PNNL-35449

W-SMART Phase-I Pathway Analysis

Case Study: City of Boston, MA

September 2023

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With support from Groundwork Data
Michael Walsh
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U.S. DEPARTMENT OF ENERGY

Prepared for the U.S. Department of Energy
under Contract DE-AC05-76RL01830

PACIFIC NORTHWEST NATIONAL LABORATORY
operated by
BATTELLE
for the
UNITED STATES DEPARTMENT OF ENERGY
under Contract DE-AC05-76RL01830

Printed in the United States of America

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Pacific Northwest National Laboratory
Richland, Washington 99354

Executive Summary

The purpose of this study is to synthesize stakeholder and research learnings to date by exercising PNNL's *Waste - Sustainability Monitoring of Alternative Reuse Options over Time* (W-SMART) sustainability protocol for the Greater Boston region. This Phase-I report serves as a basis for future discussion and final project work to characterize the costs, risks, impacts, tradeoffs, and highest uses for major waste streams in the Boston region.

W-SMART is a flexible, data-driven Waste-to-X (W2X) trade-offs model that can be used to (1) estimate the maximum potential of a single technology pathway; (2) compete multiple technologies with the same or different or overlapping feedstocks; (3) compare the impacts of alternative siting locations, including a mix of existing and proposed sites and/or different numbers of sites per technology; (4) assess impacts of variable feedstock quantity, quality, delivery costs, and market prices; and (5) evaluate different waste diversion strategies from point-of-generation to point-of-collection, transfer, or disposal to increase economies of scale in sorting and transport (i.e., fewer trips with bigger trucks).

This analysis differs from previous project work by (1) incorporating results of a newly completed detailed resource assessment for the Greater Boston area; (2) providing a head-to-head pathway comparison without any policy supports (e.g., carbon or energy credits); and (3) focusing on locally relevant critical waste streams and reuse strategies, by assessing the cost-effectiveness of two complimentary pathways, including (a) incineration of municipal solid waste (MSW) at existing incineration sites to produce baseload electricity, and (b) the conversion of blended municipal wastewater solids (i.e., sludge) and non-residential food waste to produce liquid transportation biofuels at a proposed hydrothermal liquefaction facility in Quincy, MA. The performance of each pathway is also compared to assumed business-as-usual waste (BAU) management practices as a baseline.

Key findings from this Phase-I investigation include:

- A total of **3.6 million dry metric t/y** (9,750 dry t/d) of MSW (86%), sludge (8%), and non-residential food waste (6%) solids are analyzed within the Greater Boston region. Using business-as-usual (BAU) practices, these wastes **cost $13.5 billion to manage**, as measured by 30-year net present value (NPV).

- W2X pathways, including HTL and incineration, can cost-effectively **utilize 100% of wastes** to produce **48 million gal/y of biocrude** and **3 million MWh/y of electricity**, and achieve a collective **NPV of $11.2 billion**, which represents a *reduction* in BAU waste management costs.

- Despite achieving $11.2 billion in total cost reduction, the W2X pathways could not generate enough revenue to cover the entire $13.5 billion in BAU waste management costs, resulting in a **waste system NPV of -$2.3 billion**. In other words, W2X could not simultaneously make waste producers <u>and</u> waste processors profitable, which we note does not happen presently.

- With better economies of scale and/or modest price supports, **generating a positive NPV for the entire waste system is possible with W2X, which is unprecedented**.

In Phase-I, we demonstrate the ability to exercise the W-SMART trade-off model for the Boston region using localized data. In Phase-II we will (1) add model support for anaerobic digestion and gasification technology pathways, and (2) integrate sustainability impact calculations based on modeled plant scales and locations. Sustainability accounting will highlight the tradeoffs between potentially competing waste management goals, such as reducing cost, meeting GHG targets, or maximizing a specific energy service.

Acknowledgments

We acknowledge the generous support in time and effort by our Boston regional partners.

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Waste Management
WBUR
Zoo New England
Union of Concerned Scientists

Acronyms and Abbreviations

AD	Anaerobic Digestion
ADWF	Average dry weather flow, a measure of WRRF influent flow
ATRI	American Transportation Research Institute
BAU	Business-as-Usual
BPF	Biosolids Processing Facility
C&D	Construction & Demolition (waste)
CSA	Combined Statistical Area
DITP	Deer Island Treatment Plant, Boston's largest WRRF
FW	Food Waste
HTL	Hydrothermal Liquefaction
IIC-FW	Industrial, Institutional, and Commercial Food Waste
MADEP	Massachusetts Department of Environmental Protection
MM gal/d	Million gallons per day
MSW	Municipal Solid Waste
MWRA	Massachusetts Water Resources Authority
NAICS	North American Industry Classification System
NEFCO	New England Fertilizer Company
NPV	Net Present Value
OFMSW	Organic Fraction of Municipal Solid Waste
PNNL	Pacific Northwest National Laboratory
SAM	Sustainable Adaptive Management
Sludge	Untreated wastewater solids
SSO	Source separated organics
TCI	Total Capital Investment
W2X	Waste-to-*X* (pathway)
WRRF	Water Resource Recovery Facility
W-SMART	Waste–Sustainability Monitoring of Alternative Reuse Options over Time

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1.0 Background

New England is facing historical and emerging sustainability challenges at the intersection of waste and energy that are at odds with, but also opportunities to address ambitious waste and climate goals.

Historically, the region has favored incineration as a primary waste management strategy by burning municipal solid waste to produce electricity. Massachusetts hosts five active incinerators that treat a combined 3.2 million wet short tons of solid waste annually (MADEP, 2023b) to generate 1.7 million MWh of electricity per year (EIA, 2022), which represents 3% of MA annual energy consumption of 55.3 TWh (EIA, 2015). Some of these combustion plants have been operating for more than four decades. Increasing regulation on organic waste treatment has prompted some increase in anaerobic digestion (AD) to support combined heat and power (CHP) and renewable natural gas (RNG) production, but there are many barriers to achieving cost-effectiveness. Moratoria on new or expanded landfills and accelerated closures of existing landfills require waste to be exported out-of-state for disposal as an indefinite "temporary" solution, substantially increasing disposal costs. Large-scale composting opportunities are limited in urban areas due to land availability and lack of off-takers. PFAS concerns have increased pressure to implement strategies that effectively destroy PFAS, potentially at the expense of other goals and at great expense to municipalities. Contamination and excess nutrient concerns limit the land application of treated sludges and manures. A sudden moratorium on the land application would result in an immediate solids disposal crisis for cities nationwide. The likely outcome will be an increase in exported waste to Canada and surrounding states for disposal (MADEP, 2019), thereby also increase waste related GHG emissions.

However, considering the increasing costs of regulation and decarbonization efforts, existing disposal pathways will become untenable. Five of the six New England states (CT, MA, ME, RI, VT) have legally binding net zero emissions targets by 2050 (NRRI, 2023). Regional electricity generation is rapidly shifting from coal and nuclear to natural gas and renewables, with approximately 20,000 MW of renewable solar and wind generation capacity planned by 2030 (ISO-NE, 2021). Rising natural gas prices and a shift to zero-marginal cost resources further decrease the economic performance of inflexible "always-on" incinerators. This requires incinerators to increase tipping fees to remain economically viable, but cheap landfilling limits this mitigation strategy, causing incinerators to go out of business.

Transformational waste management technologies will eventually become commercialized with substantial cost and performance implications for waste management. For example, continuous hydrothermal liquefaction (HTL) could convert wet organic wastes to biofuels while increasing carbon, energy, nutrient, and metals recovery and reducing residuals and effluent contaminant loadings. Modern gasification processes also offer sustainable pathways to biofuels and industrial precursor chemicals.

However, the siting of new clean energy and waste management infrastructure in the region will continue to be challenging. In the past several years, significant local opposition has emerged against new waste management technologies, fossil energy infrastructure, solar infrastructure, and electrical substations, even when new facilities have the potential to alleviate existing harms. This resistance is based on past harms from facilities located in or adjacent to communities that meet state and federal environmental justice criteria.

New tools are required that both facilitate the careful examination of the long-term economic, environmental, energy, and social tradeoffs of conventional and emerging technologies and characterize the potential shift of benefits and burdens associated with waste management alternatives.

2.0 Introduction

The Pacific Northwest National Laboratory (PNNL) is actively developing a "Sustainable Adaptive Management" (SAM) protocol to formalize and systematize sustainability measurement, tracking and tradeoff analysis for bioresource conversion strategies. This case study utilizes an instance of the draft SAM implementation for the waste management context called "Waste–Sustainability Monitoring of Alternative Reuse Options over Time" (W-SMART).

W-SMART is applied to evaluate the potential long-term sustainability (economic, environmental, and social) impacts of myriad organic waste reuse strategies. The concept of "adaptive (waste) reuse over time" is important because the "highest" reuse value of individual or blended waste streams is likely to change with priorities, market conditions, and technology. Therefore, continuous adaptive management approaches are required to maintain alignment between optimal waste management strategies and sustainability outcomes.

Figure 1 presents the primary components of W-SMART, including (1) a waste management technology cost and performance library; (2) a library of qualitative and quantitative sustainability metrics; (3) an inventory of waste producers and feedstock properties; and (4) a "Waste-to-X" (W2X) Pathways model.

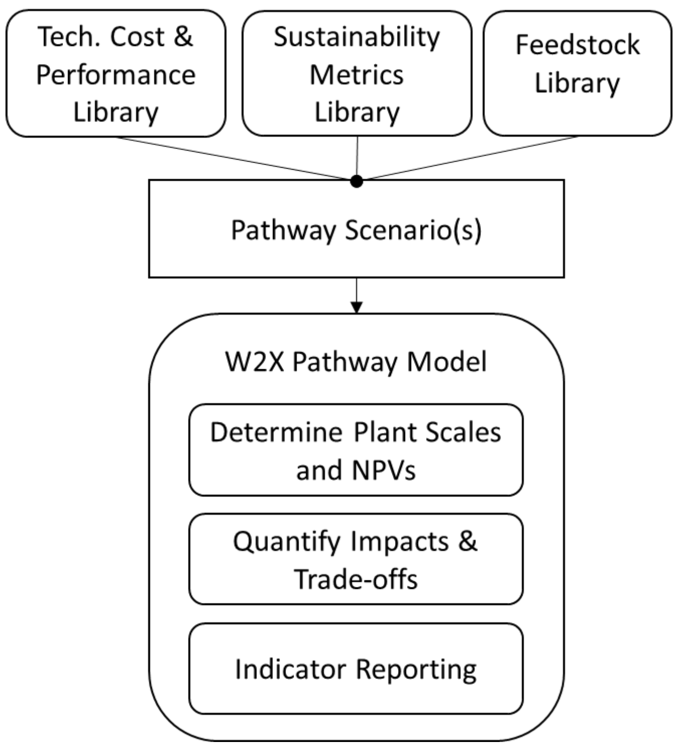


Figure 1. W-SMART system schematic diagram

The basic analysis workflow involves working with stakeholders to define a set of locally relevant waste reuse alternatives. The waste reuse strategies are represented in the model as "pathways," which are a specific combination of feedstock(s), conversion technology, and final energy endpoint (e.g., power, biofuels). Varying any component of a pathway definition constitutes a new pathway within the model. The pathways are then evaluated by the W2X Pathways model to (1) determine cost-effective plant scales and the outputs (energy, residuals, etc.) of each W2X processor; (2) quantify impacts and compare tradeoffs of one or more pathways; and (3) report impacts against a standard set of economic, environmental, and social indicators.

Because the W2X Pathways model is data-driven, it offers a high degree of flexibility to model many alternative conceptual site models. For example, the model may be used to (1) estimate the maximum potential of a single technology pathway; (2) compete multiple technologies with the same or different or overlapping feedstocks; (3) compare the impacts of alternative siting locations, including a mix of existing and proposed sites and/or different numbers of sites per technology; (4) assess impacts of variable feedstock quantity, quality, delivery costs, and market prices; and (5) evaluate different waste diversion strategies from point-of-generation to point-of-collection, transfer, or disposal to increase economies of scale in sorting and transport (i.e., fewer trips with bigger trucks).

The W2X Pathways model may be exercised in two different "modes," depending on whether the intent is for multiple technologies to compete for the same resources. In the "competitive" mode (default), various technologies are placed at the same or different locations to compete (economically) for feedstocks in same scenario. In the competitive mode, it is possible a pathway will not be realized in the model if it is not economically feasible or insufficiently competitive with other options. In the "comparative" mode, different technologies are placed at the same location but are run in separate scenarios to estimate the maximum potential of each pathway. In either case, the model will optimize similarly, regardless of the technology deployment configuration.

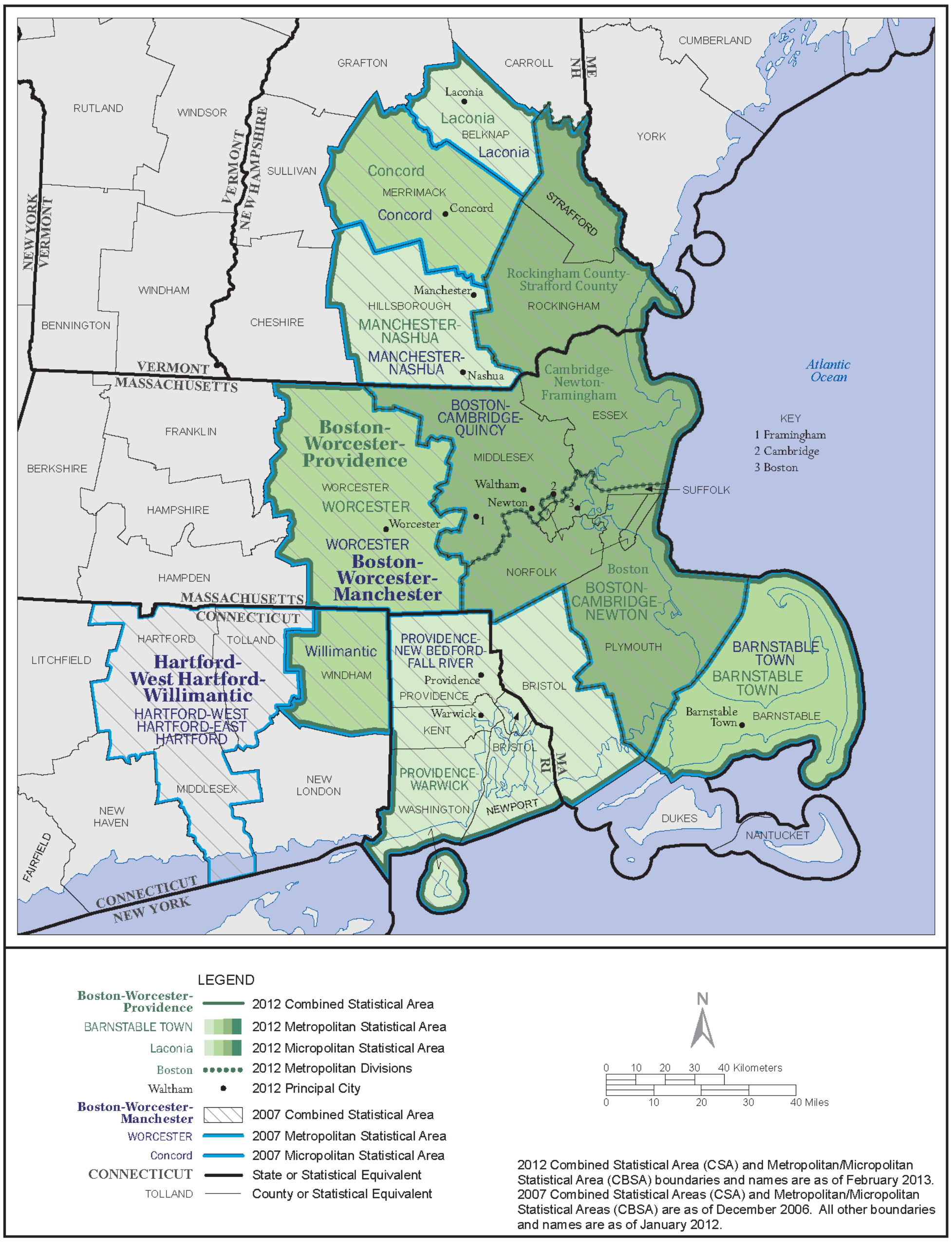
In competitive scenarios, the W2X Pathway model employs techno-economic optimization techniques to partition available waste resources among competing technologies for conversion to various energy endpoints (electricity, biofuels, biogas, etc.). The optimization process seeks the "best" overall waste utilization strategy, calculated over the specified model time horizon, by maximizing the net present value (NPV) of the entire waste management system (i.e., the NPV accounts for both waste producer waste-related costs and subsequent waste processor costs and revenues). Based on the proposed optimal mix of technology types, scales, and feedstocks, we then calculate various economic, social, and environmental impacts as the basis for performing tradeoff analysis to understand the advantages and disadvantages of each waste strategy from a sustainability perspective.

In this study, we consider the potential impacts of managing three different feedstocks, including wastewater solids ("sludge"), non-residential food waste (FW), and municipal solid waste (MSW), using three alternative management strategies, including business-as-usual (BAU) as a "baseline"; incineration of MSW for baseload electricity; and the conversion of organic wet waste (WW) (sludge + FW) to biofuels via hydrothermal liquefaction (HTL). Conversion facility locations and types are fixed in advance based on existing infrastructure and stakeholder knowledge. Waste sources (location, type, and dry mass) are also known in advance. Facility scales, waste producer-to-processor routings, and feedstock prices are determined through optimization, which forms the basis for estimating sustainability impacts.

The intended outcome of this analysis is to guide further stakeholder discussions and refine the scenarios to better meet the informational needs of the stakeholders before finalizing the model and analysis plan. Although the current scenarios (pathway definitions) are based on stakeholder input and the model implements actual local bioresource data, the results of this analysis should be considered preliminary because the Pathway model and indicators library are still under development.

3.0 Resource Assessment

This study considers (1) untreated, wastewater sludge solids, (2) non-residential food waste, and (3) municipal solid waste inventoried within the "Greater Boston" area. To best represent the social and economic interactions and waste patterns across the region, the analysis boundary for this study is represented by the 2020 U.S. Census Bureau 1:500,000 cartographic boundary for the Boston-Worcester-Providence combined statistical area ("Boston CSA") (U.S. Census, 2020), which covers >10,000 square miles and includes a total of 19 different counties covering portions of four states, including Massachusetts (MA), Rhode Island (RI), New Hampshire (NH), and Connecticut (CT), as presented in Figure 2. County-level demographic data for each county are presented in Appendix A.



U.S. DEPARTMENT OF COMMERCE Economics and Statistics Administration U.S. Census Bureau 2012 Economic Census

Figure 2. Boston-Worcester-Providence CSA (US Census, 2022)

Table 1 summarizes feedstocks occurring in the Boston CSA. All feedstock data are normalized to a dry mass basis. In total, 3.62 million dry metric tons per year of target feedstocks occur within the Boston CSA. For analysis purposes, waste producers are filtered to exclude sludge and MSW sources producing <1 dry metric t/d and FW sites producing <0.5 wet short tons per week, which is consistent with the MA commercial food waste disposal ban (310 CMR).

The remaining 3.56 million dry metric t/y (9,751 dry metric t/d) of analyzed feedstocks account for 100%, 97%, and 82% of total available MSW, sludge, and non-residential FW mass, respectively. Analyzed feedstocks are composed of 86% MSW, 8% wastewater solids, and 6% non-residential FW. Sludge and FW are similar in total mass, but sludge is more concentrated. Regional MSW feedstocks are 10 times greater than sludge but collection is more dispersed.

Table 1. Target feedstocks

Property	Unit	Sludge	FW	MSW	TOTAL
available mass	dry metric t/y	283,786	270,651	3,062,659	3,617,096
analyzed mass	dry metric t/y	274,655	223,012	3,061,202	3,558,868
analyzed mass	dry metric t/d	752	612	8,387	9,751
analyzed sources	n	71	10,808	1,913	12,792

3.1 Wastewater solids

In this study, municipal wastewater solids (“sludge”) refers to dewatered mixed solids removed from primary and secondary treatment prior to any treatment (e.g., AD or lime stabilization), which may affect solids mass and organic loading. Recoverable and disposed sludge solids are characterized by Seiple et al., 2020a and Seiple et al., 2020b for >15000 water resource recovery facilities (WRRF) in the United States. Wastewater solids estimates are already reported on a dry mass basis.

Table 2 summarizes total available and analyzed sludge resources within the Boston CSA, with comparison to the Deer Island Treatment Plant (DITP), the largest waste producer in the region. Figure 3 illustrates the spatial distribution of treatment capacity for analyzed WRRFs listed in Appendix B, in relation to the Massachusetts Water Resources Authority (MWRA) service area boundary (MWRA, 2005). Figure 4 illustrates the contribution of each WRRF to total average daily flow.

In total, 145 WRRFs occur within the Boston CSA and treat an average of 846 million gallons per day (MM gal/d) of wastewater producing approximately 283,786 dry metric t/d of wastewater solids. The WRRFs range in treatment capacity from 0.01 to 310 MM gal/d of average dry weather flow (ADWF). For analysis, WRRFs producing <1 dry metric t/d of recoverable solids are excluded from further analysis. Analyzed WRRFs represent 96% of total flow and 97% of solids.

According to reporting data, the DITP receives a long-term (29-year) average daily flow of 353 MM gal/d, and a 2020 ADWF of 310 MM gal/d, with a peak capacity of 1,270 MM gal/d. About 55 to 65 percent of the annual flow treated at DITP is sanitary flow (approximately 170 mgd) and the remaining 35 to 45 percent is comprised of stormwater from combined sewers and inflow and infiltration (I/I) that enters the regional sewer system (MWRA, 2018). The DITP accounts for 37% of total influent flow and recoverable solids in the Boston CSA. Estimates of recoverable solids for DITP exceed the reported biosolids production rate of 100 dry metric t/d because sludge feedstock is estimated prior to the application of any treatment (AD), which can reduce sludge mass by 40–50%.

Table 2. WRRF count, flow, and recoverable "untreated" solids

	Count	Flow (MM gal/d)	Flow (%)	Recoverable (dry metric t/y)	Recoverable (dry metric t/d)
DITP	1	310	37	107,785	295
Top 10 WRRFs	10	576	68	198,568	544
Boston CSA (Analyzed)	71	815	96	274,655	752
Boston CSA (Available)	145	846	100	283,786	777

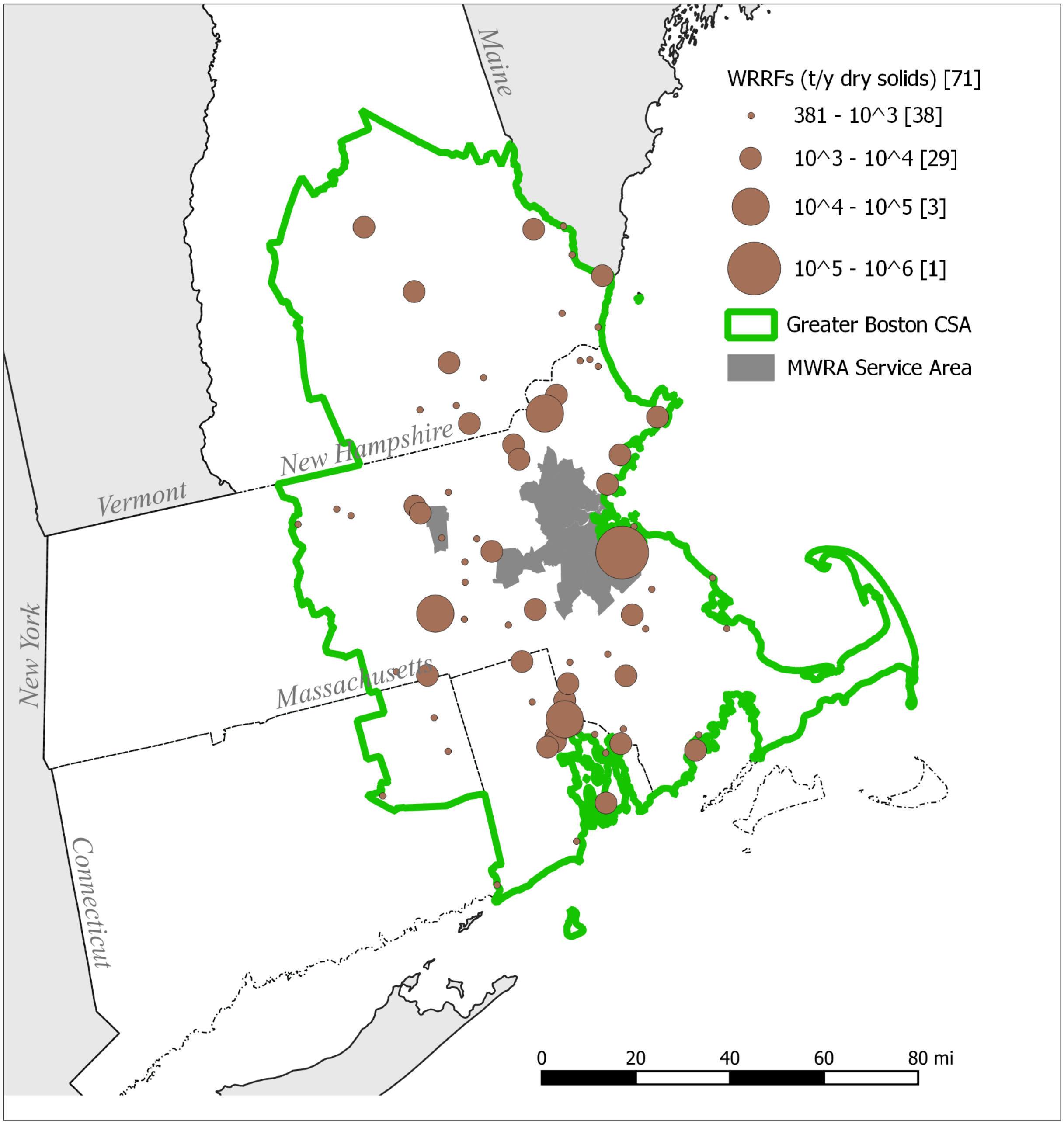


Figure 3. Spatial distribution and flow of WRRFs in the Boston CSA

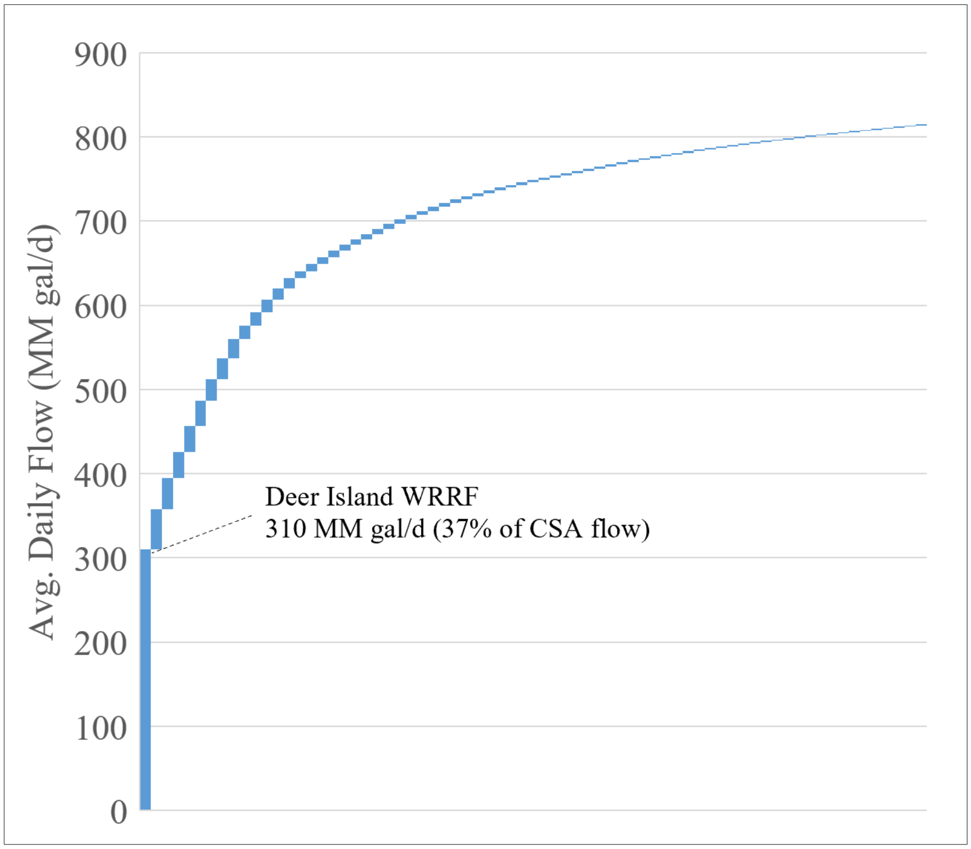


Figure 4. Waterfall diagram of WRRF flow in the Boston CSA

3.2 Municipal Solid Waste

Bulk solid waste is composed of municipal solid waste (MSW), construction and demolition (C&D) waste, sludges, contaminated soils, and other wastes. For purposes of this study, we focus on the MSW component of solid waste. In terms of W2X modeling, MSW diversion may be modeled at the point of generation, collection, transfer, disposal, or a combination thereof. The key disadvantage of using point-of-disposal data for modeling W2X potential is that it excludes waste exported out of the study area, which may be significant in the northeastern US.

Solid waste point-of-generation data are difficult to find, as the regulatory reporting focuses on monitoring disposal endpoints. Most estimates of MSW generation are often based on per capita factors, which usually only represent residential waste, or on incomplete, non-normalized survey data. For example, the Massachusetts Department of Environmental Protection (MADEP) publishes annual MSW and recycling surveys for cities and towns in Massachusetts (MADEP, 2021). The latest (2021) survey demonstrates several data analysis challenges, including (1) the precise spatial extent of the named jurisdictions is not known; (2) the response rate was 82%, with 64 of 352 listed towns failing to report; (3) compositional breakdowns by waste sector (i.e., residential, municipal, business) and type are not reliable and are sometimes contradictory; and (4) commercial and industrial sectors are not uniformly represented. For comparison, the 2021 MA cities and town MSW survey reported a total of 1.52 million wet tons of waste, accounting for only 26% of total MA state-wide annual solid waste (including exports) and 41% of in-state disposed waste. Furthermore, the analysis boundary of this study extends to other states not covered by MSW survey data.

In the absence of comprehensive waste generation data for the Boston CSA, which intersects 19 county and four state jurisdictions, MSW estimates are modeled by downscaling state-wide total MSW to census tract centroids based on population. Census tract boundaries are selected because their spatial footprint is routinely adjusted to maintain population within a specific range (1200–8000) (US. Census, 2022), which generally corresponds to the size of a typical garbage collection route of 800-850 households (SC, 2019). Therefore, the modeled estimates are designed to represent the diversion of collection trucks directly to a W2X facility instead of the current disposal endpoint (i.e., landfills). This approach provides contiguous spatial coverage, reasonable waste distribution, and enables the use of the most recent reporting data. To avoid double counting, imported waste totals are excluded from each state's balance prior to downscaling.

Table 3 summarizes state-wide estimates of total available and analyzed in-state, exported, and imported MSW for Massachusetts, Connecticut, Rhode Island, and New Hampshire. These data are developed based on public reporting data, which vary in quality by state. State-wide MSW reporting data are converted from wet mass (compacted wet short tons per year) to a dry mass basis assuming 60% solids concentration. Dry mass estimates of in-state disposed waste plus exported waste are then partitioned to 2020 U.S. Census Bureau 1:500,000 census tract boundary centroids (U.S. Census, 2020) based on the fraction of 2020 state population occurring in each tract. For analysis, tract-level MSW estimates are filtered to exclude tracts outside the Boston CSA boundary and those producing <1 dry metric t/d of waste. Analyzed tracts account for 99.95% of the total available MSW mass within the CSA. Figure 5 presents the spatial distribution of downscaled MSW by census tract.

Table 3. Total available and analyzed MSW by state

	MA	CT	RI	NH	TOTAL
MSW Data Year	2020	2018	2018	2020	
Pop. (Apr. 1, 2020)	7,029,917	3,605,944	1,097,379	1,377,529	13,110,769
State-wide MSW (compact wet short t/y)					
In-State Disposed	3,590,000	1,900,494	748,258	625,774	6,864,526
Exported	1,040,000	397,903	167,149	113,185	1,718,237
Imported	240,000	23,201	0	548,300	811,501
(In-State + Export)	4,630,000	2,298,397	915,407	738,959	8,582,763
Analyzed MSW (tracts in CSA producing ≥1 dry metric t/d)					
(In-State + Export)	2,206,534	40,389	497,590	316,690	3,061,202
(In-State + Export)	6,045	111	1,363	868	8,387
% State Population	88	3	100	79	64
MSW Reference	MADEP, 2022	NEWMOA, 2021	NEWMOA, 2021	NHDES, 2022	

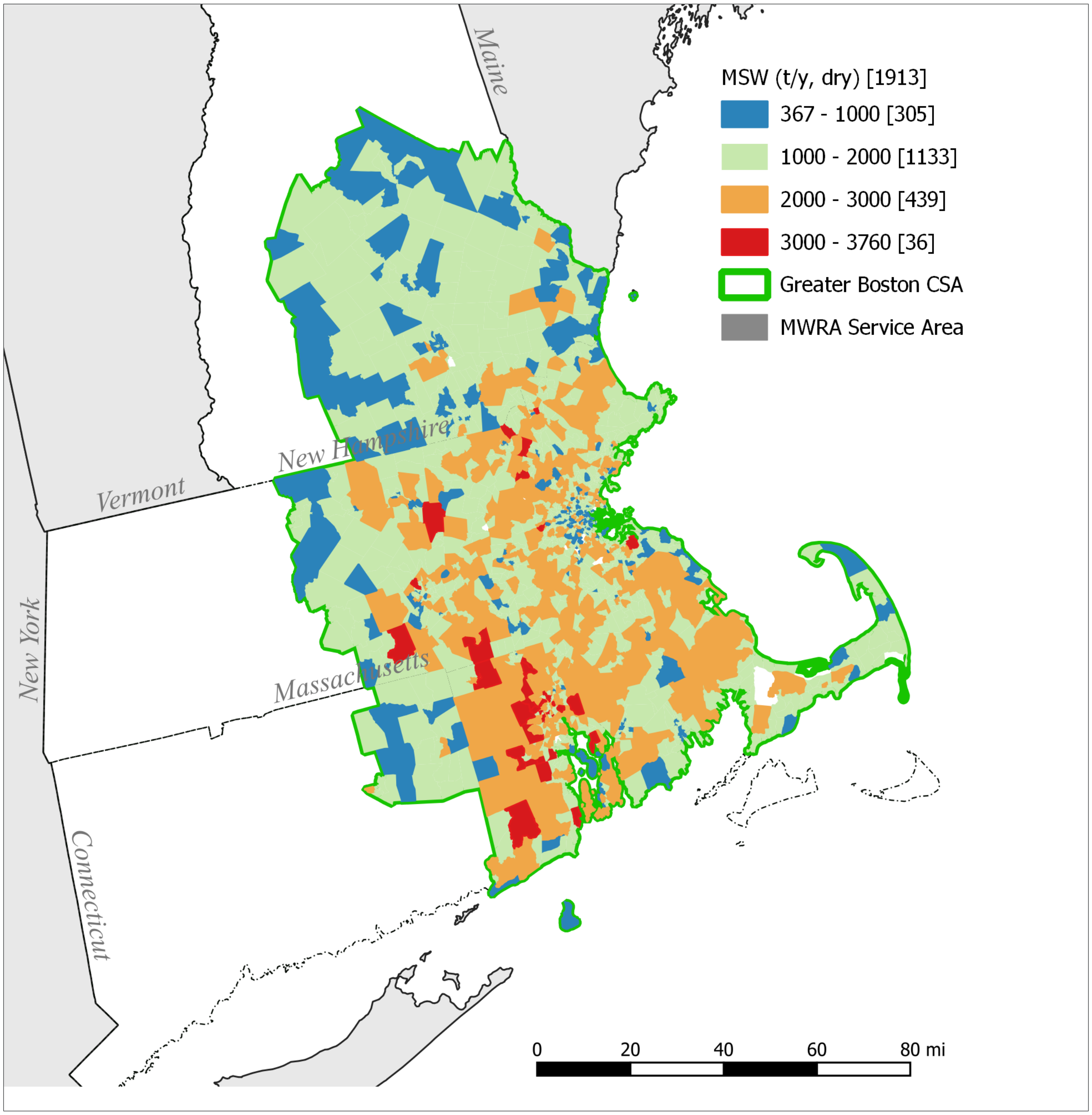


Figure 5. Spatial distribution of downscaled MSW by census tract

3.3 Non-residential Food Waste

Food waste may be classified by source as residential or non-residential. Residential waste is typically co-mingled with MSW, requiring separation. In comparison, non-residential food waste, which includes industrial, institutional, and commercial food waste (IIC-FW) waste sources, is generally less contaminated and easier to access in large quantities.

Like MSW, IIC-FW may be represented at the point of generation, collection, transfer, disposal, or some combination. The condition of IIC-FW material at each phase is dependent upon local waste management regulations and enforcement. In general, FW is handled as a mixture of source-separated organics (SSO) and the organic fraction of municipal solid waste (OFMSW). States with organics bans in place typically have a higher proportion of SSO, however, some policies may only cover yard waste and not FW, or only IIC-FW. For example, the state of Massachusetts established a commercial FW disposal ban for sources ≥0.5 wet short ton per week, which diverted 300,000 wet short tons of FW in 2020, however, FW still represents 20% to 30% of total disposed MSW (MADEP, 2022; MADEP, 2023b).

In this study we model the conversion of as-generated IIC-FW. The EPA Excess Food Opportunities Map (version 2.1) (EPA, 2020a; EPA, 2020b) characterizes establishment-level estimates (low and high) of total annual generated non-residential, post-harvest food waste for 1.2 million establishments in 76 different North American Industry Classification System (NAICS) categories, which are grouped into seven food waste sectors (Correctional, Educational, Manufacturers & Processors, Wholesale & Retail, Healthcare, Hospitality, and Restaurants & Food Services).

The dataset is prepared by applying sector-specific literature factors for waste generation (e.g., by pounds per revenue per year or similar using the number of students, beds, inmates, etc.) and composition (lipid, protein, simple or complex carbs, mixed, glycerin) to appropriate facility level attributes (revenue, employees, number of beds, number of students, etc.). Unlike disposal data, FW generation estimates do not exclude waste diverted for reuse (e.g., composting), which is better for modeling total W2X potential.

The EPA dataset is helpful, but may not reflect actual site conditions (i.e., food waste production) for a given location. Reasons for potential discrepancies relate to errors in the underlying business database or estimation errors due to the methodology and simplifying assumptions. Examples of business database issues include (1) a change in business status (i.e., permanently closed); (2) incorrectly assigning food waste production data to corporate office location; (3) grouping multiple store locations into a single business record; (4) duplicate records; (5) inaccurate business data used for scaling (e.g., number of employees, revenue, etc.); and (6) inaccurate location information. However, the EPA-modeled dataset still provides valuable information regarding which types of food waste producers are likely active in the region and the relative quantity of waste produced by each food sector.

The EPA data are first geocoded (i.e., converted from an address to latitude and longitude coordinates). In total, 96.4% of IIC-FW sites (99.2% of total IIC-FW mass) assigned to counties occurring within the Boston CSA are successfully geocoded using a combination of the US Census Geocoder (Census, 2023) and manual searches online. Geolocated FW producers are then filtered to exclude sites producing <0.5 wet short tons per week, consistent with the current implementation of the Massachusetts commercial organic material waste ban (MADEP, 2022). Lastly, a spatial intersect is performed to remove geocoded sites positioned outside the Boston CSA boundary, as some addresses have incorrect county assignments. The remaining (analyzed) IIC-FW estimates are prepared by taking the average of the EPA's low- and high-establishment level food excess values. Average values are then converted from wet short tons to dry metric dry tons per year (dry metric t/y), assuming a solids concentration of 30% for all IIC-FW types. In summary, analyzed IIC-FW includes all EPA establishments within the Boston CSA that can be geolocated and produce ≥0.5 wet short tons per week.

Table 4 presents the total available and analyzed IIC-FW within the Boston CSA by sector. Figure 6 illustrates the spatial distribution of IIC-FW. The 223,012 dry metric t/y of analyzed IIC-FW accounts for 82% of total available non-residential food waste. IIC-FW producers are quite small compared to MSW and sludge producers, ranging in size from 0.02 to 4.3 dry metric t/d with an average size of 0.06 dry metric t/d. The IIC-FW profile of Greater Boston is consistent with the EPA national average by sector, with most IIC-FW establishments and associated wastes occurring within the restaurant and food service (56%), wholesale and retail (20%), educational (11%), and manufacturing and processing (5%) sectors, with minor inputs from other sectors.

Table 4. Total available and analyzed IIC-FW by sector

Sector	Available IIC-FW Sites (n)	Available IIC-FW Sites (%)	Available IIC-FW EPA Avg. (dry t/y)	Analyzed IIC-FW Sites (n)	Analyzed IIC-FW Mass (dry t/y)	Analyzed IIC-FW Mass (dry t/d)	Analyzed IIC-FW % Avail. Mass
Correctional	92	<1	1,291	28	1,143	3	89%
Educational	3,102	10	15,334	281	8,342	23	54%
Manf. & Proc.	1,498	5	28,843	317	26,223	72	91%
Wholesale & Retail	7,820	25	150,322	7,728	148,956	409	99%
Healthcare	161	<1	2,545	110	2,412	7	95%
Hospitality	1,560	5	7,514	249	5,423	15	72%
Restaurant Service	17442	55	64,802	2,095	30,514	84	47%
TOTAL	31,675	100	270,651	10,808	223,012	612	82%

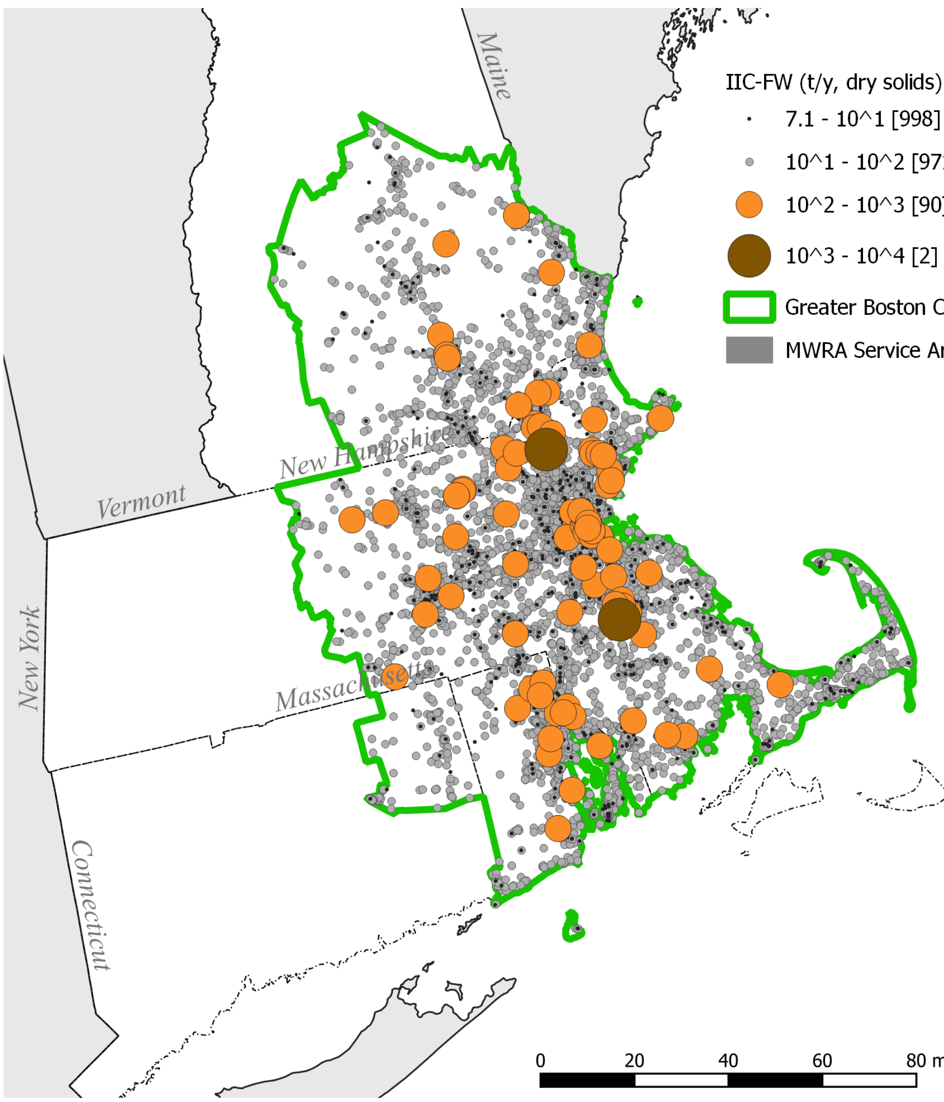


Figure 6. Spatial distribution of IIC-FW within the Boston CSA

4.0 Techno-economic Assessment

To facilitate discussion with stakeholders, we consider the cost-effectiveness of two complimentary waste conversion pathways that could manage the most critical regional waste streams. Table 5 summarizes the modeled pathways, which represent the management of three different feedstocks (i.e., sludge, IIC-FW, and MSW) using business-as-usual (BAU) "baseline" practices (P1), with comparison to two W2X alternatives including (P2) 100% incineration of MSW for electricity production and (P3) the conversion of blended sludge and IIC-FW to biofuels via HTL. In this study, carbon and energy credits are excluded to prevent policy from dominating the economic comparison.

This is accomplished by applying the W2X Pathways model, which uses techno-economic optimization techniques to partition available waste resources among competing waste treatment technologies for conversion to various energy endpoints (i.e., electricity, biofuels, biogas, etc.). The optimizer seeks the "best" overall waste utilization strategy, calculated over the specified model time horizon, by maximizing the net present value (NPV) of the entire waste management system (i.e., the NPV accounts for both waste producer waste-related costs and subsequent waste processor costs and revenues). Based on the proposed optimal mix of technology types, scales, and feedstocks, we then calculate various economic, social, and environmental impacts as the basis for performing tradeoff analysis to understand the advantages and disadvantages of each waste strategy from a sustainability perspective.

Table 5. Summary of modeled pathways

Pathway	Description	Feedstock(s)	Technology	Energy Service
P1	Baseline	All	BAU	BAU
P2	Incineration for power	MSW	Incineration	electricity
P3	HTL for biofuels	Sludge, IIC-FW	Hydrothermal Liquefaction	Renewable diesel

The BAU case (P1) is not explicitly modeled as a pathway. Rather, the BAU waste management price is multiplied by the total waste mass to estimate the total baseline waste management liability as a basis for assessing the relative cost effectiveness of alternative strategies.

4.1 Incineration for Electricity

Mass burn incineration of MSW is a commercially available, high temperature (1000–1800 °C), low pressure (0.1 MPa; 14.7 psi), rapid (1—2 sec) thermal combustion process commonly used in the Northeast to generate steam that drives a turbine to produce heat and electricity (Giraud et al., 2021). As presented in Figure 7, trash is introduced into the furnace to be burned, waste gases are scrubbed and treated for toxins and heavy metals, then fly (exhaust) and bottom ash are collected, mixed, and processed to recover metals prior to landfilling. Incineration can effectively reduce waste volumes by >90%, but community health concerns remain (NRC, 2000).

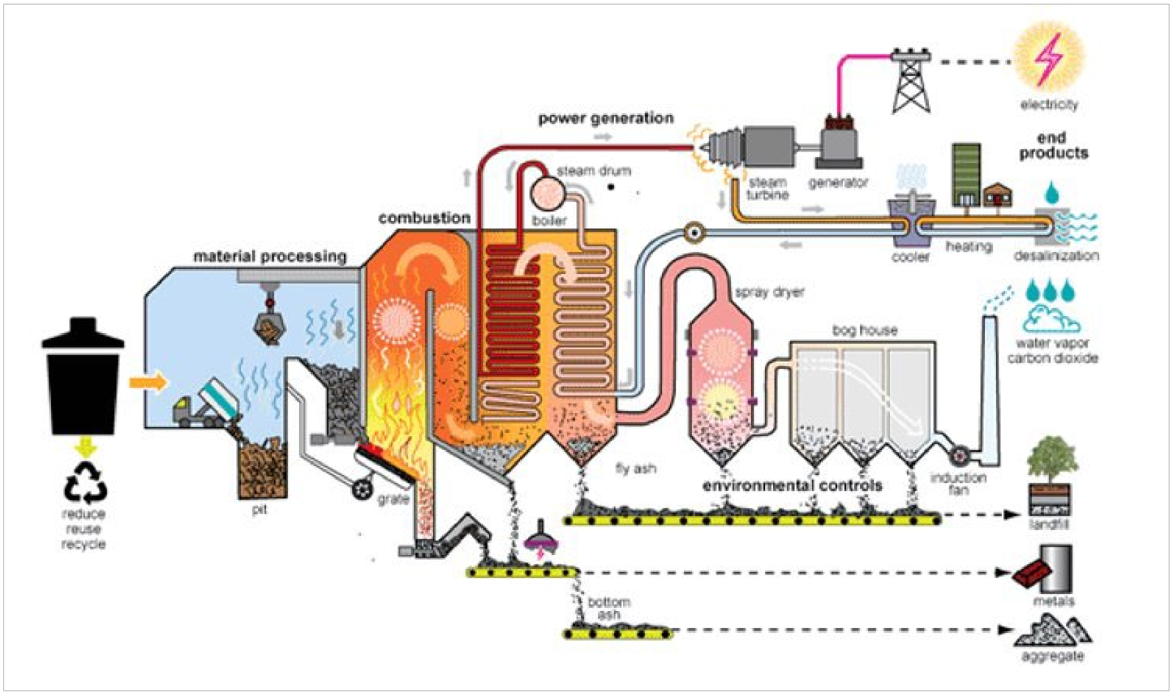


Figure 7. Schematic diagram of a mass burn incinerator (EIA, 2022)

4.2 Hydrothermal Liquefaction for Biofuels

Hydrothermal liquefaction (HTL) is an emerging, fast (10-20 mins), moderate temperature (~350 °C), high pressure (20 MPa; 2900 psi) thermochemical process that can directly convert a wide range of wet (5–35 wt% solids) organic wastes into biocrude intermediate that can be upgraded to a range of liquid biofuels (Elliott et al., 2013; Grande et al., 2021). Figure 8 illustrates the basic HTL workflow, whereby blended wet wastes are mixed in a hot pressurized reactor vessel to produce upgradeable biocrude.

The HTL reactor uses high pressure to allow water to be heated above its critical point to exploit special properties of near-supercritical water that accelerate the breakdown of complex macromolecules into smaller, more stable molecules that can be phase-separated into (1) biocrude, (2) sterilized, low-odor solids for disposal, (3) a particle and pathogen-free aqueous "effluent", and (4) a gas composed primarily of CO_2 (Sandquist et al., 2019).

As a waste treatment technology, HTL offers several key advantages compared to conventional treatment options. These benefits include high carbon recovery rates (>70%); high waste mass reduction rates (>75%); high loading rates 100 times faster than AD; concentration of metals and phosphorus in solids for easier recovery; co-liquefaction opportunities; the replacement of biological processes; reduced solids disposal costs; greater energy efficiency; ability to handle wastes with high moisture content; production of low-oxygen biocrude compared to pyrolysis; and process heat recovery (Seiple et al., 2020; Sandquist et al., 2019).

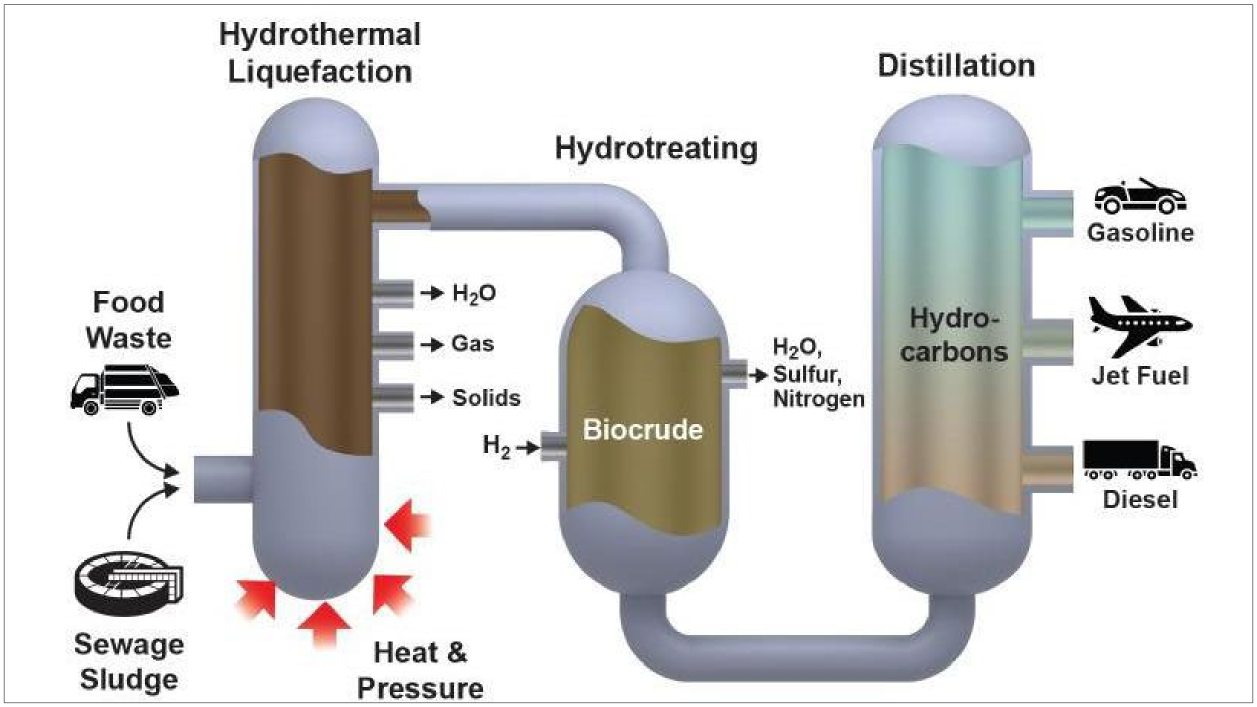


Figure 8. Schematic diagram of HTL, as modeled (Rodio, 2022)

4.3 W2X Processor Siting Locations

W2X processor facility locations are determined exogenously through either optimized siting analysis or expert local knowledge. Performing optimized siting analysis is beyond the scope of this study, so we assume that all technologies are located at active waste management sites in the region. Figure 9 presents the fixed locations for incineration and HTL used in this study.

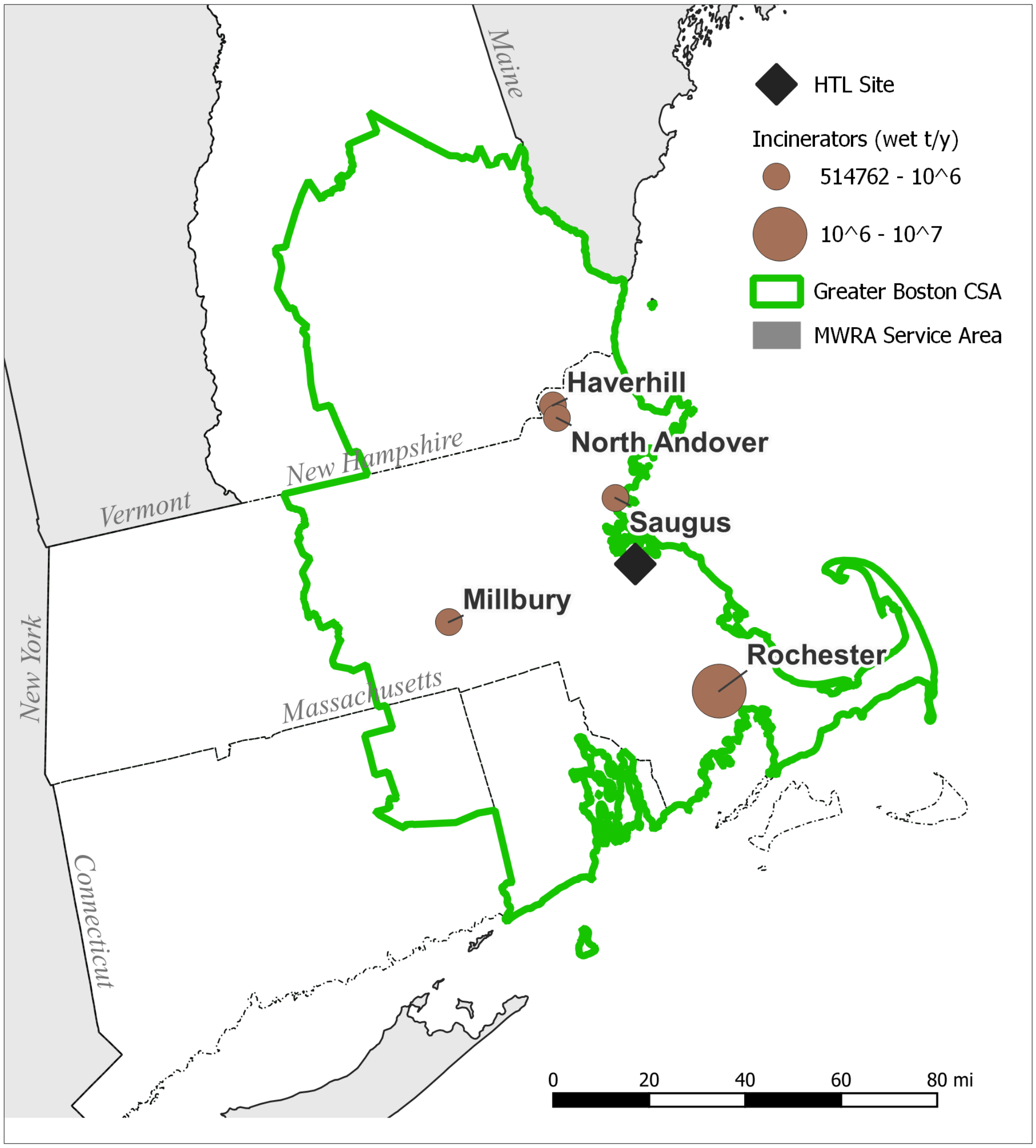


Figure 9. Waste processor locations

4.3.1 Incineration Sites

In total, 82% of MA in-state disposed MSW is treated with incineration (MADEP, 2022) to generate 1.7 million MWh/y of electricity (EIA, 2022). Table 6 lists the five active combustion facilities operating in the state of MA, all of which are located within the Boston CSA (MADEP, 2023b). In this analysis, we assume that the existing incinerators are expanded to treat 100% of available MSW to maximize electricity production. The optimization model allocates waste to maximize system-wide NPV, which may not the current capacity of each incinerator.

Table 6. Active incinerators in the state of MA

Facility	Type	Avg. Capacity (wet t/d)	2022 Capacity (wet t/y)	Nameplate Capacity (MW)	2022 Generation (MWh/y)	Capacity Factor (%)	Estimated Houses Powered
Haverhill	Mass Burn	1650	536,736	45	349,571	89	28,600
Millbury	Mass Burn	1500	514,762	48	318,641	76	29,250
North Andover	Mass Burn	1500	526,461	40	228,822	65	26,000
Rochester	RDF	3300	1,088,908	78	590271	56	50,700
Saugus	Mass Burn	1500	552,693	54	214,422	45	24,050
		9,450	**3,219,560**	**265**	**1,701,727**	**Avg. 72**	**158,600**

4.3.2 HTL Conversion Site

The MWRA sewage collection system serves 2.2 million people in 43 communities and covers over 500 square miles. Most of this waste is treated by the DITP, which is by far the largest producer of wet organic waste in the region. Therefore, it makes sense to locate a waste processing plant in proximity to DITP, as feedstock transport costs are a major determinant on the cost-effectiveness of WW conversion.

According to the 2018 MWRA Wastewater Master Plan (MWRA, 2018), biosolids from DITP are currently sent to a Biosolids Processing Facility (BPF), which is a drying and pelletizing facility in Quincy, MA operated by the New England Fertilizer Company (NEFCO) (NEFCO, 2023). Between 6 and 9 million gallons per week of liquid sludge (2–3% digested solids) are pumped from DITP to the BPF via two 14–inch, 7–mile long underground pipelines embedded within the “Inter-Island Tunnel”, and BPF centrate is returned to DITP using sewer conveyance pipelines to avoid local sewer regulations and fees (NEFCO, 2011). Finished pellets are shipped by rail or truck for beneficial reuse. During the growing season, most pellets are delivered by truck to local customers, while the rest of the year they are sent by rail to moderate climate regions.

Exploiting the existing arrangement of feedstock storage, supply and effluent treatment offers substantial cost and transportation savings for any potential W2X processor. Therefore, we assume the location of the proposed HTL facility is the same as the NEFCO managed BPF facility.

4.4 Economic Assumptions

Table 7 summarizes key feedstock properties and techno-economic assumptions. The formulation of the optimization model is presented in Appendix C.

Table 7. Key techno-economic assumptions

Parameter	Unit	Value
Ash Factor	% dry mass	[0.28, 0.15, 0.13] for [Food, Sludge, MSW]
Wet Waste Density	kg/m^3	[593, 1075, 593] for [Food, Sludge, MSW]
Dry Mass Fraction	%	[0.30, 0.30, 0.60] for [Food, Sludge, MSW]
Maximum Plant Scale	Dry metric t/d	[5.00E+03, 2.67E+03] for [HTL, Incinerator]
Transport Cost (truck + driver)	$/h	95
Truck Capacity	m^3	[20.6, 30, 20.6] for [Food, Sludge, MSW]
Truck Wait Time	hours	[0.33, 1.0, 0.25] for [Food, Sludge, MSW]
Discount Rate		0.02
Lifecycle	years	30 (3-yr build-out)
Biocrude density	g/ml	0.98
Biocrude Yield	wt% AFDW (g/g)	0.45
Biocrude Price	$/gal	1.77
Electricity yield	kWh/t, dry metric	871
Electricity Price	$/kWh	$0.05
BAU Waste Mngt. Price	$/t, dry metric	[174, 440, 174] for [Food, Sludge, MSW]

4.4.1 Feedstock Properties

The dry mass fraction and density property values reflect typical as-delivered waste conditions. Sludge values are based on average literature values for dewatered sludge (IWA, 2007), assuming other WRRFs continue to dewater their untreated sludge prior to transport. For FW and MSW, the dry mass fraction is based on average literature values while the density is equivalent to the minimum compaction rating of the model truck used to deliver the material (see "Waste Transport").

4.4.2 Biocrude Yield & Density

The biocrude conversion efficiency (yield rate) of 45 wt% AFDW feed (g/g) and biocrude density of 0.98 g/ml are harmonized with the performance assumptions in the 2023 Business Case Study for Wet Waste Hydrothermal Liquefaction and Biocrude Upgrading to Hydrocarbon Fuels report (PNNL, 2023 in publication), and experimental sludge and food waste blend data published in the 2021 HTL State of Technology report (PNNL, 2022).

4.4.3 Electricity Yield

We assume an electricity yield rate of 474 kWh per wet short ton of burned MSW (871 kWh per dry metric ton). This factor is based on the ratio of 2017 total national incinerator MSW throughput and electricity production, which is more conservative than the 17-year average of 488 kWh/t (ERC, 2018).

4.4.4 Energy Prices

4.4.4.1 Biocrude Acquisition Price

We assume any biocrude produced by HTL is purchased by the nearest conventional refinery for upgrading to renewable diesel. The refiner's acquisition cost of domestic crude represents the weighted average price refiners pay to purchase and transport domestic crude oil without considering federal or state incentives (e.g., renewable diesel credits or tax credits). Table 8 presents the average biocrude price paid to HTL plants of $1.77/gal, which represents the 15-year (2008–2022) national annual average US Crude Oil Domestic Acquisition Cost by Refiners (EIA, 2023). This price period includes temporary negative price impacts from the 2008 market crash (2009), over-production of shale oil (2015–2016) (World Bank, 2018), and the Covid-19 pandemic (2020) (BLS, 2020). If these price effects are removed, the average crude price increases to $1.99/gal.

Table 8. US Crude Oil Domestic Acquisition Cost by Refiners

Year	$/barrel	$/gal
2008	*98.47*	*2.34*
2009	59.49	1.42
2010	78.01	1.86
2011	100.71	2.40
2012	100.72	2.40
2013	102.91	2.45
2014	94.05	2.24
2015	*49.94*	*1.19*
2016	*42.41*	*1.01*
2017	52.05	1.24
2018	67.05	1.60
2019	*60.31*	*1.44*
2020	*41.23*	*0.98*
2021	69.07	1.64
2022	97.45	2.32
Avg. (all years)		**1.77**
Avg. (filtered)		**1.99**

4.4.4.2 Wholesale Price of Electricity

Incinerators are considered baseload energy providers and are not as flexible as gas turbines and renewables. They have long start-up times (hours) and must burn constantly to be efficient, which prevents them from participating in the lucrative peak power market. Revenue from producing electricity from incineration is therefore based on the average wholesale price of electricity. Electric generation from natural gas, wind, and solar power has lowered wholesale electricity prices as low as \$0.02/kWh (ASME, 2022). Recent prices have been higher and more volatile than average due to extreme weather events and higher natural gas prices, which have replaced coal as a primary fuel for electricity generation (EIA, 2023). We assume the electricity price paid to incinerators is \$0.05 per kWh, which represents the 5-year (2018–2022) US national weighted average weekly wholesale price of electricity (EIA, 2023).

4.4.4.3 Biocrude Upgrading

In the current scenario, biocrude is sold directly to a conventional refinery, and any revenue generated from refining is not considered. In future scenarios, we can account for revenues and renewable energy credits generated from conventional or biorefining. For example, if we assume that refiners utilize 100% of the biocrude to produce renewable diesel, the price paid to refiners for finished fuel is \$2.27 per gallon, representing the 15-year (2008–2022) national annual average US No 2 Diesel Wholesale/Resale Price by Refiners (EIA, 2023). This price is \$0.07/gal higher than the average resale price of jet fuel for the same period (EIA, 2023). At \$2.27/gal, refiners make an average of \$0.51/gal over the cost of domestic crude acquisition. The average diesel price increases to \$2.55/gal if the aforementioned oil market price effects are removed.

Table 9. US Wholesale/Resale Price for Diesel and Jet-A by Refiners

Year	Diesel \$/gal	Jet-A \$/gal
2008	*2.99*	*3.00*
2009	1.71	1.68
2010	2.21	2.18
2011	3.03	3.01
2012	3.11	3.09
2013	3.03	2.96
2014	2.81	2.77
2015	*1.67*	*1.60*
2016	*1.38*	*1.29*
2017	1.69	1.60
2018	2.13	2.07
2019	1.96	1.93
2020	*1.29*	*1.18*
2021	2.11	1.89
2022	2.99	2.79
Avg. (all years)	**2.27**	**2.20**
Avg. (filtered)	**2.55**	**2.48**

4.4.5 BAU Waste Management Price

The BAU waste management prices reflect the current average total cost of waste management (i.e., treatment, transport, and disposal). The optimization model uses this price as a reference to determine a waste producer's willingness-to-pay for alternative strategies.

MSW and IIC-FW are typically landfilled or incinerated, therefore BAU prices reflect the 2018 national weighted average landfill tipping fee of $52.20 per wet tonne (WBJ, 2019), or $174 per dry tonne assuming an average moisture content of 30%. The total average cost of sludge treatment and disposal of 440 USD/dry metric ton is based on the literature (Seiple, 2020). This value is consistent with Peccia and Westerhoff (2015), which reported costs as high as $800 per dry ton, most of which is for solids treatment.

In the W2X pathways model, we separate treatment, transport, and disposal (tipping fees) costs. Treatment costs are intrinsic to the scaled waste treatment technology capital and operating costs. Transport costs are modeled explicitly and assigned to waste producers. Tipping (gate) fees are assigned to the waste producers, but they may be positive or negative depending on the ability of processors to pay for feedstock. The total cost to waste producers, therefore, is the sum of the gate fee and the weighted average transport costs.

4.4.6 Waste Transport

Feedstock transport costs are calculated using a geospatial waste transport model implemented with PostgreSQL, PostGIS, and pgRouting software and a national transportation network dataset developed from the US Census Bureau's 2021 MAF/TIGER state-level edge datasets. The least cost path (in hours) is calculated from each waste source to the W2X study locations. Total annual transport costs are then calculated as the total drive time plus total wait time multiplied by the truck charge out rate and required number of trips per year. Because sludge waste from DITP is already being pumped to the proposed HTL site, the coordinates for DITP waste source were overwritten with the coordinates of the W2X site to represent the co-location of this material with the conversion site, ensuring a waste transport cost of $0.

4.4.6.1 Wait Times

For wastewater solids, we assume 30 minutes of loading wait time per trip, which includes staging, loading, verification, truck wash, and documentation, and the same amount of time for unloading, for a total wait time of one hour. For FW, we assume 5 minutes of loading wait time per load and 15 minutes of unloading. FW collection trucks are more efficient during pickup but may experience delays in offloading due to scale queues and documentation. For MSW, we assume the waste is already loaded as part of the collection service, although 15 minutes is required for unloading at the W2X facility. In some cases, the total wait time may exceed the estimated drive time. If the transport distance is zero, the wait time is also zero, assuming material can be diverted directly to the W2X facility.

4.4.6.2 Truck Charge Out Rate

The American Transportation Research Institute (ATRI) survey data indicate the average marginal cost per hour to operate a truck was $90.78 per hour in 2022 (ATRI, 2023), which we conservatively increase to $95/h. Trucking data indicate fuel and driver wages and benefits, rather than truck type and size, affect trucking costs the most from a TEA perspective, with truck payments representing only 15% of total marginal operating costs per hour and total marginal costs per mile varying by only 12% across all trucking sectors (Appendix D).

4.4.6.3 Truck Capacity

Untreated, dewatered wastewater solids are assumed to be transported using a rented covered container truck with a capacity of 30 m^3, which is typical for hauling biosolids (Marufuzzaman, 2015). A typical neighborhood garbage collection truck operating in an urban environment holds between 12 and 14 wet compact short tons of waste (SC, 2019). Therefore, we select a similarly sized, 20.6 m^3 truck (i.e., RotoPac 27R) to model maximum truck capacity for MSW and FW (New Way, 2022).

4.4.6.4 Number of Required Trips

The optimization model simulates the required number of trips per year to deliver feedstock from each source to the W2X facility. The number of trips is dependent on the total annual feedstock wet volume to be transported and the maximum truck capacity. To determine total transport volume, dry weight feedstock estimates are converted to a wet volume basis using Eq. (1) along with feedstock property values for dry mass fraction and density that reflect typical as-delivered conditions.

$$V_f = \frac{M_s}{\rho_f S_s} \qquad \text{Eq. (1)}$$

where
V_f = volume of feedstock, m^3
M_s = mass of dry solids, kg
ρ_f = density of feedstock, kg/m^3
S_s = fraction of dry solids as decimal

Alternatively, the wet volume of the truck may be converted to dry solids by rearranging Eq. 1 as follows.

$$M_t = V_t \rho_f S_s \qquad \text{Eq. (2)}$$

where
V_t = volume of truck, m^3
M_t = mass of dry solids per truck load, kg

4.5 Plant Costs & Scale

For all pathways, we assume a 3-year build-out phase followed by a 27-years of operations. We do not apply a capacity factor, as the time online is already embedded in the underlying scaled cost model assumptions for each technology.

4.5.1 Incineration

Figure 10 and empirical formula Eq. (3) present the scaled total capital expenses (CapEx) costs for incineration as reported by Waste to Energy International (WEI, 2015), where, *I* is the total investment cost in millions of dollars, *C* is the facility capacity (1000 metric t/y, wet MSW).

$$I = 2.3507C^{0.7753} \qquad \text{Eq. (3)}$$

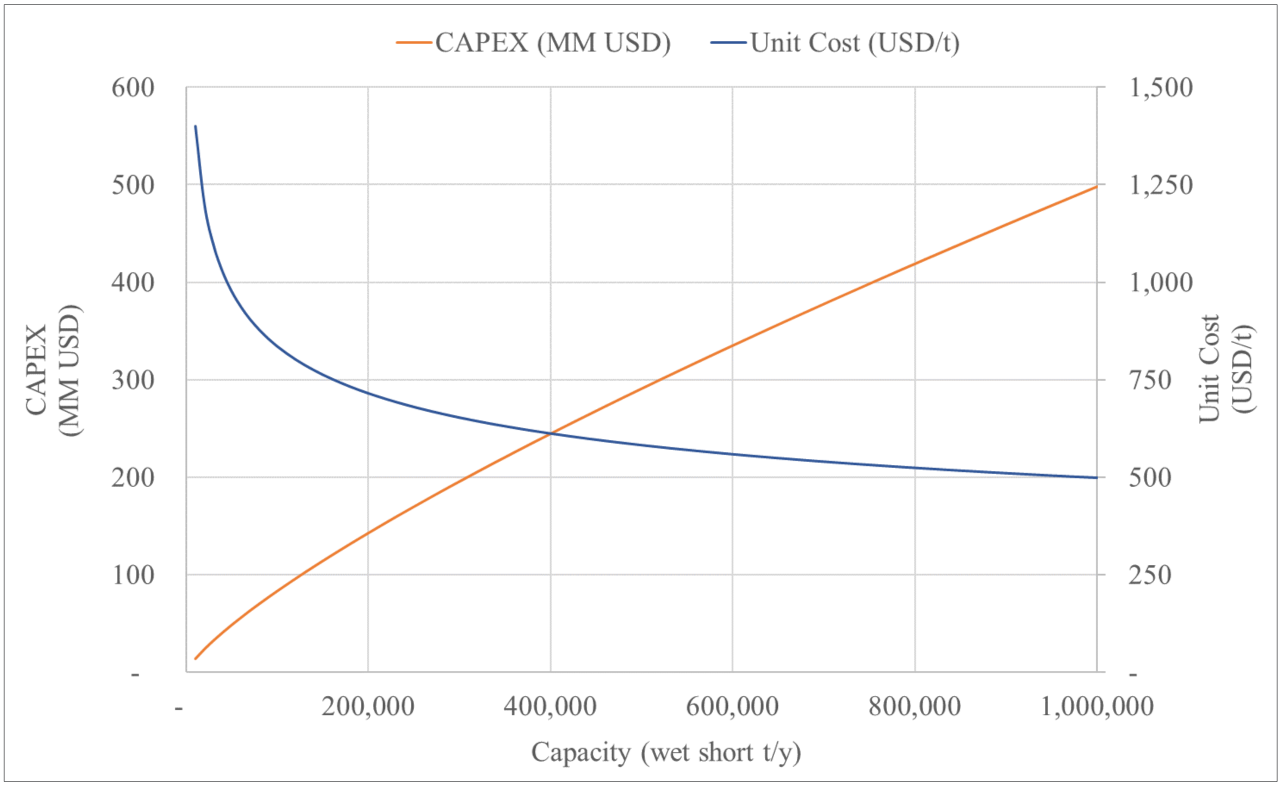


Figure 10. Scaled CapEx and unit costs for incineration (based on Eq. 3)

Applying unit conversions to the term *C* yields Eq. (4), where *q* represents dry metric tons per day of MSW.

$$I = 1.331e6q^{0.7753} \quad \text{Eq. (4)}$$

Annual operating expenses (OpEx) for incineration are assumed to equal 5% of TCI, based on input from industry (WEI interview, 2023). In the W2X model, the incinerator plant scale is limited to 3,510 wet short t/d (2670 dry metric t/d) of feedstock throughput, which is equivalent to the largest incinerator in the US.

4.5.2 HTL

The total capital investment (TCI) and annual OpEx for HTL and hydrotreating are estimated by scaling modeled costs presented in the 2022 HTL Design Study (PNNL, 2017) and adjusting the cost basis to 2020. Annual OpEx do not include credits or savings for avoided disposal.

Table 10. Scaled HTL TCI and OpEx costs

Capacity (dry short t/d)	Capacity (dry metric t/d)	TCI ($)	OpEx ($/y)
10	9.1	7,661,809	2,584,250
20	18.1	12,364,909	3,201,889
50	45.4	23,407,293	4,951,388
100	90.7	38,102,998	7,742,211
200	181.4	62,289,482	13,862,871
500	453.6	120,163,195	29,937,054
1000	907.2	198,775,337	56,256,873

Figure 11 presents the regression analysis performed for HTL on total dry metric tons per day of waste versus total CapEx and OpEx. Because the HTL CapEx and OpEx are non-linear, both alpha and gamma terms are reported.

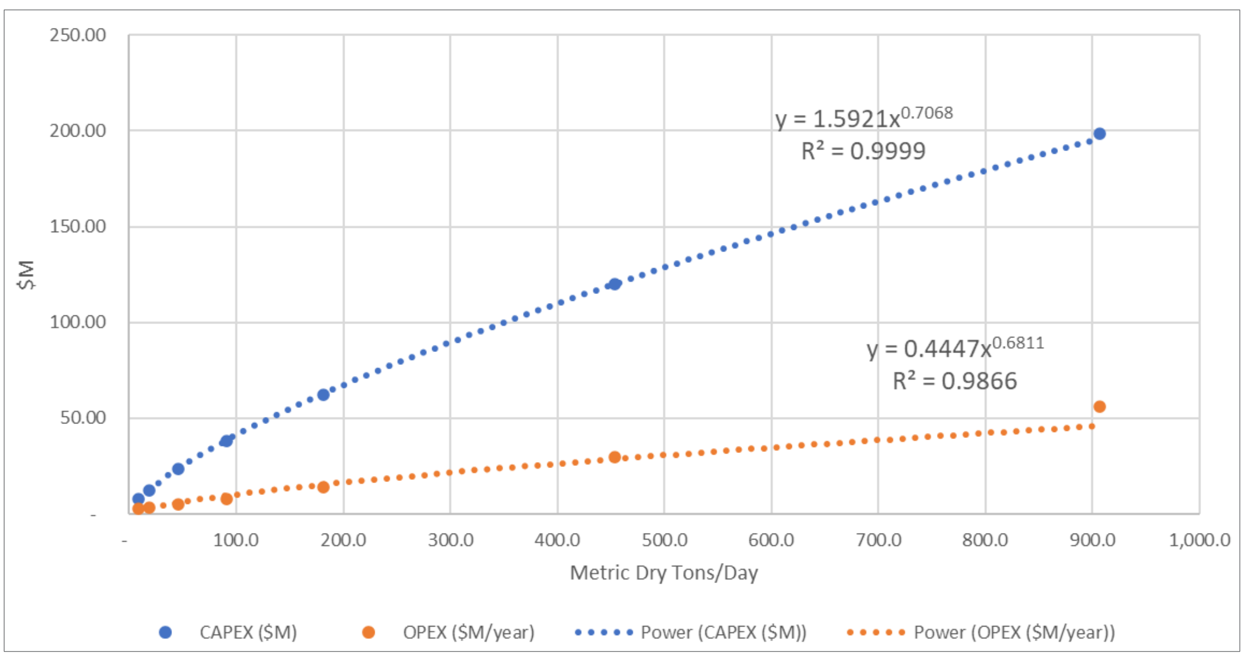


Figure 11. Regression of HTL CapEx/OpEx as a function of throughput (dry metric t/d)

4.6 Gate Fees

A powerful feature of our W2X Pathways model is that we can use different gate fee calculations to examine the potential impacts of sharing energy revenue across the entire supply chain in various ways, thereby reducing cost uncertainty for industry by providing realistic upper and lower bounds for waste producer costs and processor profits, as well as system-wide NPV.

Gate fees range from a "greedy" maximum gate fee, where the processor keeps all the profits (and waste producer costs end up the same as the BAU case), to a profit-sharing scheme that either seeks to equalize waster producer and processor NPV (both can make money) or balances producer cost reduction with processor profits (waste producers pay less tipping fees but don't make money), to a "break-even" gate fee where the producers receive all the profits (processors have an NPV of 0). Negative gate fee prices means that the processor pays the waste producer to accept waste.

Four different gate price estimation approaches are included in the model. These represent a range of potential W2X gate fee schemes to provide upper, middle, and lower bounds on W2X profitability. Each preserves the financial feasibility of waste producers and energy facilities. In other words, waste producers will send their waste to a W2X facility if the cost is less than or equal to the current cost of disposal, including transport. And W2X facilities will only operate if they make a profit or break-even.

- *Maximum gate fee by producer*: In this approach, the maximum gate "tipping" fee varies by waste producer. This calculation provides a "greedy" upper bound on W2X performance that maximizes profit with no net disposal cost reduction for waste producers.

- *Profit-Sharing Gate Fee (Equalize waste producer savings and W2X processor NPV)*: In this gate fee approach, W2X revenues are shared with waste producers such that the W2X processor NPV (per metric ton dry waste accepted) is the same as the NPV of the waste producer cost reduction (per metric ton dry waste produced). In this case, as long as the W2X pathway provides some cost reduction compared to BAU, the unit profit will always be positive.

- *Profit-Sharing Gate Fee (Equalize waste producer and W2X processor NPV)*: If even more W2X revenues are shared with waste producers, it is possible to find the point at which the waste producers and W2X processors have the same overall NPV per metric ton of dry waste (produced or accepted). This can produce negative NPVs for both producers and processors (in which case, the processors would not rationally choose such a scheme) but still represent an economic improvement to the total waste management system; such an outcome remains a useful point of comparison. Under BAU conditions, the NPV of a waste producer is already negative due to the total cost of waste management.

- *Break-Even Gate Fee*: Going a step further, the break-even gate fee tells us what could happens if W2X processors only keep enough revenue to continue operating and use the remaining profit to reduce the cost of waste management for waste producers. The break-even price could be a useful benchmark if the W2X facility is owned by the public, as the goal is to reduce cost relative to baseline total waste management prices of \$174/t for FW and MSW and \$440/t for sludge. If an energy facility only accepts a single waste type or charges a single gate fee for all waste types, then it is simple to calculate the minimum feasible gate fee where NPV equals zero. However, if gate fees vary by waste type or producer, then many gate fee combinations could yield an NPV of 0. One way to solve this is to set the gate fees such that the marginal values of additional waste of each type are equal. In other words, at the current operating point, this gate fee calculation scheme ensures that a small additional amount of waste would provide the same benefit regardless of the type of waste.

5.0 Results

Table 11 and Table 12 present the results of the economic impact assessment of selected pathways within the Greater Boston area.

Table 11. System-Wide Impacts

Parameter	Unit	Value
Total feedstock utilization	dry metric t/y	3,558,868
Total feedstock utilization	dry metric t/d	9,750
Total feedstock utilization	%	100
Total biocrude output	million gal/y	47.802
Total electricity output	million MWh/y	2.666
NPV BAU	billion USD	-13.509
NPV W2X	billion USD	-2.332
Cost Reduction	billion USD	11.176

Table 12. Pathway-Specific Impacts

Parameter	Unit	HTL	Incinerators: [Rochester, Saugus, Haverhill, North Andover, Millbury]
Feedstock utilization	dry metric t/d	1,363	8,387
Feedstock utilization by type	dry metric t/d	[Food, Sludge] [611, 752]	[MSW] [1234, 2670,1813, 0, 2670]
Maximum Gate Fee by Producer			
NPV	billion USD	3.023	[1.106, 2.736, 1.722, 0, 2.590]
ROI		17.300	[5.953, 7.616, 6.671, 0, 7.282]
Profit-Sharing Gate Fee (Producer savings = processor NPV)			
Unit profit	$/dry metric ton	155.63	[62.93, 71.93, 66.68, 0, 68.10]
NPV	billion USD	1.511	[0.553, 1.368, 0.861, 0, 1.295]
ROI		9.315	[3.642, 4.473, 4.001, 0, 4.306]
Profit-Sharing Gate Fee (Producer NPV = processor NPV)			
Unit profit	$/dry metric ton	-4.77	[-24.07, -15.07, -20.32, 0, -18.90]
NPV	billion USD	-0.046	[-0.212, -0.287, -0.262, 0, -0.359]
ROI		1.086	[0.446, 0.672, 0.516, 0, 0.504]
Break-Even Gate Fee			
Price	$/dry metric ton	[Food, Sludge] [-6.75, -34.66]	[MSW] [29.43, 17.80, 23.38, 0, 17.80]
NPV	Million USD	0	0

Analysis indicates HTL and incineration can cost-effectively treat 100% of analyzed feedstocks to produce 48 million gal/y of biocrude and 2.7 million MWh/y of base load electricity. The NPV of the total estimated waste management liability for the Greater Boston area over the scenario lifecycle (30 years) is approximately $13.509 billion. Implementing a W2X strategy with HTL and incineration could potentially reduce this liability by $11.177 billion over the same period. In other words, even without policy supports (i.e., energy and carbon credits), the W2X industry can be cost positive in the Boston region without exceeding the current waste management prices paid by waste producers.

However, when we consider the entire waste management system (i.e., waste producer costs and processor costs and revenue), the system NPV is -$2.332 billion over 30 years, which means that there is not enough profit from energy production to cover the entire costs of waste management. The primary reasons for this are (1) recent wholesale energy prices for crude oil and electricity are so low that they limit the profitability of renewable energy production in the absence of policy support, and (2) incineration is less profitable than HTL and there is a lot more MSW to treat than WW.

According to the model, cost-effectively treating 100% of sludge and food waste with a centralized HTL facility would require a plant scale of approximately 1,363 dry metric t/d of waste throughput. Although this scale is slightly higher than the maximum scaled cost value used in the regression analysis (i.e., 1000 dry metric t/d), the degree of extrapolation is acceptable. With respect to MSW treatment, the set of existing incinerators were able to handle 100% of available feedstock without exceeding the maximum plant scale, which is equivalent to the largest existing incinerator in the US.

The relative impacts of incineration versus HTL pathways on the waste management system can be seen in the gate fee calculations. Negative gate fee prices indicate the W2X processor can afford to pay the waste producers to accept waste (as feedstock). Prices for the North Andover incinerator are all zero because it was not cost-effective to send waste there, given the facilities proximity to Haverhill. Based on the maximum gate fee, which maximizes the NPV of processors, both technologies can be cost-effective, with HTL achieving an ROI >17 and an NPV of $3 billion and incinerators achieving ROI levels less than half that of the HTL with a total NPV of $8 billion USD. HTL is expected to be more profitable despite processing less waste and offers a better return on investment.

The performance gap between the two pathways is further highlighted at the other end of the NPV spectrum with the break-even gate fee, which represents the maximum potential cost reduction to waste producers (i.e., W2X processor NPV=0). If profits from the sale of biocrude and electricity are used solely to offset waste management costs then HTL can afford to pay $34.66 and $6.75 per dry metric ton of sludge and food waste, respectively, whereas incinerators must continue to charge waste producers a tipping fee ranging from $17.80 to $29.43 per dry metric ton.

The two alternative profit-sharing gate fees differentiate performance in the middle of the NPV spectrum. The first approach shares processor revenue with waste processors to equalize the NPVs (per metric ton of dry waste accepted or produced) of each entity. Under such conditions, both HTL and incinerators are required to operate at a loss, with total NPVs of -4.77 and -1.12 billion dollars, respectively. As such, the average unit profit for HTL is negative (-$4.77 per dry metric ton) but not as negative as incineration (-$15.07 to -$24.07 per dry metric ton). These findings indicate that total BAU waste management costs are relatively high compared to system revenues. Waste producers would still have to pay a tipping fee even if NPVs were equalized between producers and processors. Furthermore, since the W2X facilities have a negative NPV, it would not be economically feasible to build and operate them under this particular profit-sharing gate fee scheme without some kind of financial support (e.g., subsidized construction costs). Though, the cost reduction still represents substantial savings for waste producers.

For the second profit-sharing gate fee, W2X revenues are shared with waste producers such that the W2X processor NPV (per metric ton dry waste accepted) is the same as the NPV of the waste producer cost reduction (per metric ton dry waste produced). In this case, HTL waste producers save $155.63 per dry metric ton on waste management costs while HTL processors earn an equal amount as revenue. MSW waste producers and incinerators each earn or save $62.93 to $71.93 per dry metric ton. This profit-sharing model represents a practical middle ground for how renewable energy value may be imputed onto the waste supply chains in a market-based system.

The co-location of W2X facilities with the largest waste producers in a region is beneficial. This is especially true if there is a big difference in size between the largest and median producers and/or if the average travel time over the region in question is low relative to the load/unload time. This is because of the extra time required for loading and unloading when the waste producer is not co-located with a W2X facility. Moving the facility to another, potentially more central location might reduce total drive times, but the cost of loading and unloading the waste from a large producer might substantially outweigh the increase in travel times for smaller producers. For example, loading and unloading waste from the largest sludge producer would cost $1.06 million/year. This is equivalent to the smallest sludge producer driving over 280 hours round-trip to its nearest W2X facility.

This has implications for the gate fee calculation approaches described above. The lack of loading and unloading times, with their associated costs, increases the maximum gate fee that the co-located W2X facility can charge the co-located producer, thereby increasing the W2X facility's NPV in this case. It also produces higher NPVs, in the profit-sharing case, for *all* waste producers supplying that W2X facility (as well as the W2X facility itself). In the break-even case, nothing changes for the W2X facility, but the co-located producer sees an increased NPV.

6.0 Conclusions

Under current BAU conditions, the cost (NPV) to manage 3.6 million dry metric t/y of MSW and organic waste in the Greater Boston area over the next 30 years will be $13.5 billion. Over the same period, proposed W2X strategies, including HTL for biofuels and incineration for power, could reuse 100% of these same wastes to produce enough salable energy to substantially reduce overall waste system management costs (NPV) by $11.2 billion. Although the waste system NPV remains negative (-$2.3 billion), with better economies of scale and/or modest price supports such as energy and carbon credits, generating a positive NPV for the entire waste system is possible with W2X. In other words, W2X could potentially make waste producers and waste processors profitable simultaneously, which is unprecedented. Furthermore, the proposed HTL facility is more cost-effective than the incinerators – to the point that the HTL facility could afford to pay waste producers for their waste. The different gate fee calculation options also demonstrated an effective range of financial feasibility for both waste producers and processors to conceptualize the benefits of waste producer and processor co-location, waste blending, and profit-sharing.

Appendix A – Census Summary

Table 13. Boston CSA Census Summary (2020)

Census Designation	2020 Pop. (million)	Poverty (%)
Windham County, CT	116,418	12.3
Barnstable County, MA	228,996	6.9
Bristol County, MA	579.200	11.5
Essex County, MA	809,829	9.8
Middlesex County, MA	1,632,002	8.2
Norfolk County, MA	725,981	6.5
Plymouth County, MA	530,819	7.5
Suffolk County, MA	797,936	16.4
Worcester County, MA	862,111	10.6
Belknap County, NH	63,705	8.0
Hillsborough County, NH	422,937	6.2
Merrimack County, NH	153,808	6.8
Rockingham County, NH	314,176	5.2
Strafford County, NH	130,889	8.6
Bristol County, RI	50,793	6.7
Kent County, RI	170,363	6.5
Newport County, RI	85,643	6.2
Providence County, RI	660,741	14.0
Washington County, RI	129,839	5.4

Source: https://data.census.gov

Appendix B – Analyzed WRRFs

Table 14. List of WRRFs

FACILITY	CITY	AFDW Flow (MM Gal/d)	Cum. Flow (%)	Recoverable (Dry metric t/d)	Disposed (Dry metric t/d)
TOTAL (n = 71)		***815***		***752***	***487***
Mass. Water Resources Authority	BOSTON	310	37%	295	160
FIELDS POINT WWTF	PROVIDENCE	48	42%	46	21
Upper Blackstone Abatement District	MILLBURY	37	47%	36	31
Greater Lawrence Sanitary District	NO ANDOVER	31	50%	36	23
Fall River Public Works Department	FALL RIVER	31	54%	27	23
New Bedford Wastewater District	NEW BEDFORD	30	58%	26	22
Lynn Water and Sewer Commission	LYNN	26	61%	22	11
Manchester WWTF	MANCHESTER	24	63%	21	10
BUCKLIN PT STP	E. PROVIDENCE	24	66%	22	12
Lowell Regional Wastewater Utility	LOWELL	16	68%	14	7
Brockton Dept. of Public Works	BROCKTON	16	70%	15	13
South Essex Sewer District WWTP	SALEM	15	72%	13	11
Veolia Water - Cranston WPCF	CRANSTON	13	73%	13	11
Nashua WWTF	NASHUA	12	75%	10	5
NEWPORT WWTF	NEWPORT	8	76%	7	6
Fitchburg Wastewater System	FITCHBURG	8	77%	7	6
Haverhill Sewer Collection System	BRADFORD	8	78%	7	6
WOONSOCKET REGIONAL	WOONSOCKET	8	79%	7	6
Taunton Dept. of Public Works	TAUNTON	7	79%	6	5
East Providence WPCF	E. PROVIDENCE	7	80%	6	3
Franklin WWTF - WRBP	FRANKLIN	6	81%	5	3
Leominster Dept of Public Works	LEOMINSTER	6	82%	5	3
Webster Sewer Department	WEBSTER	6	82%	5	3
Portsmouth WWTF	PORTSMOUTH	6	83%	5	4
WARWICK WWTF	WARWICK	5	84%	4	2
West Warwick WWTF	WEST WARWICK	5	84%	4	4
Concord Hall Street WWTF	CONCORD	5	85%	4	3
Charles River WPCF	MEDWAY	4	85%	4	3
Billerica Dept. of Public Works	BILLERICA	4	86%	4	3
Marlborough Easterly WWTF	MARLBOROUGH	4	86%	3	3
Attleboro Water and Wastewater Div.	ATTLEBORO	3	87%	3	3
Rochester WWTF	GONIC	3	87%	3	3
Gloucester City Engineer's Office	GLOUCESTER	3	87%	3	3

Gardner Dept. of Public Works	EAST TEMPLETON	3	88%	3	2
Mansfield Depart. of Public Work	NORTON	3	88%	3	2
BRISTOL WWTF	BRISTOL	3	88%	2	2
North Attleborough WWTF	NORTH ATTLEBOROUGH	3	89%	2	2
KILLINGLY WPCF	DANIELSON	3	89%	2	2
WILLIMANTIC WPCF	WILLIMANTIC	3	89%	2	2
Dover WWTF	DOVER	3	90%	2	2
SOUTH KINGSTOWN REG STP	NARRAGANSETT	3	90%	2	2
Hampton WWTF	HAMPTON	3	90%	2	2
Somerset Water Pollution Control	SOMERSET	3	91%	2	2
Westerly WWTF	WESTERLY	3	91%	2	2
Marlborough Westerly WWTP	MARLBOROUGH	2	91%	2	2
Newburyport WPCF	NEWBURYPORT	2	92%	2	2
Fairhaven DPW Wastewater Dept.	FAIRHAVEN	2	92%	2	2
Templeton Dept of Public Works	BALDWINSVILLE	2	92%	2	2
Exeter WWTF	EXETER	2	92%	2	2
Hudson Dept. of Public Works	HUDSON	2	93%	2	1
Merrimack WWTF	MERRIMACK	2	93%	2	1
Derry WWTF	DERRY	2	93%	2	1
MWRA - Clinton	CLINTON	2	93%	2	1
Plymouth Dept. of Public Works	PLYMOUTH	2	93%	2	1
Athol Dept. of Public Works	ATHOL	2	94%	2	1
Milford WWTF	MILFORD	2	94%	1	1
WARREN WWTF	WARREN	2	94%	1	1
Smithfield Sewer Authority	SMITHFIELD	2	94%	1	1
Grafton Wastewater System	SO. GRAFTON	2	94%	1	1
PUTNAM WPCF	PUTNAM	2	95%	1	1
Southbridge Dept. of Public Works	SOUTHBRIDGE	2	95%	1	1
Amesbury Dept of Public Works	AMESBURY	2	95%	1	1
Ayer Water Pollution Control	AYER	1	95%	1	1
Salisbury Dept of Public Works	SALISBURY	1	95%	1	1
Rockland Sewer Dept.	ROCKLAND	1	95%	1	1
Westborough Wastewater System	WESTBOROUGH	1	96%	1	1
Somersworth WWTF	SOMERSWORTH	1	96%	1	1
Marshfield Dept of Public Works	BRANT ROCK	1	96%	1	1
Hull Collection System	HULL	1	96%	1	1
Bridgewater Sewer Department	BRIDGEWATER	1	96%	1	1
Milford Dept. of Public Works	HOPEDALE	1	96%	1	1

Appendix C – Model Formulation

The following formulations calculate the NPV of W2X operations: a capital cost (for incinerators and HTL plants) plus one year's worth of operations applied to the lifetime of the facility with future revenues/costs discounted appropriately. Note that:

$$NPV = \sum_{t=T_{build}}^{T-1} \frac{365 \times Daily\ Operating\ Profit - OPEX}{(1+\delta)^t} - \sum_{t=0}^{T_{build}-1} \frac{CAPEX}{T_{build}} \frac{1}{(1+\delta)^t}$$

$$= (365 \times Daily\ Operating\ Profit - OPEX)\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) - \frac{CAPEX}{T_{build}}\left(\frac{1-(1+\delta)^{-T_{build}}}{\delta}\right)$$

where δ is the discount rate. For descriptions of the variables, prices, and parameters used in this appendix's calculations, see Table 15, Table 16, Table 17 respectively.

Table 15. Variable Descriptions

Variable	Description
$l(j)$	Waste type l of waste producer j
$q^W_{jl(j)n}$	Waste of type $l(j)$ shipped from j to incinerator n
$q^W_{jl(j)s}$	Waste of type $l(j)$ shipped from j to HTL facility s
q^I_{ln}	Waste of type l accepted by incinerator n
q^T_{ls}	Waste of type l accepted by HTL facility s

Table 16. Price Descriptions

Price	Description
p_{ln}	Gate fee for waste type l at incinerator n
p_{ls}	Gate fee for waste type l at HTL facility s
$p_{jl(j)n}$	Gate fee paid for producer j at incinerator n
$p_{jl(j)s}$	Gate fee paid for producer j at HTL facility s

Table 17. Parameter Descriptions

Parameter	Description
δ	Discount rate
T	Total time horizon
T_{build}	W2X facility build time
$c_{jl(j)n}$	Per-unit cost for producer j to ship to incinerator n
$c_{jl(j)s}$	Per-unit cost for producer j to ship to HTL facility s
$q^W_{dispose,jl(j)}$	Total waste production by producer j
$p^W_{dispose,l}$	Per-unit disposal cost for waste type l
α_{hour}	Hourly transportation cost
h_{jn}	Travel time from producer j to incinerator n
$h_{truck,l}$	Load/unload time for waste type l
$m_{truck,l}$	Truck capacity for waste type l
p_{elec}	Price of electricity
$\eta_{elec,l}$	Conversion factor for waste type l to electricity via incineration
$\alpha_{inc,cap}$	Incineration CapEx coefficient
$\gamma_{inc,cap}$	Incineration CapEx exponent
$\alpha_{htl,op}$	HTL OpEx coefficient
$\gamma_{htl,op}$	HTL OpEx exponent
$\alpha_{htl,cap}$	HTL CapEx coefficient
$\gamma_{htl,cap}$	HTL CapEx exponent
$p_{biocrude}$	Price of biocrude
$\eta_{crude,l}$	Conversion factor for waste type l to biocrude via HTL
ρ_l	Ash-free dry mass fraction of waste type l
p_n	Marginal benefit of each feedstock at incinerator n
$p_{share,n}$	NPV per unit waste accepted at incinerator n

Each agent in the model seeks to maximize its NPV. In order to apply this to the waste producers, we need to consider the same lifetime for the entire calculation, and maximizing NPV is equivalent to minimizing the NPV of its costs. For waste producer j, which has waste type l, we have

$$\min_{q^W_{jl(j)n},q^W_{jl(j)s}} 365\left[\sum_n (p_{l(j)n} + c_{jl(j)n})q^W_{jl(j)n} + \sum_s (p_{l(j)s} + c_{jl(j)s})q^W_{jl(j)s} + \left(q^W_{dispose,jl(j)} - \sum_n q^W_{jl(j)n} - \sum_s q^W_{jl(j)s}\right)p^W_{dispose,l(j)}\right]\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)$$

$$q^W_{dispose,jl(j)} \geq \sum_n q^W_{jl(j)n} + \sum_s q^W_{jl(j)s}$$

$$0 \leq q^W_{jl(j)n}, q^W_{jl(j)s}$$

where

$$c_{jln} = \begin{cases} \dfrac{\alpha_{hour}\left(2h_{jn} + h_{truck,l}\right)}{m_{truck,l}} & h_{jn} > 0 \\ 0 & else \end{cases}$$

Analogous calculations hold for c_{jls}. For incinerator n, we have

$$\max_{q_{ln}^{I}} \left[365 \sum_{l} (p_{elec}\eta_{elec} + p_{ln}) q_{ln}^{I} - 0.05\alpha_{inc,cap} \left(\sum_{l} q_{ln}^{I}\right)^{1-\gamma_{inc,cap}}\right] \left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) - \frac{\alpha_{inc,cap}\left(\sum_{l} q_{ln}^{I}\right)^{1-\gamma_{inc,cap}}}{T_{build}} \left(\frac{1-(1+\delta)^{-T_{build}}}{\delta}\right)$$

$$q_{max}^{I} \geq \sum_{l} q_{ln}^{I}$$

$$0 \leq q_{ln}^{I}$$

Note that for incinerators, the annual OpEx costs are 5% of the CapEx costs. For HTL facility s, we have

$$\max_{q_{ls}^{T}} \left[365 \sum_{l} \left(p_{biocrude}\eta_{crude,l}\rho_{l} + p_{ls}\right) q_{ls}^{T} - \alpha_{htl,op} \left(\sum_{l} q_{ls}^{T}\right)^{1-\gamma_{htl,op}}\right] \left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) - \frac{\alpha_{htl,cap}\left(\sum_{l} q_{ls}^{T}\right)^{1-\gamma_{htl,cap}}}{T_{build}} \left(\frac{1-(1+\delta)^{-T_{build}}}{\delta}\right)$$

$$q_{max}^{T} \geq \sum_{l} q_{ls}^{T}$$

$$0 \leq q_{ls}^{T}$$

Finally, we have market clearing conditions

$$\sum_{j} q_{jln}^{W} = q_{ln}^{I}$$

$$\sum_{j} q_{jls}^{W} = q_{ls}^{T}$$

We can combine these optimizations together in a social optimum formulation

$$
\begin{aligned}
\min_{q^W_{jl(j)n},q^W_{jl(j)s},q^I_{ln},q^T_{ls}} & -365\Bigg[\sum_n c_{jl(j)n}q^W_{jl(j)n} + \sum_s c_{jl(j)s}q^W_{jl(j)s} \\
& + \left(q^W_{dispose,jl(j)} - \sum_n q^W_{jl(j)n} - \sum_s q^W_{jl(j)s}\right)p^W_{dispose,l(j)}\Bigg]\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) \\
& + \left[365\sum_l p_{elec}\eta_{elec}q^I_{ln} - 0.05\alpha_{inc,cap}\left(\sum_l q^I_{ln}\right)^{1-\gamma_{inc,cap}}\right]\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) \\
& - \frac{\alpha_{inc,cap}\left(\sum_l q^I_{ln}\right)^{1-\gamma_{inc,cap}}}{T_{build}}\left(\frac{1-(1+\delta)^{-T_{build}}}{\delta}\right) \\
& + \left[365\sum_l p_{biocrude}\eta_{crude,l}\rho_l q^T_{ls} - \alpha_{htl,op}\left(\sum_l q^T_{ls}\right)^{1-\gamma_{htl,op}}\right]\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) \\
& - \frac{\alpha_{htl,cap}\left(\sum_l q^T_{ls}\right)^{1-\gamma_{htl,cap}}}{T_{build}}\left(\frac{1-(1+\delta)^{-T_{build}}}{\delta}\right)
\end{aligned}
$$

$$q^W_{dispose,jl(j)} \geq \sum_n q^W_{jl(j)n} + \sum_s q^W_{jl(j)s}$$

$$q^I_{max} \geq \sum_l q^I_{ln}$$

$$q^T_{max} \geq \sum_l q^T_{ls}$$

$$\sum_j q^W_{jln} = q^I_{ln}$$

$$\sum_j q^W_{jls} = q^T_{ls}$$

$$0 \leq q^W_{jl(j)n}, q^W_{jl(j)s}, q^I_{ln}, q^T_{ls}$$

We can then calculate the various gate fee regimes. For each waste producer j, the maximum feasible gate fees are

$$p_{l(j)n} = p^W_{dispose,l(j)} - c_{jl(j)n}$$

$$p_{l(j)s} = p^W_{dispose,l(j)} - c_{jl(j)s}$$

Calculating the minimum feasible single price gate fees per feedstock (i.e., waste type) for HTL facilities and incinerators is an underdetermined problem if those facilities accept more than one type of waste (i.e., there is a price for each feedstock type but only a single break-even NPV criterion). However, if we impose the constraint that the marginal benefit of each feedstock must be the same, then we can solve for the prices. For the incinerator, for example,

$$f_I(q_{ln}^I) \equiv \left[365\sum_l p_{elec}\eta_{elec}q_{ln}^I - 0.05\alpha_{inc,cap}\left(\sum_l q_{ln}^I\right)^{1-\gamma_{inc,cap}}\right]\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right) - \frac{\alpha_{inc,cap}\left(\sum_l q_{ln}^I\right)^{1-\gamma_{inc,cap}}}{T_{build}}\left(\frac{1-(1+\delta)^{-T_{build}}}{\delta}\right)$$

$$0 = f_I(q_{ln}^I) + 365\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)\sum_l p_{ln}q_{ln}^I$$

$$p_n = \frac{\partial f_I}{\partial q_{ln}^I} + 365\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)p_{ln}$$

$$\Rightarrow p_{ln} = \frac{1}{365}\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)^{-1}\left(p_n - \frac{\partial f_I}{\partial q_{ln}^I}\right)$$

$$p_n = \frac{\sum_l \frac{\partial f_I}{\partial q_{ln}^I} q_{ln}^I - f_I(q_{ln}^I)}{\sum_l q_{ln}^I}$$

This means that (positive) gate fees will be lower for feedstocks that contribute more to W2X facility revenue. Analogous calculations hold for HTL facilities.

Next, we calculate the profit-sharing gate fee where producers and W2X facilities have the same overall NPV per metric ton of dry waste (produced or accepted). Let us assume that all producers, if they send waste to a W2X facility, send all of their waste to a single W2X facility; an inspection of the optimality conditions for the waste producer optimization problem will show this assumption to be justified barring various forms of degeneracy. For incinerators, we have

$$365\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)p_{share,n}\sum_l q_{ln}^I = f_I(q_{ln}^I) + 365\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)\sum_j p_{jl(j)n}q_{jl(j)n}^W$$

$$p_{share,n}q_{jl(j)n}^W = f_W(q_{jl(j)n}^W) - p_{jl(j)n}q_{jl(j)n}^W$$

where

$$f_W(q_{jl(j)n}^W) \equiv -\left[c_{jl(j)n}q_{jl(j)n}^W + \left(q_{dispose,jl(j)}^W - q_{jl(j)n}^W\right)p_{dispose,l(j)}^W\right]$$

This produces a linear system of equations in $p_{share,n}$ and $p_{jl(j)n}^W$ that we can represent (with some abuse of notation) as

$$\begin{Bmatrix} f_I\left(q_{ln}^I\right) \\ f_W\left(q_{jl(j)n}^W\right) \end{Bmatrix} = \begin{bmatrix} 365\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)\sum_l q_{ln}^I & 365\left(\frac{1-(1+\delta)^{-(T-T_{build})}}{\delta(1+\delta)^{T_{build}}}\right)q_{jl(j)n}^W \\ q_{jl(j)n}^W & q_{jl(j)n}^W \end{bmatrix} \begin{Bmatrix} p_{share,n} \\ p_{jl(j)n} \end{Bmatrix}$$

We can then solve this linear system for $p_{share,n}$ and $p_{jl(j)n}^W$. To calculate the gate fees for the profit-sharing case where W2X processor NPV (per metric ton dry waste accepted) is the same as the NPV of the waste producer cost reduction (per metric ton dry waste produced), we substitute

$$f_W\left(q_{jl(j)n}^W\right) \equiv -\left[c_{jl(j)n}q_{jl(j)n}^W - q_{jl(j)n}^W p_{dispose,l(j)}^W\right]$$

for

$$f_W\left(q_{jl(j)n}^W\right) \equiv -\left[c_{jl(j)n}q_{jl(j)n}^W + \left(q_{dispose,jl(j)}^W - q_{jl(j)n}^W\right)p_{dispose,l(j)}^W\right]$$

and repeat the calculation process as before. Analogous calculations hold for HTL facilities.

Appendix D – Transport & Disposal Costs

Table 18. Average marginal costs per hour (ATRI, 2023)

Component	CPH
Vehicle Related	
Fuel Costs	$25.84
Truck/Trailer Lease or Purchase	$13.37
Repair & Maintenance	$7.89
Truck Insurance Premiums	$3.57
Permits & Licenses	$0.60
Tires	$1.81
Tolls	$1.14
Driver Related	
Driver Wages	$29.20
Driver Benefits	$7.37
TOTAL	$90.78

Table 19. Average marginal costs per mile by sector (ATRI, 2023)

Truck sector	CPM
LTL	$2.34
Specialized	$2.44
TL	$2.15

Appendix E - Massachusetts MSW Summary

According to the 2022 MADEP Solid Waste Report, in 2020 the state of Massachusetts disposed of 5.92 million wet short tons of waste, a 7% increase over 2019 (MADEP, 2022). MSW and non-MSW waste represented 4.39 and 1.53 million t/y, respectively. Landfilling accounted for 18% of in-state disposal, while incineration accounted for the remaining 82% of in-state waste treatment. Massachusetts' long-term landfill capacity shortage, which explains the high percentage (38%) of exported waste.

Table 20. State-wide solid waste in 2020

Component	2020
Total Disposal	**5,920,000**
In-State Disposal	**3,700,000**
Landfill	660,000
MSW	570,000
C&D	-
Other	90,000
Combustion	3,040,000
MSW	3,020,000
Non-MSW	20,000
Net Exports	**2,220,000**
Exports	2,470,000
MSW	1,040,000
Non-MSW	1,430,000
Imports	250,000
MSW	240,000
Non-MSW	10,000

State-wide totals for in-state disposal and combustion are generally consistent with the latest MADEP facility reporting data for active MSA landfills (MADEP, 2023a) and combustion facilities (MADEP, 2023b). All the active landfills and combustion facilities in MA are located within the Boston CSA. There are no active landfills or incinerators in western MA and all non-recycled solid waste is exported.

Table 21. Total in-state solid waste landfilled (2022)

ID	TYPE	NAME	TPY	Lifetime
172356	MSW	Bourne	218,098	2040
172448	MSW	Dartmouth	91,670	2029
172728	MSW	Middleborough	56,402	2031
172753	MSW	Nantucket	5,055	2028
39885	MSW	Westminster	265,541	2032
			636,766	

Table 22. Total in-state solid waste combustion (2022)

Facility	Type	Avg. Capacity (wet t/d)	2022 Capacity (wet t/y)	Nameplate Capacity (MW)[1]	2022 Generation (MWh/y)[2]	Capacity Factor (%)
Haverhill	Mass Burn	1650	536,736	45	349,571	89
Millbury	Mass Burn	1500	514,762	48	318,641	76
North Andover	Mass Burn	1500	526,461	40	228,822	65
Rochester (SEMASS)	RDF	3300	1,088,908	78	590271	56
Saugus	Mass Burn	1500	552,693	54	214,422	45
		9,450	**3,219,560**	**265**	**1,701,727**	**(Avg.) 72**

1- Nameplate capacity values published on company websites
2- Annual net generation data reported on Form EIA-923 (2022 Annual Final)

Waste composition data are reported in for at all Massachusetts combustion facilities, which treat 82% of total solid waste.

Table 23. Waste composition for in-state (MA) combusted waste

Facility	Organics	Paper	Plastic	C&D	Other	Metal	HHW	Glass	Electronic	Sum
Haverhill	28%	22%	16%	13%	10%	6%	3%	2%	1%	100%
Millbury	33%	19%	16%	8%	13%	3%	6%	2%	0%	100%
N. Andover	25%	18%	14%	19%	12%	5%	5%	2%	1%	100%
Rochester	30%	24%	15%	12%	8%	5%	3%	3%	0%	100%
Saugus	32%	22%	14%	10%	11%	6%	3%	3%	0%	100%
Mean	*30%*	*21%*	*15%*	*12%*	*11%*	*5%*	*4%*	*2%*	*0%*	*100%*

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