x and volatile organic carbon (VOC), triggers a variety of health problems even at low levels, and damages plants and the ecosystem (Matthews 2003). SO_x, NO_x, and O_3 migrate over long distances, following prevailing wind patterns. Nitrous oxide (N_2O) is about 200 times more potent than CO_2 as a greenhouse gas (Kiely 1998).

Carbon monoxide (CO), a colorless and odorless gas formed from the incomplete combustion of fuel, also contributes to the greenhouse effect, smog and acidification. Particulate matter (PM), on the other hand, is a term for particulates found in the air, including dust, dirt, soot, smoke, and liquid droplets. Some particulates are directly emitted into the air; others are formed in the air from the chemical change when gases from the burning of fuel react with sunlight and water vapor. It is the major source of haze that reduces visibility; it settles on soil and water, and harms the environment by changing the nutrient and chemical balance (Matthews 2003). Other emissions to air are those that result from the burning of fuel in vehicles and during power generation, and include unburned hydrocarbons, ethylene, propylene, benzene, and toluene.

Waterborne pollutants include Biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), NH_3, and heavy metals (nickel and zinc). BOD_5, like COD, is an indicator for the concentration of biodegradable organic matter present in a sample of water. If discharged untreated, organic compounds will result in micro-organism blooms, resulting in oxygen depletion and fish kills.

Characterization methods

Energy inputs are those that are derived from non-renewable sources (diesel); energy recovery from the utilization of landfill gas (LFG) is listed separately. Hence, with respect to energy, a negative value means that the energy recovered exceeds the energy expended for the facility or the scenario. For example, for the landfill, there is energy savings associated with the conversion of the LFG to electrical energy in place of fossil-derived fuel. These savings exceed the total energy required for collection and transport of recyclables, yard trimmings and refuse, and composting and/or landfill operations.

The contribution of greenhouse gases (CH_4, CO_2, N_2O, and CO) was characterized using the global warming

Table 2 Activities associated with the SWM alternatives that contribute to energy use, air and water releases, and solid waste output

SWM activities	Environmental parameters		
	Energy use	Air and water releases	Solid waste output
Waste collection and transportation	Energy represented by fuel consumed by MSW collection vehicles	Releases from the combustion of fuels by MSW collection vehicles ($CO_{2\ fuel}$, CO, NO_x, particulates, unburned HC, alkenes)	
Composting Processes			
Windrow Composting	Energy represented by fuel consumed by heavy equipment for operations	Releases from the combustion of fuels by heavy equipment for operations ($CO_{2\ fuel}$, CO, NO_x, particulates)	Compost product
		Volatilization to air of products of aerobic decomposition of organic wastes ($CO_{2\ biomass}$, NH_3, COD, BOD_5, Ni, Zn)	
In-Vessel Composting	Energy represented by fuel consumed by equipment composting for operations	Releases from the combustion of fuels by equipment for composting operations ($CO_{2\ fuel}$, CO, NO_x, particulates)	Sludge formed from leachate treatment
	Energy represented by fuel consumed by equipment for biofilter and leachate management	Emissions associated with gas and leachate management ($CO_{2\ fuel}$, CO, NO_x, SO_x, N_2O, particulates)	Compost product
		Fugitive emissions from gas and leachate management ($CO_{2\ biomass}$, NH_3, COD, BOD_5, Ni, Zn)	
Landfill Processes (Conventional and Bioreactor)	Energy represented by fuel consumed by heavy equipment for operation	Releases from the combustion of fuels by heavy equipment for operation ($CO_{2\ fuel}$, CO, NO_x, particulates)	Landfilled material itself
	Energy represented by fuel consumed by equipment for landfill gas and leachate management	Emissions associated with landfill gas and leachate management ($CO_{2\ fuel}$, CO, NO_x, SO_x, N_2O, particulates)	
		Fugitive emissions from landfill gas and leachate management (CH_4, $CO_{2\ biomass}$, NH_3, COD, BOD_5, Ni, Zn)	

potentials (GWP) for a time frame of 20 years, which are presented as CO_2-equivalents. CH_4, the alkenes (benzene, ethene, propene, and toluene), as well as some unnamed unburned hydrocarbons, contribute to photo-oxidant formation, here represented by photochemical ozone creation potentials (POCP), expressed as ethene-equivalents.

Acidification effect is defined as the amount of protons released in a terrestrial system. The acidification potentials (AP) of NO_x, SO_x, and NH_3, expressed as SO_2-equivalents, are used in this paper. Substances contributing to aquatic eutrophication (NO_x, NH_3, and COD) are classified and characterized according to the oxygen demand when the biomass they build up is decomposed. Hence, eutrophication potentials (EP) are presented as PO_4-equivalents.

The classification of the emissions to air and water, and the subsequent characterization of their contributions as AP, EP, or POCP, made use of the Eco-Indicator 95 method available in the SimaPro 7.0 software; the Intergovernmental Panel on Climate Change (IPCC) factors with a timeframe of 20 years were used for GWP (IPCC 2001 GWP 20a).

The methodology used by Johansson (1999) in assessing ecotoxicological effects were adapted in this study. Toxicological effects on aquatic ecosystems, contributed by metals and organic matter in water emissions, are considered. These are presented as aquatic ecotoxicity potentials (AEP), expressed in zinc-equivalents. The reference substance used for characterizing human toxicity potentials (HTP) is lead in air, taking only the inhalation route of exposure into account. The characterization factors used in this study are as shown in Tables 3 and 4.

Solid waste management life cycle models

The SWM Life Cycle Models, developed for the waste management strategies considered in this study, address the aspects of waste management from the instance the materials become waste by losing value, to the moment it regains value or adds to the environmental burden or leaves the waste management system in the form of emissions. Unit operations include collection of commingled waste,

Table 3 Characterization factors based on equivalency factors from IPCC 2001 GWP 20a and eco-indicator 95

Resource	Characterization Factors
Global warming potential (GWP)	
CH_4	62
CO_2	1
CO	1.57
N_2O	275
Acidification potential (AP)	
NO_x	0.7
SO_x	1
NH_3	1.88
Eutrophication potential (EP)	
NO_x	0.13
NH_3	0.33
COD	0.022
Photochemical ozone creation potential (POCP)	
CH_4	0.007
Benzene	0.189
Ethene	1
Hydrocarbons, unspecified	0.398
Propene	1.03
Toluene	0.563

Table 4 Characterization factors based on equivalency factors from Johansson (1999)

Resource	Characterization factors
Aquatic ecotoxicity potential (AEP)	
BOD_5	0.00013
Ni	0.79
Zn	1
Human toxicity potential (HTP) for emissions to outdoor air	
CO	0.00014
NO_x	0.002
SO_x	0.0075
PM	0.0075
Human Toxicity Potential (HTP) for emissions to water	
BOD_5	0.022
Ni	0.062
Zn	0.0032

commingled recyclables, and residuals, windrow composting of yard trimmings, in-vessel composting of MSW organics, and landfilling (conventional and bioreactor).

An MRF can handle the commingled MSW, where sorters pull out corrugated cardboard, newspaper, bottles, cans and plastics. Materials pass through a screen while underflow from the screen goes into the aerobic digester, and overflow are transferred and disposed off in the conventional landfill. No LCI model is developed for the sorting operation and the subsequent processing of recyclables from this MRF.

Waste collection process model

In the Waste Collection Process Model, environmental emissions associated with the collection of commingled and separated recyclables, yard wastes, and residual waste were calculated.

Transport distances used in the scenarios, based mainly on own assumptions, are presented in Table 5. The waste was assumed to be collected at a curbside using a garbage truck. In the base scenario, the commingled waste is transported around 48 kilometers (km) to the landfill site, plus around 45 km empty return to the dispatch station. In the composting-landfill scenarios, the yard waste (and food waste and soiled paper for aerobic in-vessel composting) is transported around 16 km to the composting facility.

Collection vehicles were assumed to have a capacity of 30 cubic meters (m^3) each, requiring around 4 m^3 of diesel fuel per 100 km (White et al. 1995). Data on emissions associated with road transports by trucks were adopted from Degobert (1995), and Westerholm and Egeback (1994).

Composting process models

The composting processes considered in this study are the windrow composting for yard wastes, and the aerobic in-

Table 5 Distances within the collection and transportation system of the different scenarios considered in the study

Activity	Km
To first pickup location (commingled waste, recyclable, compostable, residual)	16
During pick-up	
Commingled waste	97
Recyclable	215
Compostable	215
Residual	97
During haul to MRF (commingled waste, recyclable)	48
During haul to composting facility	16
During haul to landfill (commingled waste, residual)	48
From MRF to dispatch station	45
From composting facility to dispatch station	45
From landfill to dispatch station	45

Figures represent one-way trips

vessel composting for yard waste, soiled paper, and food waste described previously.

Yard waste composting process model

The Yard Waste Composting Process Model uses aerated windrows, and handles a mixture of leaves, grass, and branches. Typically, the feedstock is first shredded and mixed, then placed on soil in rows called windrows that are typically 2 meters (m) high, 4–5 m wide, and up to 90 m or more in length (Tchobanoglous et al. 1993). Moisture and temperature are monitored. If needed, water is added to maintain the desired moisture level. With efficient turning by a windrow turner, minimum composting time is 1 month, followed by at least 2 months in a curing pile. The compost product is then screened after curing.

Each of the composting models calculates emissions based on the energy consumed for the composting operation, as well as those from the composting process itself. Default values for the composting processes used in this study are presented in Table 6. Airborne and waterborne emissions related to energy consumption within the facilities were included in the inventory, and were estimated using data from White et al. (1995). Carbon dioxide (CO_2) and NH_3 emissions associated with composting were estimated from Tchobanoglous et al. (1993), using the maximum fractional bioconversion of the organic matter of 75% obtained from Themelis and Kim (2002). No methane (CH_4), oxides of nitrogen (NO_x), or trace organic emissions were attributed to the composting processes in this study. Waterborne and solid environmental emissions produced during the composting process were also estimated, using data from Krogmann and Woyczechowski (2000) and White et al. (1995). It was assumed that the final compost product accounts for 50% of the input due to the composting process; the other 50% is lost through evaporation and respiration (White et al. 1995). It should be noted that the generation of liquid by-products depends on the initial moisture content of the feedstock and the composting technology.

Landfill process models

Landfill gas production is difficult to calculate as it depends on several factors such as the waste stream composition and moisture content, disposal practices (i.e., type of daily cover, degree of compaction), and the degree of decomposition of the solid waste. In estimating the total volume of LFG produced, complete conversion of the biodegradable organic waste was assumed. However, all carbon that exists in the waste may not be available for LFG production.

Outputs calculated from the Landfill Process Models were integrated over time, because the gas and leachate produced by each unit weight of waste landfilled will eventually be released. Two models were considered based on the rate of decomposition of the organic fraction: rapidly decomposing (total decomposition in 5 years), and slowly decomposing (total decomposition in 15 years). The yearly rate of decomposition of these rapidly and slowly decomposable materials was based on a triangular gas production model in which the peak rate of gas production occurs after 5 and 15 years, respectively.

Gas yields were calculated using the procedure described by Tchobanoglous et al. (1993) over the period of the landfill's active decomposition of 20 years. Gases produced from the degradation of organic matter in a landfill primarily consist of CH_4 and CO_2, and small amount of volatile organic chemicals and other hazardous air pollutants. LFG components were assumed based on the

Table 6 Summary of default parameters characterizing the composting process models

Parameter	Value	Units	References
Energy consumption			White et al. (1995)
Aerobic in-vessel	−198	kJ per kg of input to facility	
Turned windrow	−71	kJ per kg of input to facility	
Amount of liquid by-products			Krogmann and Woyczechowski (2000)
Aerobic in-vessel	−0.334	m^3/MT	
	+20	% of water raining on pile	
Turned windrow	−0.030	m^3/MT	
	+20	% of water raining on pile	
Moisture content of compost product			White et al. (1995)
Aerobic in-vessel	45.0	% of wet weight	
Turned windrow	34.8	% of wet weight	

individual component's concentration in the LFG, using median values from Tchobanoglous et al. (1993). The concentration values were assumed constant over time.

Engineered landfills have gas collection system with an efficiency of 80% for the conventional landfills; gas collection is about 90% efficient for bioreactor landfills (Barlaz et al. 2003) because gas is produced over a shorter duration. The uncollected gas enters the atmosphere as pollutant, while all captured LFG is used as fuel which replaces consumption of resources required for the production of equivalent electrical power. Electricity production was based on burning diesel fuel having calorific value around 35.6 megajoules (MJ)/liter. Data on emissions upon electricity production were adapted from White et al. (1995). The amount of energy produced from the burning of LFG was credited to the landfill process as avoided energy production from another energy source. Default emissions associated with the burning of LFG, as well as electricity production, were based on White et al. (1995). Collecting the LFG and using it for electrical energy generation avoid emissions from the anaerobic degradation of the solid waste. The energy content of the gas collected is around 20.3 MJ/m^3, and a conversion efficiency of 30% was used for simulating burning LFG to generate electricity (White et al. 1995).

Biologically induced waterborne emissions were estimated based on data reported by Kjeldsen and Christophersen (2001). BOD, which is known to peak quickly and decline as methane is formed from the decomposition of organic matter in the waste, as well as the concentrations of the other leachate components, was assumed to be constant over time. Leachate production was estimated to be around 10% of precipitation at the landfill site. It was assumed that 99% of the leachate from the conventional landfill is collected and treated, and the remaining 1% will leak to aquatic recipients (White et al. 1995). Waterborne pollutants released during conventional landfilling were estimated on the assumption that only 1% of the leachate generated will leak to aquatic recipients. The collected leachate is treated in a treatment plant within the landfill site, demanding 0.001 MJ per kilogram (kg) of leachate (Finnveden et al. 2000); sludge formed in this process is landfilled. In a bioreactor landfill, on the other hand, leachate is recirculated to achieve a sufficient and uniform moisture distribution throughout the waste, thereby supplying the nutritional needs of the microorganisms, providing a mechanism for heat transfer, and treating the leachate as a trickling bioreactor filter process. For both conventional and bioreactor landfills, the leachate treatment plant removal factors developed by Finnveden et al. (2000) were used. Leachate treatment was assumed to produce around 22 kg of sludge per m^3 of treated leachate (White et al. 1995).

Although further compaction and settlement of the landfilled material occurs during its decomposition, the total volume of the material delivered (calculated from the specific densities of the different waste materials) was used to approximate the final volume of the solid waste resulting from landfilling.

Full costs of MSW management

Default cost data used in this study, summarized in Table 7, were based on those presented in EPA (1998, 2000), and Komilis and Ham (2004), adjusted to 2005 values. "Full costs" refer to overall cost of solid waste managing system, including past and future capital outlays, costs for land acquisition and long-term care, and hidden costs (i.e., resources donated that do not result in cash outlays). The cost of additional infrastructure needed to implement accelerated degradation by recycling of the leachate was estimated to be <$1.10/metric ton, and additional operating costs to run a landfill as a bioreactor were assumed to not exceed that for a modern conventional landfill (Clarke 2000).

Results and discussion

This section presents a comparison of aerobic in-vessel composting and bioreactor landfill, followed by an analysis of the LCIs of the three alternate SWM scenarios where these waste treatment options are integrated.

Table 7 Summary of default full cost values for the different solid waste management activities

Parameter	Value, in US$ per metric ton of waste managed
Curbside collection of refuse[a]	153.20
Curbside collection of recyclables[a]	867.50
Curbside collection of yard trimmings[a]	88.20
Composting facility[b]	
Aerobic in-vessel	232.60
Windrow	18.70
Landfill	
Conventional[a]	38.60
Bioreactor[c]	39.70

[a] EPA (1998) full cost accounting in action: case studies of six solid waste management agencies

[b] EPA (2000), Biosolids technology fact sheet in-vessel composting of biosolids

[c] Komilis and Ham (2004)

Aerobic in-vessel composting versus bioreactor landfill

Life cycle inventory parameters using the Aerobic In-Vessel Composting Process Model were compared with those obtained using the Bioreactor Landfill Process Model. The estimated environmental performance of the aerobic in-vessel MSW composting and the bioreactor landfill obtained from these models are illustrated in Figure 3.

Energy consumption

Operating an in-vessel composting facility consumes about twice more energy than what is required for a bioreactor landfill, owing to the requirements for the continuous rotation of the materials in the digester, screening of materials before and after the composting process, the management of leachate and runoff, and treatment of process air. Energy is saved through the bioreactor landfill, since the LFG is burned to meet operational needs, leading to energy offset of about 9.6 MJ/kg of landfilled waste. There is no such energy recovery for in-vessel composting, which does not produce CH_4.

Global warming

Emissions of greenhouse gases are larger from the landfill alternative, mostly as CH_4 in the time period considered. Around 5% of the $CO_{2\ fuel}$ and around 3% of the N_2O are precluded from being emitted to the atmosphere when energy recovered in the bioreactor landfill operations is utilized to meet operational energy needs. N_2O quantities emitted are insignificant by comparison with CO_2. More $CO_{2\ fuel}$ from the burning of diesel fuel to meet energy requirements is released from an in-vessel composting than from a bioreactor landfill. However, because the organic fraction of MSW is treated aerobically in vessel, no CH_4 emission was attributed to the composting process.

Acidification, eutrophication, and photo-oxidant formation

The composting alternative gives the larger contribution to acidification and eutrophication, arising mainly from NH_3 emissions from the composting process. Emissions to air of NO_x and SO_x, are also greater from an aerobic in-vessel composting than bioreactor landfilling. These pollutants are mostly associated with burning of diesel fuel for the generation of electrical energy needed in the composting operations.

Waterborne organic pollutants are generally less for a bioreactor landfill than those associated with in-vessel composting. During the operational lifetime, all leachate is assumed biologically treated in situ in a bioreactor landfill. In contrast, recirculation of liquid byproduct from composting is regulated to some degree because excess water impedes the diffusion of oxygen through the compost materials and thus leads to unwanted anaerobic conditions. Thus, a large portion of the leachate needs to be routed for treatment before it is released as emissions. Due to inefficiencies in wastewater treatment, the aerobic in-vessel composting facility generates an effluent with higher COD

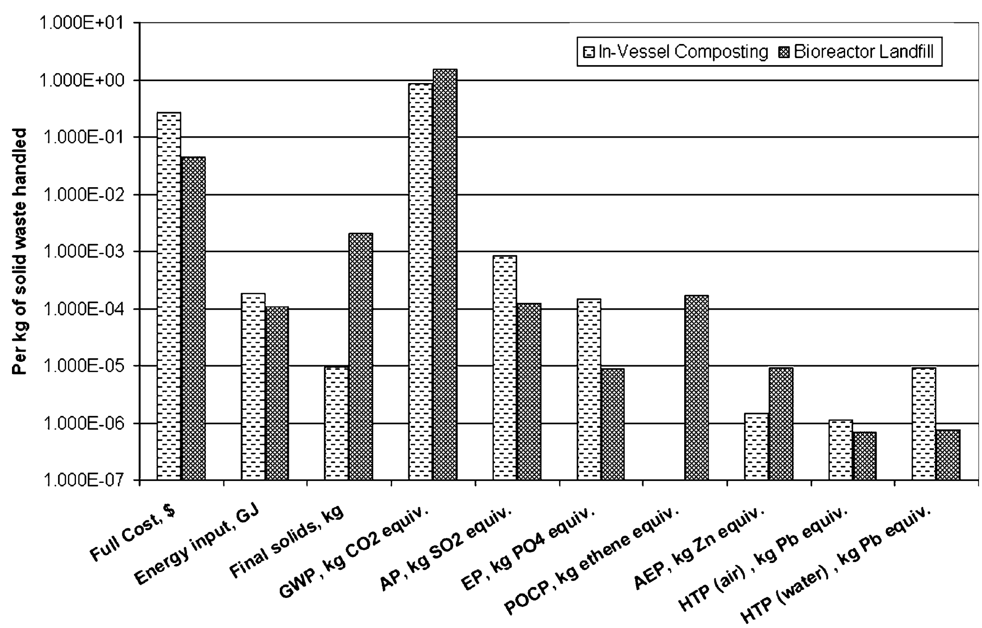

Fig. 3 Comparison of life-cycle impacts of aerobic in-vessel composting and bioreactor landfill

than the bioreactor landfill, which contributes largely to eutrophication. CH_4 emissions from landfill operations contribute to photo-oxidant formation.

Toxicological effect on aquatic ecosystems

Biochemical oxygen demand, which determines the degree of bioconversion of the organic matter in the leachate, is higher for aerobic in-vessel systems due to the assumed maximum degradation of only 75% at the end of the composting period as compared to the complete conversion in the bioreactor landfill. In spite of this, the composting alternative provides less toxicological effect on aquatic ecosystems than landfilling. The aerobic conditions in vessel limit fermentation reactions that would have occurred had conditions been anaerobic, which produce large amounts of acids and significantly reduce pH. Consequently, about 88% less metal constituents are dissolved in the aerobic composting leachate than from anaerobic landfilling.

Toxicological effect on human health

Emissions to air contributing to human toxicity are greater for aerobic in-vessel composting than for bioreactor landfill, arising from the burning of diesel fuel to supply power requirements for the composting operations. The composting option also generates more waterborne effluents that contribute to human toxicity, attributed mostly to its high BOD_5 content.

Final non-treated solid wastes

Both treatment methods cause a net generation of waste, considerably larger for bioreactor landfill than aerobic in-vessel composting. It is mostly bulk waste that is disposed to the landfill along with biodegradable wastes.

Full costs

The LCI models allowed for the estimation of the full costs of the solid waste treatment options. From the models, the calculated full cost of US$ 0.05 per kg of waste handled in a bioreactor landfill is cheaper when compared to US$ 0.27 for in-vessel composting. The additional capital cost for a bioreactor is the cost of leachate recirculation/distribution, additional gas collection pipe work, and power generating equipment compared to a conventional landfill. Because gas collection and utilization is a feature of landfill technologies, there is therefore practically no expense beyond the cost of conventional landfill operations. Aerobic in-vessel composting of MSW, on the other hand, is considered more costly with respect to capital expenditure. Odor control systems account for up to 50% of the capital and operating and maintenance costs. Additional costs may also be incurred for the containment and treatment of leachate and runoff from in-vessel composting operations. More equipment maintenance is also necessary for in-vessel systems.

Integration of aerobic in-vessel composting and bioreactor landfill in SWM systems

The LCI models were applied using typical United States MSW compositions for the three waste management scenarios. The predictions of the LCI models for these scenarios are presented in Table 8, using the default data discussed in the previous section.

The mass of wastes directed to the landfills is reduced by around 63% in Scenario 2 where recyclables are segregated at an MRF, and yard waste, food waste, and soiled paper are aerobically composted in vessel. Of the amount landfilled, only 23% is biodegradable organics. Waste diversion for Scenario 3, which integrates the separate collection of recyclables with the windrow composting of yard waste and landfilling of residual waste in a bioreactor landfill, is around 55%. In this scenario, about 61% of the mass of waste going to the bioreactor landfill is biodegradable.

Collection and transportation of waste and waste fractions

Around 91–95% of the total energy use in all three scenarios considered goes to the collection and transportation of the different waste fractions (commingled recyclables, yard trimmings, residual waste) for remanufacturing or materials recovery, composting, and landfilling. In Scenario 2 additional routes are created to transport commingled wastes to the MRF, feedstock (yard waste, food waste, and soiled paper) to the in-vessel composting facility and residuals to the conventional landfill facility; likewise for Scenario 3, where separate routes are created for the collection of yard waste, recyclables, and residuals, and their subsequent transportation to waste treatment facilities such as, respectively, windrow composting, MRF, and bioreactor landfill. Hence, waste diversion from the landfill doubled gaseous emissions contributing to global warming arising from the burning of diesel fuel used by vehicles for collection and transportation. Emissions associated with additional collection and/or transportation routes for Scenarios 2 and 3 also increased photochemical oxidant formation by around 40–50%.

Table 8 Life-cycle inventory of the three solid waste management scenarios considered

	Scenario 1			Scenario 2				Scenario 3			
	Collection	Conventional landfill	Whole system	Collection	In-vessel composting	Conventional landfill	Whole system	Collection	Windrow composting	Bioreactor landfill	Whole system
Energy use, GJ	7.7E−04	7.3E−05	8.4E−04	9.6E−04	7.2E−05	2.8E−05	1.1E−03	1.0E−03	8.5E−06	4.7E−05	1.1E−03
Energy recovery, GJ[a]	–	(1.7E−02)	(1.7E−02)	–	–	(1.9E−03)	(1.9E−03)	–	–	(7.3E−03)	(7.3E−03)
GWP, kg CO_2 equiv.	6.8E−02	5.9E + 00	6.0E + 00	1.8E−01	3.6E−01	7.0E−01	1.2E + 00	1.9E−01	8.7E−02	1.1E + 00	1.4E + 00
AP, kg SO_2 equiv.	1.7E−03	8.8E−03	1.0E−02	2.1E−03	2.6E−04	1.2E−02	1.4E−02	2.3E−03	2.3E−03	4.5E−03	9.1E−03
EP, kg N equiv.	3.2E−04	1.5E−03	1.8E−03	4.0E−04	4.5E−05	2.1E−03	2.5E−03	4.2E−04	4.1E−04	7.8E−04	1.6E−03
POCP, kg thane equiv.	8.3E−05	6.5E−04	7.3E−04	1.2E−04	–	7.5E−05	1.9E−04	1.2E−04	–	1.2E−04	2.5E−04
AEP, kg zinc equiv.		1.1E−04	1.1E−04	–	5.9E−07	3.0E−06	3.6E−06	–	1.4E−07	3.7E−06	3.8E−06
HTP (air), kg lead equiv.	5.7E−06	3.0E−05	3.6E−05	7.1E−06	4.5E−07	4.2E−05	4.9E−05	7.6E−06	5.0E−08	1.6E−05	2.3E−05
HTP (water), kg lead equiv.	–	8.5E−06	8.5E−06	–	3.7E−06	2.5E−07	4.0E−06	–	3.9E−06	3.0E−07	4.2E−06
Final solids, m^3	–	2.0E−03	2.0E−03	–	4.2E−06	6.6E−04	6.7E−04	–	8.3E−10	8.2E−04	8.2E−04
Cost, $	0.18	0.04	0.22	0.18	0.11	0.02	0.30	0.48	0.00	0.02	0.51

Scenario 1. Collection and disposal of commingled wastes in conventional landfill

Scenario 2. Collection of commingled wastes; recyclables recovered in MRF, while yard trimmings, food waste, and soiled paper are composted aerobically in-vessel, and residual wastes are disposed of in conventional landfill

Scenario 3. Separate collection of commingled recyclables, yard trimmings, and residual wastes; yard trimmings are aerobically composted in windrows, and residual wastes are disposed of in bioreactor landfill

[a] Values in parentheses are negative

Collection costs represent a large portion (around 60–95%) of the total MSW expenditures for all three scenarios. The separate collection of commingled recyclables, yard trimmings, and residuals for Scenario 3 entails additional trips that result in increased collection costs, and translates to increased environmental impacts due to the emissions from the vehicles.

Composting

Global warming potential is larger from Scenario 2, mostly as $CO_{2\ \mathrm{biomass}}$ generated during the aerobic in-vessel composting process, where a larger fraction of the organic wastes undergo biodegradation than in the windrow composting of yard waste for Scenario 3.

Composting operations for Scenario 3, however, provide the greatest contribution to acidification and eutrophication. NH_3 emission is highest for this scenario, originating mostly from the composting of yard trimmings where this and other airborne particles are released to the environment during natural processing, ventilation, and windrow turning. Total NH_3 emissions from Scenario 2, also accounted for by its composting operation, are greatly reduced by the containment and subsequent treatment of the process air by an odor control system.

The increased BOD_5 emissions associated with Scenarios 2 and 3 reflect the amount of leachate that is produced from composting accounting for 97 and 99%, respectively, of the total BOD_5. Toxicological effect on aquatic ecosystems is greater for waterborne emissions from the in-vessel composting than from that of windrow composting. More liquid by-products are generally produced when MSW is composted compared to when only yard wastes are composted. Trace metals are considerably lowered because of the reduction in the amount of solid wastes going to the waste treatment facilities in Scenarios 2 and 3.

Landfill

Consequent to the reduction in the amount of organic waste landfilled, energy recovery is decreased by around 88% for Scenario 2 and around 57% for Scenario 3. More energy is recovered in Scenario 3 relative to Scenario 2 due to the larger amount of organic wastes disposed of and a more efficient LFG collection system, which leads to a higher energy offset. However, fugitive emissions of CH_4, which provide a large contribution to global warming and photochemical oxidant formation, are reduced by around 88% for Scenario 2 and 81% for Scenario 3.

Acidification and eutrophication effects, arising mainly from NO_x and SO_x emitted from the use of diesel fuel to power landfill operations, is greater for Scenario 2, where less amount of recoverable energy is available than for Scenario 3. These air emissions also contributed to a greater toxicological effect on human health for Scenario 2.

The chemical quality of landfill leachate is influenced by several factors, including the rate of production, the composition of landfilled waste and physical, chemical, and biological properties of the waste. Most modern landfills, however, provide more efficient lining systems to avoid or minimize the seepage of leachate. Relatively small quantities of contaminants escape through the liner due to advection and diffusion. On account of these low hydraulic conductivity barriers, BOD_5 emissions to ground water are reduced by around 81%, and trace metals by around 16% from landfill liners.

The volume of final solids (residue) produced from Scenario 2 resulting from the decomposition of the waste is comparable to that from Scenario 3, and is around 60–67% of that resulting from a scenario that allows all of the solid wastes into the conventional landfill (Scenario 1). Diversion of solid wastes from the landfill clearly increases the life of the landfill, i.e., more waste can be disposed of in the landfill.

The estimated environmental performances of the three SWM scenarios considered in this study are presented in Fig. 4. In Table 9, a summary of the rankings of the waste management scenarios for the different impacts categories is presented.

Conclusions

This paper describes and compares life cycle estimates of the environmental performance of aerobic in-vessel composting and bioreactor landfilling. To assess the relative significance of energy use and environmental impacts, three SWM scenarios were established that integrate aerobic in-vessel composting and bioreactor landfilling. Estimates of cost, energy use, and environmental releases were calculated using the life-cycle models developed for the collection of wastes and the subsequent waste treatment methods considered. The key findings are presented for these four environmental parameters: solid waste output, energy use, air emissions, and waterborne wastes and relative costs.

Diverting the recyclables and organic materials from the landfill will reduce the final solid residue, and diminish the quantities of LFG produced, thereby reducing both greenhouse gas emissions and energy generation potential. However, the creation of additional collection routes for organics, and recyclables, will involve additional vehicles on the roads and could negatively impact the environment due to increased emissions from fuel use unless

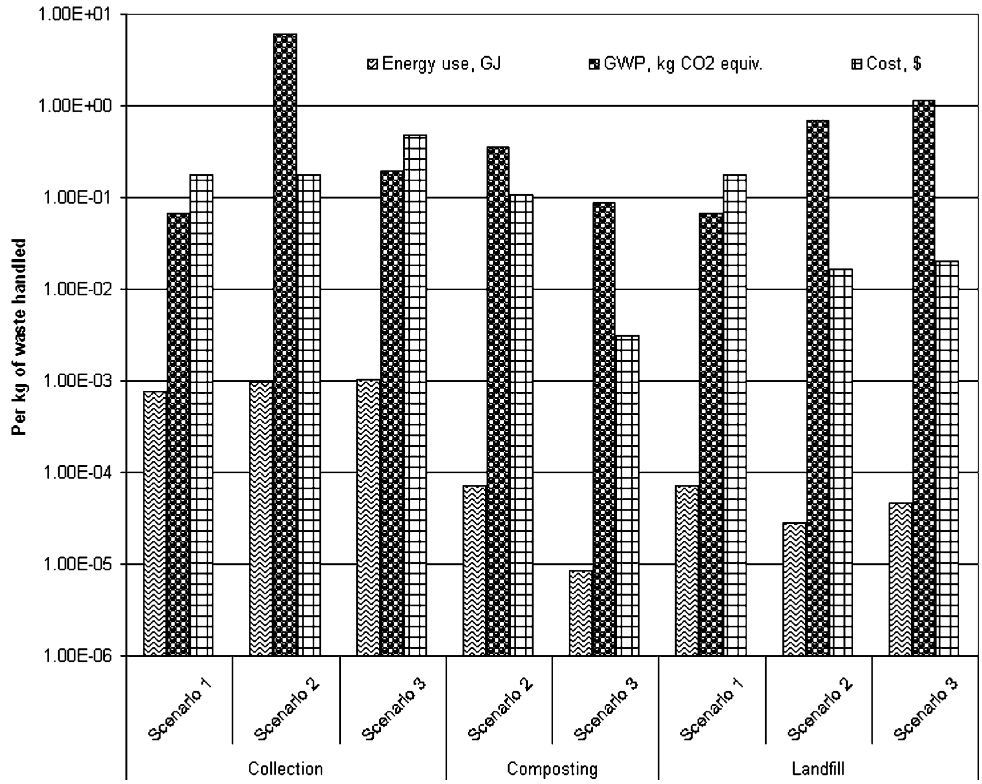

Fig. 4 Comparison of life-cycle impacts of the swm scenarios considered

Table 9 Ranking of the waste management alternatives considered for the whole system for each impact category

Impact category	Ranking
Energy use	Scenario 2 < Scenario 3 < Scenario 1
Energy recovery	Scenario 1 < Scenario 3 < Scenario 2
Global Warming	Scenario 2 < Scenario 3 < Scenario 1
Acidification	Scenario 3 < Scenario 1 < Scenario 2
Eutrophication	Scenario 3 < Scenario 1 < Scenario 2
Photochemical oxidant formation	Scenario 2 < Scenario 3 < Scenario 1
Aquatic ecotoxicity	Scenario 2 < Scenario 3 < Scenario 1
Human toxicity (air)	Scenario 3 < Scenario 1 < Scenario 2
Human toxicity (water)	Scenario 2 < Scenario 3 < Scenario 1
Final solids	Scenario 2 < Scenario 3 < Scenario 1
Cost	Scenario 1 < Scenario 2 < Scenario 3

Scenario 1. Collection and disposal of commingled wastes in conventional landfill

Scenario 2. Collection of commingled wastes; recyclables recovered in MRF, while yard trimmings, food waste, and soiled paper are composted aerobically in-vessel, and residual wastes are disposed of in conventional landfill

Scenario 3. Separate collection of commingled recyclables, yard trimmings, and residual wastes; yard trimmings are aerobically composted in windrows, and residual wastes are disposed of in bioreactor landfill

low-emission fuels are used for these vehicles (e.g., corn or liquid hydrogen-based fuels). These additional routes will also translate into increased collection costs. From a system-wide view using life-cycle inventory, separate collection of commingled recyclables, windrow composting of yard trimmings, and disposal of the remainder in a landfill operated as a bioreactor produces the lowest airborne and waterborne emissions.

Energy consumed in each of the three alternative SWM scenarios is at least 60% less than the energy generated by the recovery and use of the LFG. Reduction in the amount of solid wastes going to the waste treatment facilities through materials recovery considerably lowers waterborne emissions.

According to the results of this study, bioreactor landfill is a favorable option over in-vessel composting with regard to cost, overall energy use, airborne and waterborne emissions. The modeling of various scenarios allowed a closer evaluation of both the economic and selected environmental burdens for these scenarios. Because of lack of field data in some cases, however, assumptions were made that could impact the conclusions. Hence, such analysis using site-specific data may be needed in some cases when evaluating SWM options.

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