Evaluating landfill diversion strategies for municipal organic waste (MOW) management using environmental and economic factors

Philip A. Tominac[1], Horacio A. Aguirre-Villegas[2], Joseph R. Sanford[3], Rebecca A. Larson[2], Victor M. Zavala[1]

[1] Department of Chemical and Biological Engineering, University of Wisconsin-Madison

[2] Department of Biological Systems Engineering, University of Wisconsin-Madison

[3] Department of Soil and Crop Sciences, University of Wisconsin-Platteville

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Abstract

Municipal organic waste (MOW) contributes to greenhouse gas (GHG) emissions which lead to global climate change. Creating incentives that encourage consumers to adopt MOW disposal strategies that minimize environmental impacts. Policymakers must ensure that incentive programs align with environmental objectives and are economically competitive, else such policy will create inefficiencies. The MOW disposal infrastructure in the city of Milwaukee, WI was evaluated using a coordinated market model that captures the inherent value of using environmental indicators estimated with partial life cycle assessment methods in the context of the city's waste management supply chain. Using this approach, tax programs are identified that incentivize consumers to reduce emissions through their MOW disposal choices. Results indicate that the current MOW collection and disposal infrastructure in Milwaukee is such that consumers are already incentivized to minimize MOW GHG emissions by collecting MOW and sending this to composting and digestion sites that have lower tipping fees and are located closer to the city.

Therefore, limitations associated with organic collection likely decrease alternative landfill disposal options. When minimizing GHG emissions, anaerobic digestion systems are prioritized for allocation of MOW, but the situation can be altered depending upon tipping fees and transport distances. Composting and digestion hold few opportunities for reducing ammonia emissions from landfills, but there is a need to expand ammonia emissions measurements from landfill systems to improve model predictions.

Introduction

Increasing urbanization have led to increases in municipal solid waste (MSW) generation. Despite consistent *per capita* waste generation over the last 17 years in the US, annual generation increased 18% since 2000 to 242.9 million tonnes of MSW in 2017 (USEPA, 2019a). Currently, 52% of waste generated in the US is landfilled, and with rising MSW production the available lifetime of municipal landfills is decreasing, forcing municipalities to site new landfills or find alternatives. Aside from lifetime constraints and siting issues, landfills are a source of emissions to air and water that contribute to environmental degradation. Gaseous emissions from landfills are the third largest contributor to anthropogenic methane (CH_4) in the US, accounting for 2% of greenhouse gas (GHG) emissions (USEPA, 2017). Space constraints and environmental awareness are creating incentives for municipalities to reduce their reliance on landfills.

The organic fraction of MSW, municipal organic waste (MOW), includes food waste and yard trimmings which are biodegradable and are responsible for landfill GHG emissions. Diverting MOW from landfills reduces the volume added extending landfill life. Nearly 75% of food waste and 25% of yard waste is landfilled, representing 28% of annual MSW in the US (USEPA, 2019a). Multiple countries, states, and local governments have developed policies to promote diversion of MOW. In 1999, the European Union set the EU Landfill Directive requiring member states to achieve a 35% reduction of MOW to landfills by 2016 (EU, 1999). Some US municipalities have also implemented policies requiring the separation of MOW for curb-side collection or local drop-off (Portland City Code, 2012a, 2012b; San Francisco Environment Code, 2009; Seattle Municipal Code, 2015a, 2015b). These polices increase rates of MOW treatment, and as of 2017, an estimated 5.1 million US households have access to curbside MOW collection, and 6.7 million others have access to local drop-off MOW collection programs (Streeter and Platt, 2017). The City of Milwaukee, WI has programs that have diverted 40% of annual MOW between 2013 and 2020, with a goal of 50% by 2030, with all residents having access to curbside collection by 2025 (City of Milwaukee Public Works, 2016). Collection programs such as these have resulted in a 7% increase in diversion of MOW from landfills between 2000 and 2017 in the US (USEPA, 2019b).

MOW diverted from landfills can be processed via technologies like composting, anaerobic digestion (AD), or waste-to-energy combustion (WTE) that reduce environmental impacts. Life cycle assessment (LCA) provides a means of comparing environmental impacts through metrics like global warming potential (GWP) or eutrophication potential (EP) (Khandelwal et al., 2019a; Laurent et al., 2014; Yadav and Samadder, 2018). Composting transforms organic waste into a stable, nutrient-rich product used as a soil amendment or fertilizer to offset the use of synthetic fertilizers for crop production. Roughly 36% of MOW (69% of yard waste and 6% of food waste) in the US is composted, making it the most common alternative for MOW apart from landfilling (USEPA, 2019a). Multiple LCAs have investigated the integration of composting into MOW management (Bovea et al., 2010; Buratti et al., 2015; Hodge et al., 2016; Jaunich et al., 2019; Khandelwal et al., 2019b; Liikanen et al., 2018; Oliveira et al., 2017; Ripa et al., 2017). Composting is often found to be more favorable for reducing GWP compared to landfilling. Oliveira et al. (2017) modeled diversion of 90% of MOW in São Paulo, Brazil to an existing regional composting facility which reduced GHG emissions by 40%. However, while composting was more favorable in terms of GWP, it was often less favorable in terms of EP resulting from land application. Hodge et al. (2016) indicated similar results in the US in relation to GWP and noted that direct emission of GHG during the composting process accounted for the majority of emissions.

Anaerobic digestion (AD) produces biogas, an energy source that can offset fossil fuel consumption as well as digestate, a fertilizer for crop production to offset synthetic fertilizer. LCA studies investigating the integration of AD systems into MOW management (Bernstad and la Cour Jansen, 2012; Hodge et al., 2016; Khandelwal et al., 2019b; Liikanen et al., 2018; Rajaeifar et al., 2015; Ripa et al., 2017; Slorach et al., 2019) find AD systems are environmentally favorable compared to landfilling, composting, and WTE. Slorach et al. (2019) found that diverting food waste in the United Kingdom to AD systems rather than landfills or WTE improved 15 out of 19 impact categories assessed, and could offset 1% of industrial fertilizers needed for crop production. Hodge et al. (2016) reported AD systems significantly reduced GWP compared to landfilling and compost due fossil fuel offsets.

The environmental impacts associated with MOW in landfills and the benefits from alternative processing are often investigated through LCAs; however, the economic value of diversion schemes has not been extensively studied. Levis et al. (2014) evaluated the cost of the MSW management in a hypothetical US city using the Solid Waste Optimization Life-cycle model (SWOLF) to evaluate complete MSW diversion and found it would increase costs by 65%. When modeling to minimize GHG production, the cost increase was only of 19%. In a similar study, Jaunich et al. (2019) estimated that, for Wake County, NC to maximize food waste diversion, a carbon value of 98 USD per tonne CO_2-eq was required, and to minimize GHG production up to 839 USD per tonne CO_2-eq was required.

The goal of this study is to use LCA methods to estimate environmental indicators and optimization techniques to compare different MOW diversion strategies. The City of Milwaukee will serve as a case study to assess economic policies (taxes) that incentivize patterns of landfill diversion strategies targeted at minimizing environmental impacts.

Methods

Organic Municipal Solid Waste (MOW) Management for Milwaukee, WI

The city of Milwaukee currently uses two transfer stations for MSW collection that receive a combined 144,000 tonnes of MSW per year that is landfilled at two locations in Wisconsin (average one-way distance of 16 km from the transfer points). The landfills have a combined total capacity of 7,154,969 m^3 and are equipped with a biogas collection system which produces electricity that is supplied to the grid. It is estimated that annually 28% of MSW is MOW, based on EPA estimations of MSW fractionation (USEPA, 2019a). Based on these data, MOW was assumed to consist of 53.2% food waste and 46.8% yard waste. Food waste constitutes vegetable (29%) and non-vegetable (71%) sources, while yard waste constitutes brush (25%), leaves (25%) and grass clippings (50%). Total MOW constituents (Table 1) were estimated based on this composition and literature values for total solids (TS), volatile solids (VS), total ammoniacal nitrogen (TAN), total potassium (TK) (U.S. EPA, 2019c), total N (TN), and total P (TP) (Campuzano and González-Martínez, 2016).

Table 1. Municipal organic waste (MOW) composition

kg/tonne MOW					
TS	VS	TAN	TN	TP	K
367	341	0.25	7.9	1.7	0.4

A large-scale composting operation, a wet AD, and a dry AD system were included as alternative MOW processing systems. The composting facility utilizes a windrow composting process with windrows having a width, height, and length of 3.4, 1.8, and 45.7 m, respectively. Windrows are turned for aeration roughly every five days. The compost operation is active year-round, and material is typically processed in 16 to 24 weeks depending on seasonal temperature variation. The wet AD facility has two 4,921 m^3 tanks that operate as a continuous stirred reactor, having a retention time of 30 days, and a hydraulic capacity of 500 m^3 per day of emulsified MOW is included in the model. A dry AD facility is also included in this assessment, which operates as a batch system with a retention time of 28 days operating at mesophilic temperature. Water is added throughout the process to maintain a TS content of 37%. Biogas that is produced from the system is converted into electricity for the grid.

Life Cycle Assessment for Organic Waste Management

System boundaries, environmental indicators, and scenario definition

This work uses the SWOLF model to estimate environmental indicators for MOW management (Levis and Barlaz, 2013a, 2013b; USEPA, 2011). Evaluated indicators investigated are i) GHG emissions (measured in tonne CO_2-eq) including carbon dioxide (CO_2) from fossil energy combustion, biotic CH_4 and nitrous oxide (N_2O, direct and indirect from ammonia (NH_3) volatilization) ii) NH_3 emissions (tonne NH_3), and iii) resource depletion (RSD), measured by the consumption of fossil energy (MJ). Characterization factors for CH_4 and N_2O are 28 and 265 for a 100-year horizon (Myhre et al., 2013). Carbon storage is not considered in the study, but indirect N_2O emissions from NH_3 volatilization are included by using the emission factor 0.01 kg N_2O-N/kg NH_3-N (IPCC, 2006). Environmental indicators are indicated per tonne MOW.

Figure 1 shows the system boundaries including unit-processes from transport to landfilling or processing to land-application and product end use. The system begins with MOW production and tracks MOW through transportation and processing via landfill (LFILL), compost (COMP), dry AD (DAD), and wet AD with solid-liquid separation (WAD-SLS) technologies. The transportation network includes the city transport station. The infrastructure modeled reflects the operating characteristics of the sites in Milwaukee. Environmental impacts and benefits are quantified for each technology. Benefits include the additional products of renewable electricity (i.e. scenarios with AD) and nutrients for crop production when land applied. It is assumed that these nutrients replace the production and consumption of synthetic fertilizers. Nutrient availability for applied TAN, organic N, P, and K are 100% (after volatilization), 43%, 80%, and 80% respectively (Laboski and Peters, 2012). Grid electricity GHG emissions are specific to WI (U.S. EIA, 2018) and embedded impacts in the production of electricity, N, phosphate (P_2O_5) and potash (K_2O) are modeled using SimaPro (Pre-Consultants, 2019).

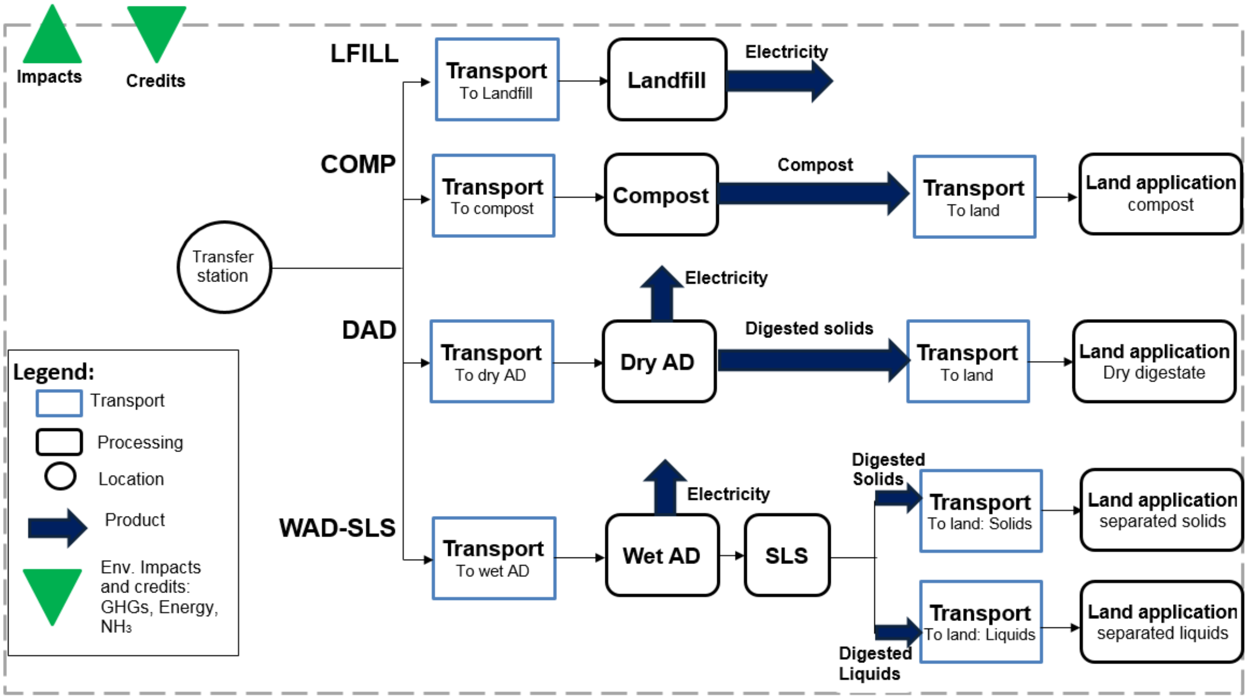

Figure 1. System boundaries for MOW scenarios of landfilling (LFILL) and four landfill diversion scenarios: composting (COMP), dry anaerobic digestion (DAD), and wet anaerobic digestion and solid-liquid separation (WAD-SLS).

Landfill

Total capacity of the existing landfill is 1,806,955 tonnes of MOW, assuming MOW density of 889 kg/m^3 and a MOW fraction of 28% MSW (U.S. EPA, 2019c). The landfill receives 461,411 tonne of MSW from many sources every year and it is assumed that a HDPE tarp is used for daily and final covers. Modeled environmental impacts occur from the transportation of MOW from the transfer station to the landfill and from pre-closure, closure, post-closure, and leachate management operations estimated based on USEPA (2011). GHGs are emitted as fossil CO_2 from the operation of equipment, with additional CH_4 fugitive emissions from the production of electricity from landfill gas. NH_3 emissions are estimated using a NH_3:CH_4 mass ratio of 0.0073 (Roe et al., 2004). Estimated material and energy inventories are presented in Table S1. No environmental impacts are modeled for landfill construction. Environmental benefits come from electricity generated from captured biogas (LHV of CH_4: 35.8 MJ/m^3, capacity factor: 0.9, electricity efficiency: 35%, thermal efficiency: 50%) that replaces grid electricity. It is assumed that all landfill leachate is recirculated to facilitate the production of biogas, avoiding treatment of leachate in-situ or in a wastewater treatment plant. Collected CH_4 for electricity production is calculated using Eq. 1 (USEPA, 2011)

$$CH_{4(produced)} = CH_{4(collected)} + CH_{4(oxidized)} + CH_{4(leaks)}, \qquad (1)$$

where $CH_{4(produced)}$ represents total production estimated from each MOW fraction (See Eq. S1), $CH_{4(collected)}$ represents electricity generation based on $CH_{4(produced)}$ multiplied by collection efficiency (assumed to be 82.3% over a period of 100 years for landfill) and equipment destruction efficiency for CH_4 (assumed to be 0.99), $CH_{4(oxidized)}$ represents CH_4 converted to CO_2 by methanotrophic organisms that reside in the cover soil (assumed to be 15% of uncollected CH_4), and $CH_{4(leaks)}$ represents CH_4 escaping the collection system and passing through the cover soil (i.e., $CH_{4(produced)}$-$CH_{4(collected)}$-$CH_{4(oxidized)}$).

Compost

MOW transported to the composting site is processed and land applied. Composting inputs are diesel and electricity (Table S1), resulting in fossil CO_2 emissions. CH_4 is emitted during composting and N_2O and NH_3 are emitted during composting and after land application. These emissions are estimated based on Levis and Barlaz (2013a) who provide emission factors for each organic waste stream. Aerobic curing is assumed after the composting process, which results in N_2 emissions. While not a GHG, it reduces compost N availability. The model assumes 70 days for windrow active composting and 30 days for curing. Water evaporates during composting and curing, reaching a final moisture of 45%. All inputs and emissions are adjusted for these moisture losses. The environmental impacts avoided through composting result from nutrients (TN, TP and TK) recovered for land application.

Anaerobic Digestion

DAD converts MOW into digestate suitable for land application. WAD-SLS produces solid and liquid fractions that are both suitable for land application but require separation. Before digestion, MOW is pretreated (involving screening, mixing, and dewatering) to maximize biogas production. Target TS is 37% for dry digestion and 10% for wet digestion. No additional drying is necessary in DAD, but additional water is needed for WAD-SLS. An average biogas yield of 85 m^3/tonne and CH_4 content of 59% are used for food and yard waste, respectively (Levis et al., 2010). Environmental benefits come from the production of electricity from biogas and the nutrient content in digestate for land application. No losses of TP and TK are considered, but the amount of TN reaching land application depends on changes in N form and on N emissions during and after AD and SLS. About 50% of organic N is mineralized during AD (Levis and Barlaz, 2013b) becoming more susceptible to N volatilization as NH_3 after land application. Fossil CO_2 is emitted from diesel consumption during MOW pre-treatment and land application (Table S1). Biogas offsets electricity and heating requirements. CH_4 leaks from the digestion process are 2.8% of the total CH_4 produced (UNFCCC, 2017). We assume digestate is surface applied. After land application, 0.0197 kg N_2O-N/kg TAN are emitted from digestate (Chadwick et al., 2011) and NH_3 emissions are estimated with Eq. S2 (in supplementary material). Table 2 and Figure S1 (in supplementary material) present the lifecycle inventory of all modeled environmental impacts and benefits.

Table 2. Life cycle inventory of GHG and NH_3 emissions, fossil energy consumption, avoided impacts, and available nutrients after land application

	LFILL[a]	COMP	DAD	WAD-SLS
Per tonne of incoming MOW				
Processing				
$CO_{2(fossil)}$ (kg CO_2-eq)	NA	5.65	1.97	1.97
CH_4 (kg)	NA	1.84	1.58	1.58
$N_2O_{(direct)}$ (kg)		0.05	-	-
$N_2O_{(indirect)}$ (kg)	NA	0.0025	-	-
Subtotal GHG processing (kg CO_2-eq)	NA	70.8	46.3	46.3
GHG avoided (kg CO_2-eq)	NA	-	-160[b]	-146[b]
NH_3 emissions				
NH_3 (kg)	NA	0.19	-	-
Fossil energy				
Fossil energy consumed (MJ)	NA	86.9	29.3	29.3
Fossil energy avoided[b] (MJ)	NA	-	-3,033[b]	-2,773[b]
Land application				
GHG emissions				
$CO_{2(fossil)}$ (kg CO_2-eq)	7.85	1.11	2.32	8.1
CH_4 (kg)	4.84	-	-	-
$N_2O_{(direct)}$ (kg)	-	0.06	0.02	0.05
$N_2O_{(indirect)}$ (kg)	0.005	0.003	0.05	0.03
Subtotal GHG land (kg CO_2-eq)	144.7	18.9	21.1	30.4
GHG avoided (kg CO_2-eq)	-57.3[b]	-25.5[c]	-33.4[c]	-47.6[c]
NH_3 emissions				
NH_3 (kg)	0.36	0.25	4.1	2.55
Resource depletion (fossil energy)				
Fossil energy consumed (MJ)	127.2	16.5	34.5	120
Fossil energy avoided (MJ)	-1,087[b]	-206[c]	-253[c]	-338[c]

Total GHG emissions (processing + land application + avoided, kg CO_2-eq)	87.4	64.3	-125.8	-117.0
Total NH_3 emissions (processing + land application, kg NH_3)	0.4	0.5	4.1	2.5
Total GHG emissions (processing + land application+ avoided, MJ)	-960.2	-102.3	-3222.4	-2961.5
Nutrients available				
TAN (kg)	NA	0.04	0.58	1.8
Organic N (kg)	NA	3.68	3.95	3.95
TP (kg)	NA	1.7	1.7	1.7
TK (kg)	NA	0.4	0.4	0.4

[a]For simplicity, LFILL impacts are all listed under the land application category, but these impacts are from landfilling activities

[b]From avoided grid electricity production

[c]From avoided fertilizer production

Market Modeling Methodology for Waste Management Policy

Modeling and Rationale

The City of Milwaukee landfill diversion goals (City of Milwaukee, 2013) are based on waste diversion from landfills to compost without modifying existing incentive structures to improve environmental sustainability. To evaluate potential alternative policies, two aspects of this problem were considered: the economic incentives present in the waste management supply chain and the environmental indicators (measured by LCA). The supply chain economics and environmental indicators were incorporated into a mathematical optimization model that treats the problem as a coordinated market. This market operates as a supply chain management system where an Independent System Operator (ISO) collects bidding information from profit-seeking stakeholders (buyers, sellers, transportation service providers, waste processing service providers, and policymakers). The ISO acts as a coordinator that pairs stakeholders together in transactions, circumventing the need for peer-to-peer negotiations. Stakeholders provide bidding information to the ISO consisting of a bid value (i.e., a sale or purchase rate) and a bid capacity (an amount available for sale or purchase). The ISO uses this information to determine the set of transactions creating the greatest market value; the maximum *social welfare*. These transactions are referred to as *allocations*.

The coordinated market model used satisfies economic axioms of *competitiveness* (Sampat, 2019b) guaranteeing that no stakeholder loses money in the market (profits will never be less than zero). The ISO sets market prices, revealing the inherent value of products in the supply chain. Unprofitable transactions do not receive allocations and are called *dry*. Competitive transactions receive a positive allocation and are said to *clear*. Sampat *et. al.* (2019a) recently developed a coordinated market model for evaluating the economic feasibility of agricultural waste management technologies which was adapted here to assess landfill alternatives and incorporate LCA metrics.

The coordinated market model is used to create a meaningful quantification of the environmental impacts of MOW management. The environment is represented by policymakers who agree to absorb MOW impacts in exchange for payments. The ISO accepts bids from the policymakers along with those of other MOW stakeholders in the supply chain. Coordination determines how MOW is disposed, and revels the market value (clearing prices) of each LCA impact factor (e.g., the dollar value per tonne of GHG). These clearing prices allow us to investigate tradeoffs between environmental impacts and disposal costs. The market price of an impact factor is interpreted as a tax. The market propagates this tax through the supply chain, influencing the effective cost (the tipping fee) of each waste management practice. Impact taxes manipulate incentives for city residents by adding the environmental cost to disposal costs.

The mathematical representation of the MOW supply chain in this study includes the city as a waste producer, transport services, processing, and environmental impacts (from waste transport and processing). Each of these stakeholders is an independent profit-seeking entity. Each stakeholder has an associated bid value and a capacity in the model that the ISO takes into consideration when assigning allocations. The process of resolving the market is called *market clearing*. The mathematical model is referred to as a *market clearing problem*, and the product prices as *market clearing prices*. The model objective is a social welfare function that maximizes the combined total profits of all stakeholders. The complete mathematical model and a detailed description are available as supplementary material. The model is implemented and solved using the Julia 1.4.1 language (Bezanson, 2017), the JuMP 0.21.2 mathematical programming framework (Dunning, 2017), and the Gurobi 9.0 solver (Gurobi Optimization, LLC, 2020). Model outputs are the set of competitive allocations and the market prices. Allocations are the distribution of MOW to the available processing systems and the resulting environmental impacts.

Assessing Sustainability Policy Using Market Coordination

Our framework provides an abstract representation of stakeholders and the physical connections between them. Waste disposal pathways are represented as part of a supply chain network. Individual supply chain stakeholders correspond to suppliers, transporters, processors, and consumers. The supply chain abstraction of the Milwaukee system boundary from Figure 1 is provided in Figure 2. City residents are modeled as a collective MOW supplier, and policymakers are consumers of LCA impact factors from all sources. It is understood that policymakers are not literal consumers; they are proxies for environmental taxes (i.e., they represent the environment).

The model includes three environmental impact factors: GHG emissions, NH_3 emissions, and RSD. Impact taxes imposed on the system modify the economics of the supply chain and change the incentives for MOW suppliers. Understanding prices and incentives allows for systematic investigation into tax policies that encourage desirable outcomes. The incentives in Milwaukee may not reflect those elsewhere; environmental impact factors will have different values in different locations.

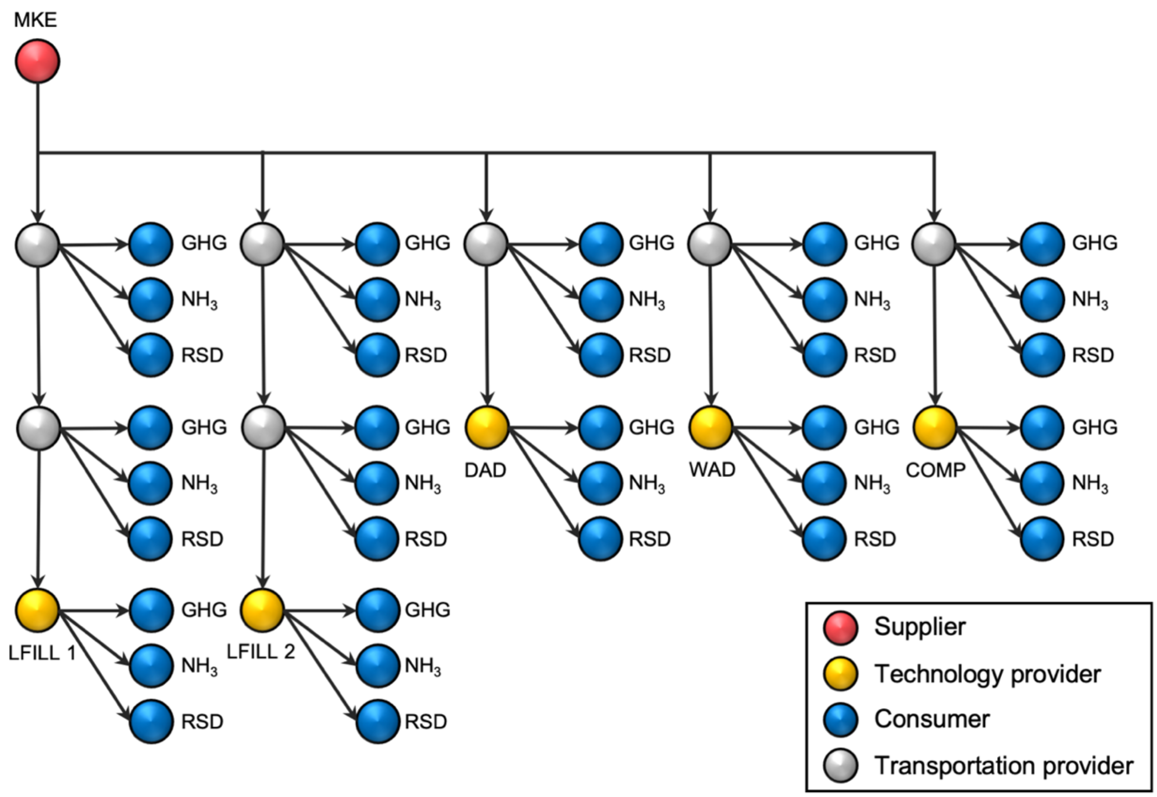

Figure 2. Schematic of the City of Milwaukee waste management infrastructure supply chain. Processing pathways are labeled and correspond to those in Figure 1.

The coordinated market model was used to examine existing incentives in Milwaukee's waste management infrastructure, and to explore how environmental taxes could be implemented to change these incentives. The model is based on Milwaukee's waste management supply chain as laid out in Figure 2 and using the data in Tables 2 and 3, and includes variations in transport distances, tipping fees, and processing capacity to examine impacts of thse on market incentives.

Table 3. MOW landfill and alternatives tipping fees, capacity estimates, and transport distances

	LFILL1	LFILL2	COMP	DAD	WAD-SLS
Tipping fee (USD/tonne MOW)	48.50	48.50	27.56	22.05	52.91
Annual capacity (tonne MOW)	15,422	22,680	15,422	4,536	28,742
Distance (km)	21	25	24	120	3

Results and Discussion

Model results are presented for GHG and NH_3 taxes. Discussion of RSD tax policy and additional discussion of LCA outcomes are included in the supplementary material.

GHG Taxation Policy

The coordinated market model subject to a GHG impact tax was solved using the supply chain and emissions data in Tables 2 and 3 (the base-case). Three variants on this case were evaluated to elucidate the underlying incentives and mechanisms at work in the market; these illustrate the ISO's decision-making properties. These variants are 1) reducing all transportation distances to a value of 10 km, 2) reducing all tipping fees to 22.05 USD per tonne MOW, and 3) combining variants (1) and (2). Variant (1) removes the effect of distance from the ISO's decisions, variant (2) removes the effect of tipping fees, and (3) removes both, leaving only environmental impact and capacity to influence the ISO. Lastly, the energy offset credits in Table 2 are assessed, which alters the market value of the impact factors.

The supply chain structure of the model allows us to directly calculate the values of an impact tax at which the ISO's MOW allocations (the model solution) will change. These critical tax values represent upper and lower bounds between which the ISO's allocations will be identical for any value of the tax. A range of tax values is produced between which there exists one incentivized market outcome. Optimal MOW allocations against identified tax ranges are provided in Figure 3.

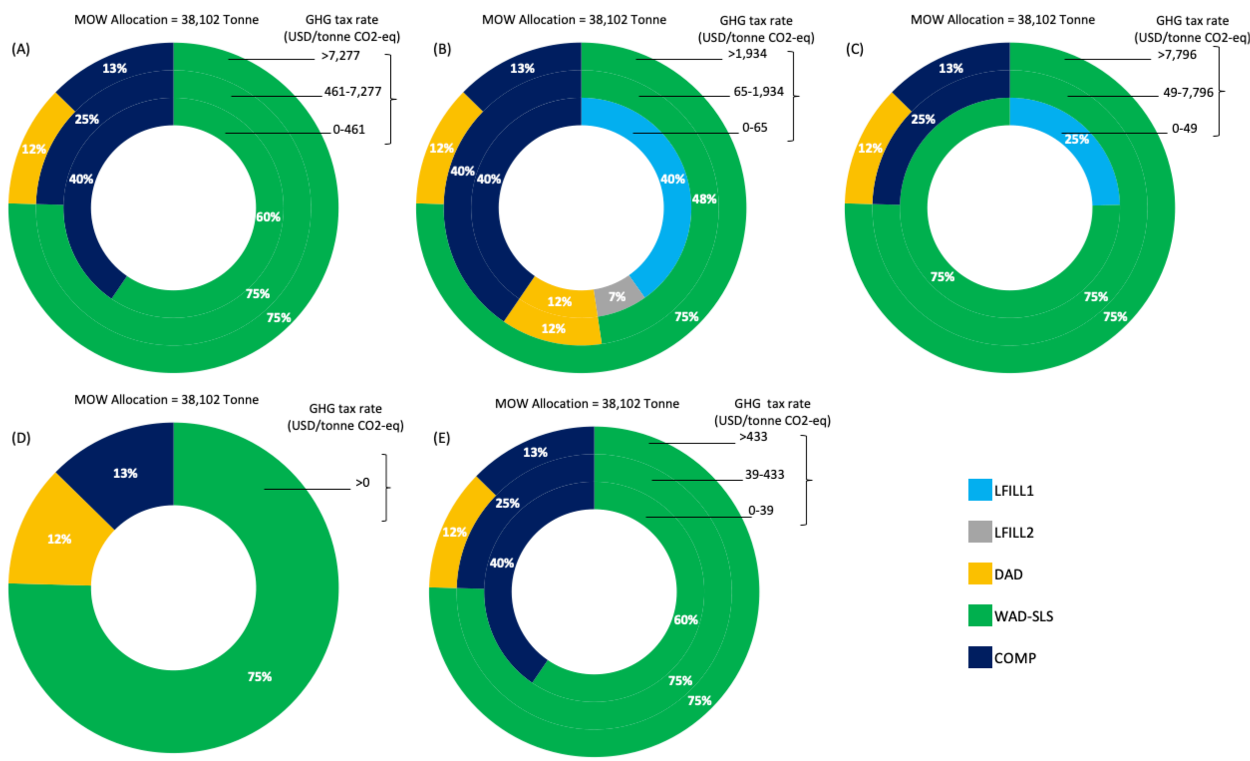

Figure 3. MOW allocations resulting from market coordination with a GHG emissions tax. (A) base-case data from Table 3, (B) all transport distances set to 10 km, (C) all tipping fees set to 22.05 USD per tonne MOW, (D) all transport distances set to 10 km and tipping fees set to 22.05 USD per tonne MOW, and (E) base-case including credits for renewable energy offsets.

The initial range of tax values in each variant in Figure 3 includes the value of zero, corresponding to no tax (inner circle). Tax values from zero up to the first upper limit do not provide the necessary economic incentives to change market behavior. Such taxes are inefficient, they raise market prices but do not create sufficient incentives. Figure 3A illustrates that the combination of low GHG emissions, tipping fees, and transport costs make COMP and WAD-SLS the optimal choices (even without a GHG tax). The indicates that, at current conditions, there is capacity for all the MOW produced in Milwaukee to move to landfill alternatives with no additional tax incentive. That landfills are still in use points to limiting conditions outside the boundaries of this study. This likely includes the additional cost and operational complexities to collect separated MOW from MSW. It should also be noted that increasing diversion to the maximum capacities of alternative processing systems (i.e., compost) could encourage landfill diversion, but also implies the need for expanding markets to absorb the finished products, a constraint highlighted by the processing system operators.

In all assessment scenarios, the highest GHG tax rates shift MOW allocations to WAD-SLS (75%) and then to DAD (12%), and the remaining to COMP (13%), which represents the lowest achievable GHG emisions (see Figure 4A). However, it should be noted that the significant increase in GHG tax rate does not shift the GHG emissions much lower than the initial allocation of 60% WAD-SLS and 40% COMP in 3A (reduces from 3,176 to 3,033 tonne CO_2-eq, Figure 4A). The tax rate required to reach the allocation of WAD-SLS (75%), DAD (12%), and (13%) in scenario 3B is much lower than in 3A (1,934 in 3B versus 7,277 USD/tonne in 3A). In 3A, the transport distance to DAD is 120 km and contributes to transportation costs of 102.96 USD/tonne (in 3B,with distance reduced to 10 km, the transport ccost is 8.58 USD/tonne) illustrating the magnitude of transportation costs in the suppl chain.

Observing that the DAD tipping fee is 22.05 USD/tonne, we note that transport presents a significant cost barrier for DAD, bringing the cumulative cost to 125.01 USD/tonne MOW. Comparing DAD in 3A to COMP (with slightly greater GHG emissions but a transport distance of only 24 km, with combined tipping and transportation costs of 48.15 USD/tonne MOW) the cost difference between the two is 76.86 USD/tonne MOW. We observe a shift away from COMP and toward DAD as the GHG tax rate increases. The GHG tax functions by increasing the cost of both technologies in proportion to their GHG emissions. Calculating the total GHG emissions rates for DAD and COMP (including both direct and transport) yields values of 0.08206 and 0.09271 tonne CO2-eq/tonne MOW. The difference between DAD and COMP is small (10.7 kg CO_2-eq/tonne MOW). We determine the tax rate required to overcome the cost barrier between DAD and COMP by dividing the 76.86 USD/tonne MOW value by the difference between the GHG emissions factors, which yields the tax value 7,277 USD/tonne CO_2-eq. This calculation demonstrates how tax rates are obtained.

Figure 3D indicates that a GHG tax can be used to overcome the incentives created by transport costs and tipping fees. However, at the high end of the tax this may not be practical as the 7,277 USD per tonne CO_2-eq threshold is almost two orders of magnitude above the 132 USD per tonne CO_2-eq charged in Sweden, the highest carbon tax in the world (Asen, 2019). Nevertheless, Figure 3A demonstrates that existing market incentives (even without a GHG tax) already encourage Milwaukee residents to use landfill alternatives. Finally, the allocations in Figure 3E are identical to Figure 3A, implying the renewable energy offsets encourage the same behavior, but at notably lower tax thresholds (39 compared to 461 USD per tonne CO_2-eq). One way to interpret this outcome is that the offsets create incentives similar to those of a GHG tax. This means that existing credit policies are creating incentives that promote GHG reductions, but at the same time, those incentives exist without credits.

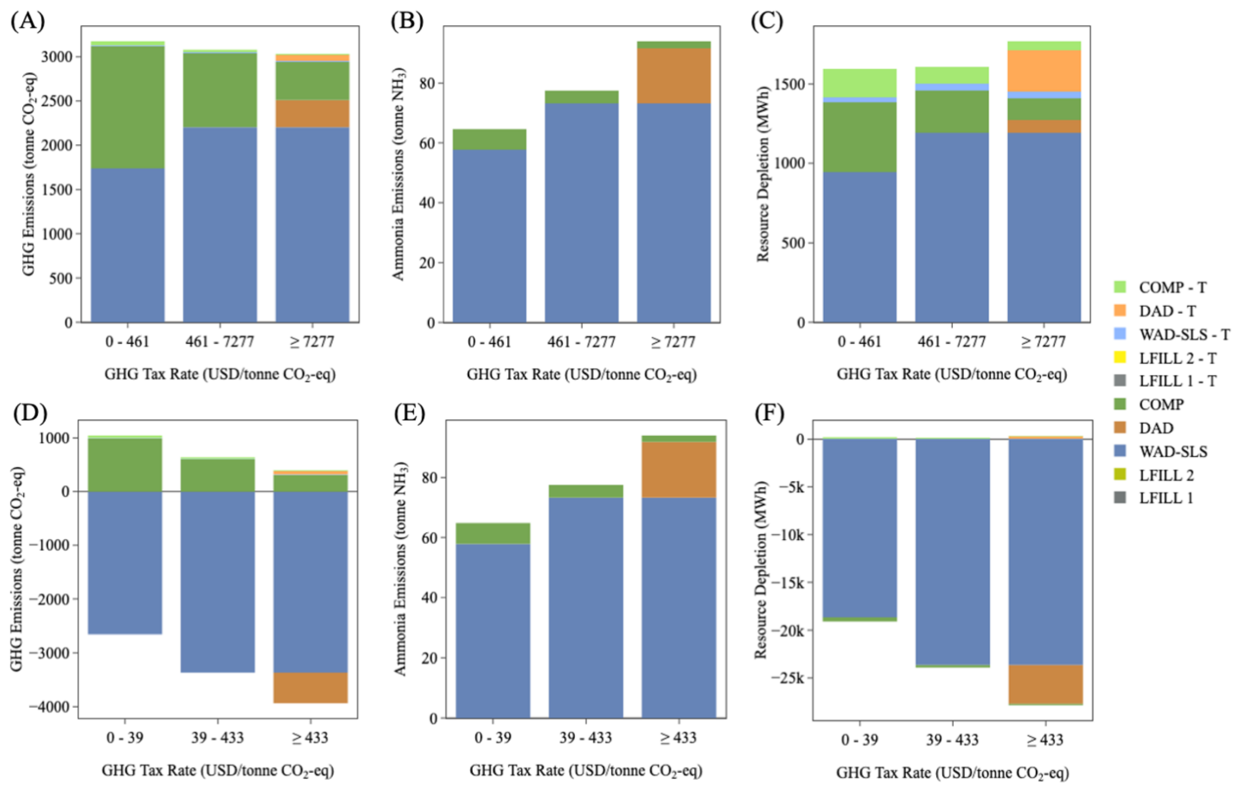

Figure 4. Impacts (GHG, NH_3, and RSD) for the base-case (A, B, C) and with energy offset credits (D, E, F) resulting from GHG taxation. Contributions from transportation indicated by "-T."

In both the base- and credited cases (Figure 4A and D) the tax scheme results in successively lower GHG emissions as the tax increases through the identified ranges. Unfortunately, the reductions in GHG emissions come at the cost of increases in NH_3 (Figure 4B and E) from 65 tonnes up to 78 and 94 with progressing tax rates. The change in RSD is different depending on the inclusion of offset credits: without the credits (Figure 4C) RSD increases by more than 175 MWh annually, whereas the credits (Figure 4F) provide a reduction of 8,600 MWh annually. Since the market approach is based solely on GHG reductions, the ISO is able to make tradeoffs between GHG reductions and substantial increases in other impacts. In most cases, except Figure 4C, the contributions from transportation are minor.

Ammonia Taxes and Incentives

The MOW allocations and tax value ranges resulting from applying the same methodology to NH_3 emissions are shown in Figure 5.

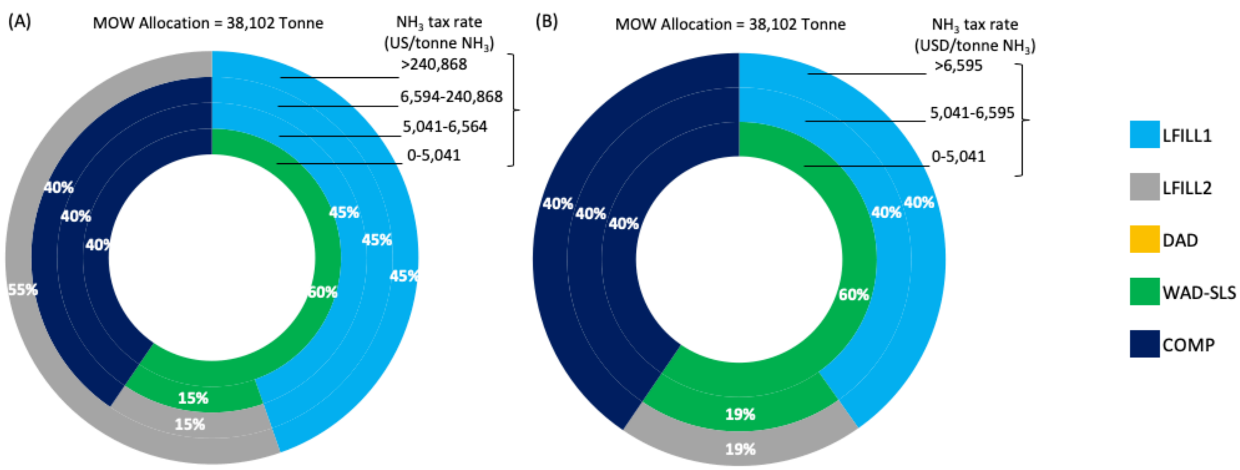

Figure 5. MOW allocations resulting from market coordination with an NH_3 emissions tax. (A) base-case data from Table 3, and (B) base-case including credits for renewable energy offsets.

The zero tax cases when minimizing NH_3 emissions are the same as those in the corresponding graphs in Figure 3; this is expected as this represents the untaxed case. The tax value ranges identified for NH_3 are interesting in that 1) the tax rate to incentivize change for NH_3 emissions is infeasibly high (more than 5,000 USD per tonne NH_3; by contrast, the market value of NH_3 fertilizer is presently between 500-600 USD per tonne (Widmar, 2020)), and 2) the incentives that an NH_3 tax creates in fact encourage the use of landfills. These results are in opposition to the incentives of a GHG tax, which creates incentives that discourage landfill use. Policymakers might consider that an NH_3 tax correlates negatively with a landfill diversion objective and would oppose the effects of a GHG tax. The emissions profiles resulting from the NH_3 tax schemes are presented in Figure 6.

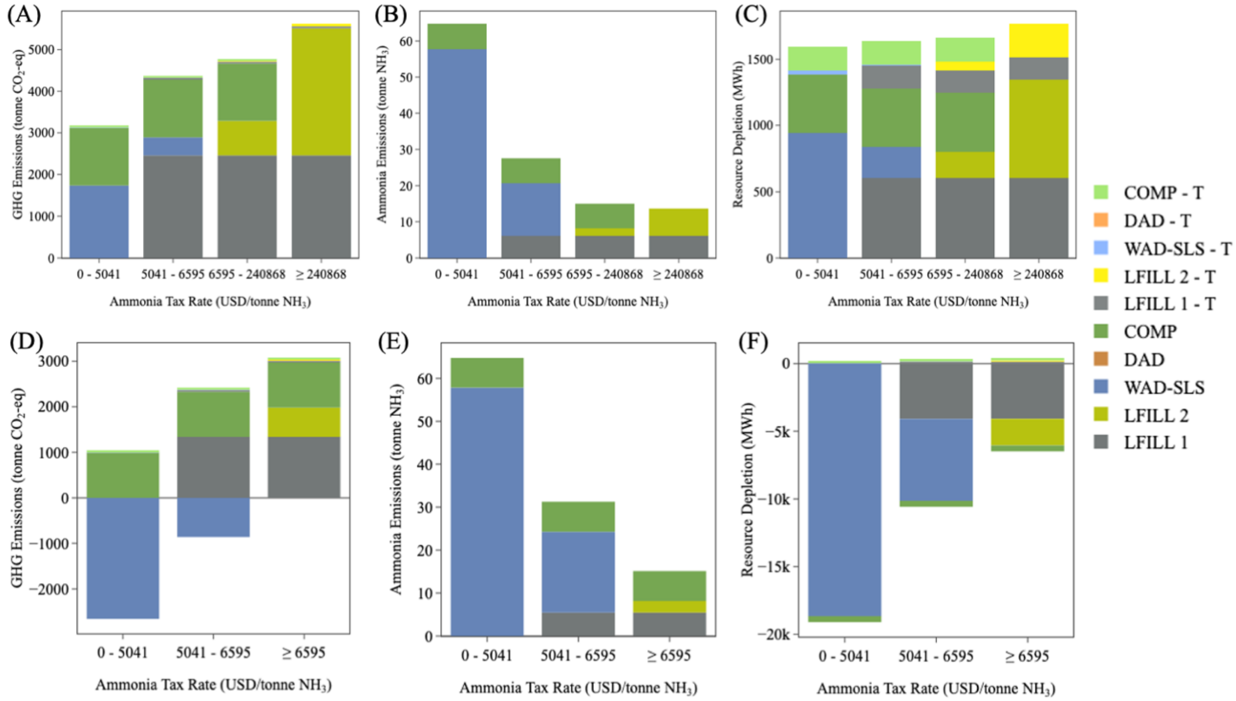

Figure 6. Impacts (GHG, NH_3, and RSD) for the base-case (A, B, C) and with energy offset credits (D, E, F) resulting from NH_3 taxation. Contributions from transportation indicated by "-T."

These emissions profiles demonstrate that (both with and without energy offset credits) an NH_3 tax incentivizes increases in GHG emissions (Figure 6A and D) and RSD (Figure 6C and F). Increases in GHG emissions are substantial, more than counteracting the effect of the offset credit (Figure 6D). Increases in RSD in the presence of energy credits (Figure 6F) reduce the amount of credit, but still maintain a net negative value.

Conclusions

A coordinated market approach was used to investigate the inherent market value of LCA metrics (GHGs, NH_3, and RSD) in a MOW supply chain that models the infrastructure of Milwaukee, WI. These inherent values are interpreted as taxes that policymakers might implement in service of a landfill diversion objective. Our results indicate that the economic landscape of the MOW supply chain in Milwaukee provides incentives that encourage residents to avoid using the city landfill.

We observe that implementing a GHG tax incentivizes behavior similar to that which is encouraged without any tax, suggesting that Milwaukee's economic landscape already has incentives for sustainable MOW management. This result points to challenges outside the scope of our analysis (operational complexities involving collection, etc.) that create incentives for continued landfill use, which is observed in the city. Additional reductions in GHG emissions could be gained through the implementation of a tax, but the value required to effect change would make Milwaukee the location of the highest carbon tax in the world. The effects of GHG and NH_3 taxes are antagonistic; reducing NH_3 emissions and GHGs simultaneously is thus a challenge.

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