Life Cycle Assessment of the Management and Disposal of Four Solid Waste Streams in Iowa

HedgeRow Analysis for SCS Engineers

Date of Report: September 30, 2026

1. Goal of the Study

The Iowa Department of Natural Resources (DNR) commissioned SCS Engineers to produce this life cycle assessment (LCA) as part of its broader effort to develop a Sustainable Materials Management – Vision for Iowa, and HedgeRow Analysis delivered it on its behalf. The primary goal of the study is to support policy evaluation and planning by providing a transparent, science-based assessment of the potential environmental impacts associated with current and potential solid waste management practices across the state. The LCA will inform internal decision-makers within DNR, regulatory agencies, and stakeholders as they work toward reducing waste and its impact on air, water, and land resources.

In alignment with the state's vision for a more integrated and forward-looking solid waste policy, the study's primary goal is to examine the present-day waste management baseline impacts for four key waste streams from residential and industrial / commercial / institutional (ICI) sources: food waste, plastic film, textiles/leather, and mixed aluminum waste (i.e., deposit beverage containers, non-deposit containers, and other aluminum waste). The secondary goal is to use those results to compare alternative waste management scenarios. Because the results are intended to support comparative assertions disclosed to the public, the study includes an independent critical review by a panel of LCA experts , consistent with the requirements of ISO 14040 and 14044 (2006).

This study first evaluates the environmental and energetic performance of Iowa's municipal solid waste (MSW) management system. To do so, we describe the flows of waste in four categories from residents and businesses: food waste, plastic films, textiles/leather, and mixed aluminum. Next, we build foreground life cycle inventory models (accounting for all direct inputs and emissions) representing waste management and treatment pathways for the system as it is at present (baseline), as well as for three alternative waste management scenarios: increased aluminum recycling, reduced fugitive methane emission using advanced methane capture and handling systems at landfills, and the development of a centralized textile recycling management system, ultimately bound for out-of-state mechanical recycling into fiber. When combined with background processes from reputable life-cycle databases (accounting for all indirect inputs from nature and emissions that occur upstream in the supply chains of the direct inputs to our models), the resulting complete LCA allows the calculation of the overall environmental impacts for all waste management scenarios, which can thus be compared in a consistent and evidence-based manner. and compares them to the alternatives.

The LCA model is constructed in a way that will enable Iowa DNR to revisit the analysis in the future to reflect changes in MSW flows, updates to the collection and transportation of waste, or test other alternative waste management scenarios.

2. Scope of the Study

2.1 Product, Product System and Function

The product system under assessment is the solid waste management system in Iowa which accepted and treated 2.4 million metric tons (2.7 million short tons) of MSW in 2024. The study evaluates the environmental performance of different management options (including recycling where applicable, and/or disposal) for four specific waste streams: food waste, plastic film, textiles/leather, and mixed aluminum; see definitions and percent of waste composition in

Table 1. These waste streams make up 33% of total residential solid waste flows and 32% of Institutional/Commercial/Industrial (ICI) solid waste (2022 Iowa Statewide Material Characterization Study).

The Iowa Department of Natural Resources (DNR) has been evaluating disposed solid waste and recycled materials in Iowa through statewide material characterization studies, facility surveys, and recycling infrastructure evaluations for many years. In 2022, the DNR held more than 30 engagement events with stakeholders representing a variety of industries, solid waste planners, elected officials, state, county, and local staff, and the public to identify, prioritize, and develop a Sustainable Materials Management Plan (Plan) for the State of Iowa. The results of these efforts informed the DNR's decision to select the following for the development of Life Cycle Assessments: Food Waste; Textiles; Plastic Film; and Aluminum. These materials represent some or all of the following: Large majority of the waste stream; Assumed greatest potential for diversion improvements; and Increasing quantities of materials in the waste stream based on historical data.

The other remaining residential and ICI waste flows (1,819,449 tons, see Figure 1) and the 1.27 million metric tons (1.4 million short tons) collected from the Construction and Demolition Debris (C&D) sector were excluded from this study, at the request of the client. The reason for the categorical exclusion of C&D waste is because these materials are generated and managed through waste management systems that are distinctly different from, and outside of the defined functional unit of, this residential and ICI MSW-focused LCA. The categories of waste excluded from residential and ICI sources include: paper, other soft and rigid plastics, ferrous and other non-ferrous metals, glass, yard waste, consumer products and e-waste, hazardous wastes, and diapers.

At a high level, we assumed a 40.6% residential and 59.4% ICI waste disposal split for the waste flows included in the study (Figure 1). These data were collected by SCS from 10 waste disposal facilities in Iowa to develop the sampling plan for the 2022 Iowa Material Characterization Study (SCS personal communication). The annual tonnage handled at

these locations accounts for 38% of statewide MSW flows and the locations chosen for the study were based on facility size, location, and types of waste handled to be a diverse and representative sample of Iowa.

The primary function of the system is the safe and effective management of solid waste from the categories chosen, involving the collection, transportation, processing, treatment, recycling, and/or disposal of post-consumer waste materials from Iowa's municipalities. This includes diversion activities (e.g., recycling) as well as final disposal (e.g., landfilling or incineration), with the goal of minimizing environmental burdens while maintaining public health and operational feasibility. These processes may produce products such as recycled aluminum ingots, textile fibers, compost, or energy for electricity.

2.2 Functional Units

The functional unit for this study is defined as:

"The management of 1 metric ton (1,000 kg) of [specific waste stream] generated in Iowa over a year"

This unit serves as the reference flow to which all inputs and outputs are normalized, providing a consistent basis for evaluating and comparing alternative waste management options. Each waste stream is modeled individually using the same mass-based functional unit to verify comparability across waste management scenarios.

The functional unit includes all processes from the point of collection through final treatment or disposal, and accounts for any outputs such as recovered energy or recycled materials. This approach is consistent with standard practice in waste management LCA studies and reflects the units commonly used by state and municipal agencies when reporting waste generation and diversion quantities.

Additionally, we provide results for the impact for the total waste flows in Figure 1 for each category and sector for Year 2024, by multiplying the impacts per Functional Unit by the tonnage for each category/sector.

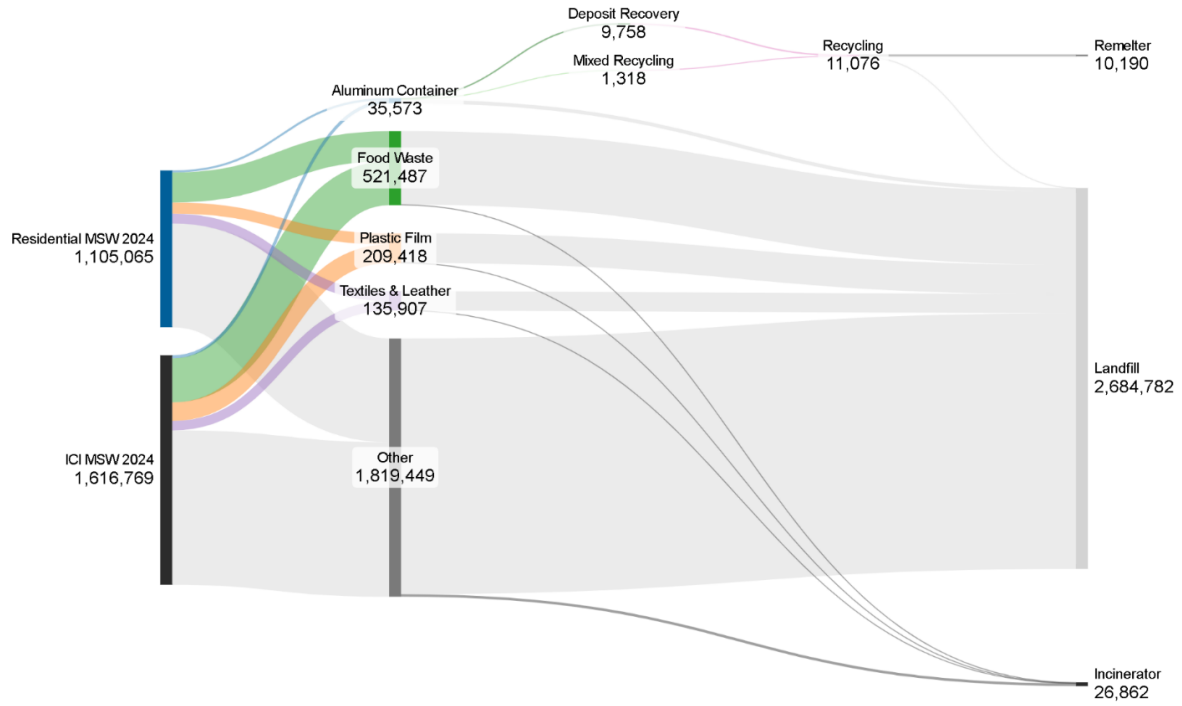


Figure 1 - Sankey diagram of residential and ICI waste flows showing the source generating the waste, the gross flows (in original units of short tons) in each of the 4 MSW categories considered and the ultimate management pathway. In the current system, 99% of waste materials considered in the study are treated by landfill, and the remainder are incinerated or recycled. Data sources include: SCS Studies for the IA DNR between 2020-2025. Flow amounts are presented in their original units of short tons.

Table 1 - Waste category definitions, examples, and percent of total waste flows in residential and ICI sectors. Source: 2022 Iowa Statewide Material Characterization Study. Complete list of waste categories and their definitions provided in Appendix A.

Waste Category	**Definition**	**% of Res. MSW flows**	**% of ICI MSW flows**
Food Waste	Food material resulting from the processing, storage, preparation, cooking, handling, or consumption of food. Includes food material still in the original packaging but does not count the mass of the packaging materials.	Loose food: 14.8% Packaged: 4.3%	Loose food: 14.5% Packaged: 4.7%

Plastic Films	Examples include plastic shopping bags, garbage bags and other types of plastic bags, painting tarps, food wrappers such as candy-bar wrappers, mailing pouches, bank bags, X-ray film, metalized film, plastic food wrap, and source contaminated commercial/industrial film.	Retail bags: 1.0% Other plastic film: 6.1%	Retail bags: 0.3% Other plastic film: 7.8%
Textiles and Leather	Items composed of at least 50% natural or manmade textile and leather. Items such as yarn, thread, clothing, apparel, shop rags, blankets, pillows, shoes, stuffed toys, backpacks; includes leather items such as wallets, purses, belts, shoes, and scrap leather.	6.3% (not disaggregated)	4.1% (not disaggregated)
Mixed Aluminum	Includes aluminum beverage containers subject to IA's container deposit law and those made from aluminum other than IA deposit containers; food containers made from aluminum and the thin aluminum foil sheeting used to cover and wrap food.	Deposit containers: 0.5% Non-dep: <0.1% Foil and food containers: 0.2%	Deposit containers: 0.5% Non-dep: <0.1% Foil and food containers: <0.1%
Total	Total flows in 4 waste categories	33%	32%

2.3 System Boundaries

The system boundaries for this LCA begin at the point of waste generation—defined as the moment a material is discarded by a residence, business, or institution within an Iowa municipality —and extend through the entire downstream waste management system within the state and transport to final processing destinations out-of-state (if required). Overall, the temporal and geographic boundaries of this LCA reflect the modern MSW management system of Iowa in 2025.

This includes:

- Collection and transportation of waste to a transfer station or sorting facility, or directly to a landfill or incinerator
- Handling, sorting, and consolidation at intermediate processing facilities
- Transportation to and treatment at final destinations, including:
    - Landfills (with or without energy recovery)

    - Incineration
    - Recycling facilities
- Rejected materials from recycling facilities

Materials entering the system are treated as zero-burden at the point of disposal in accordance with the cutoff approach for LCA. That is, any upstream environmental burdens associated with the production and use of the discarded materials are excluded from this assessment.

2.3.1 System expansion with substitution

In addition, potentially displaced products and energy as a result of end-of-life energy recovery and/or material recycling were included within the system boundary through the application of system expansion. As a result, the study boundary encompasses two distinct but linked systems: (1) the waste management system, including collection, sorting, processing, and final disposal; and (2) the expanded (displacement) system, representing the production of materials and energy displaced by recovered outputs.

Accordingly:

- Recyclable materials recovered from the waste stream are modeled to displace the production of equivalent virgin materials or synthetic fertilizers, and associated credits are allocated to the system.
- Energy recovered from incineration or landfill gas recovery is modeled to offset average grid mix electricity from the Midwest Reliability Organization (MRO) NERC region, and credits for avoided emissions are accounted for.
- Material losses and process inefficiencies (e.g., yield losses in recycling) are explicitly modeled.

All upstream inputs to the modeled waste management and material recovery processes (e.g., collection vehicles, infrastructure including buildings, sorting and processing equipment and their operation, including electricity, fuels, and ancillary supplies) were modeled on a cradle-to-gate basis, from raw material extraction through delivery of the product or service to the point of use within the waste management system. Labor, travel by staff, and maintenance were excluded. This approach was applied consistently to foreground waste management processes, associated manufacturing and processing activities, and products and energy included through system expansion.

2.3.2 Geographic and Temporal Boundaries

The system is geographically bound to waste materials generated by an Iowa person or business that are ultimately processed by and managed at Iowa waste transfer facilities, landfills, and the Ames Resource Recovery Plant incinerator, including energy recovery (electricity only) (2.3.3 Cut-off rules and Exclusions (with justification)

The focus of this LCA is on the solid waste flowing to landfills and incinerator in Iowa and the diversion of such materials to recycling facilities, where applicable. On-site waste management and disposal, such as backyard composting, private septic systems, or industrial self-managed waste systems, are excluded from the system boundaries due to the limited availability of standardized, verifiable data and to maintain comparability across publicly managed waste streams. No processes other than the above were excluded from the assessment. Completeness and truncation were evaluated by verifying that all material and energy flows contributing more than 1% individually to mass, energy, or environmental relevance were included in the life cycle inventory, and that the cumulative contribution of excluded flows remained below 5% across these same dimensions. This approach verifies that the aggregate effect of minor exclusions does not materially influence results or conclusions.

2.4 Methodological treatment of multi-functionality

In accordance with ISO 14040/14044, the study addressed all multi-output processes using a hierarchical decision framework designed to minimize allocation bias and preserve physical causality. Wherever feasible, allocation was avoided at the foreground level by applying system expansion with substitution, whereby the provision of recovered materials, fuels, or energy from waste management processes was modeled as displacing equivalent products in the broader economy (e.g., primary aluminum, textile fibers, or grid electricity). This approach verifies that environmental credits associated with material and energy recovery are attributed based on actual physical substitution, rather than through arbitrary partitioning of process burdens.

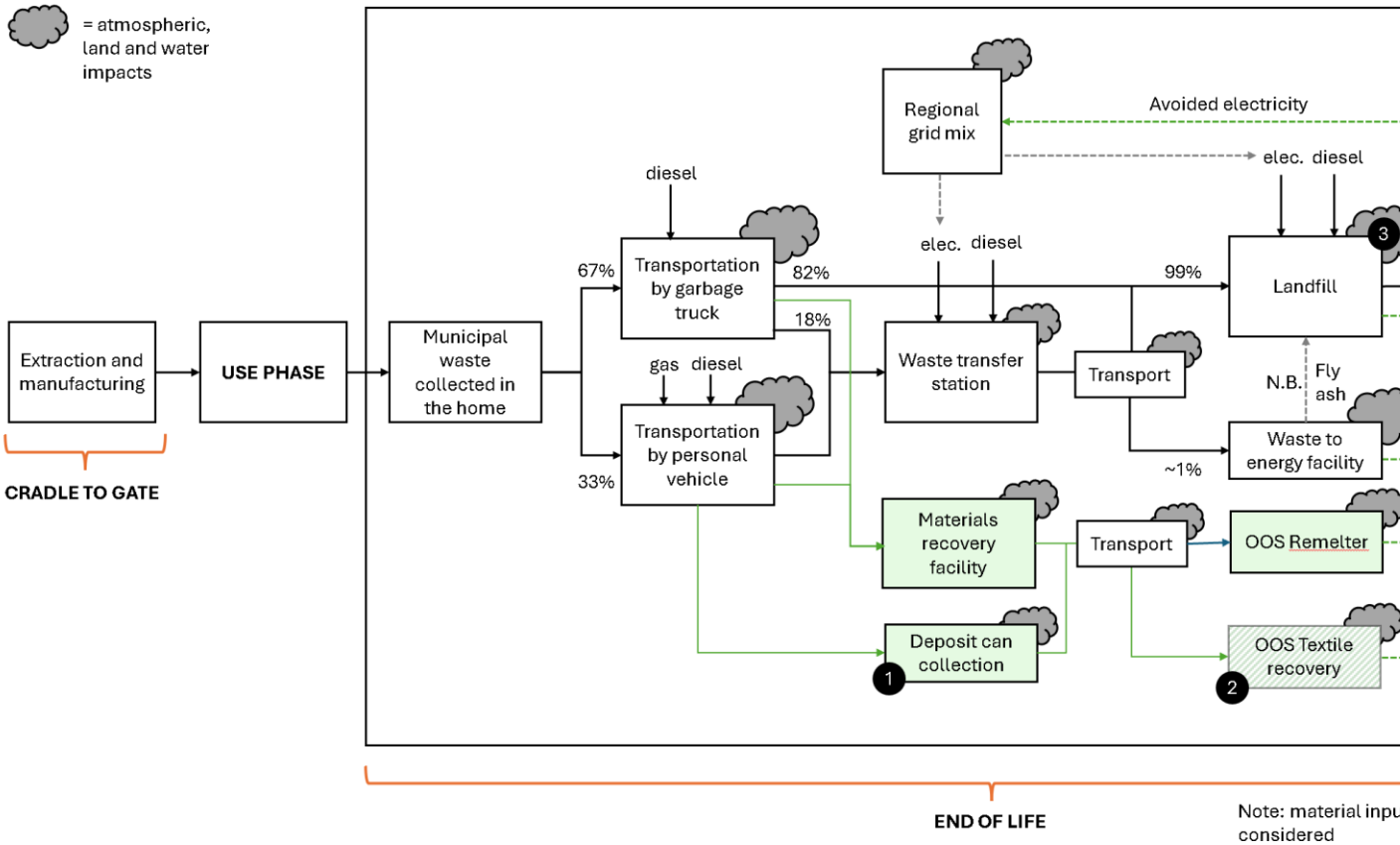


Figure 2). Since no aluminum remelter facilities or textile fiber recyclers operate within Iowa state limits, aluminum and textiles separated and prepared for recycling in-state are sent to out-of-state recyclers. Transport to and processing at these facilities is included in the boundaries of our analysis.

As noted above, the contemporary MSW system model underlying this LCA is also time-bound – temporally reflective of 2020-2025 – using the most current and defensible process-level insights for a statewide solid waste management operation.

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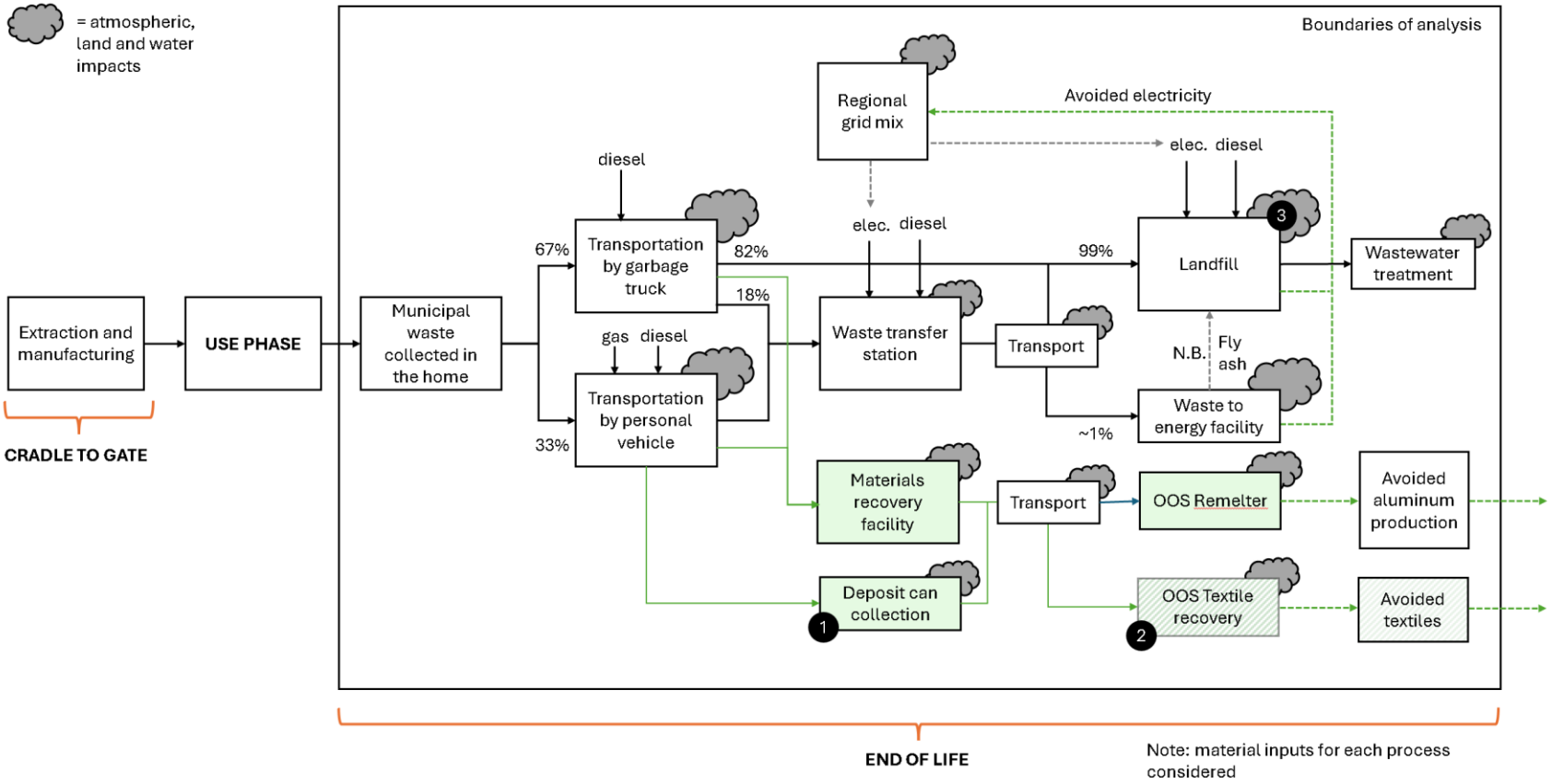


Figure 2 - Basic system boundary diagram. The LCA system boundary includes activities related to collection, transportation, processing, and waste management, in addition to activities displaced by the products of municipal solid waste management. The green pathways indicate material recovery. Note: for clarity sake, rejected materials pathway from materials recovery facilities, textile recovery, and remelter to landfill not shown. Alternative treatment scenarios (increased aluminum deposit recycling, textile recycling, and landfill gas capture improvements as described in Section 2.5) indicated by numbers 1-3.

For multi-output processes where system expansion was not technically feasible—either due to non-substitutable co-products or because allocation was already embedded in the background life cycle inventory database—the study followed the ISO 14044 hierarchy by applying allocation prioritized based on underlying physical relationships. Specifically, at the foreground level, mass-based allocation was applied to material co-products not intended for energy use, and energy-based allocation was applied to fuels and energy carriers. Economic allocation was not used for any multi-output processes modeled at the foreground level, to avoid distortion from price volatility and market effects unrelated to physical causality. However, economic allocation is still embedded in some database processes used to model background processes (e.g., co-production of virgin metals).

This combined use of system expansion and (primarily physically based) allocation is consistent with the requirements of ISO 14044 §4.3.4 and §4.3.5 and verifies that all reported life cycle results reflect causal, transparent, and reproducible relationships between waste management activities and their environmental outcomes.

2.5 Scenarios for Alternative Waste Management

The alternative scenarios evaluated in this study were developed in accordance with the principles and requirements outlined in ISO 14040/14044 for Life Cycle Assessment, with particular emphasis on goal and scope definition, transparency of assumptions, and consistency in system boundaries.

Iowa Department of Natural Resources wishes to understand the potential impacts of alternative waste management strategies within the state while maintaining methodological rigor and avoiding speculative infrastructure or policy assumptions. In alignment with ISO guidance, alternative scenarios were selected to be both technically representative and decision-relevant, providing modeled outcomes that can reasonably inform near-term policy and stakeholder discussions. Sensitivity analyses were incorporated to evaluate the robustness of results to key parameter uncertainties, consistent with ISO recommendations for interpretation and uncertainty evaluation.

In defining alternative scenarios, special consideration was given to maintaining closed system boundaries and avoiding burden shifting, as required under ISO 14044. ISO standards emphasize the need for clearly defined reference flows and fate of materials across all scenarios. In cases where diverted waste streams could not be reliably traced without introducing speculative assumptions, the scenarios were excluded to preserve the integrity and comparability of results. Instead, the assessment focuses on interventions where changes can be directly attributed to operational or policy-level actions, and where the baseline system remains clearly defined.

Based on these criteria, the three scenarios evaluated were:

1. Increase Aluminum Deposit Container Recycling: Using incentives, education and/or policy changes to improve recovery rates for aluminum deposit beverage

cans from the state's current estimate of 41.8% (2022 levels) to 64%, which is the 2024 average among states with deposit programs (Container Recycling Institute, 2025.)

2. Centralized Textiles Collection for Out-of-State Recycling: Diverting 15% of residential textile waste from landfills to a centralized processor in-state, and then to out-of-state mechanical recycling facilities to recover usable fibers. The national level recycling rate for textiles is 14.7% (EPA 2018 Fact Sheet)

3. Reducing Fugitive Methane Emissions from Landfills: A 10% reduction in relative terms by increasing the efficiency of current landfill gas collection and beneficial use systems, and improving landfill operations (i.e., improved daily cover, compaction densities, etc.)

These scenarios were identified as feasible within current policy and operational frameworks and capable of delivering measurable emissions impact reductions within the assessment timeframe. For each scenario, the functional unit, system boundaries, and allocation approaches were applied consistently with the baseline case. Where required, we include the additional infrastructure required, (e.g. for collecting and processing textile waste.) For the increased aluminum container recycling and landfill gas reduction, we assume no additional infrastructure investments are required to meet these objectives. Credits for avoided virgin material production were applied where appropriate, following ISO-consistent substitution principles, while care was taken to prevent double counting of emissions or benefits. The selected scenarios reflect a balance between environmental relevance, data availability, and practical feasibility of implementation, supporting that results are both scientifically robust and suitable to inform decisions at the state-level.

2.6 Impact Categories and LCIA Methods

Life cycle impact assessment (LCIA) was conducted to translate the life cycle inventory results into a limited set of midpoint environmental indicators consistent with the goal and scope of this study and the requirements of ISO 14044. Indicator selection was intentionally constrained to focus on impact categories most relevant to the evaluated waste streams, management pathways, and displacement mechanisms, and to align with Iowa DNR criteria for decision making. This study applied a U.S.-policy-relevant Life Cycle Impact Assessment (LCIA) framework based on the EPA's Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) 2.1 methodology and Cumulative Energy Demand (CED) metric to quantify environmental impacts across all modeled LCA stages.

Note: This report applies the U.S. EPA's TRACI 2.1 life-cycle impact assessment method, which includes standardized impact categories such as "global warming potential." These terms are retained solely to accurately reference the established scientific nomenclature of the TRACI method and do not imply policy positions or interpretations. Throughout the

narrative sections of this report, the analysis refers to these results using neutral terminology such as “emission-related impacts” or “warming-related emission impacts” to align with federal communication requirements while preserving the scientific integrity of the LCIA framework.

The LCIA metrics applied in this study are emissions impacts (in kg CO2-eq) and eutrophication (in kg N-eq), characterized using TRACI 2.1 (Bare 2011), and cumulative energy demand (CED), reported as total and non-renewable primary energy use (in MJ, 1 x 10^6 J, or TJ, 1 x 10^{12} J) (Frischknecht et al. 2007). Emissions impacts are reported as kg or t CO_2-equivalent using a 100-year time horizon, eutrophication impacts are reported as kg N-equivalent, and cumulative energy demand reflects the total primary energy embodied in materials, fuels, and electricity, including both renewable and non-renewable sources.

- TRACI 2.1 was used to quantify emissions impacts (kg CO_2-eq) and Eutrophication (kg N-eq). These indicators are developed and maintained by the U.S. Environmental Protection Agency and are widely used in state-level environmental policy analysis, waste management planning, and regulatory LCA applications, verifying methodological alignment with U.S. decision-making contexts.
- Cumulative Energy Demand (CED) was used to quantify total life cycle energy use, disaggregated by renewable and non-renewable energy resources and expressed in MJ oil—equivalent (“energy harvested” approach, as per Frischknecht et al., 2015). This provided a consistent measure of the primary energy intensity and energy resource dependence of alternative waste management pathways.

Impact category selection was intentionally limited to air emissions impacts, eutrophication, and cumulative energy demand, as these indicators are most directly responsive to the modeled waste management and displacement processes, while expansion to additional TRACI categories was not expected to change comparative outcomes and would increase uncertainty relative to the study objectives. All selected indicators are midpoint metrics describing physicochemical mechanisms linking emissions and resource use to potential environmental impacts.

Reporting is structured to distinguish gross emissions, avoided emissions associated with displaced virgin production and electricity generation, and net results, consistent with ISO 14044 transparency requirements and established waste-sector LCA practice. The 100-year time horizon aligns with prevailing federal and state emissions policy frameworks, independent and objective accounting standards, and decarbonization planning horizons, making the results directly interpretable for policy-relevant decision support. Endpoint or damage-oriented indicators were not evaluated as they are not required for the comparative objectives of this study and would introduce additional value-based assumptions.

TRACI 2.1 provides two eutrophication indicators: a standard formulation that includes both short-term nutrient releases and characterization factors representing long-term mobilization of nitrogen from soils and waste repositories, and a “no long-term (no-LT)”

variant that reflects only near-term aquatic nutrient loading. In OpenLCA, these indicators are implemented as separate impact categories that apply alternative characterization factor sets to the same life cycle inventory, no changes are made to foreground processes, background data, allocations, or system boundaries. Accordingly, differences between the two metrics arise solely from the inclusion or exclusion of long-term fate assumptions embedded in the characterization factors. We chose the no-LT approach to distinguish immediate, operational eutrophication pressures from longer-term legacy effects, consistent with ISO 14044 transparency requirements.

Cumulative Energy Demand (CED) results are reported using two complementary indicators: "Energy resources: non-renewable" and "Total CED." Both metrics are calculated in openLCA from the same life cycle inventory and differ only in their characterization factor sets, with Total CED representing the sum of renewable and non-renewable primary energy inputs, and the non-renewable indicator isolating fossil, nuclear, and other finite energy resources. Electricity produced from MSW through the capture of landfill gas or combustion in a waste to energy incinerator is burden free under the cutoff method, and a credit is applied representing displaced energy from the grid. No changes are made to foreground processes, background data, allocations, or system boundaries between these indicators. Results are presented for both metrics to distinguish overall system energy intensity from dependence on finite energy resources, consistent with ISO 14044 transparency principles. Electricity produced from captured landfill gas and from combusting waste in incinerators is considered burden-free when using the cut-off method. Electricity output from waste management is given a credit for the avoided use of electricity from the grid and therefore the impact is that reduces both total and non-renewable CED for those end-of-life processes.

Together, TRACI 2.1 and CED provided a complementary and ISO-consistent LCIA framework: TRACI captured the environmental consequences most relevant to U.S. waste and materials policy, while CED captured the upstream primary energy requirements embedded in material recovery, processing, and disposal. Additional impact categories not considered in this study are provided in Appendix E.

The LCIA results presented in this study are relative expressions of potential environmental impacts and are not intended to predict actual impacts at the category endpoint level, nor the exceeding of environmental thresholds, safety margins, or risks.

No normalization, weighting, or aggregation of impact categories was performed. Results are presented by indicator individually, consistent with ISO 14044 requirements.

2.6.1 LCIA calculation tools and implementation

LCIA calculations were performed using OpenLCA 2.5.0, combining project-specific foreground inventories with background life cycle datasets from the ecoinvent 3.11 Cutoff database to generate complete inventories for each waste stream and management

pathway. Characterization factors corresponding to the selected LCIA indicators were applied consistently across all scenarios and reporting levels.

The characterization models, characterization factors, category indicators, and reference units associated with the applied LCIA methods were obtained directly from the corresponding LCIA method implementations within OpenLCA.

Full documentation of the characterization methodologies, characterization factors, and impact category development for the applied LCIA methods is available in the original method publications (Bare, 2011; Frischknecht et al., 2007). LCIA implementation within this study was conducted using the corresponding method libraries in OpenLCA 2.5.0 (GreenDelta GmbH, 2025).

Characterization factors represent relative expressions of potential environmental impacts and do not predict actual environmental impacts, category endpoint exceedances, safety margins, or risks.

LCIA results were exported to Microsoft Excel for quality control, aggregation, and visualization. Internal checks were conducted to confirm consistency of material flows across pathways and to verify that characterization was applied uniformly throughout the model. Table 2 provides an overview of the software tools, versions, databases, and LCIA method files used in the study.

2.6.2 Result structure and reporting conventions

LCIA results are reported using a gross–credit–net structure to verify transparency in the treatment of avoided burdens. Gross impacts represent emissions and resource use associated with waste collection, transportation, processing, and waste management activities. Gross avoided impacts (credits) represent displaced upstream production and energy generation resulting from material recovery or energy recovery.

Net impacts are calculated as the difference between gross impacts and avoided impacts:
Net impact = Gross impacts − Avoided impacts

Negative net values indicate a net environmental benefit relative to the displaced reference system, while positive net values indicate a net increase in environmental burden (Figure 3).

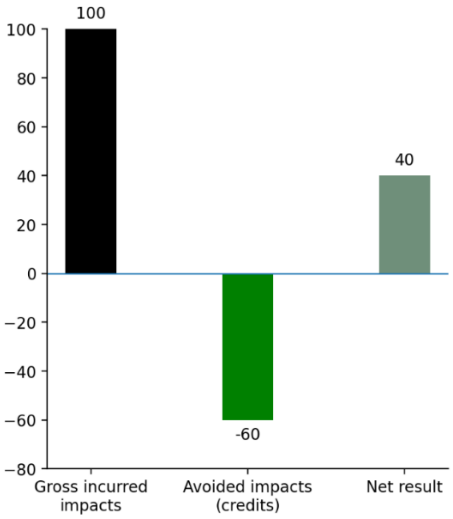


Figure 3 - Conceptual illustration of gross impacts, avoided impacts, and net results.

Table 2 - Software tools, versions, and LCIA methodologies used in the study.

Software tool / LCIA method	**Version**	**Comments**
OpenLCA	2.5.0	Primary LCA modeling platform used to construct product systems, implement system expansion (avoided burden), and perform life cycle impact assessment calculations. All scenarios were evaluated using identical model structures, foreground processes, and calculation settings. GreenDelta GmbH. (2025). *openLCA 2.5.0*. GreenDelta GmbH. openLCA official website
Ecoinvent	3.11 Cutoff	Background life cycle inventory database used for upstream production of fuels, electricity, materials, and services. Cutoff system model applied; both system and unit process datasets were used as appropriate. Providers were selected to best represent U.S. conditions where available.
TRACI 2.1	2.1 (openLCA LCIA method package)	Bare, J. C. (2012). *Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI), version 2.1: User's manual* (EPA/600/R-12/554). U.S. Environmental Protection Agency. Access document
Cumulative Energy Demand	2021	Cumulative Energy Demand (Total and Non-renewable). Frischknecht, R., Jungbluth, N., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Humbert, S., Margni, M., Nemecek, T., & Spielmann, M. (2007). *Implementation of Life Cycle Impact Assessment Methods*. ecoinvent report No. 3, v2.0. Swiss Centre for Life Cycle Inventories.
Custom foreground processes	Project-specific	Waste generation, collection, sorting, processing, disposal, and recovery processes developed using project data, engineering assumptions, and literature sources.

Allocation / system expansion	–	Avoided burden approach used to credit displaced virgin material production and electricity generation.
Electricity grid modeling	–	Regional (MRO) or national Ecoinvent electricity providers used unless otherwise specified.
Functional unit	–	Defined per waste stream and sector as described in the Methods section.
Normalization / weighting	–	No normalization or weighting applied.
Uncertainty analysis	–	Non-parametric bootstrapping was used to estimate empirical uncertainty bounds for GIS-derived transportation distances and aluminum recycling residual rates. Additional deterministic sensitivity scenarios were evaluated in OpenLCA for key operational and behavioral parameters e.g., methane emissions, recovery efficiencies, allocation assumptions, and LCIA method choice.

2.6.3 Functional unit and scaling approach

Two complementary reporting bases are used in this study:

- Pathway-level results, reported per 1 metric ton of material entering a specific waste management pathway for each waste stream; and
- State-level results, calculated by scaling pathway results per metric ton by statewide material flow quantities.

The per-metric-ton pathway basis enables direct comparison of waste management options within and across waste streams. Statewide results reflect the aggregate environmental implications of baseline and alternative management configurations.

Statewide impacts were calculated by scaling pathway-specific LCIA results (expressed per metric ton of material managed) by corresponding statewide material flows and summing across all Residential and ICI waste management pathways. Formally, statewide impacts were computed as:

$$\text{Statewide Impact} = \sum_{i \in P} (\text{Impact per ton}_i \times \text{Statewide tons}_i)$$

where Prepresents the set of all modeled Residential and ICI pathways, including landfill, incineration, and recycling routes for aluminum containers, food waste, plastic films, textiles, and leather.

2.7 Data Sources

Data used in this study consisted of a combination of primary data collected by the SCS/HRA project team and secondary data obtained from established life cycle inventory

(LCI) databases and peer-reviewed literature. Primary data included Iowa's waste characterization, transfer station, materials recovery facility (MRF), and landfill facility surveys, tonnage reports, operational data, energy use, and GIS facility point layers to calculate average transport distances, mostly from SCS Engineers. Secondary data were sourced primarily from Ecoinvent version 3.11, EPA's WARM and MSW Decision Support tool, Version 16, IA-specific meteorological data from the National Weather Service (NWS), industry and company disclosures, Doka WasteTools 2023 model, and relevant academic and technical literature (Table 3).

All data were selected to verify appropriate temporal, geographic, and technological representativeness for the study context. Temporal representativeness targeted data from 2019–2024. Geographic representativeness reflected, in order of priority: State of Iowa, U.S. Midwest, and continental U.S. Technological representativeness reflected current U.S. landfill practices, typical MRF technology, modern aluminum remelting, Iowa or U.S. vehicle fleet mixes, the Midwest Reliability Organization electricity generation grid mix and contextualized with IA temperature and precipitation data.

A data selection hierarchy was applied to verify consistency and transparency (Table 4Table 4). Where available, primary, site-specific data were used in preference to generic datasets. When primary data were unavailable, U.S.-specific secondary datasets were prioritized over non-U.S. data, and peer-reviewed or government sources were prioritized over industry or grey literature. Where multiple secondary datasets were available for the same process, selection was based on geographic representativeness, technological representativeness, temporal representativeness, data completeness, and consistency with other model elements.

Data gaps and missing information were addressed using proxy datasets that best matched the intended process in terms of technology, geography, and scale. When proxies were required, they were selected to be representative of typical operations, and when in doubt, we used a conservative approach when there was uncertainty about the degree to which the provider or process in Ecoinvent matched those employed in Iowa (i.e., favor those that had a greater energy consumption and/or emissions impact.) All major proxy uses were documented in Appendix B, Data Assumptions Table.

The modeling framework was further designed to verify consistent system coverage across all waste streams and management pathways, thereby avoiding systematic bias between product categories. For the same reason, preference was given to adapted life cycle inventory models from Ecoinvent, rather than mixing processes from other databases (e.g., Sphera or USLCI), which may apply different system boundaries, upstream assumptions, or end-of-life modeling conventions that could introduce unintended inconsistencies.

Consistency and reproducibility were maintained by applying the same data sources, modeling assumptions, and calculation procedures across all scenarios and waste streams unless explicitly justified. All LCI data, assumptions, and calculation steps were

1 archived in an openLCA database, Excel workbooks, and an LCI project repository to
2 support internal quality assurance and external critical review.

3 *Table 3 - Technical and peer-reviewed publications used.*

Source	Material Stream	Implications and Notes
Ecoinvent Association. Ecoinvent database version 3.11. Zürich, Switzerland: Ecoinvent Association. 2024. Retrieved August 2025, from https://ecoinvent.org	All	
Brown, Martin J., & Canepa, Peter. Technical Overview of the Waste Impact Calculator. Oregon Department of Environmental Quality, 11 April 2023.	MSW Waste Management Framework	Comparison point for LCI model construction
Doka, G. Guidance on How to Consider Waste Disposal in Inventories of Waste Producing Activities, Zurich, May 2023.	Food Waste, Textiles & Leather	Used to produce food waste and textile landfill LCI models
U.S. Environmental Protection Agency. Waste Reduction Model (WARM), Version 16 – Excel User's Guide (EPA 530 R 23 024). Office of Resource Conservation and Recovery, Washington, DC. Released December 2023.	All	Transportation assumptions
U.S. Environmental Protection Agency. (2012). Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI) version 2.1 user's manual. U.S. EPA, Office of Research and Development.	All	EPA developed this U.S.-centric chemical and environmental impact LCIA Methodology
Frischknecht, R., (2015). Cumulative energy demand in LCA: The energy harvested approach. The International Journal of Life Cycle Assessment, 20(7).	All	Basis of Ecoinvent CED impact assessment method
Krause, M., Kenny, S., Stephenson, J., & Singleton, A. Quantifying methane emissions from landfilled food waste (EPA/600/R-23/064). U.S. Environmental Protection Agency, Office of Research and Development. 2023.	Food Waste	Fugitive methane emissions
US EPA. From Field to Bin: The Environmental Impacts of U.S. Food Waste Management Pathways. U.S. Environmental Protection Agency Office of Research and Development EPA/600/R-23/065. October 2023.	Food Waste	Fugitive methane emissions
IPCC. 2019 Refinement to the 2006 IPCC Guidelines. Volume 5: Waste. 2019.	Food Waste	Fugitive methane emissions
Haupt, Melanie, et al., "Life Cycle Inventories of Waste Management Processes," Data in Brief 19. August 2018	All	Backbone of Ecoinvent waste management process-level data
New Jersey Department of Environmental Protection, Division of Sustainable Waste Management, Bureau of Planning and Licensing. (2022, August 5). Municipal solid	Textiles and Leather	Proportion of leather in MSW, differentiated from textiles

waste quantification and characterization of Burlington, Mercer & Passaic Counties		
US Department of Energy. Alternative Fuels Data Center – Vehicle registration counts by state. 2024.	All	Choice of process representing dropoff of MSW and recycling via personal vehicles
U.S. Environmental Protection Agency & RTI (MSW-DST). (2000). Data and Data Input Requirements for the Municipal Solid Waste Management Decision Support Tool. PDF: https://mswdst.rti.org/docs/Inputs_Document_OCR.pdf.	All	The MSW-DST default of 50% dedicated trips is numerically equivalent to attributing one-way travel (half the round trip) to the drop-off function when trip chaining is unknown
Circular Thrift and SWACO. 2025 Columbus Textile Collection Pilot Case Study. 2025.	Textiles and Leather	15% Losses in sorting and preparing textiles for recycling
Enking, J., Becker, A., Schüe, G., Gausmann, M., et. al. (2025). Recycling processes of polyester-containing textile waste – A review. Resources, Conservation & Recycling, 219, Article 108256.	Textiles and Leather	5-40% Losses in mechanical recycling of textiles
Moazzem, S., Wang, L., Daver, F., & Crossin, E. (2021). Environmental impact of discarded apparel landfilling and recycling. Resources, Conservation and Recycling, 166, Article 105338.	Textiles and Leather	20% losses in mechanical recycling of cotton, 55% recycled/45% virgin cotton requirement for quality improvement of recycled yarn

1

Table 4 - Data Quality Pedigree Matrix

DQ Dimension	Target DQ Goal	Target Score	Scoring Logic (1-5)
Reliability	Major impact-driving parameters are derived from documented, traceable, and technically sound sources.	≤3.	1 = Independently verified primary measured data for major drivers; fully documented methodology and QA/QC; 2 = Measured or calculated data with strong documentation and traceable sources for major drivers (e.g., SCS operational data, state datasets); 3 = Established, well-documented secondary LCI dataset (e.g., Ecoinvent, EPA); 4 = Proxy, engineering estimate, or literature-derived factor with limited documentation affecting major drivers; 5 = Unknown origin, undocumented assumption, or untraceable data affecting major drivers.
Temporal	Major impact-driving parameters reflect conditions during the study reference year or recent representative conditions.	≤3.	1 = All major drivers within +-3 years of study year; 2 = Major drivers within +- 5 years; 3 = Major drivers within +- 10 years; 4 = At least one major impact-driving parameter > 10 years old; 5 = Major drivers outdated or temporal coverage unknown.
Geographic	Data reflect Iowa-specific or regionally representative conditions.	≤3.	1 = Iowa-specific operational data; 2 = Midwest regional data; 3 = U.S. national average data; 4 = Non-U.S. proxy (e.g., EU ecoinvent dataset); 5 = Global average or geography unknown.
Technological	Modeled processes reflect actual technology deployed in Iowa or relevant U.S. systems.	≤3.	1 = Facility-specific or confirmed Iowa technology configuration; 2 = Representative U.S. technology consistent with Iowa deployment; 3 = Generic industrial average technology; 4 = Technology differs materially from Iowa context; 5 = Technology unknown or clearly mismatched
Completeness	All relevant material and energy flows influencing major impact categories are included.	≤3.	1 = All relevant flows included; no material gaps; 2 = Minor omissions unlikely to affect results; 3 = Some missing flows with limited influence; 4 = Known omissions affecting major drivers; 5 = Significant data gaps affecting results
Process Review	Data and modeling decisions have been reviewed and documented.	≤3.	1 = Independently peer-reviewed or externally verified dataset; 2 = Third-party reviewed dataset (e.g., ecoinvent); 3 = Internal technical review with documentation; 4 = Limited informal review; 5 = No documented review

2.8 Assumptions and Limitations

This LCA was developed based on a set of explicit structural, data, and methodological assumptions that define the scope, representativeness, and interpretability of the results. These assumptions are required to represent real-world waste management systems in Iowa within a quantitative life cycle modeling framework and should be considered when interpreting the findings.

The quality of the results depends on the quality, representativeness, and completeness of the underlying data. LCI data were compiled using a combination of primary data collected by the SCS and HRA project team, secondary data from established LCI databases, and publicly available technical and peer-reviewed literature (Table 3). Data sources were selected to reflect appropriate geographic, temporal, and technological relevance and to verify internal consistency across waste streams and scenarios. Where important parameters were uncertain or variable, sensitivity analyses were conducted to evaluate their influence on reported results (Table 5).

2.8.1 Structural and system assumptions

Total annual waste generation for Iowa was compiled from the Iowa Department of Natural Resources 2024 Tonnage Report Data, and the proportion of waste in the four waste stream categories flowing from residential and ICI sources using data from the 2022 Iowa Statewide Material Characterization Study by SCS, as well as the 2022 Recovery Rate Report for Deposit Beverage Containers by CRI.

Average collection distances for residential curbside pickup and commercial waste routes were assumed to be 32.2 km (20 miles; derived from EPA WARM v16) and average drop-off distances were calculated using a GIS analysis of the distance between population centers and the nearest landfill or transfer station. Transport was modeled using a combination of different capacity heavy diesel trucks (collection and interfacility transportation) and passenger vehicles (residential drop-off). Facility location shapefiles were derived from publicly available facility data hosted by the Iowa Department of Natural Resources Solid Waste Program (accessed in 2025). Coordinates were extracted and converted to ESRI shapefile format (NAD83 projection) for transport distance modeling.

In the absence of local trip-chaining data, we model residential self-haul travel to drop-off facilities using the MSW-DST default approach: a round-trip travel distance multiplied by the fraction of trips assumed to be dedicated to the drop-off purpose (default 0.50). This results in attributing 50% of the round-trip passenger vehicle travel to the waste or recycling system, with the remaining travel treated as multipurpose/incidentally shared with other travel purposes. This approach is documented in MSW-DST input defaults and equations and is applied in published LCI/LCA studies and datasets that cite MSW-DST defaults.

Material sorting and processing efficiencies were assumed to be 92% in aluminum recycling facilities (average of 8% residuals among Aluminum Waste Collection Facilities in Nov 2025 Survey), 0.9% additional loss at aluminum remelters (Ecoinvent v3.11). Textile recycling losses include 15% overall rejection of soiled or improper materials in the sorting phase (Circular Thrift for SWACO, 2025), and an average of 20% in the mechanical recycling phase (Moazzem, et. al., 2020). End-of-life technologies were assumed to represent current U.S. landfill practice with and without gas recovery, modern aluminum processing and remelting, grid-connected incineration, and fiber recycling.

A complete list of model assumptions can be found in Appendix B.

2.8.2 Data and proxy assumptions

Where primary facility-specific data were unavailable, secondary or proxy datasets were used that best represented typical U.S. technologies, regional averages, or conservative estimates. Proxy data were selected to reflect typical industry operations rather than optimized conditions. These assumptions may affect absolute impact magnitudes but are not expected to materially alter relative comparisons between management pathways. Influential parameters were evaluated through sensitivity analyses.

2.8.3 System expansion and substitution assumptions

Environmental credits from material and energy recovery were calculated using system expansion, assuming that recovered materials and energy displaced primary aluminum production, primary textile fibers and average MRO grid mix electricity, respectively on a 1:1 mass or energy basis. These assumptions are consistent with ISO 14044 but do not reflect short-term market behavior or future system evolution.

Specifically, the default assumption of 1:1 substitution of 100% virgin materials is consistent with a clear-cut “0:100” = “avoided burden” approach, which is widely employed in the LCA literature on waste management systems. It is acknowledged that this assumption arguably provides an indication of the upper bound of the range of possible environmental credits that could be attributed to material recycling.

Table 5 - Sensitivity analysis parameters and scenarios

Parameter	Description	Default	Low	High	Units	Notes / Impacted waste streams
Fugitive methane emissions	Emissions factors for landfill unit processes Note that for each of these scenarios, different assumptions were made for electricity generation per metric ton of food waste landfilled. See Section 3.3.	Food: 0.036	Food: 0.027 Non-food: 0.75x	Food: 0.047 Non-food: 1.3x	kg CH4/ kg waste	Landfilling of food, textiles, plastics. For non-food emissions, we used the same high and low proportion as food, multiplying the default value by 0.75 and 1.3 to determine low and high emissions rates respectively
Drop off distance	Distance traveled (round-trip) for residential waste and recycling drop off	MSW: 46 Mixed Al: 27 Deposit: 17	MSW: 42 Mixed Al: 22 Deposit: 14	MSW: 49 Mixed Al: 33 Deposit: 21	km	All residential MSW and recycling drop off. Derived from non-parametric bootstrap of primary data.
Aluminum recovery rates	Percent of total aluminum waste recovered through recycling deposit containers and mixed aluminum waste	Deposit: 41.6% Mixed Al: 10.8%	Deposit: 33% Mixed Al: 9%	Deposit: 50% Mixed Al: 13%	%	Uncertainty unknown, used +/- 20% relative change. Post hoc analysis
Aluminum processing losses	Percent residuals from aluminum sorting and processing	8%	6.6%	9.8%	%	Deposit and Mixed Al. Derived from non-parametric bootstrap of primary data.
Trip allocation	Percent of residential waste or recycling drop-off trip allocated to that purpose	50%	37.5%	62.5%	%	All residential waste streams
Curbside pickup participation rate	% of residences with curbside pickup	MSW: 67% Recycle: 45%	MSW: 55% Recycle: 32%	MSW: 78% Recycle: 64%	%	Derived from low and high estimates of urban and rural curbside pickup

Marginal vs MRO grid mix electricity	Avoided product for electricity produced by energy recovery during landfill or incineration	Default: electricity production, Midwest Reliability Organization, US Alternative: electricity production, natural gas, combined cycle power plant	Default: electricity production, Midwest Reliability Organization, US Alternative: electricity production, natural gas, combined cycle power plant	Default: electricity production, Midwest Reliability Organization, US Alternative: electricity production, natural gas, combined cycle power plant	All waste streams with energy recovery
LCIA Method	Comparison of chosen LCIA methods to alternatives	Default: TRACI 2.1 Climate and Eutrophication Alternative: Environmental Footprint (EF) 3.1 Climate, Eutrophication, Resource Use Fossils	Default: TRACI 2.1 Climate and Eutrophication Alternative: Environmental Footprint (EF) 3.1 Climate, Eutrophication, Resource Use Fossils	Default: TRACI 2.1 Climate and Eutrophication Alternative: Environmental Footprint (EF) 3.1 Climate, Eutrophication, Resource Use Fossils	Project level analysis of results

2.8.4 Limitations and interpretation context

This study does not model dynamic market responses, price effects, future technology or transportation changes, infrastructure expansion, or behavioral shifts associated with the current or alternative waste management options. All these effects would require a shift from the Attributional approach selected for this study to a consequential approach, with associated changes in terms of goal and scope, and inevitably increased uncertainty.

The use of geographically averaged background data may not capture facility-specific performance variability. Accordingly, results are most appropriate for comparative evaluation of waste management strategies under current conditions, rather than for predicting future market or technology outcomes.

The results are considered robust for relative ranking of, for example, aluminum recycling versus landfill disposal, textile waste diversion versus landfill disposal, and impacts under the current recycling regime compared to an expanded effort to achieve nationwide redemption rates. However, the results are more sensitive to assumptions regarding landfill gas capture efficiency, transportation distances and material recovery rates.

2.9 Criteria for inclusion in LCIA impact results

Individual contributions that fall below 1% of environmental significance have been discarded.

2.10 Critical Review

This report underwent critical review by EarthShift Global, who convened an independent third-party review panel consisting of a chair, Mariana Ortega, EarthShift Global, and two LCA experts: Terrie Boguski, Harmony Environmental and Simon Aumonier, Aumonier Consulting Limited.

3. Life Cycle Inventory

3.1 Introduction

This section presents the data sources, unit processes and their associated inventory data, and specific modeling assumptions used to construct the LCI for the waste management systems evaluated in this study. Individual unit processes are treated as modular building blocks and combined to represent complete waste management pathways for each waste stream and sector. Each unit process is defined by an associated inventory of material, energy, and ancillary inputs and outputs, which together form the quantitative basis for the LCA.

Unit process inventories used in this study include both pre-existing "off-the-shelf" datasets obtained from the Ecoinvent life cycle inventory database (v3.11) and custom foreground unit process inventories developed using project-specific data, direct observations, and engineering judgment. In all cases, foreground unit processes are linked to background life cycle data representing upstream production of fuels, electricity, materials, and services. Where available and appropriate, unit processes were selected directly from the Ecoinvent database based on geographic relevance, technological representativeness, and consistency with current waste management practices in Iowa and the broader U.S. context as outlined in Section 2. Custom foreground unit process inventories were developed to represent waste generation, collection, transportation, sorting, processing, and final disposition activities specific to the study scope where pre-existing datasets were unavailable or insufficient.

The following subsections describe the foreground unit process inventories developed for each modeled waste stream, along with the shared collection and transportation models applied across multiple systems. Together, these unit processes define the complete life cycle inventory used to quantify the modeled waste management pathways and provide the quantitative foundation for the life cycle impact assessment results presented in Section 4.

3.2 Collection and Transportation

Solid waste bound for landfills or incinerators, and source-separated or mixed materials being recycled must be collected and transported through the waste management system. In some waste management pathways, there are intermediate transportation steps which must be accounted for, such as the transport of waste between facilities prior to its final disposal. Additionally, byproducts of the waste management process, such as soiled or contaminated aluminum rejected from recycling centers or waste leachate collected from landfills must also be managed, transported and treated at landfills or wastewater treatment plants. In the following subsections, we describe the transportation processes that are common across the four waste streams being studied, as well as transportation processes that are specific to a waste management pathway, if they differ from the status quo.

3.2.1 Solid waste collection from residences and businesses to landfill

Waste collection begins at the point of generation. We assume that all four of our waste stream categories are transported via the same methods with some nuances for recycling and incineration. Residential waste is collected through two methods: curbside pickup by rear-loading waste-hauling trucks or drop-off via personal vehicles. ICI waste is collected primarily via front- or side-loading trucks. Mixed MSW is hauled in waste-hauling trucks from residential and ICI sources an average of 32.2 km (20 miles; EPA WARM, 2023). We used a GIS analysis to determine that households transporting their own MSW in

passenger vehicles travel 23 km one-way (46 km, or 28.6 mi round trip) to the nearest landfill or transfer station. For the portion of the analysis focused on passenger vehicle drop-off we assumed that:

1. Rural households were more likely to lack curbside pickup than urban households, and therefore we exclude population centers with populations greater than 50,000.
2. Households transport 44.5 kg (98 lbs.) per trip (SCS calculation using Iowa residential waste generation rate, US Census data, and assuming weekly trips).
3. 100% is allocated to errands, out of which 50% of the trip is for the purpose of dropping off the waste (U.S. EPA MSW-Decision Support Tool default 0.50 factor for dedicated drop-off trips).
4. The average passenger vehicle in Iowa has a fuel economy of 24.3 MPG equivalent to the EURO5 passenger vehicle mix (SCS analysis based on data from DOE Vehicle Registration Counts by State).

The formula used is:

$$(dropff_msw_\% \times dist_dropoff \times allocation_dropoff) \div dropoff_mass_calc$$

The result of this formula is a distance in km that is used in the unit process:

transport, passenger, car, EURO 5, fleet average | transport, passenger, car, EURO 5, fleet average | Cutoff, S

Other distances traveled, Ecoinvent processes used to model transportation, and related assumptions are provided in Table 6.

3.2.2 Solid waste collection from residences and businesses to Ames incinerator

The Ames incinerator serves the City of Ames and surrounding area and processes less than 1% of the state's MSW. According to annual reports, 97% of waste is delivered by waste-hauling trucks and the remaining 3% is dropped off by personal vehicles (RRP Annual Report 2024). We assume that MSW is hauled in waste-hauling trucks from residential and ICI sources an average of 32.2 km (20 miles; derived from EPA WARM), and that residences transporting their own waste travel 23 km (14.3 mi.) one-way (Table 6).

3.2.3 Mixed Recycling

In some areas of Iowa, curbside collection of residential recycling by waste haulers is provided (45%), but a majority of Iowa residents (55%) must collect and transport their recycling to drop-off collection centers (SCS calculations). All ICI recycling, except for aluminum deposit beverage containers, is assumed to be collected and transported by front- or side-loading waste-hauling trucks. Since there are no primary data available on the transportation of aluminum collected curbside from residential and ICI sources, we

use the most recent EPA WARM estimate of 32.2 km (or 20 mi.) for the average distance that the recycling is transported by waste-hauling trucks to Iowa MRFs.

For residential drop-offs we used GIS to calculate the population weighted average distance between population centers and waste collection locations that collect metal recycling (transfer stations, some landfills, MRF). The average distance is 26.9 km (16.7 mi) roundtrip. We use the assumptions about residential drop off from section 3.2.1.

3.2.4 Recycling of aluminum deposit beverage containers

We assume that aluminum deposit beverage containers are transported from residential sources via personal vehicle to 135 redemption centers located throughout the state. We assume that commercial businesses redeeming deposit containers transport them by "light commercial vehicles" such as a pickup truck, work van, or box truck. Using GIS, we calculated the population-weighted average distance between population centers (as proxies for household and business locations) and the list of operating deposit beverage container redemption centers distributed throughout the state. The population-weighted average distance is 17 km (10 miles) roundtrip. Following EPA MSW Decision Support Tool defaults, we allocate 50% of the trip for the purpose of redeeming the deposits and 50% to other errands. We assume that on average 17% of the 200 deposit containers returned per redemption trip are aluminum cans and the remainder are plastic and glass (calculated from 2022 Recovery Rate Report).

3.2.5 Interfacility Transportation

There are six key instances of interfacility transportation: MSW moved between transfer stations and landfills, leachate moved from landfills to wastewater treatment plants (WWTP), non-combustibles and ash moved from the incinerator to landfills, mixed aluminum recycling from drop-off centers to aluminum sorters, sorted and pressed aluminum from mixed recycling collection centers and redemption centers moved to out-of-state remelters, and rejected aluminum waste from collection and redemption centers to the nearest landfill.

Table 6 - Collection and Transportation Parameters by Sector and Waste Stream. Location data from 2024, Ecoinvent data 2011-2024.

Sector	Waste Stream	Destination	Collection Mode	Share of Stream (%)	Vehicle Type	Kilometers per trip (one way)	Miles per trip (one way)	Ecoinvent Process Used
Res.	All waste streams excl. recycling	Landfill	Curbside	67%	Waste-hauling truck	32.2	20	municipal waste collection service by 21 metric ton lorry
Res.	All waste streams excl. recycling	Landfill	Drop-off	33%	Personal vehicle	23	14.3	transport, passenger, car, EURO 5, fleet average
Res.	All waste streams excl. recycling	Incinerator	Curbside	97%	Waste-hauling truck	32.2	20	municipal waste collection service by 21 metric ton lorry
Res.	All waste streams excl. recycling	Incinerator	Drop-off	3%	Personal vehicle	23	14.3	transport, passenger, car, EURO 5, fleet average
Res.	Aluminum mixed recycling	Aluminum sorting	Curbside	45%	Waste-hauling truck	32.2	20	municipal waste collection service by 21 metric ton lorry
Res.	Aluminum mixed recycling	Aluminum sorting	Drop-off	55%	Personal vehicle	26.9	16.7	transport, passenger, car, EURO 5, fleet average
Res.	Aluminum dep. containers	Redemption center	Drop-off	100%	Personal vehicle	17	10	transport, passenger, car, EURO 5, fleet average
ICI	All waste streams excl. recycling	Landfill	Curbside / Contract	100%	Waste-hauling truck	32.2	20	municipal waste collection service by 21 metric ton lorry
ICI	Aluminum mixed recycling	Aluminum sorting	Curbside / Contract	100%	Waste-hauling truck	32.2	20	municipal waste collection service by 21 metric ton lorry
ICI	Aluminum dep. containers	Redemption center	Drop-off	100%	Light Comm. vehicle	32.2	20	transport, freight, light commercial vehicle
Both	All waste streams excl. recycling	Landfill from transfer station		17.7%	Large waste-hauling truck	38	24	transport, freight, lorry, >32 metric ton, diesel, EURO 5
Both	Leachate	Landfill to WWTP		32%	Medium waste truck	33.7	20.1	freight, lorry, 16-32 metric ton, diesel, EURO 5
Both	Aluminum recycling	Sorting rejects to landfill		8%	Medium waste truck	15.1	9.4	freight, lorry, 16-32 metric ton, diesel, EURO 5
Both	Aluminum recycling	Sorting to remelter		100%	Large waste-hauling truck	563	350	transport, freight, lorry, >32 metric ton, diesel, EURO 5
Res.	Textile recycling	Landfill to central processor		61%	Medium waste truck	154	96	freight, lorry, 16-32 metric ton, diesel, EURO 5

Res.	Textile recycling	Central processor to OOS recycler		100%	Large truck	1,835	1,140	transport, freight, lorry, >32 metric ton, diesel, EURO 5

Additionally, for the alternative waste management scenarios, we assume that in a textile recycling pathway, sorted and bundled textiles would be transported from landfills and transfer stations to a central processing center in Des Moines, and then again to out-of-state mechanical textile recyclers. Using GIS, we estimated average distance traveled from landfills and transfer stations to the central Des Moines location to be 154 km (96 mi).

3.2.5.1 Transport between transfer stations and landfills

From the 2024 Iowa Tonnage Report data, we calculated that 17.7% of MSW first goes to a transfer station prior to being transferred to landfills. We used a GIS analysis to determine that the average distance between transfer stations and the nearest landfills (Solid Waste Program Data Portal) is 38.2 km (24 mi) one-way. We assume that larger trucks are used to move MSW to the landfill, and, therefore, we model interfacility transportation using >32 metric ton (>35 short ton) diesel semi-trucks.

3.2.5.2 Transportation of leachate

Leachate is generated in landfills as water percolates through waste. In modern landfills, it must be collected and properly treated or disposed of to protect groundwater and surface waters. Most leachate generated in Iowa's landfills is transported via gravity fed pipelines to WWTPs. However, approximately 1/3rd of leachate collected from Iowa landfills (32%) is transported via tanker truck to the nearest WWTP (SCS internal study of leachate handling in active landfills). Ecoinvent processes for the treatment of leachate account for transport via pipeline, but the transport via tanker truck must be explicitly modeled. To do so, we calculated the average weighted distance between landfills using trucks and the nearest WWTP to be 33.65 km one-way (21 mi). Since the landfill processes used in our model report the leachate output produced in cubic meters, we must calculate the mass of leachate moved. To do so, we use the density of seawater, 1.025 metric tons (1.13 short tons) per cubic meter. For example, 3.2 cubic meters of leachate is produced per ton of food waste:

$$3.2\ m^3 leachate \times 1.025 \frac{metric\ tons}{m^3} \times\ 33.7\ km\ =\ 111\ t * km$$

3.2.5.3 Transportation of non-combustibles and ash from Ames incinerator to landfill

A large portion (47.5%) of all material brought to the Ames incinerator cannot be burned and is rejected by plant operators (RRP Annual Report 2024). Some of this rejected material is from specific loads that are unsuitable for incineration, while the remainder consists of mixed waste. Nearly all this material must be landfilled. The distance from the incinerator to the nearest landfill is 48.3 km (30 miles) one-way. We assume that, much like interfacility transport from transfer stations to landfills, large semi-trucks (>32 metric ton) haul the rejected waste to its final destination.

Ash and other byproducts from the incinerator must also be disposed of. We assume that ash from the incinerator is also hauled 48.3 km (30 miles), the street driving distance from Ames RRP to the nearest landfill, via large semi-trucks. For the much smaller amounts of residues and spent activated carbon we use the default market Ecoinvent processes for transporting and treating these wastes (i.e. market for residues, MSWI[F], biowaste).

3.2.5.4 Transportation of mixed aluminum recycling from drop-off center to sorting facility

A survey of recyclers in Iowa revealed that mixed aluminum recycling drop-off centers transported their aluminum to sorters and preparers. We used GIS to calculate the weighted average distance between drop-off centers and their reported aluminum sorter/preparer. The average distance between these facilities was 51.1 km (31.8 mi) one-way.

3.2.5.5 Transportation of aluminum prepared for recycling to out-of-state remelters

Transport distance was estimated by averaging the shortest road routes from a representative central Iowa origin (Metro Waste Authority in Des Moines) to three nearby aluminum remelting facilities in Illinois, Indiana, and Minnesota (Table 7). Transport routes were obtained via Google Maps and the distance of shortest route was selected for each plant. The mean transport distance of 559 km was rounded to 560 km (~347mi) for modeling purposes.

Table 7 - Distances to out-of-state Aluminum remelting facilities from Metro Waste Authority (MWA), Des Moines, IA 50309.

Facility / Area	Address	Distance
Wabash cluster of secondary Al producers	4525 US-24 BUS, Wabash, IN 46992	753 km (468 miles)
EGA Spectro Alloys	13220 Doyle Path, Rosemount, MN 55068	377 km (239 miles)
Trialco Inc	900 E Lincoln Hwy, Chicago Heights, IL 60411	547 km (336 miles)

3.2.5.6 Transportation of rejected aluminum from recycling and redemption centers to nearby landfills

Some aluminum sent to recycling and redemption centers is rejected. An SCS survey of recycling facilities found that responding facilities (n = 17) produced 8% residuals on average. We assume this material is collected and transported to the nearest landfill for disposal. We used a GIS analysis to determine the average distance traveled one-way is 15.1 km (9.4 mi) via a 16-32 metric ton (17.6 to 35.3 short ton) truck.

3.2.6 Transportation of textiles collected for recycling

We assume that in the alternative waste management scenario, 15% of textile waste would be diverted from landfills, and this waste would come solely from residential waste. Under this scenario, residents would separate textiles and deliver them to nearby landfills and transfer stations by passenger vehicles, where they would be collected and transported to a central location in Des Moines for sorting and baling, before being shipped to out of state mechanical textile recyclers. We assume the distance from households to landfills and transfer stations is 46 km roundtrip (28.6 mi), and that 50% of the trip is allocated to recycling used clothing (Note: we used the MSW-DST default approach: a round-trip travel distance multiplied by the fraction of trips assumed to be dedicated to the drop-off purpose (default 0.50). We used GIS to determine that the average distance from active landfills and transfer stations to a central Des Moines location is 154 km (96 miles) and assume a 7.5–16 metric ton-sized (8.3 to 17.6 short ton) truck is used to transport the separated textiles. The average distance to out-of-state (OOS) mechanical recyclers in Welford, SC (Leigh) and Brownsville, TX (Martex) is 1,835 km (1,140 miles). We assume large semi-trucks (>32 metric ton (>35 short ton)) haul the rejected waste to its final destination.

3.3 Food Waste

This section describes the life cycle inventory developed for food waste, including food waste origin, food waste collection and transportation, food waste management pathways, and associated material and energy flows.

3.3.1 Waste origin and composition

The total quantity of food waste modeled in this study was 473,085 metric tons (521,487 short tons) per year, representing 19.2% of the total MSW generated within Iowa from residential and ICI sources. Food waste generation was allocated between the Residential and ICI sectors with 191,477 metric tons (211,067 short tons) generated from residential sources and 281,608 metric tons (310,420 short tons) coming from ICI sources.

The food waste stream modeled included edible and inedible food waste and excluded any packaging. There is not currently a “food waste” option in the Ecoinvent database or the WasteTools 2023 LCA model developed by Gabor Doka (hereafter the Doka model (2023)). Therefore, it was necessary to identify the most appropriate alternative option along with modification methods to represent food waste to the best degree possible. The Doka model (2023) provided an option for “biowaste at 65% average moisture”, which represents a typical mix of garden, yard, and food/kitchen waste disposed of as part of municipal waste. The moisture content of 65% is the closest available moisture assumption to that used in EPA WARM for food waste (72% moisture) (WARM V16, 2023).

However, biowaste at 65% moisture and food waste at 72% moisture are not perfectly interchangeable due to differences in degradability. Therefore, additional steps detailed below were taken to: (1) adjust the fugitive methane emission factor (MEF) of "biowaste at 65% average moisture" to better represent the methane generation characteristics of food waste and related electricity production, and (2) incorporate sensitivity analysis for MFE, an important but uncertain parameter. Fats, oils, and grease (FOG) are excluded from this study as these waste types are managed outside of the MSW program.

Iowa has nascent alternative treatments for food waste. In 2024, approximately 2,653 metric tons (2,925 short tons) of food waste were managed via composting and three WWTPs accepted 2,267 metric tons (2,500 short tons) of food waste. These alternative treatments represent 0.1% of food waste in 2024 and were excluded from our study.

3.3.2 Collection and transportation

Food waste was collected using the collection and transportation models described in Section 3.2. Residential food waste was assumed to be collected within MSW via curbside and drop-off, while commercial food waste was assumed to be collected within MSW via contracted pickup in waste-hauling trucks.

3.3.3 Sorting and preprocessing

We did not assume that food waste undergoes any preprocessing.

3.3.4 An Iowa-specific LCI model for the landfilling of food waste

The Ecoinvent database does not include a process model for sanitary landfilling of food waste. The most representative waste stream for food waste included in the Ecoinvent database is biowaste. Treatment options for biowaste in the database include the following: municipal incineration, anaerobic digestion, industrial composting, clinker production, open dumping, and home composting in heaps. Table 8Since there is no default model in Ecoinvent for the sanitary landfilling of food waste, we utilized the Doka model (2023), which serves as the foundation for many of the waste management models in Ecoinvent. This open-source spreadsheet includes several biowaste options. We selected the "biowaste, 65% water" category as a proxy for food waste using default values and providers. Some key inputs to the Doka model (2023) are provided in Table 8. We exported an LCI from the Doka model (2023) for sanitary landfilling of this biowaste stream into OpenLCA and further adjusted the fugitive methane emission factor (MEF) and electricity produced from captured landfill gas to better represent the methane generation characteristics of food waste and the status of landfill technology in Iowa. For more details on how the Doka model was used to build a process that works within Ecoinvent, see Appendix C.

Table 8 - Parameter inputs into Waste Tools 2023 Model by Doka for the LCI for treatment of food waste by sanitary landfill. Where not indicated specifically, default values and providers were used. Note that the resulting fugitive methane emission factor and electricity produced from captured landfill gas were adjusted in OpenLCA to better represent food waste and Iowa landfill technology.

Parameter	Value	Justification
Landfill Characteristics		
Landfill height	20 m	Default value
Lifespan	20 years	Default value
Biowaste Characteristics		
Water content	65 w%	Default value
Carbon content	14 w%	Default value
Biogenic carbon	100 %	Default value
Hydrogen content (without H in H_20)	2 w%	Default value
Oxygen content (without O in H_20)	11 w%	Default value
Degradability (100 years)	27 %	Default value

3.3.4.1 Fugitive Methane Emissions from Landfilled Food Waste

A 2023 EPA meta-analysis found that fugitive landfill methane emissions are the most important contributor to the total global warming potential of landfilled food waste. They also note that there is a high degree of uncertainty in food waste landfill methane emissions, and that recent studies indicate that food waste landfill methane emissions may be underestimated (*From Field to Bin: The Environmental Impacts of U.S. Food Waste Management Pathways*, see pages 2-30). Though we could not ascertain an average fugitive methane emission factor from this EPA meta-analysis, other sources report estimates of fugitive methane losses from landfilled food waste. Some of these are detailed below in Table 9.

Given the importance of the landfill methane emission factor in the overall LCA for food waste and the variability in estimates in Table 9, we generated a range of estimates for this parameter (0.027 to 0.047 kg CH_4/kg landfilled food waste), which were used for sensitivity analysis.

Table 9 – Survey of methane emission factors from landfilled food waste.

Value (original source)	Units (original source)	Value (kg CH_4/kg landfilled food)	Reference
0.0478	kg CH_4/kg biowaste	0.0478	Ecoinvent treatment of waste paperboard by sanitary landfill (CH)
44	Metric ton CH_4/1000 short tons food waste	0.0485	EPA Avoided Landfilled Food Waste Methane Emissions Calculator

34	MT fugitive CH_4 per 907 MT food waste landfilled	0.037	Quantifying Methane Emissions from Landfilled Food Waste
54	kg CH_4/metric ton wasted food landfilled with gas collection	0.054	US EPA LandGEM model, as cited in *From Field to Bin: The Environmental Impacts of U.S. Food Waste Management Pathways*, see pages 2-30
36.2	kg CH_4/Mg food waste landfilled	0.0362	Food Waste Management Life Cycle Inventory Tool V.1.0.0, US EPA 2024 Note: lifetime gas capture efficiency = 28%

Scenarios based on Assumptions for the Methane Emission Factor (MEF)

Table 9Table 10Table 9Table 9

The scenarios examined were:

- **Scenario 1 (High MEF):** A pessimistic scenario that assumes a relatively low but realistic recovery (35%) of methane at Iowa landfills with landfill gas capture systems in place and a landfill gas methane oxidation factor of 10%.
- **Scenario 2 (Low MEF):** A more optimistic scenario that assumes a relatively high but realistic recovery (70%) of methane at Iowa landfills with landfill gas capture systems in place and a landfill gas methane oxidation factor of 25% (Krause et al. 2023).
- **Scenario 3:** A scenario based on the total volume of landfill gas captured according to the available Iowa landfill data and an assumption of total landfill gas generated based on Iowa population data and IPCC assumptions for municipal solid waste in the North America region. In this scenario, a recovery rate was estimated rather than assumed.
- **Scenario 4 (Baseline):** A baseline scenario that assumes a methane emissions factor equal to the mean of MEF values determined in Scenarios 1 through 3.

For these scenarios, we used the IPCC Waste Model (Volume 5, Waste) to estimate fugitive methane emission factors from landfilled food waste. In the IPCC Waste Model, the methane correction factor (MCF, maximum possible value = 1.00) is based on landfill management leading to either fully anaerobic conditions (MCF = 1.00) or other conditions leading to semi-aerobic management (MCF = 0.50 to 0.70), active aeration (MCF = 0.40 to 0.70), or unmanaged conditions (MCF = 0.40 to 0.80). Here we assumed MCF = 1.00 (full

anaerobic conditions). We then varied the recovery fraction (R) and oxidation factor (OX) for Scenarios 1-3. All other assumptions followed IPCC defaults. The general equation used was:

$$CH4\ Emissions = (CH4\ Generated) * (1 - R) * (1 - OX)$$

where CH_4 Generated was estimated using the Food tab in the IPCC Waste Model.

Iowa-specific data available included municipal waste composition (2022 Iowa Statewide Material Characterization Study) and an Iowa state agency database (Iowa Landfill LFG and Capacity Information spreadsheet) with information on tonnage received, presence/absence of landfill gas (LFG) capture at each landfill, and LFG volume captured and flared at each landfill. These data indicated that approximately 70% of the landfilled tonnage in 2024 occurred at landfill facilities with a LFG collection system in place. This Iowa-specific information was leveraged in the scenarios described in more detail below.

Scenario 1 Methods (High MEF)

Here, we make the following assumptions to develop a relatively pessimistic estimate for the methane emission factor (MEF) for landfilled food waste using IPCC Waste Model results:

- All Iowa landfills are managed sanitary landfills (methane correction factor = MCF = 1.00)
- All Iowa landfills have an oxidation factor of 0.10 (i.e., 10% of methane generated is oxidized within cover soils)
- 70% of landfilled food waste goes to facilities with LFG recovery with a recovery factor of 0.35, 30% of landfilled food waste goes to facilities without LFG recovery (recovery factor = 0.00).
- Other standard assumptions used in the IPCC Waste Model (including that LFG is 50% methane)
- A 30-year time horizon for emissions.

Using those assumptions and outputs from scenarios run in the IPCC Waste Model, we can estimate a weighted Iowa-specific fugitive MEF to be 0.047 kg CH_4/kg landfilled food. This was calculated as:

$$Weighted\ Iowa\ specific\ fugitive\ methane\ emission\ factor\ =$$

$$(0.70 * 0.04079\ kg\ CH4/kg\ landfilled\ food) + (0.30 * 0.06276\ kg\ CH4/kg\ landfilled\ food)$$

$$= 0.047\ kg\ CH4/kg\ landfilled\ food$$

This estimate is within the range of values presented in Table 9.

Scenario 2 Methods (Low MEF)

Here, we make the following assumptions to develop a relatively optimistic estimate for the MEF for landfilled food waste using IPCC Waste Model results:

- All Iowa landfills are managed sanitary landfills (methane correction factor = MCF = 1.00)
- All Iowa landfills have an oxidation factor of 0.25 (i.e., 25% of methane generated is oxidized within cover soils)
- 70% of landfilled food waste goes to facilities with LFG recovery with a recovery factor of 0.70, 30% of landfilled food waste goes to facilities without LFG recovery (recovery factor = 0.00).
- Other standard assumptions used in the IPCC Waste Model (including that LFG is 50% methane)
- A 30-year time horizon for emissions

Using those assumptions and outputs from scenarios run in the IPCC Waste Model, we can estimate a weighted Iowa-specific fugitive methane emission factor to be 0.027 kg CH_4/kg landfilled food. This was calculated as:

$$Weighted\ Iowa\ specific\ fugitive\ methane\ emission\ factor\ =$$

$$(0.70 * 0.0157\ kg\ CH4/kg\ landfilled\ food) + (0.30 * 0.0523\ kg\ CH4/kg\ landfilled\ food)$$

$$= 0.027\ kg\ CH4/kg\ landfilled\ food$$

This optimistic estimate is below the range of values presented in Table 9.

Scenario 3 Methods

We then took a third approach that leveraged additional data available on landfill gas recovery at Iowa facilities. We used the following information within the IPCC Waste Model to generate estimates of cumulative MSW disposed of during 1975-2024 and CH_4 generation in 2024 (essentially modeling a hypothetical single landfill representing all of Iowa):

- Iowa state-level population data (Federal Reserve Economic Data, Federal Reserve Bank of St. Louis) for 1975-2024 (the period corresponding to the lifetime for landfills for which data were provided)
- The IPCC regional default assumption for North America of 960 kg solid waste per capita per year, IPCC regional default assumptions for solid waste composition (20% food waste, see favorable comparison to Iowa data in Table 10 below), and
- An assumption of no industrial waste (no information on industrial waste available).

Using this approach, the IPCC Waste Model indicated 93 Gg of methane generated in 2024, equivalent to 12.85 million standard cubic feet per day (mmscfd) of CH_4 or 25.71 mmscfd LFG (assuming LFG is 50% methane by volume). Recent Iowa state agency data indicated that LFG recovery systems collected a total of 11.67 total mmscfd LFG. We then estimated an Iowa-specific recovery factor of 0.45 (=11.67/25.71). Importantly, this recovery factor represents all Iowa landfills, including those with and without LFG recovery systems. If we focus on the landfills where all the LFG recovery occurs, which accounted for ~70% of waste tonnage in 2024, the estimated recovery factor at only Iowa facilities with LFG capture is approximately 0.65 (=11.67/(25.71*0.7)). Next, we used the IPCC Waste Model with assumptions of 0.45 for the recovery factor, 0.10 for the oxidation factor, 1.00 for the methane correction factor, and 30 years for the time horizon to estimate an Iowa-specific fugitive methane emission factor to be 0.035 kg CH_4/kg landfilled food.

$$Weighted\ Iowa\ specific\ fugitive\ methane\ emission\ factor\ =$$

$$\left(69.73\frac{metric\ tons\ CH4}{Gg\ landfilled\ food}\right) * (1-0.45)\ *\ (1-0.1) * (\frac{1}{1000})$$

$$= 0.035\ kg\ CH4/kg\ landfilled\ food$$

Table 10 - Comparison of IPCC regional assumptions for solid waste composition with data available in the 2022 Iowa Waste Characterization study.

	Food	Garden	Paper	Wood	Textile	Nappies (Diapers)	Plastics, other inert	Total
IPCC North America Region Assumptions	20%	7%	23%	4%	4%	0%	42%	100%
Iowa-Specific Data (Overall Statewide MSW)	19%	4%	22%	7%	5%	3%	40%	100%
Iowa-Specific Data (Residential MSW)	19%	5%	18%	4%	6%	5%	43%	100%

Scenario 4 Methods (Baseline)

For the baseline scenario, we assumed a methane emissions factor of 0.036 kg CH_4/kg landfilled food, which is equal to the arithmetic mean of estimates produced in Scenarios 1 through 3 (= (0.047 + 0.027 + 0.035)/3). This baseline estimate aligns with the assumption used in the Food Waste Management Life Cycle Inventory Tool V.1.0.0 (US EPA 2024) (Table 9).

Given that the baseline estimate for the methane emission factor (MEF) was nearly identical to that in Scenario 3, results are only shown in this report for Scenarios 1, 2, and 4. From this point forward, Scenarios 1, 2, and 4 are referred to as High MEF, Low MEF, and Baseline, respectively.

3.3.4.2 Electricity Offset from Landfilled Food Waste

For all three approaches, in addition to modifying the fugitive methane emission factor (MEF), we also added a credit for electricity produced from the capture of landfill gas, which is assumed to offset the production of the equivalent amount of electricity on the regional grid (US-MRO Midwest Reliability Organization in this case). Based on the methane emission factors for Iowa derived above, we calculated the amount of methane recovered and used for energy projects using the following approaches:

We estimated methane recovery for energy projects as the MEF under no LFG recovery minus the estimated methane emission factor for each scenario, multiplied by a factor of

0.66 to account for the fact that approximately 33% of landfill gas collected in Iowa is flared (based on Iowa Landfill LFG and Capacity Information spreadsheet provided by Iowa state agency).

$$\left[\left(\frac{kg\ CH4}{kg\ landfilled\ food}\ without\ LFG\ recovery\right) - \left(\frac{kg\ CH4}{kg\ landfilled\ food}\ with\ LFG\ recovery\right)\right] * \frac{0.66\ kg\ LFG\ not\ flared}{kg\ LFG\ recovered} =$$

$$kg\ CH4\ recovered\ for\ energy\ projects\ per\ kg\ landfilled\ food\ waste$$

High MEF Scenario:

$$(0.06276\ kg\ CH4/kg\ landfilled\ food - 0.047\ kg\ CH4/kg\ landfilled\ food) * 0.66 =$$

$$0.0104\ kg\ CH4\ recovered\ for\ energy\ projects\ per\ kg\ landfilled\ food\ waste$$

Low MEF Scenario:

$$(0.0523\ kg\ CH4/kg\ landfilled\ food - 0.027\ kg\ CH4/kg\ landfilled\ food) * 0.66 =$$

$$0.0167\ kg\ CH4\ recovered\ for\ energy\ projects\ per\ kg\ landfilled\ food\ waste$$

Baseline Scenario:

Here, we used an average MEF under no LFG recovery of 0.0575 kg CH_4/kg landfilled food (= (0.06276 + 0.0523)/2).

$$(0.0575\ kg\ CH4/kg\ landfilled\ food - 0.036\ kg\ CH4/kg\ landfilled\ food) * 0.66 =$$

$$0.0142\ kg\ CH4\ recovered\ for\ energy\ projects\ per\ kg\ landfilled\ food\ waste$$

We next estimated kwh of electricity offset using the equation:

$$kWh\ offset\ per\ kg\ landfilled\ food =$$

$$(kg\ CH_4\ /kg\ landfilled\ food\ waste\ )/density\ of\ methane\ \times LHV_CH4 \times \eta_e \times (0.2778\ kWh/MJ\ )$$

LHV_{CH4} = lower heating value of methane = 35.8 MJ/m3 (source: US EPA Food Waste LCI Tool V1.0.0)

η_e = average electrical efficiency = 30% = 0.30 (operator-reported values from EPA LMOP and GHGRP datasets are commonly near ~30%)

- High MEF scenario estimate of kWh electricity offset =

$$\frac{0.0104\ kg\ CH4/kg\ landfilled\ food\ waste}{0.7\ kg\ CH4/m3} \times 35.8\ MJ/m3 \times 0.3 \times 0.2778\ kWh/MJ$$

$$= 0.04433\ kWh\ per\ kg\ landfilled\ food\ =\ 44.33\ kWh\ per\ metric\ ton\ food\ waste$$

- Low MEF scenario estimate of kwh electricity offset =

$$\frac{0.0167\ kg\ CH4/kg\ landfilled\ food}{0.7\ kg\ CH4/m3} \times 35.8\ MJ/m3 \times 0.3 \times 0.2778\ kWh/MJ$$

$$= 0.07118\ kWh\ per\ kg\ landfilled\ food\ =\ 71.18\ kWh\ per\ metric\ ton\ food\ waste\ landfilled$$

- Baseline scenario estimate of kwh electricity offset =

$$\frac{0.0142\ kg\ CH4/kg\ landfilled\ food}{0.7\ kg\ CH4/m3} \times 35.8\ MJ/m3 \times 0.3 \times 0.2778\ kWh/MJ$$

$$= 0.06052\ kWh\ per\ kg\ landfilled\ food\ =\ 60.52\ kWh\ per\ metric\ ton\ food\ waste\ landfilled$$

Note that this electricity offset estimates account for *all* food waste landfilled in Iowa, including that for which landfill gas is not captured or is captured and flared. Considering this fact, our estimates are in good agreement with the electricity offset range reported in the EPA meta-analysis referenced above, which states: “Across 34 reviewed landfill study scenarios, 25 specify that they include avoided electricity production benefits. The

quantity of electricity generated from landfill gas per metric ton of wasted food landfilled varies between 34 and 224 kWh" (*From Field to Bin: The Environmental Impacts of U.S. Food Waste Management Pathways*, see pages 3-7). A more recent EPA product, the US EPA Food Waste LCI Tool V1.0.0 yields a default of 88 kWh recovered per metric ton food waste, and references literature values ranging from 55 to 166 kWh per metric ton (from 50 to 150 kWh per short ton) of food waste. Note that the US EPA Food Waste LCI Tool assumes an average electrical efficiency of 39%, higher than the assumption used here.

Summary of Assumptions Used in Scenarios

Table 11 compiles the assumptions used for each scenario analyzed.

Table 11 - Scenarios based on various assumptions related to the food waste methane emissions factor (MEF).

Scenario	Methane Emission Factor (kg CH_4/kg landfilled food)	kWh per metric ton food waste landfilled
High MEF	0.047	44.33
Low MEF	0.027	71.18
Baseline	0.036	60.52

3.3.5 Incineration of Food Waste in Iowa

Approximately 1% of food waste, or 4,731 metric tons (5,215 short tons), was assumed to be managed via incineration at a resource recovery facility with electricity generation in Ames, Iowa. Of this, we used the statewide breakdown of 40.6% residential and 59.4% ICI to estimate that 1,915 metric tons (2,111 short tons) were from residential sources and 2,815 metric tons (3,104 short tons) were from ICI sources.

We built the process for incineration of food waste in Iowa by modifying the existing Ecoinvent process, *treatment of biowaste, municipal incineration FAE* to include Iowa-specific management inputs and providers (Table 12).

Table 12 – Iowa-specific inputs and providers into incineration of food waste life cycle inventory, per kg treated. Where not indicated specifically, default values and providers were used.

Parameter	Value	Justification
Delivery of food waste to incinerator	1.0 kg	treatment of biowaste, municipal incineration FAE does not include transportation
Transport of non-combustibles to landfill	0.475 kg	Transport of non-combustible waste not included
Market for electricity, low voltage	-0.4 MJ	Credit for avoided product

Remainder of inputs kept as default entries from treatment of biowaste, municipal incineration FAE as no specific data on inputs/outputs available from Ames RRP; see Appendix B for providers chosen.

3.3.5.1 Electricity Offset from Incinerated Food Waste

We added a credit for electricity produced from the resource recovery plant, which is assumed to offset the production of the equivalent amount of electricity on the regional grid (US-MRO in this case). Since the plant's recorded electricity output cannot be disaggregated into production from specific types of waste added to the incinerator, we use the Ecoinvent value provided for electricity produced for each kg of biowaste treated by incineration.

3.3.5.2 Non-combustible waste and ash disposal

While food waste itself is burnable, 47.5% of MSW delivered to the Ames RRP is rejected. Because the handling and disposal of rejected waste is part of normal operation, we allocate transportation and handling costs of non-combustible waste to each ton of waste incinerated at the plant based on its mass share. Additionally, for every metric ton (1.1 short ton) of food waste consumed, the plant produces 0.129 metric tons (0.142 short tons) of ash and 0.012 metric tons (0.013 short tons) of residues, which must be treated in a landfill. Iowa-specific transportation was used for disposal of ash. Ecoinvent default processes were used for transport and treatment of residues and spent activated carbon due to uncertainty about how these end products are disposed of and treated.

3.3.6 Inventory checks and consistency

Mass balance checks were performed to confirm that total outputs across all food waste management pathways equaled total food waste inputs within ±5%. Energy and material flow assumptions were applied consistently with those used for other waste streams unless explicitly noted. Key sources of uncertainty for the food waste inventory include diversion rates, food waste characteristics, preprocessing losses, biogas yield, landfill methane emission factors, and landfill methane capture rates. Specifically, they include the use of 'biowaste, 65% water' as a proxy for food waste compared to 72% in EPA WARM, NWS-derived precipitation and moisture, and the fugitive methane emission factor for landfilled food waste, which is both highly variable in the literature and a dominant driver of GWP for this waste stream. Additional uncertainty exists in LFG capture-to-energy and flaring assumptions and in electricity offset credits for both LFG recovery and incineration, since facility-level electricity outputs cannot be attributed to specific waste types. All of these were evaluated through sensitivity analysis as described in Section 4.

3.4 Plastic Film

This section describes the life cycle inventory developed for plastic film waste, including waste origin, collection and transportation, management pathways, and associated material and energy flows.

3.4.1 Waste origin and composition

The total quantity of plastic film waste modeled in this study was 189,981 metric tons (209,418 short tons) per year, representing 7.7% of the total MSW generated within Iowa from residential and ICI sources. Plastic film waste generation was allocated between the Residential and ICI sectors with 71,117 metric tons (78,460 short tons) generated from residential sources and 118,803 metric tons (130,958 short tons) coming from ICI sources.

3.4.2 Collection and transportation

Plastic film waste was collected using the collection and transportation models described in Section 3.2. Residential plastic film waste was assumed to be collected within MSW via curbside and drop-off, while commercial plastic film waste was assumed to be collected within MSW exclusively via contracted pickup in waste-hauling trucks.

3.4.3 Sorting and preprocessing

We did not assume that plastic film waste undergoes any preprocessing.

3.4.4 An Iowa-specific LCI model for the landfilling of plastic film waste

While there are many types of plastic film waste present in Iowa's MSW stream, such as polypropylene (PPE), high and low-density polyethylene (HDPE/LDPE), and multi-layer films, all types of plastic films have similar impacts from their end-of-life treatment in landfills and incineration. During the 2022 Waste Characterization Study, SCS collected data on total mass of plastic films but did not separate into sub-categories by type (personal communication). We chose waste polyethylene as a representative product for all plastic films treated in Iowa's landfills.

We selected the existing Ecoinvent process treatment of waste polyethylene, sanitary landfill for Switzerland, and chose Iowa-specific providers or their proxies to reflect technology status and conditions in Iowa (Table 13).

Table 13 - Iowa-specific inputs and providers into the process of landfilling plastic film waste, per kg treated. Where not indicated specifically, default values and providers were used.

Parameter	Value	Justification
Delivery of plastic film waste to nearest landfill or transfer station	1.0 kg	treatment of waste polyethylene, sanitary landfill does not include transportation

Transport from transfer station to nearest landfill	0.177 kg	Interfacility transport waste not included in Ecoinvent process
Market for electricity, low voltage	-0.004 kWh	Credit for avoided product, calculated as Ecoinvent default x 67% to represent CH4 flaring in Iowa landfills
Leachate, SLF, waste polyethylene (SLF = sanitary landfilling)	0.0032 m3	Adjusted leachate amount to reflect Iowa average annual precipitation
Transport leachate via tanker truck to WWTP	0.0353 kg*km	Reflects 32% of leachate trucked x 33.7 km * volume of leachate * density of leachate (kg/m3)
Remainder of inputs kept as default entries from treatment of biowaste, municipal incineration FAE as no specific data on inputs/outputs available from Iowa landfills; see Appendix B for providers chosen.		

3.4.4.1 Electricity Offset from Landfilled Plastic Film Waste

The Ecoinvent process for the treatment of waste polypropylene indicates that a small amount of electricity is produced from the capture of landfill gas generated by waste plastics. We reduce this electricity generation by 1/3rd to reflect the flaring of landfill gas in Iowa.

3.4.5 Incineration of Plastic Film in Iowa

Approximately 1% of plastic film waste, or 1,900 metric tons (2,094 short tons), was assumed to be managed via incineration at a resource recovery facility with electricity generation in Ames, Iowa. Of this, we used the statewide breakdown of 40.6% residential and 59.4% ICI to estimate that 712 metric tons (785 short tons) were from residential sources and 1,188 metric tons (1,310 short tons) were from ICI sources.

We built the process for incineration of plastic film waste in Iowa by modifying the existing Ecoinvent process, *treatment of waste polyethylene, municipal incineration FAE* to include Iowa-specific management inputs and providers (Table 14).

Table 14 – Iowa-specific inputs and providers into incineration of plastic film waste life cycle inventory, per kg treated. Where not indicated specifically, default values and providers were used.

Parameter	Value	Justification
Delivery of plastic film waste to incinerator	1.0 kg	treatment of waste polyethylene, municipal incineration FAE does not include transportation
Transport of non-combustibles to landfill	0.475 kg	Transport of non-combustible waste not included
Market for electricity, low voltage	-4.98 MJ	Credit for avoided product, value from Ecoinvent treatment of waste polyethylene, municipal incineration FAE

Remainder of inputs kept as default entries from treatment of biowaste, municipal incineration FAE as no specific data on inputs/outputs available from Ames RRP; see Appendix B for providers chosen.

3.4.5.1 Electricity Offset from Incinerated Plastic Film Waste

We added a credit for electricity produced from the resource recovery plant, which is assumed to offset the production of the equivalent amount of electricity on the regional grid (US-MRO in this case). Since the plant's recorded electricity output cannot be disaggregated into production from specific types of waste added to the incinerator, we use the Ecoinvent value provided for electricity produced for each kg of plastic film treated by incineration.

3.4.5.2 Non-combustible waste and ash disposal

Like food waste, we allocate transportation and handling costs of the 47.5% non-combustible waste to each metric ton of waste incinerated at the plant based on its mass share. Additionally, for every metric ton of plastic film consumed, the plant produces 0.027 metric tons (0.03 short tons) of ash and 0.005 metric tons (0.006 short tons) of residues, which must be treated in a landfill. Iowa-specific transportation is used for disposal of ash. Ecoinvent default processes used for transport and treatment of residues and spent activated carbon due to uncertainty about how these end products are disposed of and treated.

3.4.6 Inventory checks and consistency

Mass balance checks were performed to confirm that total outputs across all plastic waste management pathways equaled total plastic waste inputs within ±5%. Energy and material flow assumptions were applied consistently with those used for other waste streams unless explicitly noted. Key sources of uncertainty for the plastic film inventory include waste composition (material) heterogeneity, use of waste PE as a representative proxy, diversion and end-of-life waste management-related travel distance assumption. Additional sources due to technological limitations with plastics recycling, Iowa-specific data gaps and facility-level inconsistencies introduce uncertainty to plastic waste processing and related downstream impacts. Specifically, uncertainty exists from the representative diverse mix of chemicals in plastic films (e.g., LDPE, HDPE, PPE, and multi-layered films) using a single waste PE dataset, along with a lack of Iowa-specific data on local influences on plastic waste degradation and landfill gas generation behavior.

Additional uncertainty exists in LFG capture-to-energy and flaring assumptions and in electricity offset credits for both landfill gas recovery and incineration, since facility-level electricity outputs cannot be attributed to specific waste types. Finally, we note there are assumptions regarding incineration ash and residue processing, along with leachate

generation. Their end-of-life treatment -- subsequent transport and disposal -- rely on the most granular Ecoinvent processes due to gaps with facility-specific data.

3.5 Textiles and Leather

This section describes the life cycle inventory developed for textile and leather waste, including waste generation, pre-processing, collection and transportation, management pathways, and associated material and energy flows.

3.5.1 Waste generation and composition

The total quantity of textile and leather waste modeled in this study was 123,293 metric tons (135,907 short tons) per year, representing 5% of the total MSW generated within Iowa from residential and ICI sources. Textile and leather waste generation was allocated between the Residential and ICI sectors with 63,157 metric tons (69,619 short tons) generated from residential sources and 60,135 metric tons (66,288 short tons) coming from ICI sources.

3.5.1.2 Textile portion of waste stream

The composition of products contributing to used textiles is shown in Figure 4, showing that jackets, blazers, and trousers (i.e., pants) make up 31%; sweaters and cardigans, making up 24%; and T-shirts and singlets, making up 16%, together constitute over two-thirds of used textiles.

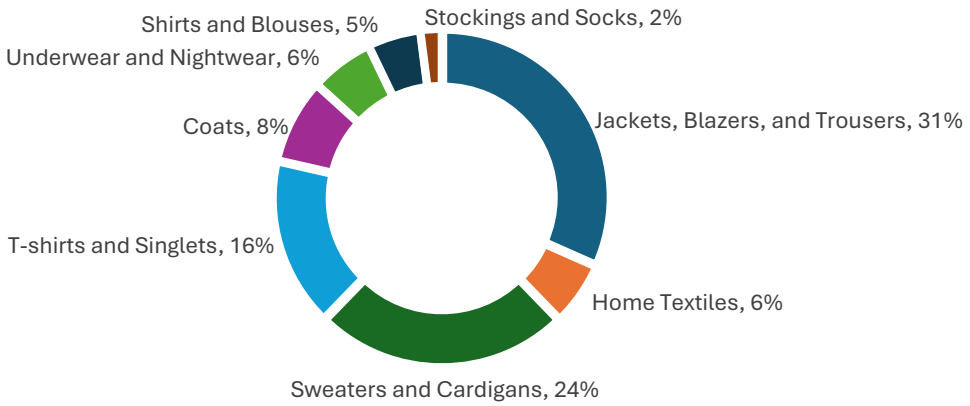


Figure 4. Textile and leather waste by garment type, based on data from 2024 IA Recycling Facility Study by SCS Engineers.

Regarding the difficulties of recycling via mechanical shredding, the presence of disruptors (e.g. buttons, fasteners, zippers, etc.) is shown in Figure 5 with around 2/3rd of items having some form of disruptor with almost 50% have disruptors that cannot be removed.

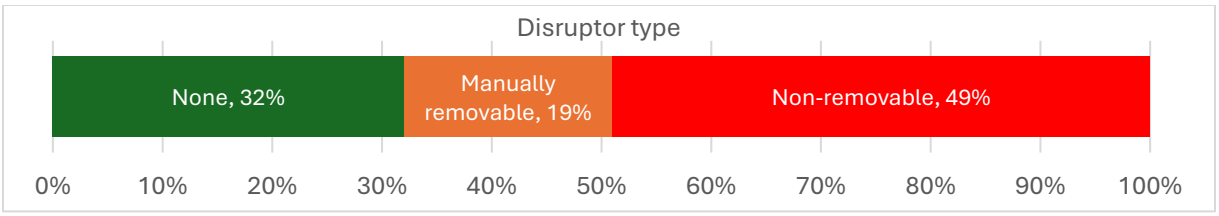


Figure 5. Prevalence of different types of disruptors on clothing items to recycling using data from 2024 IA Recycling Facility Study by SCS Engineers.

The environmental impact of textiles throughout the lifecycle of a product and the ease with which it can be recycled are also critically dependent on the type of material the textile is made from. In the IA Recycling Facility Study by SCS in 2024, we see the mass breakdown of textiles by fiber type, with cotton being the highest at 50%, followed by polyester, and acrylic. Based on Iowa Recycling Facility Study, Iowa Department of Natural Resources (URL to DNR website) default blend is 50% cotton, 18% polyester, 7% acrylic, 6% viscose, 3% wool, 2% polyamide, and 4% other fibers (assumed synthetic), and 10% disruptors. Mass of disruptors are assumed to be half metal and half plastic (see Figure 6). The textile recycling process (alternative waste treatment scenario #2, described in section 2.5) is based on the Oregon DEQ Waste Impact Calculator model: https://www.oregon.gov/deq/mm/pages/waste-impact-calculator.aspx

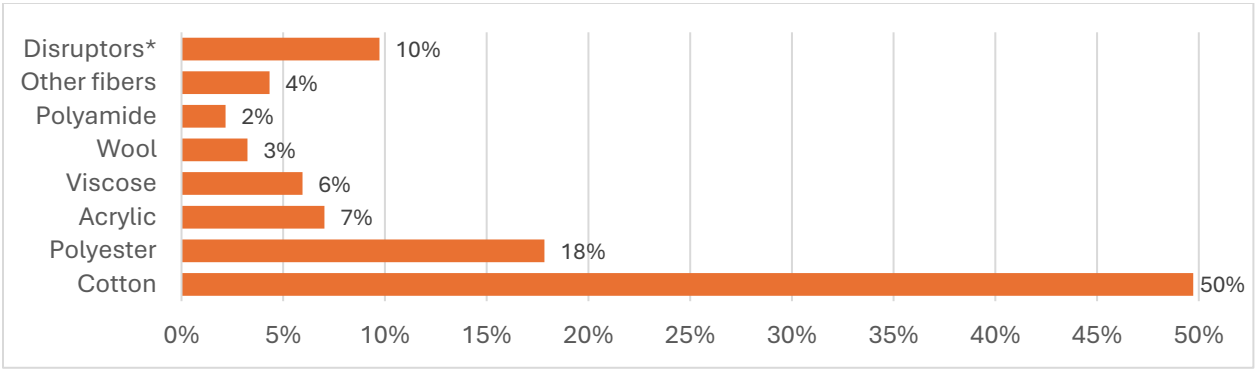


*Figure 6. Textile waste by type of fiber from the 2024 IA Recycling Facility Study by SCS. Data from source has been amended to account for the mass of disruptors that were not accounted for in the study. *Disruptors were assumed to make up 10% of the mass of any product.*

3.5.1.2 Leather portion of waste stream

Data are sparse regarding the breakdown of waste between the textiles and leather categories. Iowa did not collect subcategory data for this waste stream. In other cases, leather is often combined with textiles or with rubber waste (EPA, 2020). New Jersey's Department of Environmental Protection (NJ DEP) did produce a waste characterization study for 3 counties in 2022, which broke out leather in a separate category (NJ DEP, 2022). In this study, the leather category did not exceed 0.4% of residential MSW from urban

sources and only reached 0.2% in one season of commercial waste for one county. The remaining counties and seasons were less than 0.1%. Leather made up between 3.3% and 8.3% of combined residential textiles and leather, and between 1.5% and 3.9% of combined commercial textiles and leather (NJ DEP, 2022).

For this analysis we assumed that annual combined leather waste was 5,594 metric tons (6,166 short tons), of which residential leather waste was 3,789 metric tons (4,177 short tons) (6% of combined textiles/leather category) and commercial leather waste was 1,804 metric tons (1,989 short tons) (3%).

3.5.2 Collection and transportation

Textile and leather waste was collected using the collection and transportation models described in Section 3.2. Residential textile and leather waste was assumed to be collected within MSW via curbside and drop-off, while commercial textile and leather waste was assumed to be collected within MSW exclusively via contracted pickup in waste-hauling trucks.

3.5.3 Sorting and preprocessing

We did not assume that textile and leather waste bound for landfills or the resource recovery plant undergoes any preprocessing. For the textile recycling alternative waste management scenario, we assume that textiles are separated at the source and then delivered separately to recycling collection centers located at nearby landfills or transfer stations. From there, they are shipped to a centralized location in Iowa before being shipped for out-of-state-processing.

3.5.4 An Iowa-specific LCI model for the landfilling of textile and leather waste

There are no options for the treatment of waste textiles via sanitary landfill available in the Ecoinvent database for modeling the end of life of textiles. Two related processes are available: (1) *treatment of waste textile, soiled, municipal incineration*; and (2) *treatment of waste yarn and waste textile, unsanitary landfill*. A search of nexus.openlca.org found only one dataset for unsanitary landfill of textile within the Environmental Footprint database. A search within broader literature found only two references: Morell-Delgado et al. (2024) and Moazzem et al. (2021) none of which had sufficient data to build a comprehensive LCI dataset.

Gabor Doka's WasteTools 2023 model, which was used to build a great many of the waste treatment datasets available in Ecoinvent, enables a user to build an LCI database by specifying a waste stream, location, and treatment option (Doka WasteTools, 2023). As such, the tool was able to model the treatment of waste yarn and textiles by sanitary landfill. Inputs to the Waste Tools 2023 model are provided for textiles in Table 15. For more details on how WasteTools was used to build a process that works within Ecoinvent,

see Appendix C: Using WasteTools to create datasets for Ecoinvent. We use this process to represent the three natural fibers present in textile waste in Iowa: cotton, wool and viscose, which make up approximately 59% of textiles. Synthetic fibers and fasteners make up the remainder (41%) of textile waste. We use the existing Ecoinvent process *treatment of waste plastic, mixture, sanitary landfill* to represent the end-of-life treatment of synthetic textiles and fasteners.

Similar to the end-of-life for textiles, there is a complete lack of data for leather waste treatment, not only in Ecoinvent, but also in any other publicly accessible databases. Unfortunately, there is also no pre-built leather waste flow within the WasteTools model. A literature review turned up some references related to leather waste (Jiang et al. 2016, Van Rensberg et al 2020, Navarro et al. 2020), which highlighted the potential problem of heavy metals release in landfills, especially from chrome-tanned leather. However, few of these references provided any substantive data and none were sufficient to build an LCI.

Degradation of tanned leather in standard landfills is minimal; rather, the literature emphasizes stabilization of chromium, risk of chromium leaching, and the need for alternative end-of-life strategies to reduce environmental impact (Hemati et al., 2023; , Popita et al., 2020; , Fathima et al., 2012). As such, leather waste was modeled as inert waste.

Table 15 - Parameter inputs into Waste Tools 2023 Model by Doka for the LCI for treatment of textile waste by sanitary landfill. Where not indicated specifically, default values and providers were used.

Parameter	**Value**	**Justification**
Landfill height	20 m	Default value
Lifespan	20 years	Default value
Gas capture	50 w%	Default value
Gas flared	34 w%	Default value
Electric conversion efficiency	27.8 %	Default value
Heat conversion efficiency	13.5%	Default value
Lower heating value	15.7 MJ	Default value
Water content	2 w%	Default value
Carbon content	42 w%	Default value
Biogenic carbon	100 %	Default value
Hydrogen content (without H in H_20)	6 %	Default value
Oxygen content (without O in H_20)	46 w%	Default value
Degradability (100 years)	12 %	Default value

Table 16 - Iowa-specific inputs and providers into the process of landfilling leather waste, per kg treated. Where not indicated specifically, default values and providers were used.

Parameter	Value	Justification
Delivery of leather waste to nearest landfill or transfer station	1.0 kg	treatment of inert waste, inert material landfill does not include transportation
Transport from transfer station to nearest landfill	0.177 kg	Interfacility transport waste not included in Ecoinvent process
Market for electricity, low voltage	N/A	No credit applied. Assumed no electricity produced as inert materials unlikely to produce meaningful CH4
Leachate, SLF, inert waste	0.0032 m3	Adjusted leachate amount to reflect Iowa average annual precipitation
Transport leachate via tanker truck to WWTP	0.0353 kg*km	Reflects 32% of leachate trucked x 33.7 km * volume of leachate * density of leachate (kg/m3)
Remainder of inputs kept as default entries from treatment of biowaste, municipal incineration FAE as no specific data on inputs/outputs available from Iowa landfills; see Appendix B for providers chosen.		

3.5.4.1 Electricity Offset from Landfilled Textiles and Leather Waste

The model for the landfilling of textile waste, based on the process *disposal, waste yarn and waste textile, to sanitary landfill* indicates that a small amount of electricity is produced from the capture of landfill gas. We reduce this electricity generation by 1/3 to reflect the flaring of landfill gas in Iowa.

We do not assume that electricity is produced from the landfilling of leather (inert) waste.

3.5.5 Incineration of Textile and Leather in Iowa

Less than 1% of textile and leather waste, or 1,232 metric tons (1,359 short tons), was assumed to be managed via incineration at a resource recovery facility with electricity generation in Ames, Iowa. We used the statewide data from the waste characterization study: textiles and leather waste being 6.3% of residential MSW and 4.1% of ICI MSW, to estimate that 631 metric tons (696 short tons) were from residential sources and 601 metric tons (663 short tons) were from ICI sources.

For the incineration of organic textile waste, we used the existing process *treatment of waste textile, soiled, municipal incineration FAE (*Table 17*)*. For incineration of synthetic textile waste in Iowa, we used the same Iowa-specific Ecoinvent process used for plastic

films, *treatment of waste polyethylene, municipal incineration FAE*, with the same Iowa-specific providers and proxies selected (Table 14).

Table 17 - Iowa-specific inputs and providers into incineration of organic textile waste life cycle inventory, per kg treated. Where not indicated specifically, default values and providers were used.

Parameter	Value	Justification
Delivery of organic textile waste to incinerator	1.0 kg	treatment of waste textile, soiled, municipal incineration FAE does not include transportation
Transport of non-combustibles to landfill	0.475 kg	Transport of non-combustible waste not included
Market for electricity, low voltage	-1.78 MJ	Credit for avoided product, value from Ecoinvent treatment of waste textile, soiled, municipal incineration FAE
Remainder of inputs kept as default entries from treatment of waste textile, soiled, municipal incineration FAE as no specific data on inputs/outputs available from Ames RRP; see Appendix B for Iowa-specific providers chosen.		

We excluded flows of leather waste into the resource recovery plant since they make up only 54 metric tons (60 short tons) of waste incinerated annually.

3.5.5.1 Electricity Offset from Incinerated Textile Waste

We added a credit for electricity produced from the resource recovery plant, which is assumed to offset the production of the equivalent amount of electricity on the regional grid (US-MRO in this case). Since the plant's recorded electricity output cannot be disaggregated into production from specific types of waste added to the incinerator, we use the Ecoinvent value provided for electricity produced for each kg of textile waste (organic and synthetic) treated by incineration.

3.5.5.2 Non-combustible waste and ash disposal

Like food waste and plastic films, we allocate transportation and handling costs of the 47.5% non-combustible waste to each ton of waste incinerated at the plant based on its mass share. Additionally, for every metric ton of organic textile waste treated, the plant produces 0.037 metric tons (0.041 short tons) of ash and 0.008 metric tons (0.009 short tons) of residues, which must be treated in a landfill. Each ton of synthetic textile waste produces 0.027 metric tons (0.03 short tons) of ash and 0.005 metric tons (0.006 short tons) of residues.

3.5.6 Inventory checks and consistency

Mass balance checks were performed to confirm that total outputs across all textile and leather waste management pathways equaled total textile waste inputs within ±5%. Energy

and material flow assumptions were applied consistently with those used for other waste streams unless explicitly noted.

Due to complexities involved with specifying the exact textile and leather waste data for present day Iowa, there are several sources of data uncertainty associated with the life cycle inventory and assessment for this MSW stream. The most significant source of uncertainty arises from limitations in waste characterization data, notably, the lack of Iowa-specific sub-category data distinguishing the material composition and mass characteristics for more granular analysis of impacts resulting from the disposal pathways in the functional unit of MSW. In the absence of state-level textiles and leather composition data, assumptions regarding the natural or synthetic materials in textile fibers, and leather fraction in the combined leather and textiles category, were derived from a limited range of relevant textile waste characterization studies. Most notably, we rely on the New Jersey Department of Environmental Protection's 2022 study that looked at textile and leather waste disposal across three counties. While these data provide the best empirical basis for leather content estimation, they might not reflect Iowa-specific consumption and disposal patterns, industrial composition, and seasonality. As such, estimations of leather waste, namely, the allocation between commercial and residential sources, represent a moderate level of uncertainty and should be interpreted as indicative rather than definitive.

Additionally, we note that uncertainty through the modeling of end-of-life treatment pathways for textile and leather waste is due to substantial data gaps in existing peer-reviewed Ecoinvent databases. No existing Ecoinvent datasets for sanitary landfill processes of textiles or leather are appropriate for this LCA. Broader LCA literature provides limited and incomplete inventories for these materials. For textiles, this study attempts to address this data gap through the construction of an Iowa-specific landfill process, using the Doka 2023 WasteTools model, which disaggregates textile waste into natural and synthetic fractions and applies proxy data sets where necessary. This approach is consistent with established waste modeling practice and the WasteTools model is interoperable with the methodological foundations of Ecoinvent's waste treatment datasets. Even so, uncertainty remains regarding fiber composition, degradation rates (in response to climate, processing, and technological variation), landfill gas generation potential, and the long-term emissions behavior of mixed textile waste containing fasteners, coatings, and blended fibers, as well as contamination.

3.6 Mixed Aluminum Waste

This section describes the life cycle inventory developed for mixed aluminum waste, including waste generation, collection and transportation, sorting and processing, management pathways, and associated material and energy flows.

3.6.1 Waste generation and composition

The total quantity of mixed aluminum waste modeled in this study was 32,271 metric tons (35,573 short tons) per year, representing 1.3% of the total MSW generated within Iowa from residential and ICI sources. Mixed aluminum waste generation was allocated between the Residential and ICI sectors with 13,102 metric tons (14,443 short tons) generated from residential sources and 19,169 metric tons (21,130 short tons) coming from ICI sources. Unlike the other waste streams studied in this life cycle assessment that are mainly treated by landfilling or sent to an incinerator, Iowa residents have two additional pathways for recycling their aluminum waste: mixed (blue bin) recycling and redemption of aluminum deposit beverage containers.

3.6.1.1 Calculation of aluminum waste flows in Iowa

Primary data on aluminum waste is scarce. We rely on three sources of data from which we derive the mass of aluminum waste treated by landfill and by recycling via mixed recycling collection or deposit container redemption:

- **2022 Iowa Statewide Material Characterization Study**, which provides an accounting of the prevalence of Aluminum deposit beverage containers (0.5%), non-deposit beverage containers (<0.1%), and other aluminum products (0.2%) in MSW sent to landfills.
- **2023-2024 Iowa Recycling Facility Survey** that handles mixed recycling, which provides data on tonnage recycled, percentage of Aluminum deposit beverage containers (0.6% of total recyclables), non-deposit beverage containers (0.9%), and other aluminum (1.1%). Residuals, on average, were 8% among providers surveyed. We were also able to calculate the average distance between recycling collection centers and where their aluminum is sent to be processed for recycling.
- **2022 Recovery Rate Report for Deposit Beverage Containers** which provides a calculated estimate of the recovery rate (41.8%).

From these data, we can infer the total tonnage of aluminum deposit beverage containers landfilled and the amount recovered via redemption centers. We also calculate the sum of the non-deposit containers and other aluminum waste landfilled and the amount recycled through the mixed recycling system (Table 18).

Table 18 – Aluminum (Al) waste flows (in short tons) to landfill and recycling by container type

	Total Al MSW	**Recovered Al**	**Landfilled Al**	**% Recovery Rate (Recovered / Total)**	**% Non-Recovery Rate (1 – Recov. Rate)**
Deposit Al Beverage Containers	23,367	9,758	13,609	41.8%	58.2%

Other Al Beverage Containers	3,308	586	2,722	17.7%	82.3%
Other Al Waste	8,898	732	8,166	8.2%	91.8%
Total	35,573	11,076	24,497	31.1%	68.9%

3.6.2 Collection and transportation

For consistency, mixed aluminum waste was collected using the collection and transportation models described in Section 3.2. Residential aluminum waste bound for landfills and mixed recycling centers was assumed to be collected via curbside and drop-off, while commercial aluminum waste was assumed to be collected exclusively via contracted pickup in waste-hauling trucks. For aluminum deposit containers, we assumed that households dropped off residential sources to redemption centers by their personal vehicles, and commercial and industrial firms used light commercial vehicles, such as pick-up trucks, work vans or box trucks.

3.6.3 Sorting and preprocessing

We did not assume that aluminum waste bound for landfills undergoes any preprocessing.

Aluminum collected for recycling undergoes a processing step where it is sorted, cleaned and pressed to prepare it for the remelter. On average 8% of aluminum collected for recycling is rejected (SCS 2024 survey of recycling facilities). These rejected materials are sent to the landfill, and we account for their transportation.

Since Iowa does not collect data from redemption centers, we do not know the rejection rate at these facilities. We assume that like the mixed recycling centers, 8% of containers are similarly rejected and sent to the nearest landfill.

We use the existing Ecoinvent process *treatment of used beverage cans, for recycling, unsorted, sorting* to capture the impact from the sorting and processing of aluminum in preparation for sending to out-of-state remelters. We selected the most appropriate providers of upstream processes and inputs for Iowa.

3.6.4 An Iowa-specific LCI model for the landfilling of aluminum waste

To model the treatment of aluminum waste by landfill, we adapted the existing Ecoinvent process *treatment of waste aluminum, sanitary landfill* for Switzerland by choosing Iowa-specific providers and adding in representative transportation processes (Table 19).

Table 19 - Iowa-specific inputs and providers into the process of landfilling aluminum waste, per kg treated. Where not indicated specifically, default values and providers were used.

Parameter	Value	Justification

Delivery of aluminum waste to nearest landfill or transfer station	1.0 kg	treatment of inert waste, inert material landfill does not include transportation
Transport from transfer station to nearest landfill	0.177 kg	Interfacility transport waste not included in Ecoinvent process
Market for electricity, low voltage	N/A	No credit applied. Assumed no electricity produced from landfill gas as aluminum unlikely to produce meaningful CH4
Leachate, SLF, waste aluminum	0.0032 m3	Adjusted leachate amount to reflect Iowa average annual precipitation
Transport leachate via tanker truck to WWTP	0.0353 kg*km	Reflects 32% of leachate trucked x 33.7 km * volume of leachate * density of leachate (kg/m3)
Remainder of inputs kept as default entries from treatment of biowaste, municipal incineration FAE as no specific data on inputs/outputs available from Iowa landfills; see Appendix B for providers chosen.		

3.6.4.1 Electricity Offset from Landfilled Aluminum

Because aluminum in landfills is effectively inert and does not produce methane, we do not model production of electricity from landfill gas.

3.6.5 Incineration of Aluminum Waste in Iowa

We do not model the incineration of Aluminum waste. First, aluminum accounts for approximately 1% of total MSW, and of that, only ~1% is sent to the Ames RRP. Secondly, at the Ames RRP, metals are recovered prior to incineration. This recovery pathway would therefore represent just ~0.01% of total aluminum waste.

3.6.6 Recycling of Aluminum Waste in Iowa

Within the recycling scenario, we evaluated multiple unit process options (e.g. processes for mixed metals, used beverage cans) and different choices for the displaced material to be credited (e.g., primary aluminum or wrought aluminum alloy). We selected the representative processes to build a product system that best reflects aluminum recycling practices in Iowa and out-of-state remelters (Table 20). We selected processes (1-A) for sorting used beverage cans for recycling; (2) for treatment of prepared post-consumer scrap by remelting; and (3) an avoided product of aluminum ingot to the North American market. Default parameters for the Ecoinvent processes were left unchanged. To better represent the local waste management system, the product system was configured to reflect Iowa-specific rejection rates and processing losses by routing a portion of material flows to landfill (1-B).

Combined, total losses from waste generation through transport to out-of-state recycling facilities summed to 8.9%. This is a line-level rejection rate applied to the aluminum stream as a proxy due to a lack of data about the composition of rejected material and the mass of contaminants rejected along with the waste aluminum. To account for this in our model we modified the aluminum waste flow from the sorting process to 0.08 kg per kg of aluminum sorted, aligning with Iowa MRF reports (SCS, 2024), and we kept the default aluminum waste flows from the Ecoinvent remelting process (0.00898 kg per kg remelted). For every 1 metric ton of waste aluminum collected for recycling then, 0.912 metric tons are produced at the remelter, and 0.088 metric tons are sent to the landfill. Therefore, the total recycled aluminum ingot credit is 0.912 metric tons per 1 metric ton of waste entering the recycling pathway.

The recycling model included a transport sub-model specific to aluminum, representing shipment from Iowa MRFs to out-of-state aluminum melters. Based on locations of remelters provided by SCS, we estimate an average transport distance of approximately 560 km (~348 mi.), with a range of 380 to 750 km (236 to 466 mi.), to facilities located in Illinois, Minnesota, and Indiana (Table 7). We assume that 100% of this transport occurs via truck (EURO 5, >32 metric ton (>35 short ton) lorry), consistent with our research on current practices and SCS comments.

At the remelter, the prepared aluminum scrap is remelted into aluminum ingots. We assume a 1:1 replacement rate and assume that the recycled aluminum replaces virgin aluminum from the IAI-2, North America region (noted as market for aluminium, primary, ingot - IAI Area, North America - aluminium, primary, ingot in Ecoinvent).

Table 20 - Description of Ecoinvent processes for Aluminum recycling.

Step	Process	Notes	Link to Data
1	treatment of used beverage cans, for recycling, unsorted, sorting	This option best represents a waste stream of well sorted aluminum containers, and foils. .	https://ecoquery.ecoinvent.org/3.11/cutoff/dataset/23491/documentation
2	Treatment of aluminum scrap, post-consumer, prepared for recycling, at remelter	This represents the second step of a two-step aluminum recycling process.	https://ecoquery.ecoinvent.org/3.11/cutoff/dataset/5002/documentation
3	market for aluminum, primary, ingot	Avoided product: primary aluminum displaced from the North American market by recycling, applied as a credit.	https://ecoquery.ecoinvent.org/3.12/cutoff/dataset/23037/documentation https://ecoquery.ecoinvent.org/3.11/cutoff/dataset/23037/documentation.

Table 21 - Iowa-specific inputs and providers into the aluminum recycling process, per kg treated.

Parameter	Value	Justification
Delivery of mixed aluminum waste or deposit beverage containers to recycling sorting sites	1.0 kg	treatment of used beverage cans, for recycling, unsorted, sorting does not include transportation
Transport of rejected aluminum from sorting to nearest landfill	1.21 kg*km	Interfacility transport waste not included in Ecoinvent process, reflects rejected mass and average distance to nearest landfill
Output: Municipal solid waste	0.08 kg	Generic waste product used to represent rejected aluminum, most likely contaminated with food or non-recyclables. Provider: treatment of municipal solid waste, sanitary landfill
Remainder of inputs kept as default entries from treatment of used beverage cans, for recycling, unsorted, sorting; see Appendix B for providers chosen.		

3.6.7 Inventory checks and consistency

Mass balance checks were performed to confirm that total outputs across all mixed aluminum waste management pathways equaled total aluminum waste inputs within 5%. Energy and material flow assumptions were applied consistently with those used for other waste streams unless explicitly noted. Key sources of uncertainty for the aluminum waste inventory include the derivation of mixed aluminum waste flows from multiple secondary data sources, recovery and rejection rate assumptions, as well as sorting and processing losses, and transport distance estimates from curbside pick-up and drop off pathways, hauling to MRFs/transfer facilities, and recycling pathways via out-of-state remelters.

Specifically, uncertainty arises from limited primary data on curbside and dropoff pathways of mixed aluminum waste generation in Iowa and the implicit need to infer landfill and recycling mass and material flows using a combination of primary data collected by SCS Engineers through surveys of MRFs and waste management facilities. As such, assumptions regarding rejection rates at MRFs, transfer stations, and redemption centers introduce additional uncertainty, particularly because redemption center–specific rejection data are not systematically reported, and gaps remain due to non-responsiveness. Further uncertainty is associated with recycling losses and the assumption of a 1:1 substitution of recycled aluminum ingot for primary aluminum production in the North American market.

Finally, uncertainty exists in transportation modeling assumptions, as average long-distance transport to out-of-state remelters was estimated based on reported facility

locations by SCS Engineers in nearby states without assigning allocations based on actual tonnage data. Recycled aluminum is transported by rail for certain high-volume long-distance applications (e.g. urban areas, filling multiple rail cars, shipping more than 800 mi), however, available stakeholder data from Iowa MRFs supported modeling transport to out-of-state remelters exclusively by heavy-duty truck. Additional end-of-life treatment uncertainty remains regarding the actual distribution of aluminum tonnage leaving the system boundary, as mixed aluminum waste bound for recycling may be processed across multiple facilities in Illinois, Indiana, and Minnesota. These uncertainties were addressed through conservative parameterization of material losses and routing to landfill and were evaluated through sensitivity analysis as described in Section 5.

3.7 Modeling Alternative Waste Management Scenarios

At the request of the client, we modeled three alternative waste management scenarios: 1) Increased aluminum deposit container recycling by 53% to national average; 2) Centralized Textiles Recycling: Diverting 15% of residential textile waste from landfills to a centralized collection system for in-state recycling pre-processing before being shipped to recyclers out-of-state; and 3) Reduced Fugitive Methane from Landfills: reduction of fugitive methane emissions at landfills by 10% through improvements in advance methane management techniques and capture technology.

To model the impact of additional aluminum deposit can recycling, we decreased the amount of aluminum deposit containers being landfilled by 4,700 metric tons (5,180 short tons, Table 22), and increased deposit container recycling by that amount, and then multiplied by respective impact factors by Functional Unit. We do not assume any increased programmatic costs associated with the increase in consumer participation in the deposit container program, however, associated transportation and processing costs increase proportionally.

Table 22 – Updated assumptions for aluminum (Al) waste flows (in short tons) to landfill and recycling by container type in alternative scenario.

	Total Al MSW	**Recovered Al**	**Landfilled Al**	**% Recovery Rate (Recovered / Total)**	**% Non-Recovery Rate (1 – Recov. Rate)**
Deposit Al Beverage Containers	23,367	14,955 (+5,180)	8,412 (-5,180)	64%	36%
Other Al Beverage Containers	3,308	586	2,722	17.7%	82.3%
Other Al Waste	8,898	732	8,166	8.2%	91.8%

Total	35,573	16,273	19,300	45.8%	54.2%

To model the residential textile recycling, we built Iowa-specific processes for collecting and transporting textile waste. We assumed that collection centers would be staged at landfills and transfer stations and that passenger vehicles would be used to transport residential textile waste 23 km (14.3 mi) one-way (the average distance from population centers to the nearest landfill or transfer station). We built a custom unit process to represent the sorting and processing stage. During this step, cotton and polyester textiles are separated, and the remaining textiles are sent to the landfill. We also assume that fasteners are removed during this stage. The output of this step is 0.45 metric tons (0.496 short tons) of cotton and 0.162 metric tons (0.179 short tons) of polyester prepared for recycling for each ton of recyclables process, and 0.388 metric tons (0.428 short tons) of rejected materials sent to the landfill.

We also assume that the processing center was built near the landfill and that no to minimal transportation was required for rejected materials. From the processing center, cotton and polyester materials are transported via semi-trucks an average of 1,835 km (1,140 miles) to out of state mechanical textile recyclers. We built bespoke unit processes for converting prepared cotton to cotton fiber, and prepared polyester textiles to recycled PET granulate, see Table 23 for details. We accounted for 20% losses in cotton and PET recycling processes, and lost material was assumed to be transported to and treated by landfill. Ultimately, the result of recycling one metric ton of residential textile waste is the avoided use of 0.36 metric tons (0.397 short tons) of cotton fibers and 0.13 metric tons (0.143 short tons) of PET granulate.

Finally, to model the reduction in fugitive methane emissions achieved through alternative landfill management techniques and technologies, we reduce the fugitive methane emissions factor for landfilling each of the waste streams treated by landfill by 10%. We do not assume any increased infrastructure or operational costs associated with the increased capture of methane emissions.

Table 23 - Description of Ecoinvent processes for textile recycling.

Step	Process	Notes	Link to Data
1	Treatment of waste polyethylene, for recycling, unsorted, sorting	This process was used as a proxy for sorting and processing textile waste, since no direct match available in the Ecoinvent database	https://ecoquery.ecoinvent.org/3.11/cutoff/dataset/16716/documentation

2	Cotton recycling to cotton fiber	This custom process includes inputs and outputs from the academic literature to represent the processing of cotton waste textiles into cotton fiber	Cotton recycling was modeled as a mechanical fiber recovery process including shredding, fiber separation, and dust extraction. Electricity demand was assumed to be 1.0 kWh per kilogram of cotton processed, based on reported industrial ranges for mechanical textile recycling. A base-case fiber recovery yield of 80% was assumed. Residual materials were modeled as disposed of via municipal solid waste landfill. No washing step was included
3	Polyethylene terephthalate, granulate, amorphous, recycled	Used as a proxy for making secondary PET from recycled synthetic textiles	https://ecoquery.ecoinvent.org/3.11/cutoff/dataset/16830/documentation

1

4. Life Cycle Impact Assessment

The following the results of the baseline analysis are presented in this section. LCIA results are organized and presented at multiple aggregation levels to support interpretation and decision-making. Results are reported:

- By waste stream: food, plastic films, textiles/leather, aluminum;
- By management pathway: landfill, incinerator with energy recovery, recycling; and
- At both per-metric-ton and statewide total scales.

In the first section, gross impacts, credits, and net impacts are reported per metric ton of waste treated in Iowa. These per-ton impact factors are then combined with annual tonnage estimates for each stream and management pathway to calculate statewide impacts. We also provide information about important contributors to emission impacts, eutrophication, and energy impacts by category. Lastly, alternative waste management scenarios selected by Iowa DNR are compared to our baseline results.

As with all attributional LCA studies, the reported LCIA results are subject to uncertainty arising from variability in foreground system parameters, assumptions associated with waste management behavior and transportation, background life cycle inventory datasets, and the selection and implementation of LCIA characterization methods. The results presented in this section therefore represent modeled baseline estimates rather than precise predictive values. Consistent with ISO 14044 interpretation requirements, uncertainty and sensitivity analyses were conducted to evaluate the robustness of comparative conclusions and identify parameters with the greatest influence on study outcomes. Detailed discussion of uncertainty sources and sensitivity analysis results is provided in Section 5.2 and Section 5.3.

4.1 Per-Ton Environmental Performance by Material and End-of-Life Pathway

Aluminum recycling provides the largest net environmental benefits observed in the study and represents the most consequential recovery pathway within the current system (Figure 7, Figure 8, Figure 9, Figure 10). In contrast, all landfill pathways exhibit net environmental burdens across the evaluated indicators, reflecting the absence of material recovery and the limited energy offsets associated with present disposal-based management. Incineration provides modest reductions in eutrophication and non-renewable energy demand through energy recovery but remains a net emissions impact burden for food waste, plastics, and textiles, as combustion-related emissions exceed recovered energy benefits (Table 24).

For clarity and readability, figures and tables in this section present baseline point estimates only and do not display uncertainty ranges or parameter variation bounds. The influence of uncertainty on comparative conclusions is evaluated separately through the sensitivity analyses presented in Section 5.

Residential landfill pathways are consistently more impact-intensive per ton than their ICI counterparts (Figure 7,

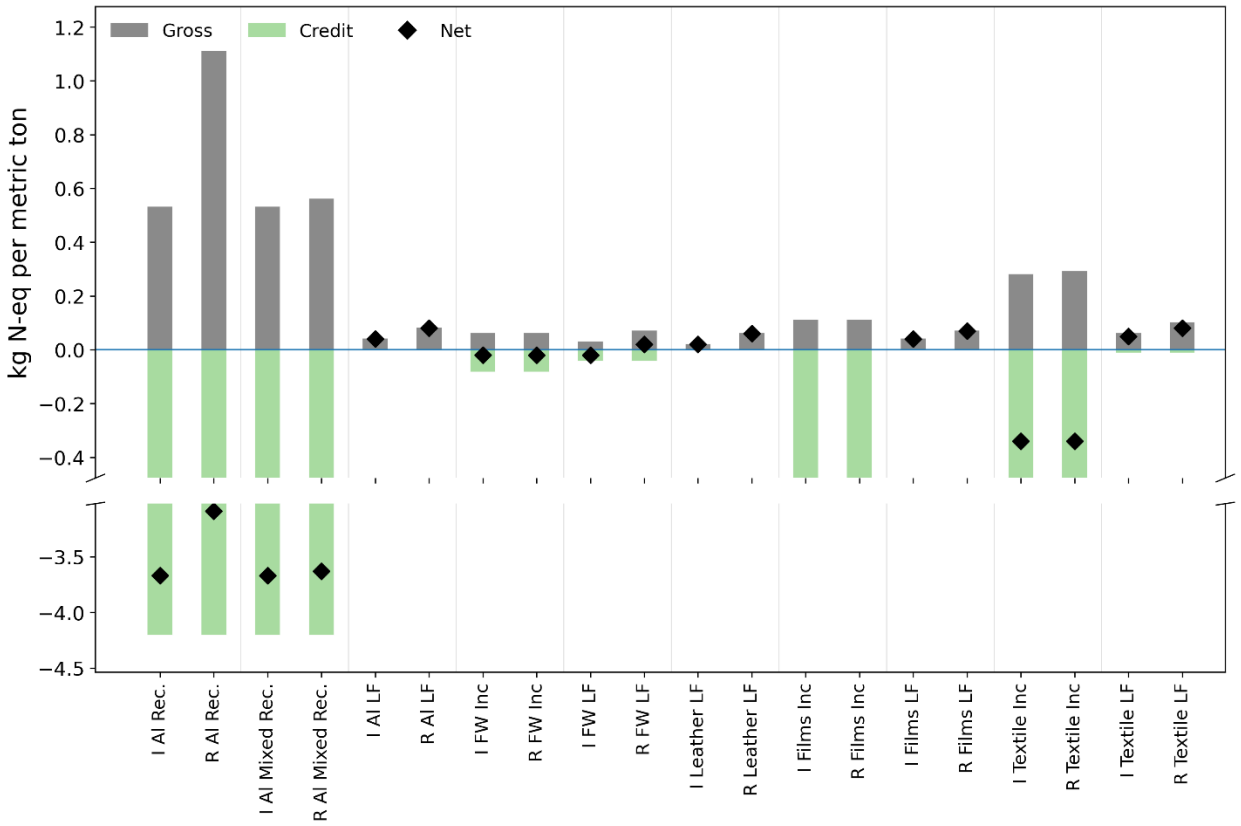


Figure 8, Figure 9, Figure 10; Table 24). This reflects higher collection burdens, lower operational efficiencies, smaller average loads, and longer effective transport distances per ton of material managed associated with residential landfill pathways.

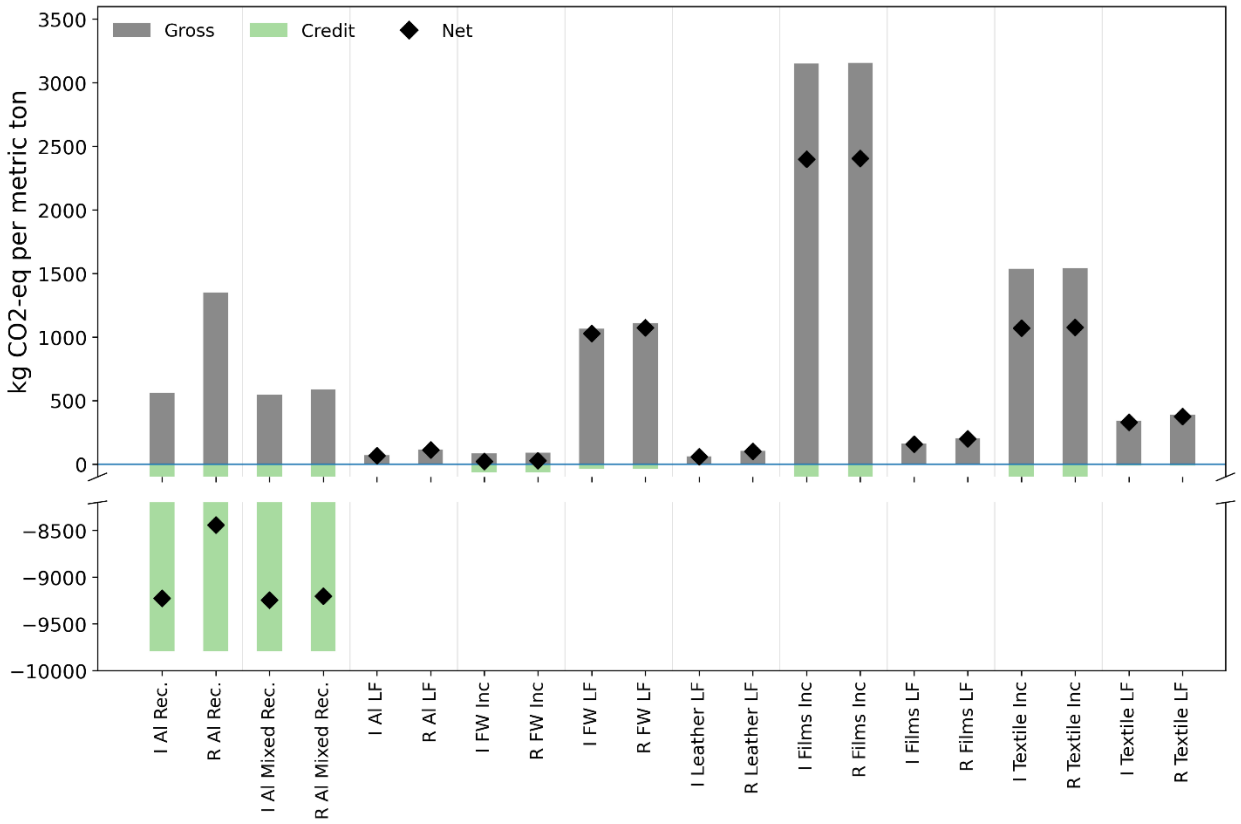


Figure 7 - Emissions impacts per metric ton treated, by waste stream and treatment. Gray bars indicate gross emissions impacts in t CO_2-eq/t waste treated, green bars indicate credits from avoided products, black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum. Please see section 5.4 for impacts of uncertainty on key findings.

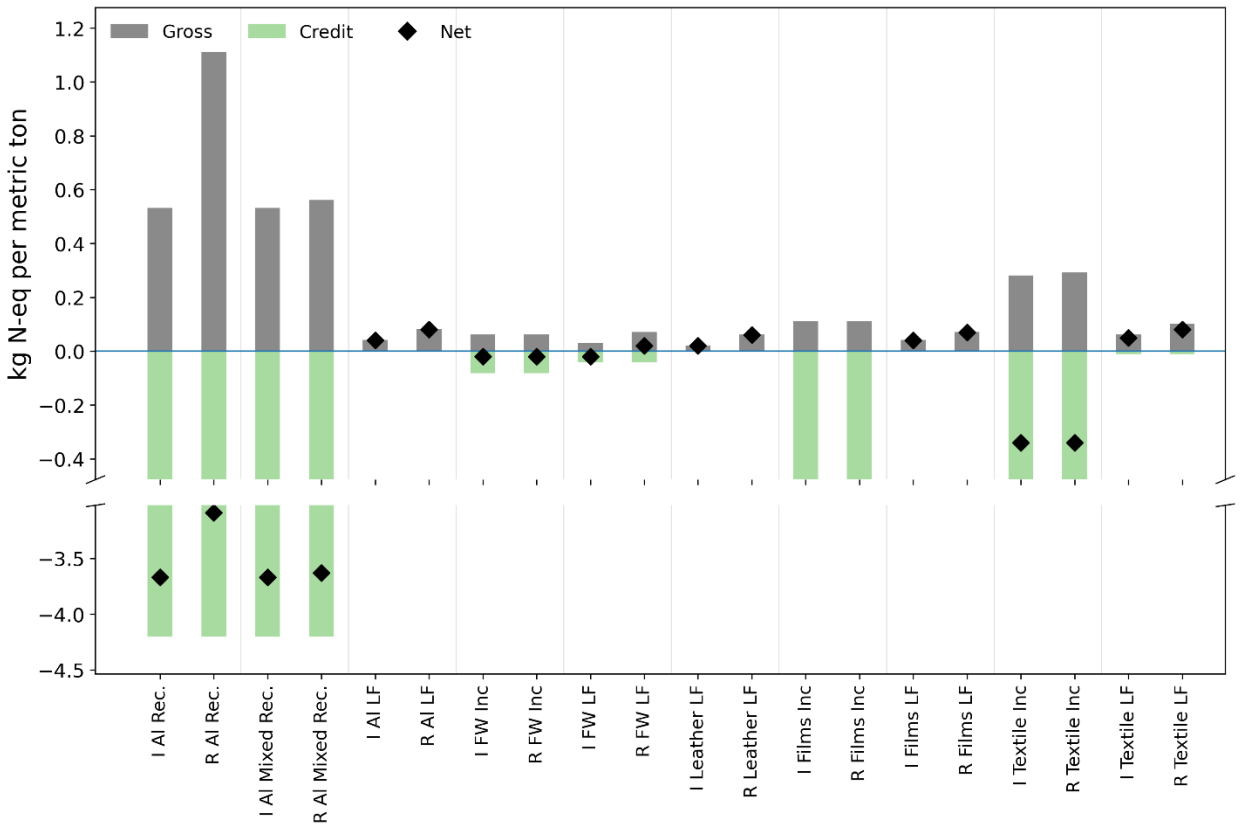


Figure 8 - Eutrophication impacts per metric ton treated, by waste stream and treatment. Gray bars indicate gross impacts in kg N-eq/t waste treated, green bars indicate credits from avoided products, black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.

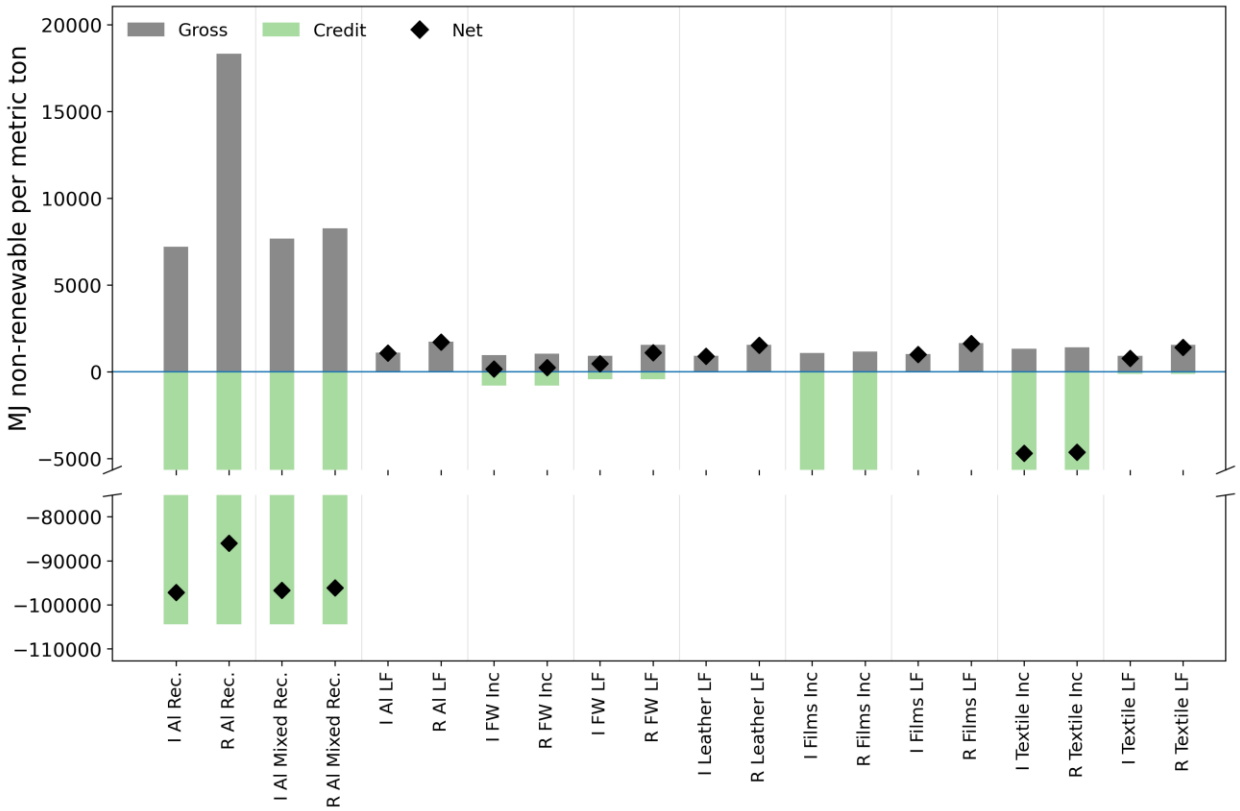


Figure 9 - Non-renewable energy demand per metric ton treated by waste stream and treatment. Gray bars indicate gross energy impacts in MJ non-renewable primary energy/t waste treated, green bars indicate credits from avoided products, black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.

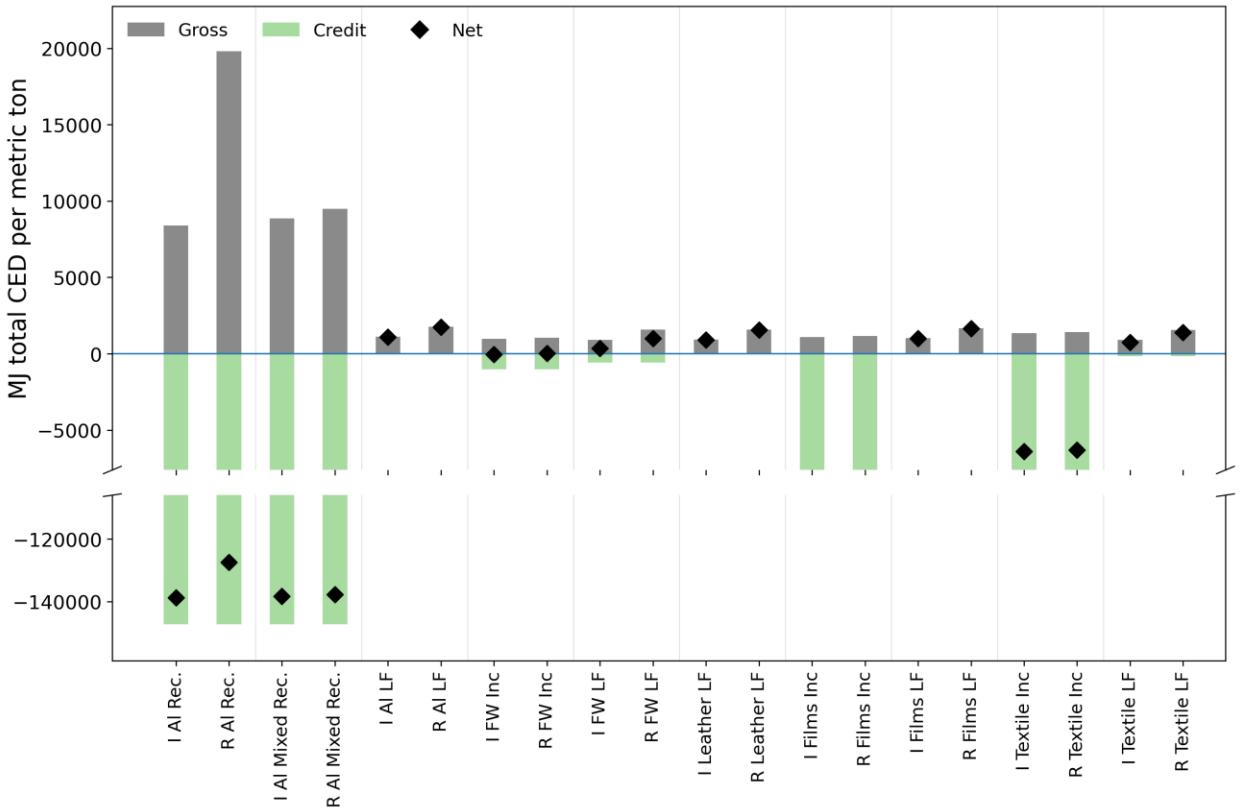


Figure 10 - Total CED per metric ton treated by waste stream and treatment. Gray bars indicate gross energy impacts in MJ non-renewable primary energy/t waste treated, green bars indicate credits from avoided products, black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.

Table 24 - Gross, credit, and net impact factors by sector, waste stream, and treatment for emissions impact (t CO_2-eq), eutrophication (kg N-eq), and energy (MJ). (ICI = Institutional/Industrial/Commercial, Res = Residential)

			Emissions (kg CO_2-eq / t)			Eutrophication (kg N-eq / t)			CED Non-renew (MJ / t)			Tot CED (MJ / t)		
Waste Type	**Treatment**	**Sector**	**Gross**	**Credit**	**Net**	**Gross**	**Credi**	**Net**	**Gross**	**Credit**	**Net**	**Gross**	**Credit**	**Net**
Aluminum	Recycling Dep.	ICI	560	-9,787	-9,227	0.53	-4.20	-3.67	7,167	-104,353	-97,186	8,363	-147,192	-138,829
Aluminum	Recycling Dep.	Res	1,348	-9,787	-8,440	1.11	-4.20	-3.09	18,304	-104,354	-86,050	19,782	-147,192	-127,410
Aluminum	Recyc. Mixed	ICI	542	-9,787	-9,245	0.53	-4.20	-3.67	7,636	-104,353	-96,717	8,831	-147,191	-138,360
Aluminum	Recyc. Mixed	Res	584	-9,787	-9,203	0.56	-4.20	-3.63	8,246	-104,353	-96,108	9,463	-147,192	-137,729
Aluminum	Landfill	ICI	67	0	67	0.04	0.00	0.04	1,072	0	1,072	1,099	0	1,099
Aluminum	Landfill	Res	112	0	112	0.08	0.00	0.08	1,705	0	1,705	1,752	0	1,752
Food Waste	Incineration	ICI	83	-60	23	0.06	-0.08	-0.02	940	-777	163	955	-998	-43
Food Waste	Incineration	Res	88	-60	28	0.06	-0.08	-0.02	1,016	-777	239	1,033	-998	35
Food Waste	Landfill	ICI	1,063	-33	1,030	0.03	-0.04	-0.02	887	-423	464	894	-543	351
Food Waste	Landfill	Res	1,107	-33	1,075	0.07	-0.04	0.02	1,520	-423	1,097	1,548	-543	1,005
Leather	Landfill	ICI	58	0	58	0.02	0.00	0.02	899	0	899	905	0	905
Leather	Landfill	Res	103	0	103	0.06	0.00	0.06	1,532	0	1,532	1,558	0	1,558
Plastic Film	Incineration	ICI	3,147	-748	2,399	0.11	-1.00	-0.90	1,053	-9,675	-8,622	1,065	-12,419	-11,355
Plastic Film	Incineration	Res	3,153	-748	2,405	0.11	-1.00	-0.89	1,129	-9,675	-8,546	1,143	-12,419	-11,277
Plastic Film	Landfill	ICI	157	0	157	0.04	0.00	0.04	990	0	990	998	0	998
Plastic Film	Landfill	Res	201	0	201	0.07	0.00	0.07	1,623	0	1,623	1,651	0	1,651
Textile	Incineration	ICI	1,535	-464	1,070	0.28	-0.62	-0.34	1,298	-6,007	-4,709	1,319	-7,711	-6,392
Textile	Incineration	Res	1,540	-464	1,076	0.29	-0.62	-0.34	1,374	-6,007	-4,633	1,397	-7,711	-6,314
Textile	Landfill	ICI	339	-8	331	0.06	-0.01	0.05	883	-108	775	890	-139	751
Textile	Landfill	Res	384	-8	375	0.10	-0.01	0.08	1,516	-108	1,408	1,543	-139	1,404

4.1.1 Aluminum

On a per-ton treated basis, aluminum recycling provides the largest net environmental benefits observed in this study across emissions impacts, eutrophication, and non-renewable energy demand (

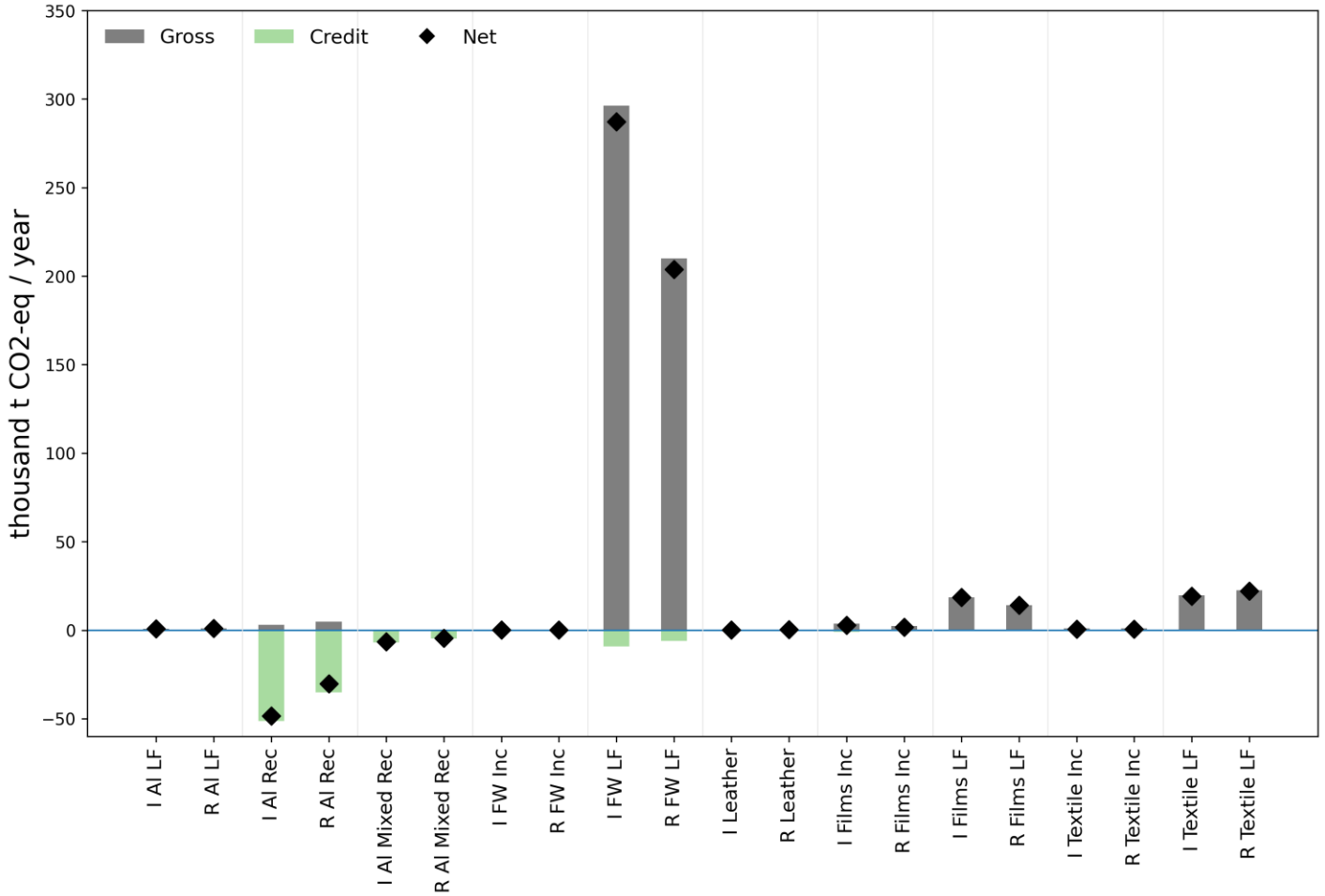


Figure 11 to Figure 14; and Table 24. Net emissions impact benefits range from approximately –8.4 to –9.2 t CO_2-eq per ton recycled, net energy benefits from roughly –88 to –97 GJ per ton recycled, and eutrophication credits from about –3.1 to –3.7 kg N-eq per ton recycled. These outcomes are attributable primarily to displacement of primary aluminum production and reflect the upper bounds of benefits from avoided products due to 1:1 replacement assumption.

Recycling operations and processing require approximately 7.1 to 15 GJ per ton of non-renewable energy, exceeding the direct energy demands of landfill and incineration. Gross emissions impacts associated with recycling range from approximately 0.6 to 1.3 t CO_2-eq per ton. In both cases, these incurred burdens are small relative to the avoided primary

aluminum production credits of approximately 9.8 t CO_2-eq and 89 to 97 GJ per ton, which determine net outcomes.

These results confirm aluminum recycling as the single most environmentally beneficial pathway in the entire system. By comparison, landfilled aluminum exhibits modest but non-zero impacts, approximately +67 to +112 kg CO_2-eq per ton and +1.1 to +1.7 GJ per ton, which is reflective of the absence of material recovery and avoided production credits (Table 24).

4.1.2 Food Waste

Landfilling represents the highest-impact management pathway for food waste, with emissions impacts of approximately +1.03 to +1.08 t CO_2-eq per ton of food waste treated and net non-renewable energy burdens of roughly +464 to +1,097 MJ per ton. These impacts are driven primarily by fugitive methane emissions from anaerobic decomposition, coupled with limited energy recovery. Eutrophication impacts without long-term consideration are comparatively small and near net-neutral on a per-ton basis.

Food waste incineration results in substantially lower emissions impacts on a per-ton basis, producing approximately +23 to +28 kg CO_2-eq per ton of waste treated, along with modest net non-renewable energy burdens of +160 to +240 MJ per ton and slightly net-negative eutrophication impacts attributable to energy recovery. While recovered energy partially offsets combustion-related emissions, it does not fully compensate for them in emissions impact terms (Table 24).

4.1.3 Plastic Films

Landfilled plastic films exhibit emissions impacts of approximately +160 to +200 kg CO_2-eq per ton of waste treated and net non-renewable energy burdens of roughly +1.0 to +1.6 GJ per ton, reflecting the absence of both material recovery and energy offsets (Table 20). Landfilled plastic films result in net eutrophication burdens.

Plastic film incineration results in substantially higher emissions impacts on a per-ton basis, approximately +2.4 t CO_2-eq per ton of waste treated, alongside substantial net non-renewable energy credits of approximately –8.5 to –8.6 GJ per ton and strongly net-negative eutrophication impacts attributable to energy recovery. Emissions impacts are dominated by fossil carbon emissions, and while recovered energy provides significant credits for avoided electricity production from the grid, these energy credits are insufficient to offset the increased emissions (Table 24).

4.1.4 Textiles and Leather

Landfilling textile waste results in emissions impacts of approximately +330 to +380 kg CO_2-eq per ton of waste treated, net non-renewable energy burdens of +0.8 to +1.4 GJ per

ton, and net eutrophication burdens under the no–long-term characterization (Table 24). These impacts reflect the absence of material recovery and avoided production credits associated with disposal-based management.

Textile incineration results in higher emissions impacts on a per-ton basis, approximately +1.07 t CO_2-eq per ton treated but provides net non-renewable energy benefits of roughly –4.6 to –4.7 GJ per ton and net-negative eutrophication impacts attributable to energy recovery. As observed for plastics, combustion-related emissions drive emissions impacts, while recovered energy provides compensating credits that are not sufficient to offset increased emissions .

Leather waste was modeled only under landfill treatment and exhibits relatively low air emissions impacts, approximately +58 to +103 kg CO_2-eq per ton of waste treated, reflecting its comparatively inert material properties. However, leather disposal remains a net environmental burden due to the absence of recovery pathways, including net non-renewable energy demands of approximately +900 to +1,530 MJ per ton (Table 24).

4.1.5 ICI Versus Residential Performance

Across nearly all materials, residential landfill pathways show approximately 20–60% higher air emissions and energy impacts per ton relative to ICI. Recycling benefits remain comparable between sectors, but residential gross burdens are consistently higher. These differences reflect structural characteristics of residential collection systems, including less efficient logistics, smaller generators, and higher per-ton collection intensity especially for waste and recyclables transported by personal vehicles.

4.2 Statewide Life Cycle Impact Assessment Results

Statewide impacts were calculated by scaling per-ton LCIA results by annual material tonnage for ICI and residential streams, as described in Section 4.1 (See overview in Table 25). Reported values reflect modeled average conditions under the baseline parameter set and represent the aggregate environmental consequences of the modeled waste management system for the study year.

As with all attributional life cycle assessment results, statewide impacts are subject to uncertainty arising from multiple sources, including waste composition assumptions, transport distances, landfill gas management performance, material recovery efficiencies, and background life cycle inventory data. Accordingly, the results presented in this section are intended to support comparative interpretation of material streams and management pathways rather than precise prediction of absolute impacts.

Quantitative uncertainty and sensitivity analyses are presented in Section 5. These analyses include:

- Deterministic sensitivity testing of high-leverage parameters (e.g., fugitive methane emissions, aluminum recovery efficiency, transportation allocation, and participation rates),
- Bootstrap-derived empirical bounds for GIS-based travel distances,
- Data-informed variability in aluminum processing residuals, and
- Structural modeling sensitivity (e.g., marginal versus average electricity displacement assumptions).

Sensitivity and uncertainty results are applied in the Interpretation phase to evaluate the robustness of statewide comparative findings, assess the stability of pathway rankings, and identify parameters that materially influence air emissions, energy, and eutrophication outcomes. These analyses are not used to revise baseline point estimates reported in Section 4 but rather to contextualize them within plausible bounds of system performance and behavioral variability.

4.2.1 Landfill Pathways

Landfilling represents the dominant end-of-life management pathway (99% of tonnage) for the waste streams evaluated in this study and is the primary driver of statewide environmental impacts. Accordingly, results for landfill pathways are presented first, with emphasis on their contribution to air emission impacts, eutrophication, and non-renewable energy demand.

4.2.1.1 Food Waste Landfill

Landfilled food waste represents the single largest contributor to statewide air emissions impacts (

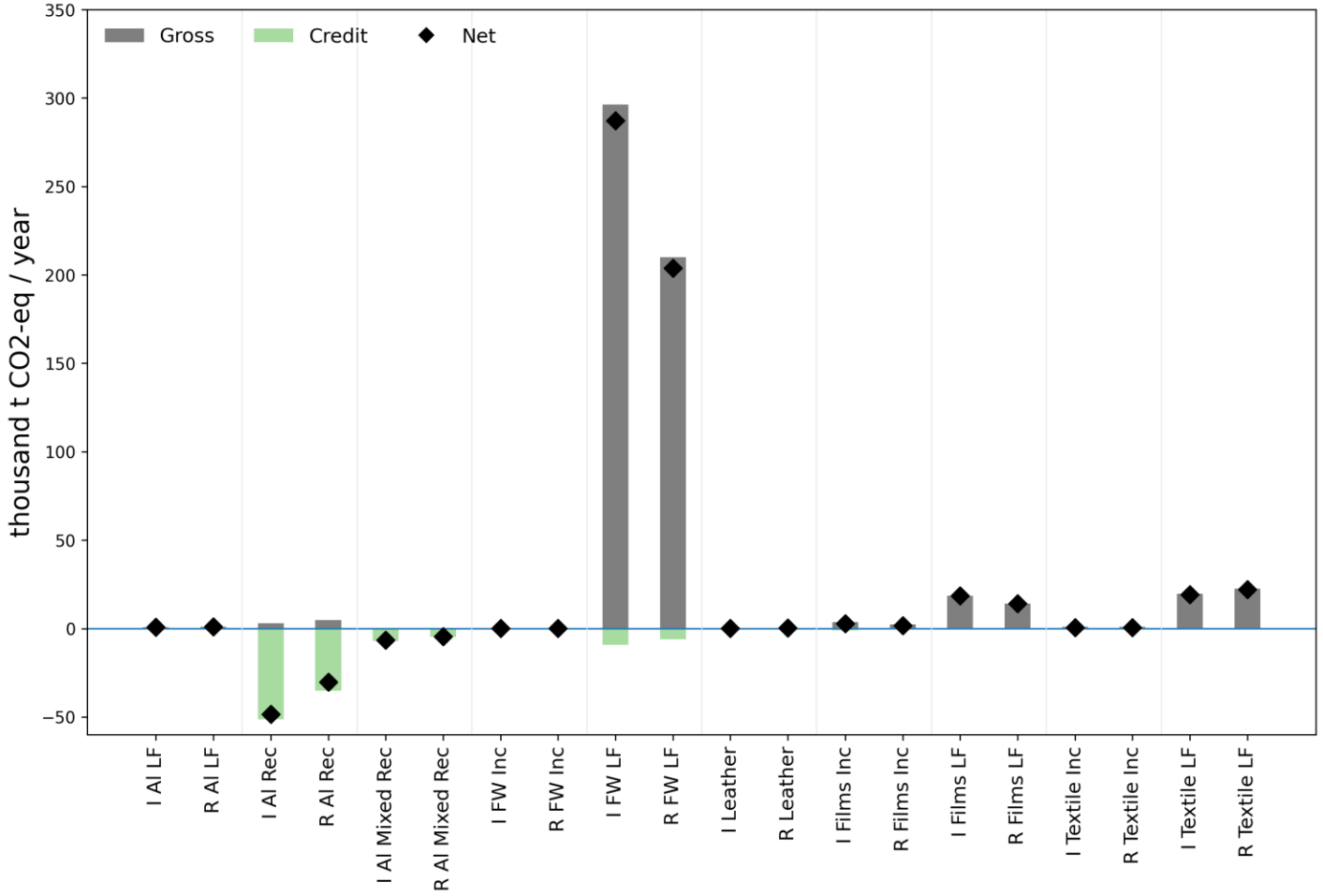


Figure 11, and Table 21). Combined ICI and Residential food waste disposal generates approximately 491,000 metric tons CO_2-equivalent per year, accounting for the majority of landfill-related emissions impact burden. These impacts are driven primarily by methane emissions from anaerobic decomposition. Eutrophication impacts are comparatively small and near net-neutral at the statewide level (

Figure 12). Non-renewable energy demand totals approximately 337 TJ per year, reflecting collection and disposal operations with limited offsetting recovery (Figure 13). Total CED is slightly lower at 288 TJ, reflecting larger non-renewable credits for electricity production from captured landfill gas (Figure 14).

4.2.1.3 Textile Waste Landfill

Landfilled textiles contribute approximately 41,000 metric tons CO_2-equivalent per year, making textiles the second-largest landfill-related air emissions driver after food waste (

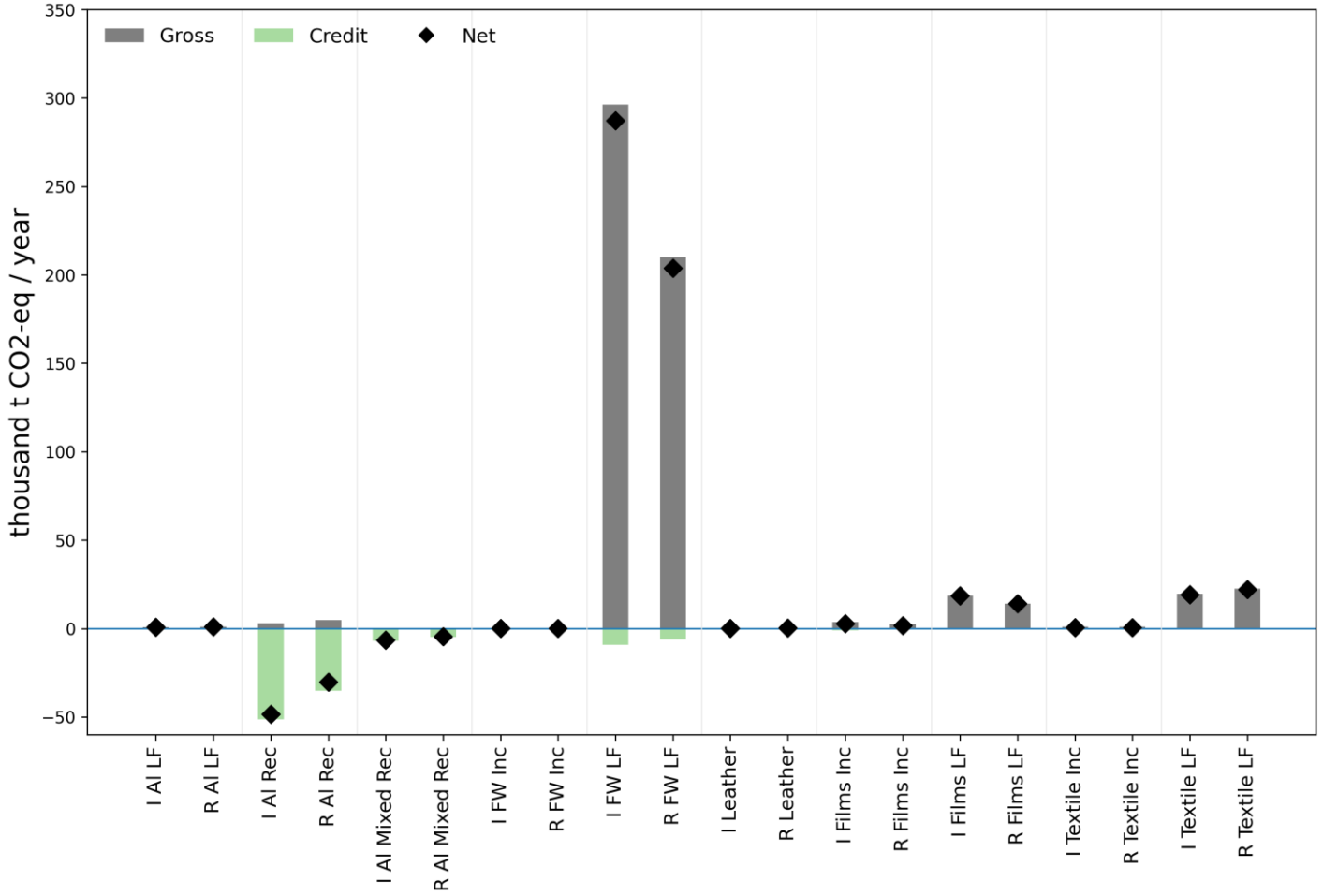


Figure 11, Table 21). Non-renewable energy demand from textile disposal totals approximately 128 TJ per year, with residential systems contributing a disproportionate share due to higher per-ton collection burdens (Figure 13). Eutrophication impacts total approximately 7,600 kg N-equivalent per year, with residential textiles exceeding ICI impacts despite similar tonnage, reflecting structural differences in collection efficiency (

Figure 12).

4.2.1.2 Plastic Films Landfill

Plastic films constitute the third-largest landfill-related air emissions burden, contributing approximately 33,000 metric tons CO_2-equivalent per year statewide (

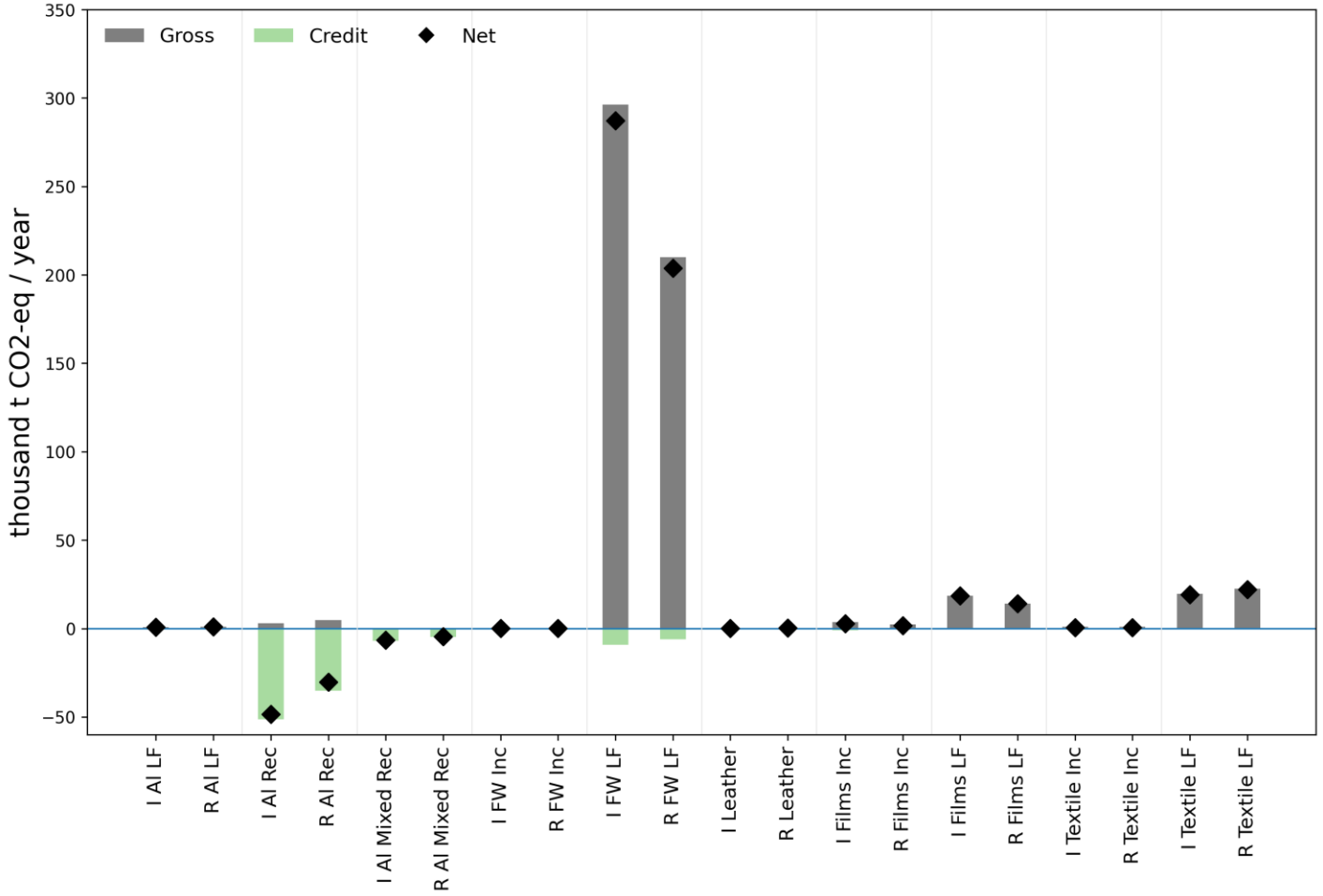


Figure 11, Table 21). Landfilled films also generate substantial non-renewable energy demand, totaling approximately 230 TJ per year, driven by the transportation and disposal operations with no material recovery and the fact that plastic films make up the second largest waste flow studied. Eutrophication impacts total approximately 9,300 kg N-eq per year (Table 21).

4.2.1.4 Aluminum and Leather Landfill

Aluminum waste treated by landfill makes up less than 3% of the waste flows studied. It contributes approximately 1,900 metric tons CO_2-equivalent per year of net emissions impact statewide and approximately 29 TJ per year of non-renewable energy demand. Leather disposal contributes less than 500 metric tons CO_2-equivalent per year and approximately 7 TJ per year of energy demand.

While these streams represent relatively minor direct landfill burdens, aluminum disposal carries a substantial opportunity cost when compared to recycling outcomes (Table 24, Table 25).

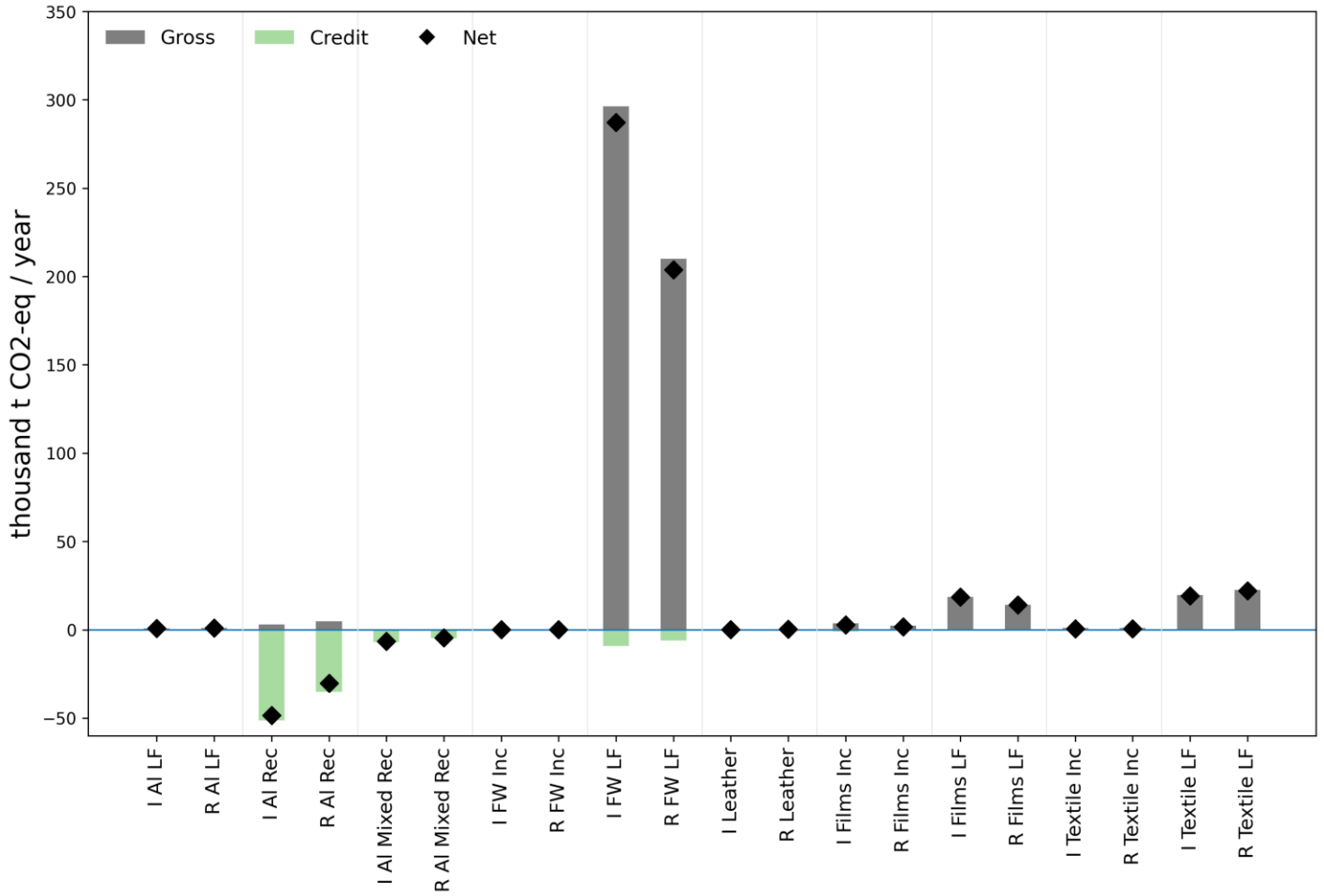


*Figure 11 - Statewide air emissions impacts by waste stream and treatment. Gray bars indicate gross **air emissions impacts** in t CO_2-eq/year, green bars indicate credits from avoided products, and black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.*

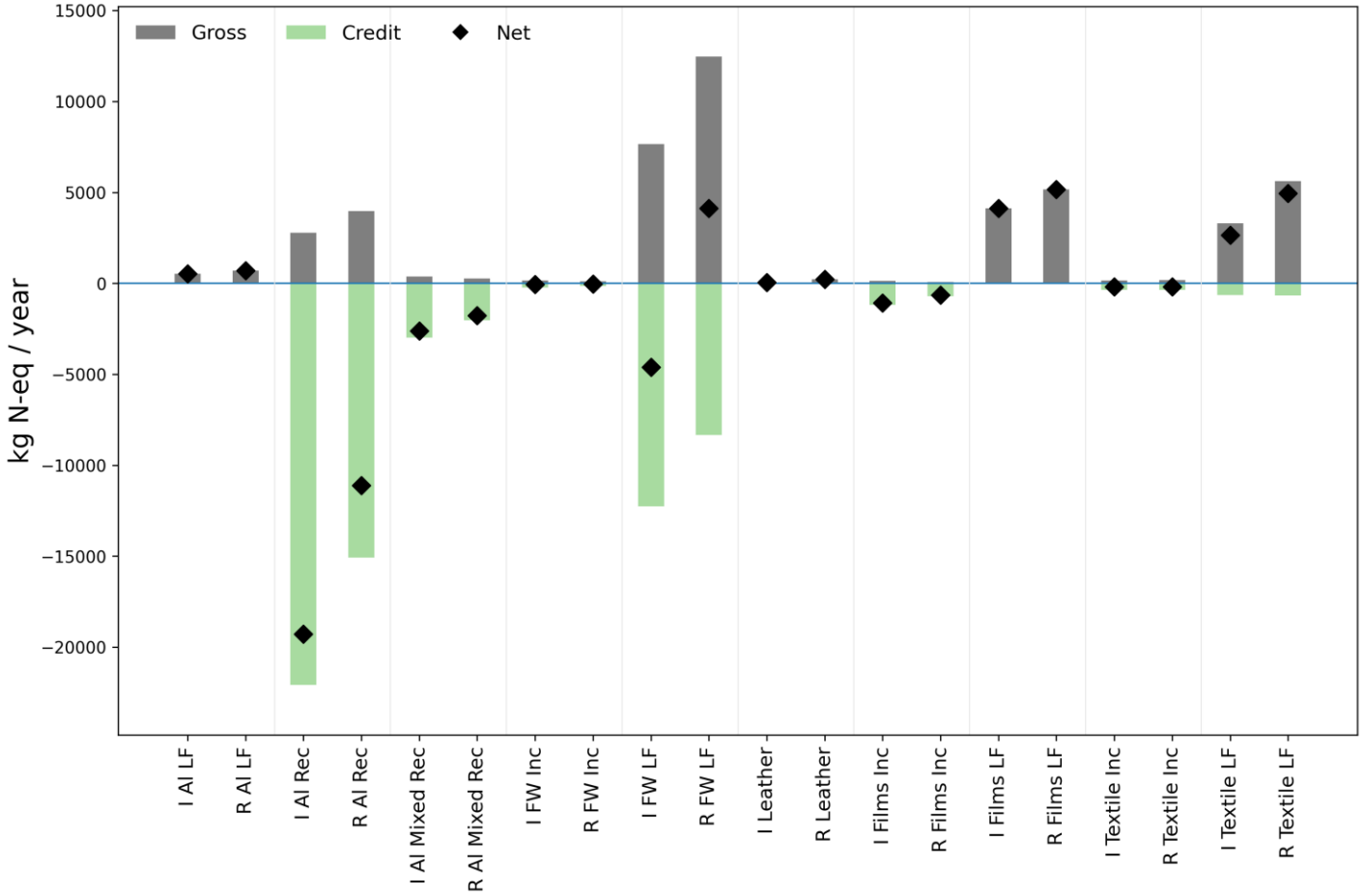


Figure 12 - Statewide eutrophication impacts by waste stream and treatment. Gray bars indicate ***eutrophication*** *impacts in kg N-eq/year; green bars indicate credits from avoided products; black diamond indicates net impact (gross – credit). Please see Section 5.1.3 for an explanation for eutrophication impact difference for food waste from ICI and Residential sources. I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.*

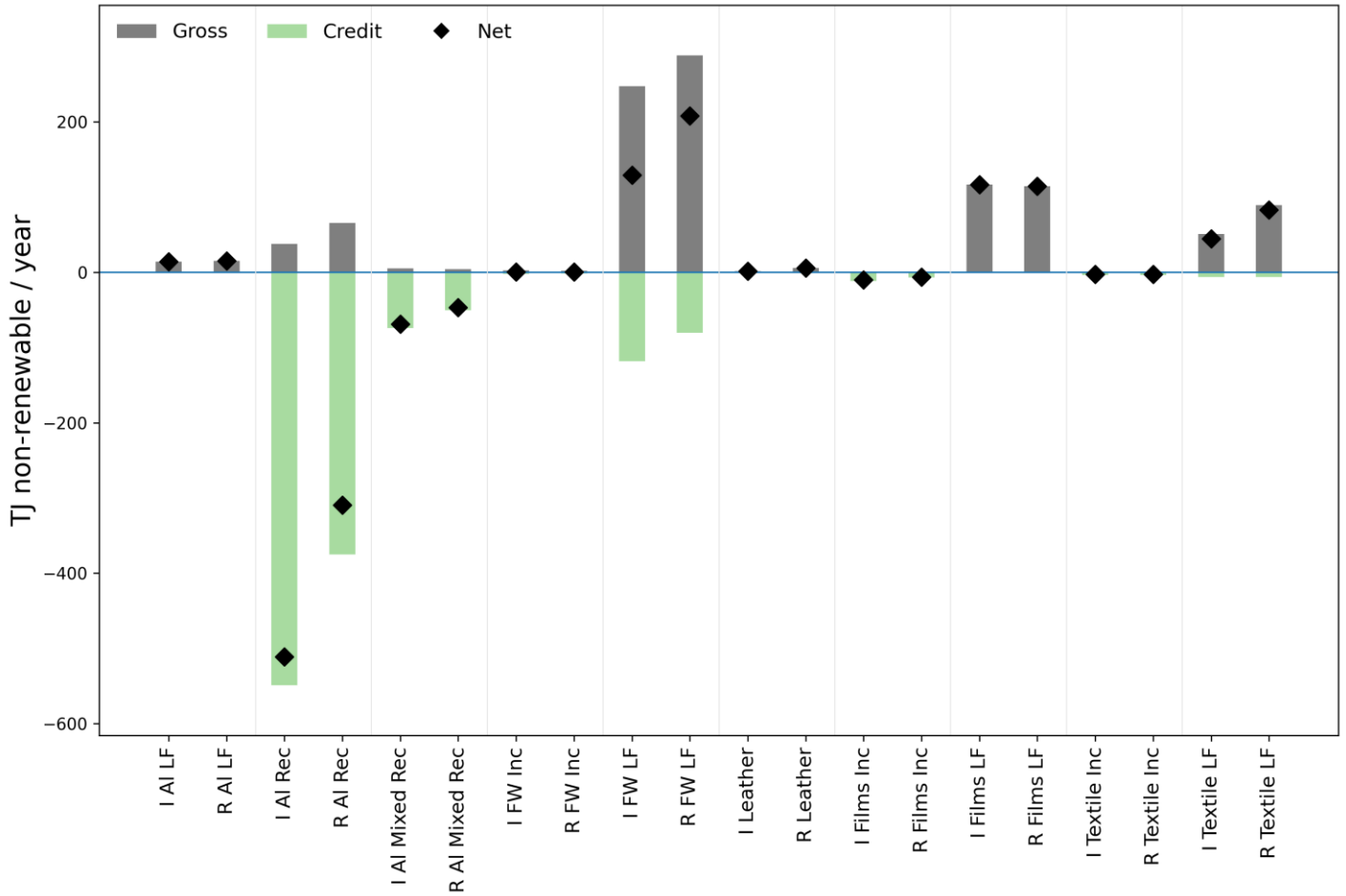


*Figure 13 - Statewide energy impacts by waste stream and treatment. Gray bars indicate gross **non-renewable** energy (CED) impacts in TJ/year (1 x 10^6 MJ); green bars indicate credits from avoided products; black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.*

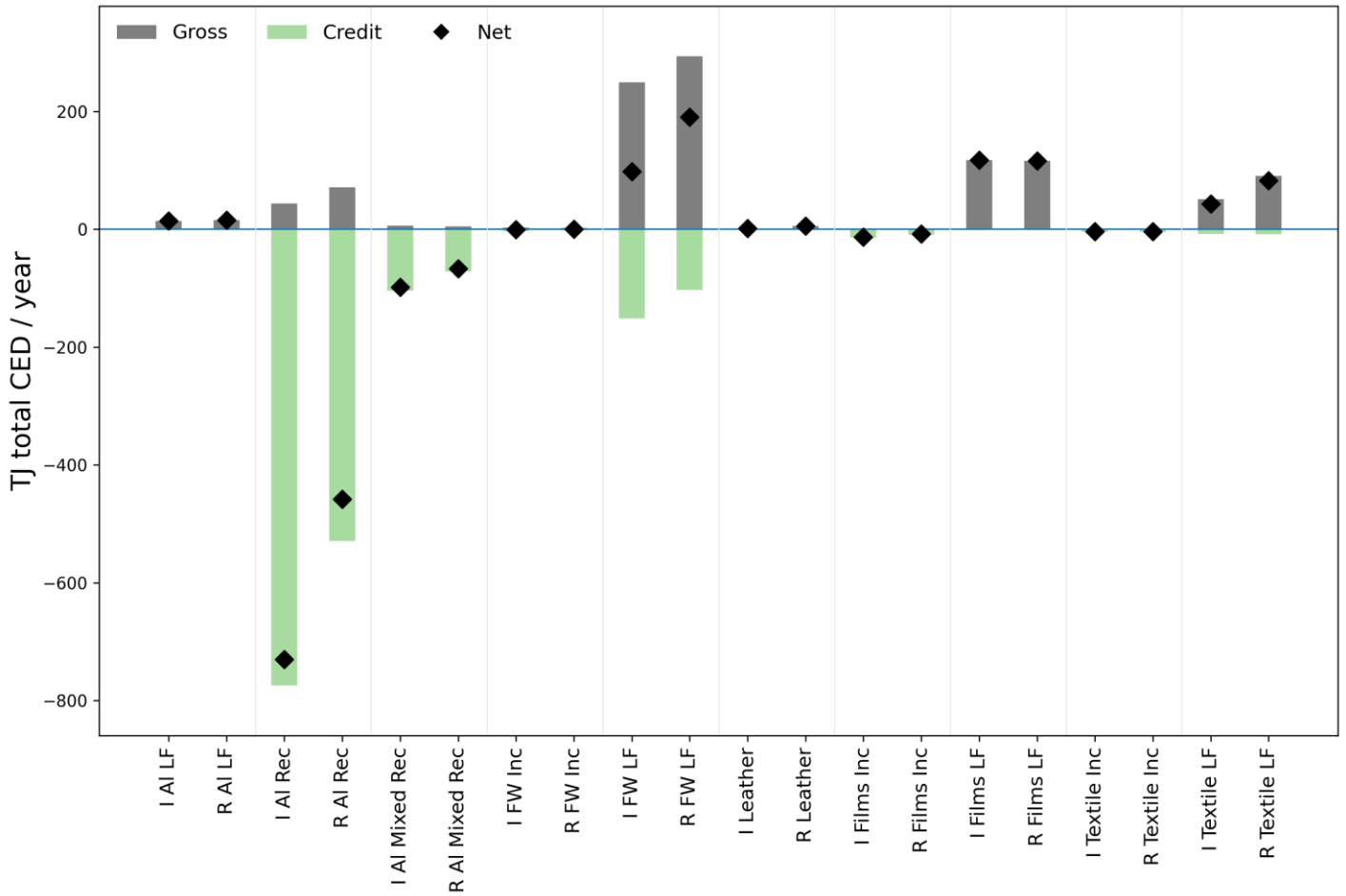


*Figure 14 - Statewide energy impacts by waste stream and treatment. Gray bars indicate gross **Total CED** impacts in TJ/year (1 x 10^6 MJ); green bars indicate credits from avoided products; black diamond indicates net impact (gross – credit). I = ICI, R = Residential, LF = Landfill, Inc = Incineration, FW = Food Waste, Al = Aluminum.*

Table 25 – Overview of Statewide LCIA results by material and pathway. For air emissions and eutrophication impacts, values are rounded to the nearest 10 or 100 to reflect the uncertainty inherent in life cycle inventory data and modeling assumptions. Rounding does not affect interpretation of relative magnitudes or comparative results.

Material / Pathway	Air emissions (t CO2-eq/yr)	Eutrophication (kg N-eq/yr)	CED Non-renewable (TJ/yr)	CED Total (TJ/yr)
Food Waste – Landfill (Total)	490,900	−470	337	288
└ ICI	287,200	−4,600	129	98
└ Residential	203,700	4,100	208	190
Textiles – Landfill (Total)	41,200	7,600	127	126
└ ICI	19,100	2,700	45	43
└ Residential	22,100	4,900	83	83
Plastic Films – Landfill (Total)	32,600	9,300	231	234
└ ICI	18,400	4,100	116	117
└ Residential	14,200	5,200	114	116
Aluminum – Landfill (Total)	1,900	1,200	30	30
└ ICI	890	530	14	15
└ Residential	1,000	710	15	16
Leather – Landfill (Total)	490	270	7	8
└ ICI	110	40	2	2
└ Residential	390	230	6	6
Aluminum – Recycling (Total)	−89,900	−34,800	−936	−1,353
└ ICI	−55,100	−21,900	−580	−828
└ Residential	−34,800	−12,900	−356	−525
Incineration (All Materials)	5,900	−2,200	−21	−29

Note: CED Non-renew. represents cumulative demand for non-renewable primary energy resources, while Total CED represents cumulative demand for all primary energy resources, including renewable sources such as biomass, hydropower, wind, and solar.

4.2.2 Recycling Pathways

Recycling provides the largest net environmental benefits observed in this assessment, driven almost entirely by recovery of aluminum deposit containers. Statewide aluminum recycling (deposit and mixed streams across ICI and residential sectors) yields approximately 79,000 metric tons CO_2-equivalent per year in avoided emissions impacts and approximately 820 TJ per year in avoided non-renewable energy demand (Table 24).

These benefits exceed direct landfill burdens for aluminum by more than fifty-fold and are substantially larger than impacts associated with all other modeled individual recovery or waste management pathways (

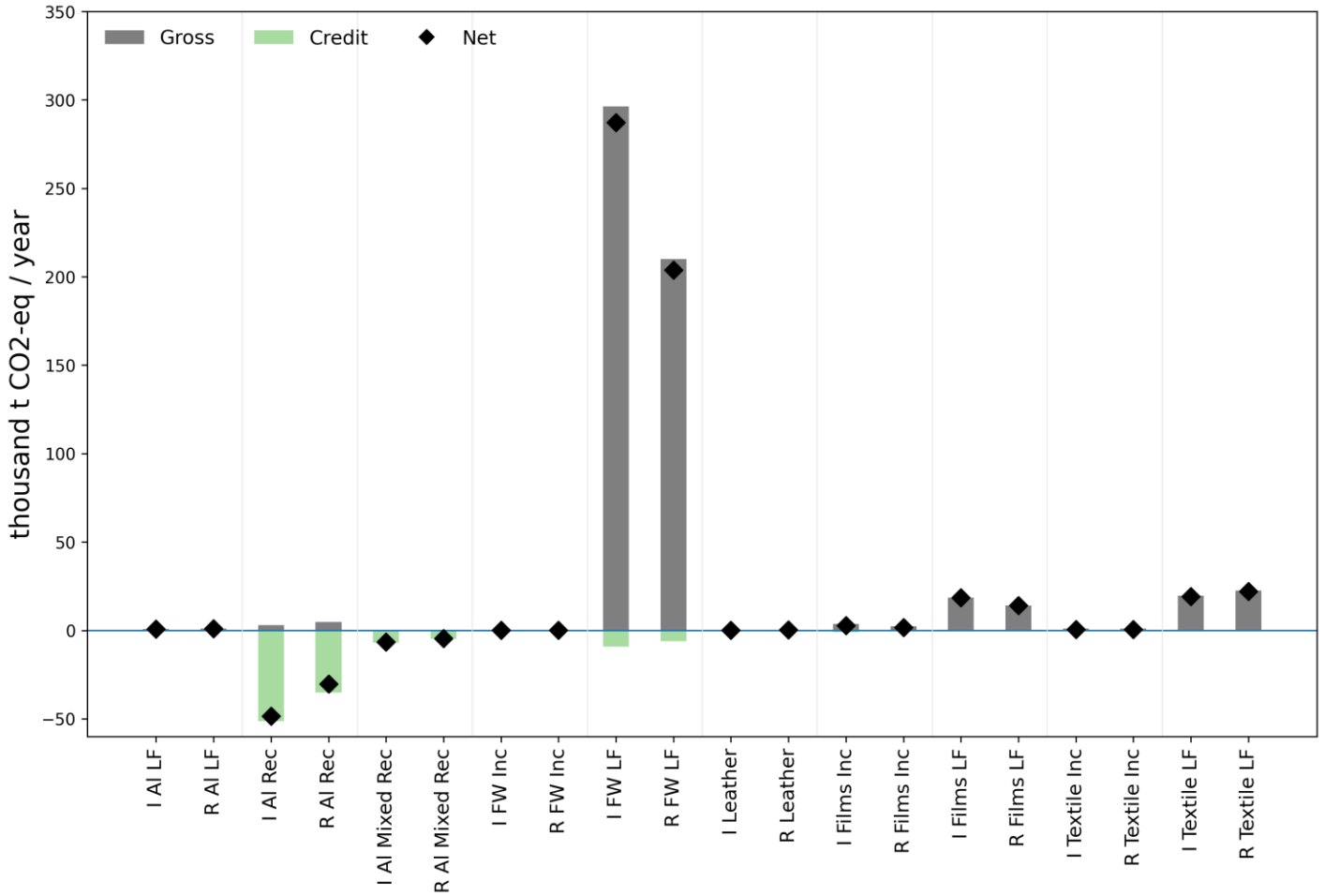


Figure 11 to Figure 14).

Deposit aluminum streams account for the majority of recycling benefits due to higher throughput, while mixed aluminum streams provide smaller but still meaningful contributions. Differences between ICI and residential recycling performance are attributable primarily to upstream collection and transportation burdens rather than differences in avoided material production.

Recycling of plastic films and textiles is currently limited in scale and does not materially influence statewide outcomes under existing system conditions. Potential impacts of textile recycling are evaluated in the alternative waste management scenarios presented below.

4.2.3 Incineration

Incineration represents approximately one percent of modeled tonnage statewide and therefore plays a limited role in overall environmental performance (

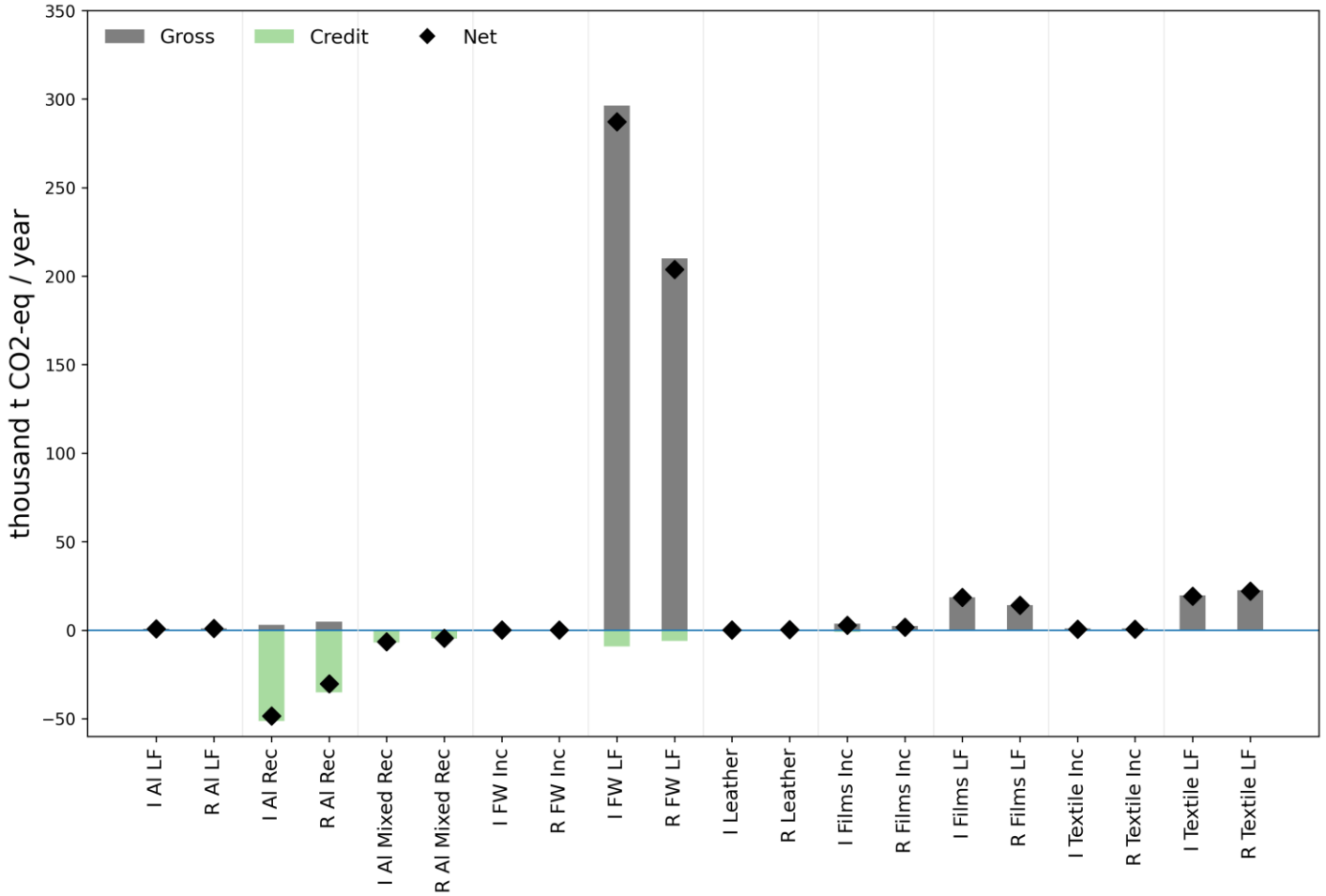


Figure 11 to Figure 14, Table 25). Incineration provides modest energy offsets from electricity production, approximately 20 TJ per year statewide, but increases air emissions impacts by approximately 5,900 metric tons CO_2-equivalent per year, primarily from plastics and textiles. Given its limited current deployment and lack of planned expansion, incineration does not materially influence statewide outcomes relative to landfill and recycling pathways.

4.3 Key Contributing Factors by Material and Management Pathway

Figures Figure 15 to Figure 17 present stacked contribution results by material and end-of-life pathway, showing the relative contributions of collection and transportation, treatment processes, and avoided production or energy recovery to net impacts for air emissions, eutrophication, and non-renewable energy demand. *Direct* refers to direct impacts from the waste management process, e.g. direct methane emissions from landfilling food waste. *Leachate* refers to impacts from the leachate transportation or treatment process.

Transport encompasses all collection and interfacility movement of MSW or recycled materials. *Treatment* includes impacts from inputs into the waste management process, including upstream cradle-to-gate impacts, e.g. silicon used the aluminum recycling process. *Others* refers to the sum of all the smaller impact processes not yet captured in the categorization major drivers or below the 1% threshold used in this analysis. Finally, *credit* refers to the credit taken for the displacement of energy or materials produced by the waste management process, e.g. electricity or aluminum ingot.

Contribution analyses were used to identify the dominant life cycle processes driving net impacts across air emissions, eutrophication, and non-renewable energy demand for each modeled material and management pathway. Results indicate that overall waste management system performance is governed by a small number of high-impact processes, with relative contributions varying substantially by material type and end-of-life option.

4.3.1 Landfill Pathways

For food waste, air emissions impacts are dominated by methane emissions from anaerobic decomposition in landfills, which account for the majority (90%+) of net burdens and were nearly 12 times greater than contributions from collection/transportation and treatment related emissions (Figure 15). Energy impacts for food waste landfill are driven primarily by collection activities and landfill operations, with limited offsetting recovery under modeled conditions (Figure 16). Eutrophication impacts are comparatively small on a per-ton basis (Figure 17) and near net-neutral at the statewide level.

For plastic films and textiles, landfill-related air emissions impacts are lower per ton than food waste. Impacts remain positive due to disposal emissions and the near absence of material recovery credits for the capture and combustion of landfill gas for plastic films, and in small amounts for the organic textile portion of textile waste. In these pathways, net impacts are driven primarily by upstream collection and landfill operations, which together account for most energy and eutrophication burdens. Residential collection contributes a larger share of per-ton impacts than ICI due to smaller load sizes and the reduced efficiency of transportation by personal vehicle, a pattern consistently observed across landfill-managed materials

Across all landfill pathways, collection and transportation represent a modest but persistent contribution to total impacts, while treatment-stage emissions (particularly methane for food waste) drive statewide air emissions impacts. The absence of avoided production credits results in uniformly positive net burdens for landfill-managed plastic films, aluminum waste, and leather.

4.3.2 Recycling Pathways

Aluminum recycling is characterized by avoided primary aluminum production overwhelmingly determining net outcomes across all impact categories. Avoided production credits for aluminum dominate air emissions and energy results, exceeding gross recycling burdens by 7x to 17x. Gross recycling impacts arise primarily from material handling, sorting, and remelting energy requirements, which are visible in the stacked bars but remain small relative to avoided production.

Differences between deposit and mixed aluminum streams reflect variation in collection and sorting intensity rather than differences in material substitution, as both pathways receive equivalent avoided virgin aluminum credits. Residential recycling exhibits higher gross collection contributions per ton than ICI, consistent with observed differences in collection efficiency.

Recycling contributions for plastics and textiles are currently negligible at the statewide scale due to limited recovery volumes, and therefore, they do not materially influence aggregate results under existing system conditions.

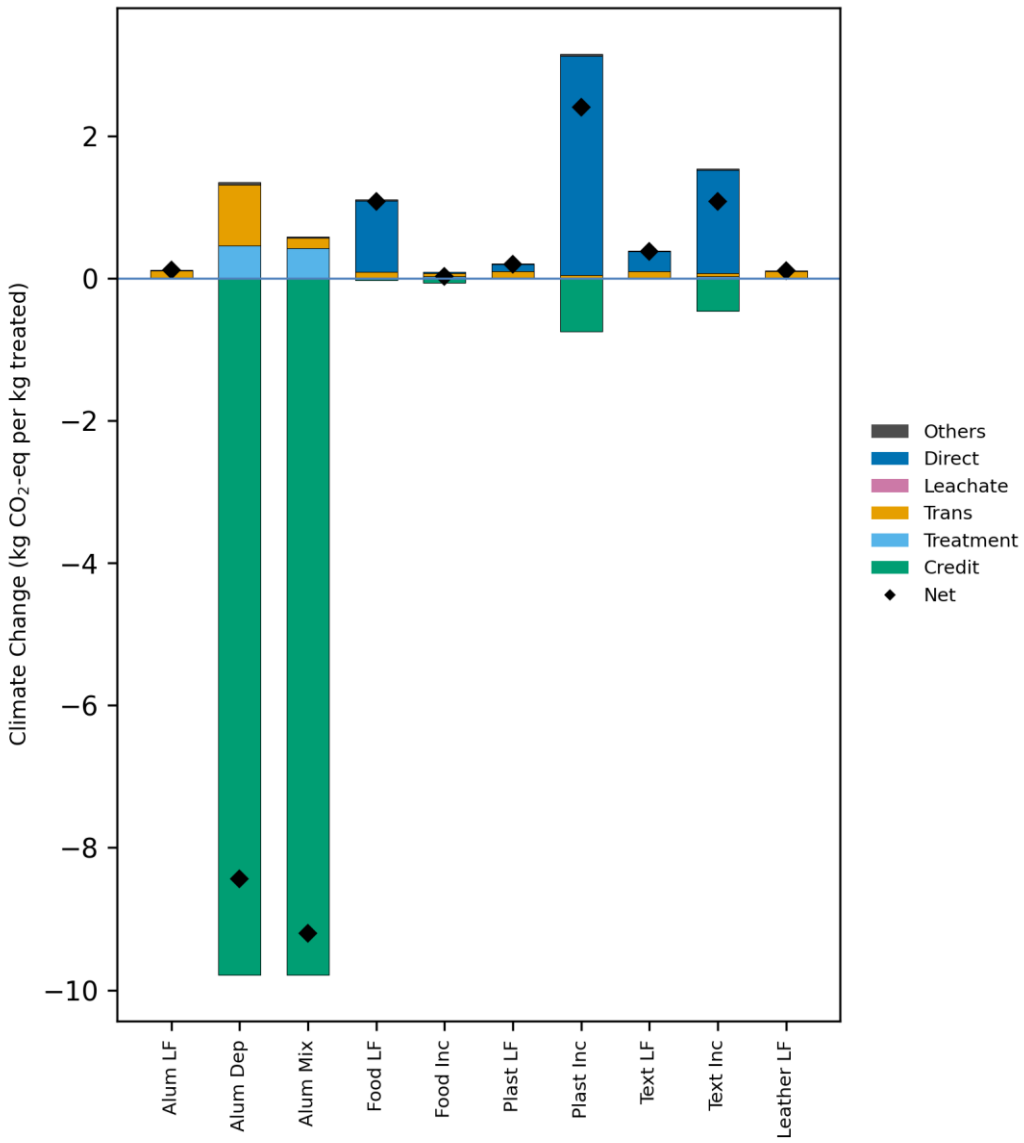


Figure 15 - Key contributing factors to air emissions impacts (kg CO_2-eq / ton treated) from residential sources by material and management pathway. Gross impacts are shown in grayscale by category and have positive values. Credits from avoided products shown in green and have negative values. Net impacts indicated by black diamond. LF = Landfill, Inc = Incineration. ICI sources follow a similar pattern but have consistently lower relative transportation impacts.

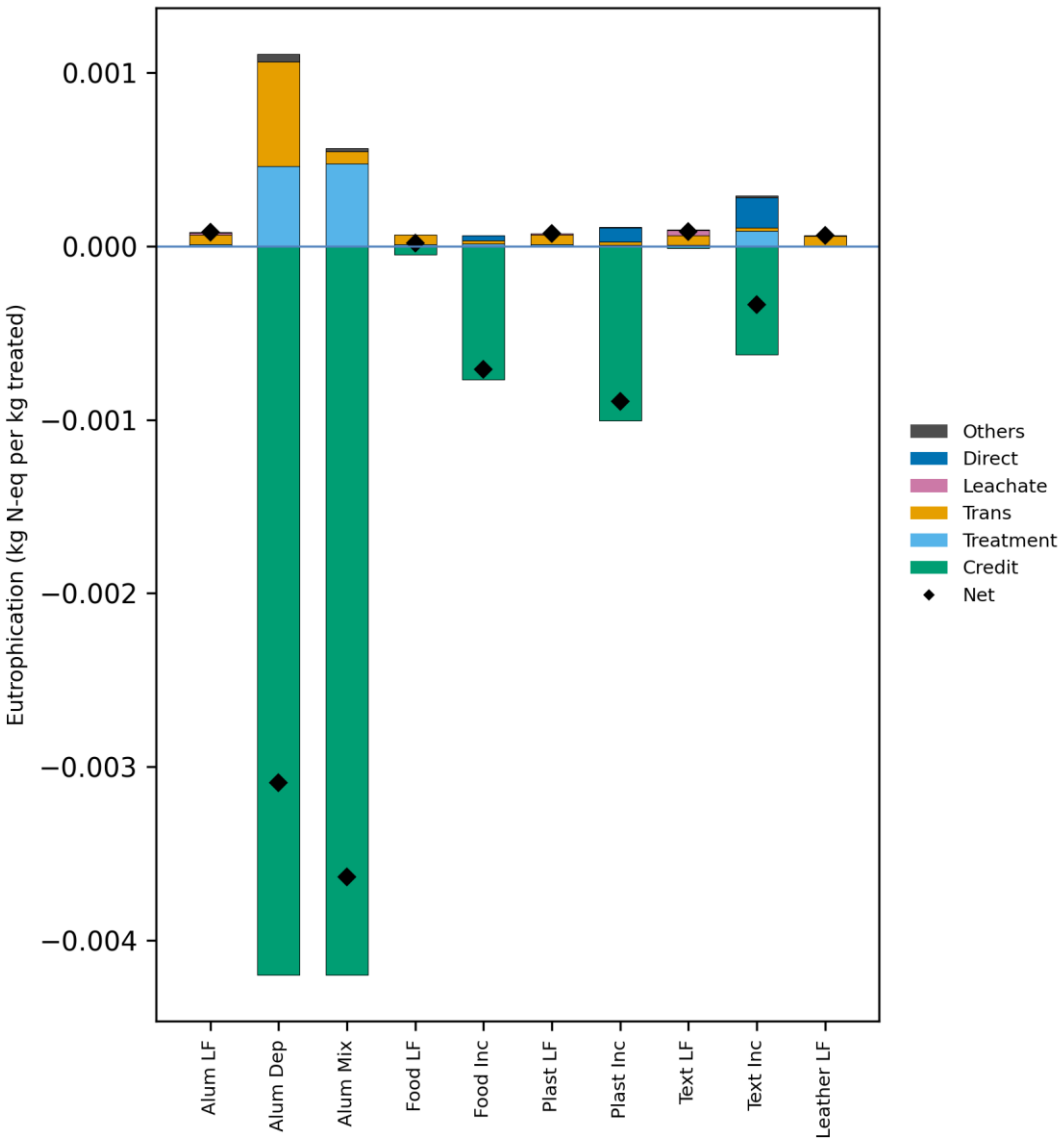


Figure 16 - Key contributing factors to Eutrophication impacts (kg N-eq / ton treated) from residential sources by material and management pathway. Gross impacts are shown in grayscale by category and have positive values. Credits from avoided products shown in green and have negative values. Net impacts indicated by black diamond. ICI sources follow a similar pattern but have consistently lower relative transportation impacts.

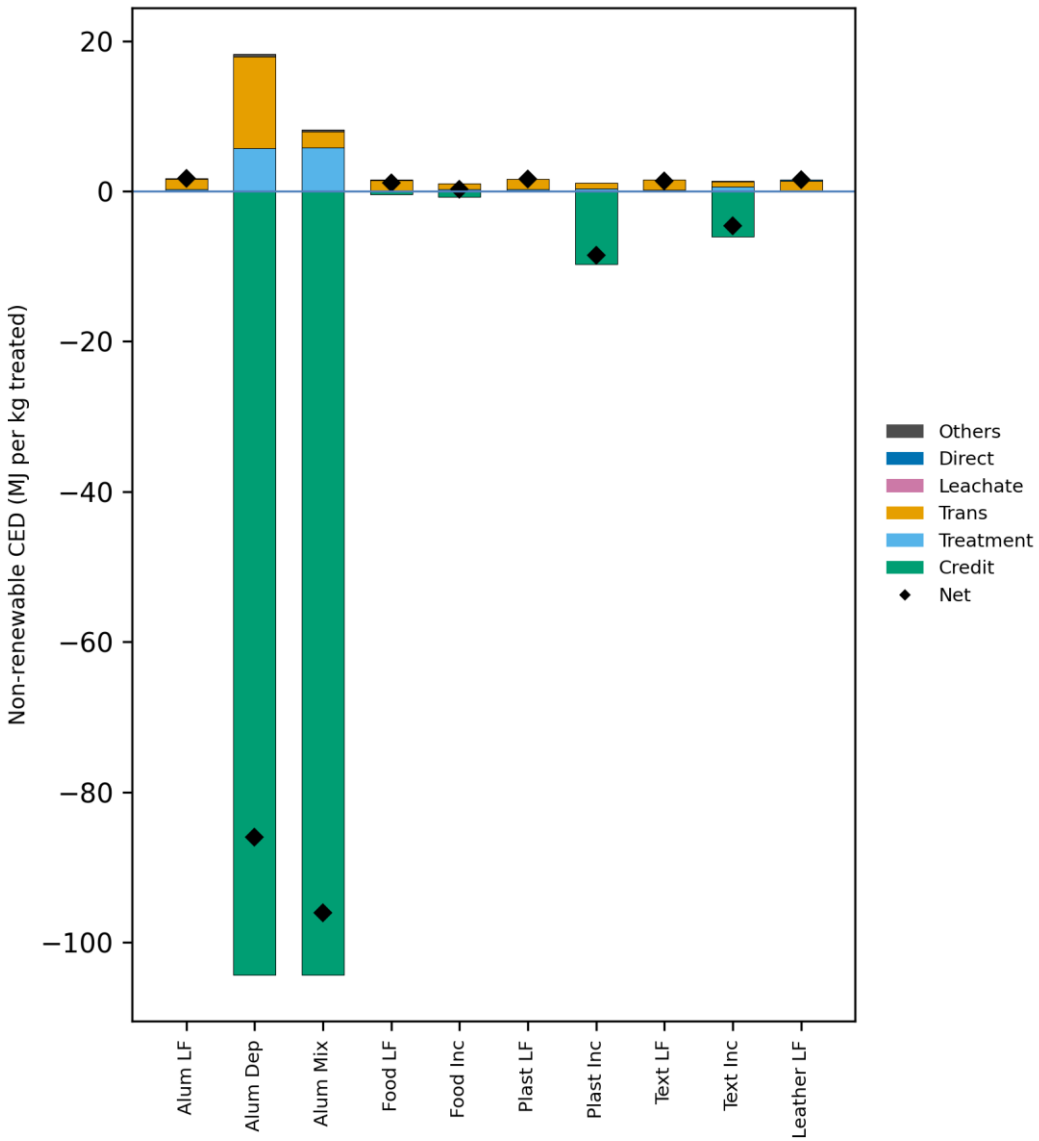


Figure 17 - Key contributing factors to Cumulative Energy Demand (CED) impacts (MJ / ton treated) from residential sources by material and management pathway. Gross impacts shown in grayscale by category and have positive values. Credits from avoided products are shown in green and have negative values. Net impacts indicated by black diamond. Y-axis broken in negative direction to allow reader to discern contributions of different gross impact categories. ICI sources follow a similar pattern but have consistently lower relative transportation impacts.

4.3.3 Incineration Pathways

For incineration pathways, air emissions impacts are driven primarily by combustion-related emissions, particularly fossil carbon in plastics and textiles. Energy recovery provides non-renewable energy credits that partially compensate for gross energy

demand, but these credits do not overcome combustion emissions in terms of air emissions. As illustrated in the stacked bars, incineration pathways exhibit a tradeoff between energy recovery benefits and air emissions burdens. Transportation burdens are minimal. However, due to limited deployment, incineration contributes only a small fraction of statewide impacts.

4.5 Quantitative Context

Across landfill pathways, fugitive methane emissions account for the majority of air emissions impacts associated with food waste, while collection and disposal operations dominate cumulative energy demand and eutrophication impacts for plastics and textiles. For aluminum recycling pathways, avoided primary aluminum production exceeds gross recycling burdens by approximately 7x to 17x for both air emissions and energy indicators, explaining the strong net environmental benefits observed. Residential collection systems consistently exhibit higher per-ton impacts than ICI systems across all material categories, reflecting more decentralized logistics and lower collection efficiencies.

These contribution patterns indicate that statewide results are primarily governed by a limited number of high-leverage mechanisms: landfill methane emissions for degradable materials, avoided primary production for recycled aluminum, and collection efficiency across residential and ICI systems. Together, these drivers largely determine both the magnitude and direction of net impacts and benefits across waste management pathways.

5. Interpretation

The quantitative findings on contributing factors described in Section 4.3 provide the foundation for interpreting system-level performance and evaluating the robustness of results under alternative assumptions. Accordingly, Section 5 examines the sensitivity of key outcomes to uncertainty in transportation distances, methane emissions, recovery efficiencies, and other influential operational parameters, and assesses the implications for comparative pathway performance.

5.1 Overview of Key Findings

Consistent with the state's objective of advancing a more integrated and forward-looking solid waste policy, this study evaluates the present-day baseline environmental impacts associated with management of four key waste streams – food waste, plastic film, textiles/leather, and mixed aluminum waste (including deposit beverage containers, non-deposit containers, and other aluminum waste). The primary purpose of the analysis is to identify the mechanisms that drive environmental performance across management pathways and also to assess how alternative strategies influence those outcomes.

Across the 20 modeled waste management pathways (10 waste streams evaluated for both Residential and ICI sources), environmental performance is governed by a limited set of structurally dominant mechanisms. These mechanisms operate differently across impact categories but consistently explain the majority of observed variation in both pathway-level and statewide results.

The principal drivers identified in this study are:

1. Fugitive methane emissions from landfill processes
2. Material recovery efficiency and associated avoided primary production credits
3. Electricity displacement assumptions for energy-recovering pathways
4. Transportation intensity and collection structure

Each of these mechanisms influence environmental performance through distinct causal pathways. Methane emissions directly determine the magnitude of emissions impacts for degradable materials. Material recovery efficiency governs the scale of avoided upstream production burdens, particularly for aluminum and textiles. Electricity displacement assumptions affect the magnitude of avoided generation credits in energy-recovering pathways. Transportation structure and participation behavior influence fossil energy demand and eutrophication through fuel combustion and associated upstream emissions.

The relative importance of these mechanisms varies by impact category. Air emissions results are primarily governed by methane emissions from landfill processes. Fossil energy demand is strongly influenced by aluminum recovery efficiency and, to a lesser extent, by residential drop-off travel allocation assumptions. Eutrophication outcomes reflect a combination of transportation-related emissions and avoided upstream production burdens. Electricity displacement assumptions affect magnitude in certain pathways but do not alter comparative rankings.

The following subsections examine each impact category in turn, identify the dominant mechanisms shaping baseline performance, and evaluate the robustness of those findings under sensitivity and alternative scenario testing.

5.1.1 Air Emissions Impacts

Emissions impact results are primarily governed by two mechanisms: fugitive methane emissions from landfill processes and avoided primary material production in recycling pathways.

Contribution analysis (Section 4.3) indicates that for degradable landfill streams (e.g., food waste and textiles), fugitive methane emissions account for more than 90% of total emissions impacts under baseline assumptions. In contrast, for aluminum recycling pathways, the credit for avoided primary aluminum production is approximately 8 times the gross air emissions impact of recycling, reflecting the intensity of primary aluminum manufacture.

Baseline net impacts range from −9,245 to 2,405 kg CO_2-eq per ton of waste treated across pathways. Landfill pathways for degradable materials are dominated by methane emissions, while aluminum recycling pathways exhibit substantial net benefits due to avoided primary aluminum production. Textile recycling pathways show narrower margins between gross burdens and avoided credits, resulting in greater sensitivity to recovery yield assumptions. Residential pathways exhibit somewhat higher transportation burdens due to decentralized collection behavior; however, transportation remains secondary to methane emissions and material substitution in determining comparative outcomes across pathways.

Although food waste does not exhibit the highest per-ton air emissions intensity among the four waste streams and waste management options analyzed, it represents approximately 19% of the 2.4 million metric tons per year managed statewide each year. When scaled to annual generation, food waste landfill emissions contribute approximately 491,000 t CO_2-eq per year, exceeding the aggregate contribution of other individual waste streams. This distinction between impact intensity (per ton) and aggregate contribution (total mass × intensity) is important under ISO 14044 interpretation, which requires consideration of both magnitude and decision relevance when identifying significant issues.

Overall, emissions impact outcomes at both the pathway and statewide levels are structurally governed by methane control performance and material recovery efficiency, with methane emissions representing the dominant driver of aggregate results.

5.1.2 Energy Use (Non-Renewable CED and Total CED)

Energy-related indicators are governed primarily by material production intensity, collection fuel use, and electricity generation assumptions.

Non-renewable CED is strongly influenced by avoided primary aluminum production. Because primary aluminum production is energy-intensive, recovery efficiency directly determines the magnitude of fossil energy savings in recycling pathways. Aluminum recycling therefore produces the largest net energy savings relative to disposal options.

For landfill and incineration pathways, non-renewable CED reflects a combination of:

- Collection and transportation fuel consumption
- Facility electricity use
- Avoided electricity credits (where applicable)

Total cumulative energy demand (Total CED) reflects overall primary energy use, including both fossil and non-fossil sources. Differences between non-renewable CED and total CED results are driven by the energy composition of displaced electricity and upstream production processes. Across pathways, net non-renewable energy results range from

approximately -97,200 to +1,700 MJ per ton, resulting in statewide energy impacts per waste stream between -906 (Aluminum) to +337 TJ (Food waste) per year[1].

Energy outcomes are therefore structurally influenced by aluminum recovery efficiency, electricity displacement modeling, and fuel use intensity.

5.1.3 Eutrophication

Eutrophication impacts arise primarily from upstream fuel combustion, transportation emissions, electricity generation, and landfill-related releases. Unlike air emissions impact and energy indicators, eutrophication is not dominated by a single mechanism but reflects the cumulative influence of multiple processes.

Per-ton eutrophication impacts range from approximately -3.7 to +0.1 kg N-eq/ton across modeled pathways. The highest per-ton intensities are associated with landfill pathways for residential textile, aluminum, and plastic film waste streams, reflecting upstream burdens and transportation-related emissions. Aluminum recycling pathways exhibit substantially lower eutrophication impacts, with net values of approximately -3.1 to -3.7 kg N-eq/ton, due to avoided primary aluminum production.

When scaled to statewide tonnage, aggregate eutrophication impacts range from approximately -33,531,400 for deposit aluminum containers to 7,600 kg N-eq for plastic films. Residential plastic film and residential textile treated by landfill each contribute nearly 5,000 kg N-eq per year, while residential food waste treated by landfill contributes approximately 3,200 kg N-eq per year due to its larger generation volume. The residential waste management pathways consistently have higher eutrophication impact, stemming from the impacts associated with transportation of waste by personal passenger vehicles. Interestingly, the statewide eutrophication impact of ICI food waste is nearly –4,600 kg N-eq per year, while Residential food waste is +4,100. This is the only ICI and residential waste pairing that has opposite signs. The underlying reason for this is the eutrophication credit given for electricity production which displaces grid electricity. Both Residential and ICI have this credit, but the Residential waste credit for avoided electricity production is overwhelmed on a per unit basis by the eutrophication burdens from transportation.

Overall, eutrophication outcomes are governed primarily by transportation intensity and upstream fuel use rather than methane emissions or avoided aluminum production. Landfill pathways generally exhibit higher burdens than recycling alternatives. However, differences across pathways are smaller in magnitude than those observed for air

[1] 1 terajoule (TJ) is 1 x 10^6 megajoules (MJ) or 1 x 10^{12} joules. The primary energy contained in a gallon of gas is around 120 MJ, therefore 1 TJ is equivalent to around 8,300 gallons of gasoline which could power about 15 averaged-sized passenger vehicles in the US for a year.

emissions impacts and, fossil energy indicators, and the overall eutrophication burden for these four waste streams is net negative due to credits for avoided products.

5.1.4 Cross-Category Interpretation

Across all impact categories, several consistent structural themes emerge:

- Recovery efficiency strongly influences air emissions and energy benefits in recycling pathways.
- Methane emissions dominate landfill air emissions impact performance.
- Electricity displacement assumptions influence energy-recovery pathways but do not independently determine comparative rankings.
- Transportation and behavioral parameters influence magnitude but are not primary ranking drivers.

Taken together, these findings indicate that statewide environmental outcomes are primarily governed by methane management and material recovery performance, with transportation structure and electricity modeling assumptions exerting secondary influence. The dominance of these mechanisms is consistent across Residential and ICI sectors, although differences in mass distribution and collection logistics influence relative magnitude.

Importantly, the relative importance of pathways differs when viewed on a per-ton basis versus when scaled to statewide tonnage. Some pathways exhibit high impact intensity per functional unit but contribute modestly at the system scale due to limited generation volume. Conversely, pathways with moderate per-ton impacts may dominate aggregate statewide results when associated with large material flows. Evaluating both perspectives is necessary to identify significant issues in accordance with ISO 14044, which requires consideration of both magnitude and decision relevance in interpretation.

5.1.5 Comparison of Per-Ton and Statewide Rankings

To further evaluate the relative significance of modeled waste management pathways, rank ordering was performed using both per-ton impact intensity (PT) and aggregate statewide impacts (SW) for each LCIA category. This comparison distinguishes between impact severity per functional unit and total system contribution driven by annual tonnage. The resulting rank orders for all 20 pathways across air emissions impacts, eutrophication, non-renewable CED, and total CED are presented in Table 26.

5.1.5.1 Air Emissions Impacts

Per-ton air emissions impact rankings identify residential and ICI plastic film incineration as the highest-intensity pathways (PT-Cl ranks 1 and 2). These results reflect direct combustion CO_2 emissions without substantial offsetting recovery credits. Landfill

disposal of food waste also ranks highly on a per-ton basis (PT-Cl ranks 3 and 4), driven primarily by methane generation from degradable organic material.

When scaled to statewide tonnage, the ranking structure shifts materially. ICI food waste landfill ranks first statewide (SW-Cl = 1), followed by residential food waste landfill (SW-Cl = 2). Although plastic film incineration exhibits the highest per-ton intensity, it ranks only 7th (residential) and 8th (ICI) statewide because its total mass is comparatively small. In contrast, the large generation volume of food waste drives its dominance in aggregate emissions.

This divergence demonstrates that significance of air emissions impacts at the system level is governed more by tonnage distribution than by intensity alone. Methane generation from food waste landfilling therefore represents the single largest leverage point for reducing statewide emissions within the modeled system.

5.1.5.2 Energy Demand (Non-Renewable and Total CED)

Per-ton non-renewable energy demand rankings are structurally different from air emissions impacts. Residential aluminum landfill ranks highest in per-ton non-renewable energy intensity (PT-ENR = 1), followed by residential plastic landfill (rank 2) and residential leather landfill (rank 3). These results reflect the lack of offsets from electricity production from captured landfill gas when compared to organic materials that produce higher amounts of methane.

Incineration pathways for plastics and textiles rank substantially lower on a per-ton energy basis (PT-ENR ranks 13–16), indicating partial offset through energy recovery. Food waste landfill does not rank highest in per-ton energy intensity (PT-ENR ranks 6 and 10), as its impacts are driven less by embodied production energy and more by biogenic degradation dynamics and offsets from electricity production.

At the statewide scale, tonnage effects again dominate. Residential food waste landfill ranks first for both non-renewable and total CED (SW-ENR = 1; SW-ETot = 1), and ICI food waste landfill ranks second in non-renewable CED. Plastic landfill pathways follow closely (SW-ENR ranks 3 and 4 for residential and ICI plastics). The transition from moderate per-ton ranking to leading statewide contributor reflects the large volume of food waste entering the landfill system.

Across both energy indicators, aluminum recycling pathways consistently rank among the lowest burden options at both per-ton and statewide scales (ranks 17–20). This stability confirms that avoided primary aluminum production delivers substantial and persistent energy savings, regardless of scaling.

5.1.5.3 Eutrophication

Eutrophication rankings exhibit a distinct structural pattern relative to air emissions impacts and energy demand. On a per-ton basis, residential textile landfill ranks highest (PT-Eu = 1), followed by residential aluminum landfill (rank 2) and residential plastic landfill (rank 3). These results reflect upstream production burdens and landfill-related emissions.

When scaled statewide, residential plastic landfill emerges as the largest contributor (SW-Eu = 1), followed by residential textile landfill (SW-Eu = 2) and ICI plastic landfill (SW-Eu = 3). Food waste landfill does not dominate this category; residential food landfill ranks 4th statewide, while ICI food landfill ranks 18th. This contrast indicates that eutrophication impacts are influenced more strongly by material composition and upstream processes than by degradable mass alone.

Unlike air emissions impacts, where methane drives dominance of food waste landfill, eutrophication outcomes are distributed more broadly across synthetic and textile materials.

5.1.5.4 Structural Interpretation

Comparison of per-ton and statewide rankings reveals three consistent structural findings. First, high per-ton intensity does not automatically translate to statewide significance. Plastic film incineration provides the clearest example: it ranks highest in per-ton emissions impact intensity yet contributes less at the system level due to limited tonnage.

Second, statewide air emissions impacts and energy outcomes are strongly governed by mass distribution. Food waste landfill does not consistently exhibit the highest per-ton burdens, yet it dominates aggregate emissions and energy demand results because of its large generation volume.

Third, material recovery pathways—particularly aluminum recycling—maintain favorable rankings across scales and impact categories. Both deposit and mixed aluminum recycling consistently rank among the lowest burden pathways at per-ton and statewide levels, demonstrating the robustness of avoided primary production benefits.

This dual-scale ranking approach strengthens interpretation under ISO 14044 by explicitly identifying significant issues at both the functional unit and system scales. It demonstrates that policy-relevant conclusions are shaped not only by pathway intensity but also by the structural distribution of waste volumes within the modeled system.

Table 26 - Rank ordering of waste management pathways, sorted by statewide air emissions impact rank. Abbreviations: SW = Statewide aggregate impact; PT = Per-ton impact intensity; LF = Landfill; INC = Incineration; REC = Recycling; Cl = Air emissions impacts; Eu = Eutrophication; ENR = Non-renewable CED; ETot = Total CED. The top 10 contributors in each indicator are in bold.

Sector	Waste	Treat	SW-Cl	PT-Cl	SW-Eu	PT-Eu	SW-ENR	PT-ENR	SW-ETot	PT-ETot
ICI	Food	LF	1	4	18	11	2	10	5	10
Res	Food	LF	2	3	4	9	1	6	1	7
Res	Textile	LF	3	7	2	1	5	4	4	4
ICI	Textile	LF	4	8	5	5	6	9	6	9
ICI	Plastic	LF	5	10	3	7	3	7	3	6
Res	Plastic	LF	6	9	1	3	4	2	2	2
ICI	Plastic	INC	7	2	15	16	16	16	16	16
Res	Plastic	INC	8	1	14	15	15	15	15	15
Res	Aluminum	LF	9	11	6	2	7	1	7	1
ICI	Aluminum	LF	10	13	7	6	8	5	8	5
Res	Textile	INC	11	5	13	13	14	13	14	13
ICI	Textile	INC	12	6	12	14	13	14	13	14
Res	Leather	LF	13	12	8	4	9	3	9	3
ICI	Leather	LF	14	14	9	8	10	8	10	8
ICI	Food	INC	15	16	11	12	11	12	12	12
Res	Food	INC	16	15	10	10	12	11	11	11
Res	Al Mixed	REC	17	18	16	18	17	18	17	18
ICI	Al Mixed	REC	18	20	17	20	18	19	18	19
Res	Al Deposit	REC	19	17	19	17	19	17	19	17
ICI	Al Deposit	REC	20	19	20	19	20	20	20	20

5.2 Identification of Key Uncertainty and Variability Sources

Uncertainty in this study arises from four principal domains:

1. Parameter uncertainty (e.g., methane emission factors, recovery rates, processing losses)
2. Behavioral variability (e.g., trip allocation, participation rates, self-haul distances)
3. Structural modeling assumptions (e.g., electricity displacement approach)
4. LCIA method selection

Parameters were selected for sensitivity analysis based on one or more of the following criteria:

- High contribution to baseline impacts
- High empirical uncertainty or variability
- Decision relevance
- Structural modeling importance

Parameters were included if they satisfied one or more of these criteria (Table 27).

Table 27 - Criteria Used to Select Parameters for Sensitivity Analysis

Parameter / Assumption	High Contribution	High Uncertainty / Variability	Decision-Relevant	Included in Sensitivity	Baseline	Lower bound	Upper bound
Fugitive landfill methane	✓	✓	✓	✓	0.036 kg/kg waste	0.75x baseline	1.3x baseline
Drop-off distance	—	✓	—	✓	MSW: 46 km Mixed Al: 27 km Deposit: 17 km	MSW: 42 km Mixed Al: 22 km Deposit: 14 km	MSW: 49 km Mixed Al: 33 km Deposit: 21 km
Aluminum recovery (deposit)	✓	✓	✓	✓	41.8%	33%	50%
Aluminum recovery (mixed)	✓	✓	✓	✓	10.8%	9%	13%
Aluminum processing losses	—	✓	✓	✓	8%	6.6%	9.8%
Residential trip allocation	—	✓	—	✓	50%	37.5%	62.5%

Parameter / Assumption	High Contribution	High Uncertainty / Variability	Decision-Relevant	Included in Sensitivity	Baseline	Lower bound	Upper bound
Curbside availability	—	✓	✓	✓	MSW: 67% Recycl: 45%	MSW: 55% Recycl: 32%	MSW: 78% Recycl: 64%
Grid vs. Marginal electricity producer	Structural	—	✓	✓	NA	NA	NA
LCIA method (TRACI vs EF)	Structural	—	✓	✓	NA	NA	NA
Textile recycling substitution factor	—	✓	✓	✓		Baseline: 100% Alt.: 40-50%	

In addition to the specific parameters evaluated quantitatively, broader sources of uncertainty are inherent to LCA modeling. These include limitations in the representativeness of secondary datasets, temporal and geographic variability in waste management practices, simplifications required for system modeling, and uncertainty embedded within characterization models and background life cycle inventories. Many background datasets incorporated from the Ecoinvent database contain their own underlying uncertainties associated with industrial process data, allocation approaches, and regional assumptions. While these uncertainties cannot be fully eliminated, the study was designed to evaluate whether they materially influence the comparative interpretation of management pathways and statewide impact conclusions.

5.3 Sensitivity Analysis Framework

Sensitivity analyses were conducted using a one-at-a-time deterministic framework implemented through global parameters and parameter sets in OpenLCA. This approach isolates the influence of individual assumptions while maintaining full system consistency.

Changes in net impact magnitude, pathway ranking, or sign were evaluated to determine whether conclusions were sensitive to tested assumptions. For consistency in interpretation, variations exceeding ±20% in net impact were treated as potentially material, particularly where such changes affected comparative conclusions. This threshold does not represent an ISO-prescribed standard, but rather a study-specific decision criterion established to support systematic evaluation of robustness in accordance with ISO 14044 §4.5.

The purpose of the sensitivity analysis was not to establish precise predictive bounds for all modeled outcomes, but rather to evaluate the robustness and directional stability of

the study conclusions under plausible alternative assumptions. In accordance with ISO 14044 interpretation requirements, emphasis was placed on identifying whether reasonable changes in influential parameters altered comparative findings, pathway rankings, or the overall interpretation of dominant impact drivers.

Results indicated that some categories, particularly climate change impacts associated with landfill methane emissions and energy-related indicators associated with transportation allocation assumptions, exhibited meaningful sensitivity to selected parameters. However, the overall directional conclusions of the study remained generally robust across evaluated scenarios. In particular, landfilling remained the dominant contributor to statewide climate impacts, while material recovery pathways—especially aluminum recycling—continued to provide substantial avoided burden benefits under all tested conditions. The sensitivity analysis therefore supports the conclusion that the principal findings of the study are not dependent on a single modeling assumption or isolated parameter choice.

5.4 Sensitivity Analysis Results

5.4.1 Fugitive Methane Emissions

Fugitive methane emissions from landfill processes were confirmed as the most influential parameter affecting air emissions impacts for degradable waste streams.

Baseline methane emissions from the landfilling of food waste were modeled as 0.036 kg CH_4 per kg of food waste, with sensitivity bounds defined as 0.027 and 0.037 kg CH_4 per kg of food waste. These bounds reflect calculated high and low methane emission scenarios based on alternative landfill gas capture and oxidation assumptions (see Section 3.3.4.1). Methane emissions from other landfill-treated waste streams were adjusted proportionally to maintain internal consistency, 0.75x the baseline flow for low and 1.3x for high.

At the pathway level, methane variation produced substantial changes in air emissions impacts for landfill pathways. Under the low-methane scenario, net impacts decreased by approximately 9-22% depending on the waste stream, corresponding to absolute reductions of up to 230 kg CO_2-eq per ton of waste treated by landfill. Under the high-methane scenario, these impacts increased by approximately 11–28%, corresponding to increases of up to 287 kg CO_2-eq per ton.

In this scenario we did not assume that the reduced fugitive methane emissions would necessarily translate into increased electricity production, therefore the model produced negligible changes in non-renewable cumulative energy demand and Total CED. Eutrophication results were likewise unaffected, reflecting the limited role of methane in eutrophication characterization under TRACI 2.1.

When evaluated at the statewide level, methane sensitivity produced the largest observed variation in air emissions impact results of any tested parameter, increasing statewide impacts by 31% under the high case and decreasing them by 25% under the low case. No measurable statewide changes were observed in energy or eutrophication indicators. No changes in rank order were observed in the per-ton impact factors or statewide impacts per waste stream.

These findings confirm that landfill methane management is the dominant determinant of statewide air emissions impact performance, while exerting minimal influence on energy demand or eutrophication outcomes. Accordingly, methane emissions represent the primary emissions-relevant uncertainty and the most consequential decision-sensitive parameter for degradable landfill pathways within the modeled system.

Fugitive methane emissions from landfill processes for the treatment of food waste were confirmed as the most influential parameter affecting air emissions impacts for degradable waste streams.

5.4.2 Aluminum Recovery Rates

Aluminum recovery efficiency directly determines the quantity of secondary aluminum produced and therefore the magnitude of avoided primary aluminum production credited to recycling pathways. Because primary aluminum production is highly energy- and emissions-intensive, recovery rates represent a high-leverage parameter affecting air emissions impacts, fossil energy demand, and eutrophication outcomes. Secondary effects also arise from diversion of aluminum from landfill and associated avoided disposal impacts, though these are comparatively minor. Baseline recovery assumptions and sensitivity bounds are summarized in Table 5.

Sensitivity analysis varied both deposit and mixed recovery simultaneously by ±20% relative to baseline. These bounds reflect plausible variation associated with program performance, participation rates, sorting effectiveness, and uncertainty in calculated recovery rates, rather than structural modeling extremes.

At the pathway level, increasing recovery proportionally increased avoided primary aluminum production credits and strengthened net environmental benefits across air emissions impacts and energy indicators. Decreasing recovery reduced avoided burdens and increased net impacts. Recycling pathways remained environmentally preferable to disposal options across all tested bounds.

When scaled to the statewide level, aluminum recovery emerged as a dominant driver of fossil energy outcomes and a significant contributor to eutrophication variation, while exerting comparatively modest influence on air emissions impacts (Table 28). No changes in rank order were observed in statewide impacts per waste stream.

Table 28 - Statewide Impact Variation Under Aluminum Recovery Sensitivity

Impact Category	High Recovery	Low Recovery
Air emissions impacts	−4%	+4%
Eutrophication (no LT)	−37%	+38%
Non-renewable CED	−84%	+87%
Total CED	−379%	+40%

These results demonstrate that:

- Air emissions impacts outcomes remain primarily governed by methane emissions rather than aluminum recovery (±4% statewide variation).
- Non-renewable CED is highly sensitive to recovery efficiency due to the fossil energy intensity of primary aluminum production.
- Total CED exhibits substantial but moderated sensitivity relative to non-renewable CED.
- Eutrophication is materially influenced by avoided upstream aluminum production burdens.

Because recovery was varied simultaneously across deposit and mixed systems, the results represent a true system-wide scaling of secondary aluminum production rather than isolated pathway behavior.

Aluminum recovery sensitivity reflects realistic policy-relevant variation in system performance. Accordingly, aluminum recovery represents one of the most influential practical statewide levers for fossil energy demand reduction and a significant driver of eutrophication outcomes, while remaining secondary to methane emissions for air emissions impacts.

5.4.3 Aluminum Processing Losses

Baseline aluminum processing losses were modeled at 8% during sorting and pre-processing and 0.9% during remelting. Sensitivity bounds for sorting and processing losses were defined as 6.6% (low case) and 9.8% (high case). These bounds reflect the observed variance in reported residual rates from metals recyclers and material recovery facilities (*data from 2024 IA Recycling Facility Study by SCS Engineers*). Additionally the low case reflects a sensitivity check for the proportion of the mass contaminants rejected alongside the rejected aluminum. Remelting losses were held constant to isolate the influence of upstream material preparation efficiency.

At the pathway level, variation in processing losses altered net air emissions impacts per ton of aluminum waste treated by recycling by approximately −1.8% to +2.3%. Non-renewable CED exhibited a similar proportional response, reflecting changes in avoided

primary aluminum production associated with marginal differences in secondary yield. Total CED variation was of comparable magnitude, as reduced secondary output proportionally decreased avoided upstream primary energy burdens. Eutrophication impacts shifted by approximately −1.8% to +2.5%, driven by scaled changes in upstream mining, refining, and electricity-related emissions rather than by changes in direct recycling emissions.

Statewide impacts under the aluminum processing loss sensitivity are summarized in Table 29, which shows modest variation across energy and eutrophication indicators and no measurable change in air emissions impact outcomes under either the high- or low-loss scenarios. No changes in rank order were observed in the per-ton impact factors or statewide impacts per waste stream.

These results confirm that aluminum processing losses exert measurable but secondary influence on statewide fossil energy and eutrophication outcomes, and negligible influence on air emissions impact performance. Importantly, because the sensitivity bounds reflect documented variability in recycler residual rates rather than arbitrary perturbation, the observed magnitude of impact variation (±6–8% for non-renewable CED and ±3–4% for Total CED) represents a realistic range of system performance variability.

Table 29 - Statewide Impact Variation Under Aluminum Processing Loss Sensitivity

Impact Category	High-Loss Scenario	Low-Loss Scenario
Air emissions impacts	+0.4%	-0.3%
Eutrophication	+4%	−3%
Non-renewable CED	+9%	−7%
Total CED	+4%	−3%

Compared with aluminum recovery sensitivity (±75–77% non-renewable CED; ±37–38% Total CED), processing losses represent a materially smaller source of statewide variability. Accordingly, while accurate characterization of recycler residuals improves model precision, recovery rate assumptions remain the dominant aluminum-system parameter influencing statewide energy outcomes.

5.4.4 Textile Recycling Substitution Factor

In order to estimate a more realistic potential environmental benefit of end of life (EoL) recycling, we also perform a Sensitivity Analysis whereby we adopt a “quality degradation factor” to account for the reduced quality of secondary (recycled) materials *vs*. their primary (virgin) counterparts. Doing so is deemed particularly relevant in the case of waste textiles, whose technical quality is often found to degrade more significantly during recycling.

From a methodological point, this Sensitivity Analysis is implemented via the "Resource Use and Emissions Profile (RUaEP) per unit of analysis" formula (originally introduced in Annex V of the Product Environmental Footprint (PEF) Guide[1] and later modified in the 2021 revision of the same guide[2]).

This formula, also commonly referred to as the "Circular Footprint Formula" (CFF) was devised to provide a generalized, flexible and fully parametrized modelling approach to EoL recycling, within an internally-consistent whole-life-cycle framework that also includes different options for modelling the material inputs to the system under study, as well as other EoL waste management options.

Specifically, as illustrated in the box below, by setting the A and B parameters to 0 in the CFF, the modelling approach simplifies back to the "avoided burden" approach, which provides methodological consistency with our default modelling assumption, with the only difference that it retains an additional parameter (or more precisely, a ratio of two parameters: Q_S/Q_P) to account for material quality degradation due to recycling.

Conversely in all cases, the use of the average grid mix to model displaced electricity is selected as the default option, in order to align with the study's intended Attributional approach, whereby the assumption is made that the average composition of the electricity grid mix into which the waste management system is embedded will not be significantly affected by the (comparatively minor) additional input of electricity from landfill energy recovery or waste incineration.

Displacement of marginal electricity generated by combined cycle gas turbines is assessed as an alternative option.

Box 1. Circular Footprint Formula as per PEF guidelines (when A = 0 and B = 0)



$$E_V + R_2 \times (E_{rec,EoL} - E^*_V \times Q_S/Q_P) + R_3 \times (E_{ER} - LHV \times X_{ER} \times E_{SE}) + (1 - R_2 - R_3) \times E_D$$

Where:

E_V = specific emissions and resources consumed for virgin material production in input to product system generating the waste (this is OUTSIDE of our system boundary)

R_2 = share of input material that is recycled at EoL

$E_{rec,EoL}$ = specific emissions and resources consumed for material recycling at EoL, including collection and sorting

$\mathbf{E^*_V}$ = specific emissions and resources consumed for production of alternative virgin material that is assumed to be displaced (if the same material is displaced, then: $E^*_V = E_V$)

$\mathbf{Q_S/Q_P}$ = dimensionless ratio representing the loss in "quality" of the recycled material at EoL vs. its virgin counterpart

$\mathbf{R_3}$ = share of input material that is incinerated for energy recovery

$\mathbf{E_{ER}}$ = specific emissions and resources consumed for energy recovery

LHV = lower heating value [J/kg] of material

$\mathbf{X_{ER}}$ = efficiency of energy recovery. Calculated separately for heat and electricity.

$\mathbf{E_{SE}}$ = specific emissions and resources consumed for alternative energy source that is assumed to be displaced. Estimated separately for heat and electricity.

$\mathbf{E_D}$ = specific emissions and resources consumed for final disposal (e.g., landfilling)

5.4.5 Transportation and Behavioral Parameters

Drop-off distance uncertainty was derived via bootstrap resampling of GIS-derived population-weighted distances (10,000 iterations) for MSW, mixed aluminum recycling, and aluminum deposit redemption. The 2.5th and 97.5th percentiles define empirical bounds of 42 to 49 km for MSW, 22 to 33 km for mixed aluminum, and 14 to 21 km for deposit container redemption, respectively.

Residential trip allocation sensitivity evaluated alternative allocation fractions of 37.5% and 0.625% relative to the baseline 50%.

Curbside participation rates were varied from 55% to 78% for MSW, and 32% to 64% for mixed recycling.

Across these parameters:

- Net air emissions impacts shifted by less than approximately 1% in most pathways.
- Non-renewable CED and Total CED shifted by less than 4% in all scenarios except:
    - Curbside pickup % high and low, in which total annual CED for the 4 waste streams changed by -10% to +11% respectively; and
    - Trip allocation high and low, in which total CED changed by an average of ±14%
- Eutrophication impacts varied by less than 5% in all scenarios, except:
    - Curbside pickup % high and low, in which total annual eutrophication for the 4 waste streams changed by -20% to +21% respectively; and
    - Trip allocation high and low, in which eutrophication changed by an average of -±13%

These findings indicate that transportation and participation assumptions influence absolute magnitude but do not materially alter comparative conclusions across air emissions impacts, energy, or eutrophication indicators.

Additionally, removing transportation burden of non-burnable waste in incineration pathways impacts environmental emissions <0.1% for CED, air emissions and eutrophication at the statewide level.

5.4.6 Electricity Displacement Assumptions and Associated Credits, Average Grid Mix vs. Marginal Generators

Baseline avoided electricity credits were modeled using the US-MRO average grid mix (EcoInvent 3.11), consistent with the attributional structure of the study. A dedicated sensitivity analysis was then also performed, substituting marginal natural gas combined cycle (NGCC) electricity generation, to evaluate the influence of electricity displacement assumptions on avoided burdens.

At the unit electricity level, NGCC electricity is characterized by 15% lower greenhouse gas emissions per kWh, vs. the average US-MRO grid. This is because the latter includes coal and other higher-emission sources (Table 30). As a consequence, the emissions substitution credit is lower, when assuming that NGCC electricity is substituted, instead of the grid mix.

Conversely, the non-renewable cumulative energy demand credit increases by approximately 23% under the NGCC assumption. Because NGCC relies almost entirely on fossil fuel inputs, avoided electricity credits reflect a greater displacement of fossil primary energy compared to the mixed grid, which includes non-fossil generation. Despite this shift in fossil composition, the total cumulative energy demand (CED) credit changes by only 4%, indicating that overall primary energy requirements per kWh remain broadly similar (Table 30).

The Eutrophication credit decreases substantially (≈95%), as electricity from natural gas is associated with much lower eutrophication impacts.

To summarize, under the NGCC displacement assumption:

- Emissions credits per kWh decrease (because NGCC is less carbon-intensive than the average grid), reducing the emissions benefit attributed to electricity recovery.
- Non-renewable energy credits increase (because NGCC displaces fossil primary energy more directly than a mixed grid), increasing the energy benefit assigned to electricity recovery.
- Total primary energy credits change only modestly, indicating that structural conclusions about overall energy recovery are relatively stable.
- Eutrophication credits decline sharply.

Table 30 - Unit level comparison (1 kWh replaced) between the baseline US MRO Market and the marginal producer in the MRO, a natural gas powered, combined cycle electricity plant

Impact Category	LCIA Method	US MRO Market (Baseline)	US MRO NGCC (Sensitivity)	Absolute Difference (NGCC – Market)	Percent Difference
Emissions impacts (kg CO_2-eq)	TRACI v2.1	0.5407	0.4607	-0.0799	-15%
Energy resources: non-renewable (MJ-eq)	CED	6.9939	8.5925	+1.5986	+23%
Eutrophication no LT (kg N-eq)	TRACI v2.1 no LT	0.0007	0.0000	-0.0007	-95%
Total CED (MJ-eq)	CED	8.9778	8.6111	-0.3667	-4%

5.4.6.1 Per-Ton Waste System Effects

When propagated through the waste model, sensitivity to displacement assumptions is concentrated in electricity-generating pathways:

- Air emissions impacts changed minimally across all waste streams, approximately 2%, reflecting reduced avoided CO_2 under NGCC displacement.
- Non-renewable energy credits for plastic film and textile incineration increase by approximately 1,370–2,210 MJ per metric ton (≈25%), consistent with the higher fossil intensity of NGCC.
- Changes in Total CED across all waste streams, however, is minimal, approximately 0.3%.
- Although eutrophication credits are reduced under the NGCC scenario, total eutrophication impacts were minimal in absolute terms

In practical terms, electricity displacement assumptions primarily alter the magnitude of avoided burdens rather than their direction. Incineration pathways experience measurable shifts in emissions and fossil energy credits, but the changes are insufficient to reverse pathway rankings or convert net benefits to net burdens (or vice versa).

Recycling and landfill pathways remain comparatively stable because they are less directly influenced by electricity displacement assumptions.

5.4.6.2 Interpretation and Methodological Framing

This sensitivity addresses a goal modeling choice rather than parameter uncertainty. The baseline study adopts an attributional perspective and therefore appropriately applies average grid displacement consistent with system accounting conventions. Testing

marginal NGCC displacement provides a bounded exploration of consequential interpretation without changing the scope of the assessment.

Across all indicators evaluated, comparative conclusions remain robust under both electricity displacement assumptions. While credit magnitudes shift—particularly for non-renewable energy demand—the relative ordering of pathways and policy-relevant conclusions are unchanged.

This result indicates that the study's findings are not dependent on a single electricity modeling perspective. Instead, conclusions are stable across both average and marginal displacement assumptions, strengthening interpretive confidence under ISO 14044.

5.4.7 Statewide Sensitivity Patterns

Evaluation of sensitivity results at the statewide level highlights distinct structural drivers across impact categories. In this section, "statewide % change" refers to the percent change in the statewide total net impact across all modeled waste streams and pathways relative to the baseline statewide total for the same impact category. Unless otherwise noted, results reported as "0%" indicate changes smaller than 1% rounding threshold and are therefore treated as negligible at the statewide scale.

5.4.7.1 Methane Sensitivity (Dominant Emissions Impact Driver)

Variation in landfill methane emissions produces the largest statewide change in air emissions impacts. Under the high-methane case, statewide impacts increase by approximately 19%, while under the low-methane case they decrease by approximately 22%. Changes in non-renewable CED, Total CED, and eutrophication indicators under methane variation are negligible, consistent with the limited role of methane emissions in primary energy accounting and eutrophication characterization for the selected LCIA method.

These results confirm that methane management assumptions are the principal determinant of statewide outcomes for degradable waste streams.

5.4.7.2 Transportation and Behavioral Attribution Sensitivity

Residential collection structure parameters—particularly allocation of drop-off vehicle travel—exert a measurable influence on statewide energy and eutrophication indicators. The baseline model allocates 50% of residential drop-off vehicle travel to waste management activities, reflecting the assumption that many drop-off trips are combined with other household errands (US EPA MSW-DST). Consistent with the attributional modeling framework applied in this study, only the fraction of shared vehicle travel attributable to waste management is assigned to the waste system.

To evaluate sensitivity to this attribution decision, allocation was varied between 37.5% and 62.5%, representing reasonable variation in the share of drop-off trips undertaken primarily for waste management versus multipurpose travel. Vehicle travel associated with drop-off is modeled exclusively within residential drop-off unit processes and is not represented elsewhere in the system, thereby avoiding double counting.

Within this allocation range, statewide non-renewable CED varies by approximately ±32%, Total CED by approximately ±11%, and eutrophication by approximately ±19%. The effect is concentrated in residential drop-off pathways, where passenger-vehicle travel burdens are large relative to the mass of waste transported. In contrast, statewide air emissions impacts vary by approximately ±1%, indicating that transportation attribution has a comparatively smaller influence on outcomes than on energy and nutrient indicators. Methane emissions and avoided production effects remain the dominant drivers of air emissions impact results across the tested range.

Under the higher allocation assumption (62.5%), statewide non-renewable energy (CED-ENR) ranks show only minor reordering relative to the 50% baseline. Changes are concentrated within residential landfill and incineration pathways and are generally limited to one-position shifts. For example, Residential Plastic Films – Landfill shifts from rank 4 under baseline to rank 2 under the higher allocation case, while Residential Food Waste – Incineration and Residential Textile Waste – Incineration each move by one rank position. Most ICI pathways retain their baseline positions, and dominant contributors such as Residential Food Waste – Landfill (rank 1) remain unchanged. Aluminum recycling pathways continue to occupy the lowest-burden ranks. No reversals occur between landfill and recycling options, and no pathway changes sign.

Overall, while residential drop-off travel allocation materially influences the magnitude of statewide non-renewable energy and eutrophication results, it does not alter pathway rankings in a substantive way or change the broader structural conclusions of the study.

5.4.7.3 Curbside Participation Sensitivity

Variation in curbside participation rates produces moderate statewide sensitivity in energy and eutrophication indicators (approximately ±28–30% for non-renewable energy and ±19–21% for eutrophication), with comparatively small influence on air emissions impacts (approximately ±1%). These differences reflect shifts in collection logistics and transportation fuel use associated with curbside collection versus self-haul behavior.

The limited air emissions impact sensitivity indicates that outcomes are dominated by downstream treatment and avoided production effects rather than collection mode. In contrast, energy and eutrophication impacts are more responsive to transportation intensity and vehicle efficiency, particularly where passenger vehicle trips substitute for consolidated curbside collection.

Because these sensitivities are driven by assumptions regarding household behavior and travel distances, improved empirical data would materially strengthen future modeling. The state could refine estimates through:

- Surveys of household recycling participation and drop-off frequency
- Data collection on average trip distances and vehicle types used for self-haul
- Program-level participation tracking across municipalities
- Waste composition audits paired with behavioral surveys

Collecting such data would allow refinement of transportation emission factors and better alignment between modeled assumptions and observed behavior. This would improve confidence in energy and eutrophication results while supporting more targeted program optimization.

These findings have practical implications for program design. Policies that increase participation in optimized curbside systems—while minimizing incremental vehicle miles traveled—can reduce energy and nutrient-related burdens without materially affecting air emissions impact outcomes. Conversely, increased self-haul behavior may erode energy and eutrophication benefits even if total diversion remains constant.

5.4.7.4 Aluminum Residual and Recovery Sensitivity

Aluminum processing residual assumptions produce comparatively modest statewide variation (generally <±8% across energy indicators and negligible change in air emissions impacts), confirming that statewide outcomes are more strongly influenced by recovery performance than by moderate shifts in processing loss fractions.

5.4.7.5 Summary Interpretation

At the statewide scale, conclusions about the impact of air emissions are primarily governed by methane management assumptions, while energy and eutrophication outcomes are more dependent on residential collection structure and travel attribution decisions. These findings support robust air emissions impact-oriented conclusions and indicate that energy and eutrophication indicators for residential drop-off systems should be interpreted with explicit recognition of the underlying behavioral allocation assumptions.

5.5 LCIA Method Sensitivity

Comparison of LCIA methods indicates strong consistency for air emissions impacts and non-renewable energy demand, but notable divergence for eutrophication (Table 31). Air emissions impacts rankings were identical across methods, reflecting their shared reliance on IPCC global warming potentials. Dominant contributors are driven by large CO_2 and CH_4 emissions, and minor characterization differences do not alter pathway ordering.

This consistency indicates that conclusions for this indicator are structurally determined by inventory magnitudes rather than method selection.

Non-renewable cumulative energy demand rankings were likewise unchanged, as both comparisons apply the same cumulative energy demand framework. Energy demand conclusions are therefore method-invariant within the scope of this study.

Eutrophication rankings, however, differ substantially between TRACI and EF 3.1. This divergence reflects differences in indicator structure rather than inventory inconsistencies. TRACI reports eutrophication as a single nitrogen-equivalent midpoint indicator, whereas EF 3.1 separates freshwater (kg P-eq), marine (kg N-eq), and terrestrial (mol N-eq) components, each based on distinct nutrient limitation assumptions and fate-exposure modeling.

Because freshwater systems are typically phosphorus-limited and marine systems nitrogen-limited, pathways characterized by differing nitrogen and phosphorus emission profiles shift in relative importance between methods. Landfill leachate and combustion-related emissions therefore influence rankings differently depending on the nutrient framework applied.

The stability of air emissions impact and energy rankings alongside divergence in eutrophication results indicate that air emissions and energy conclusions are robust across LCIA methods, while nutrient-related impacts are more sensitive to methodological framing. These differences reflect alternative scientific representations of ecological response rather than uncertainty in modeled emissions.

Table 31 - Rank Ordering of Waste Management Pathways Across LCIA Methods (TRACI 2.1 and EF 3.1)

Pathway	EF-Cl	TRACI-Cl	EF-ENR	CED-ENR	EF-EuM	EF-EuF	TRACI-Eu
Res Plast INC	1	1	15	15	15	15	15
ICI Plast INC	2	2	16	16	16	16	16
Res Food LF	3	3	6	6	5	1	9
ICI Food LF	4	4	10	10	6	4	11
Res Text INC	5	5	13	13	3	13	13
ICI Text INC	6	6	14	14	4	14	14
Res Text LF	7	7	4	4	1	9	1
ICI Text LF	8	8	9	9	2	10	5
Res Plast LF	9	9	2	2	7	5	3
ICI Plast LF	10	10	7	7	8	8	7
Res Al LF	11	11	1	1	11	2	2

Pathway	EF-Cl	TRACI-Cl	EF-ENR	CED-ENR	EF-EuM	EF-EuF	TRACI-Eu
Res Leath LF	12	12	3	3	13	3	4
ICI Al LF	13	13	5	5	12	6	6
ICI Leath LF	14	14	8	8	14	7	8
Res Food INC	15	15	11	11	9	11	10
ICI Food INC	16	16	12	12	10	12	12
Res Al Dep REC	17	17	17	17	17	17	17
Res Al Mix REC	18	18	18	18	18	18	18
ICI Al Dep REC	19	19	20	20	20	19	19
ICI Al Mix REC	20	20	19	19	19	20	20

Abbreviations: LF = Landfill; INC = Incineration; REC = Recycling; Al = Aluminum; Plast = Plastic Films; Text = Textile; Leath = Leather; EF = Environmental Footprint 3.1; TRACI = TRACI 2.1; ENR = Non-renewable Energy Resources; CED = Cumulative Energy Demand; EuM = Marine Eutrophication; EuF = Freshwater Eutrophication; no LT = excluding long-term emissions.

Overall, primary system-level conclusions remain stable under both LCIA methods, with eutrophication results requiring compartment-specific interpretation.

5.6 Alternative Waste Management Scenarios

At the request of the client, three alternative waste management scenarios were evaluated relative to the baseline statewide configuration described in Section 3:

1. Increased aluminum deposit container recycling
2. Centralized residential textile recycling (15% diversion from landfill)
3. A 10% reduction in fugitive methane emissions from landfill processes

The functional unit (1 metric ton of waste managed) remains unchanged. Scenario impacts were calculated by adjusting statewide tonnage allocations and applying the per-ton impact factors developed in the baseline model, as well as from the new textile recycling LCIA results. Results are presented as net differences relative to baseline statewide totals. No economic costs or behavioral rebound effects were modeled.

5.6.1 Scenario 1 – Increased Aluminum Deposit Container Recycling

This scenario reallocates 4,700 metric tons (5,180 short tons) of aluminum from landfill to deposit container recycling. Mass balance consistency was maintained by reducing landfill tonnage by an equivalent amount and increasing recycling output accordingly.

From an air emissions impact perspective, this reallocation results in a statewide reduction of approximately 42,400 t CO_2-eq per year, representing a reduction of baseline

aluminum-related impacts by 9%. The reduction is driven almost entirely by avoided primary aluminum production, which is highly emissions-intensive. The avoided primary aluminum production credit constitutes over 99% of the total scenario benefit. A small benefit arises from reduced landfill burdens associated with diverting aluminum from disposal, while transportation and processing contributions remain comparatively minor.

Energy indicators show parallel improvements. Non-renewable cumulative energy demand decreases by approximately 443 TJ/year, reflecting the substantial fossil energy intensity of primary aluminum production. Total cumulative energy demand also declines by approximately 639 TJ, indicating reductions in both fossil and non-fossil primary energy use.

Eutrophication impacts decrease by approximately 16,400 kg N-eq/year, reflecting avoided upstream emissions associated with primary aluminum production. No offsetting increases were observed in other processes.

Across all impact categories, this scenario produces consistent directional improvements with no evidence of burden shifting. The magnitude of benefit scales proportionally with additional recovery volume. These findings reinforce the structural importance of aluminum recovery efficiency identified in baseline and sensitivity analyses.

5.6.2 Scenario 2 – Centralized Residential Textile Recycling (15% Diversion)

Under this scenario, 15% of residential textile waste is diverted from landfill to a centralized collection and mechanical recycling system. Collection centers are assumed to be located at existing landfill and transfer station sites, with residential drop-off transport averaging 23 km (one-way). Materials are transported from these collection centers to a theoretical central processing location in Des Moines, from where recovered cotton and polyester fractions are transported an average of 1,835 km to out-of-state mechanical recyclers. Recycling yields result in 0.36 metric tons of cotton fiber and 0.13 metric tons of PET granulate displaced per ton of residential textile waste processed, after accounting for 20% processing losses.

In the baseline case, recycled fiber is assumed to displace virgin production at a 1:1 ratio (100% substitution), consistent with a conventional attributional avoided burden framework.

5.6.2.1 Baseline Results (100% Substitution)

At the per-ton level, textile recycling yields :

- Air emissions impacts: -718 kg CO_2-eq
- Non-renewable CED: -6,150 MJ-eq
- Eutrophication (no LT): -65.7 kg N-eq (due to large avoided fertilizer use for growing cotton)

- Total CED: -19,410 MJ-eq (includes the gross caloric value of the biomass)

At the statewide level, diversion of 15% of residential textiles and recycling them into secondary fibers yields a substantial reduction in Eutrophication impact statewide. In the baseline scenario, the waste management of food waste, plastic films, mixed aluminum waste, and textiles and leather produces approximately a net credit of -21,000 kg N-eq per year. With textile recycling, this credit is 29x greater at -604,850 kg N-eq/year. This is primarily due to the large eutrophication impacts of growing cotton fiber (over 180 kg N-eq per ton of cotton), and the credit taken for displacing those virgin cotton fibers.

Additionally, there is a net reduction of approximately 9,700 t CO_2-eq/year. Benefits arise from two primary mechanisms:

1. Avoided landfill methane emissions; Landfill emissions decreased by 3,300 t CO-eq/year
2. Avoided virgin cotton and PET production; Net benefit of recycling is -6,400 t CO-eq/year

Transport and processing burdens partially offset avoided production credits but do not negate them under baseline assumptions. Contribution analysis indicates that collection and interfacility transport represent a substantial share of gross burdens. Passenger vehicle transport (~0.36 kg CO_2-eq), medium-voltage electricity (~0.24 kg CO_2-eq), and long-distance freight transport (~0.12 kg CO_2-eq) are among the largest contributors to gross impacts.

Non-renewable cumulative energy demand decreases by approximately 67 PJ/year, driven primarily by avoided energy-intensive virgin fiber production, though moderated by long-distance freight transport.

Under these assumptions, the scenario produces major eutrophication benefits combined with moderate air emissions impact and energy benefits without cross-category trade-offs.

5.6.2.2 Substitution Sensitivity (40% Displacement)

Because mechanically recycled fibers are shorter and may require blending with virgin materials, full substitution likely represents an upper-bound assumption. A sensitivity analysis was therefore conducted using a 40% substitution factor. From a methodological perspective, this was implemented by setting the “quality” ratio (Qs/Qp) in the CFF formula to 0.4 (see Section 2.8.3). All other modelling parameters remained unchanged vs. the baseline scenario.

Under this scenario, avoided virgin production credits decrease proportionally by 60%, while gross collection, transport, and processing burdens remain unchanged.

At the per-ton level, results shift materially:

- Air emissions impacts: +0.466 kg CO_2-eq
- Non-renewable CED: +6.76 MJ-eq
- Eutrophication (no LT): –0.0259 kg N-eq
- Total CED: +1.84 MJ-eq

Under such a reduced substitution assumption, avoided cotton and PET production is no longer sufficient to offset collection and transport emissions. Emissions and energy results therefore reverse sign, while eutrophication remains strongly beneficial, though reduced in magnitude by around 60%.

This indicates that scenario performance is highly sensitive to the assumed substitution rate. Unlike aluminum recycling, where avoided burdens greatly exceed processing impacts, textile recycling exhibits narrower margins between gross burdens and avoided credits.

5.6.2.3 Processing loss sensitivity

A sensitivity analysis was conducted to evaluate the influence of textile processing losses on the environmental performance of textile recycling. The baseline model assumes that 20% of collected textile material is rejected during sorting and processing, with the remaining 80% converted into marketable recycled textile products. Additional scenarios increased processing losses from 30% to 90% to evaluate the effect of declining material recovery efficiency.

Textile recycling remained environmentally beneficial across all impact categories under the baseline assumption. At 20% processing losses, net impacts were -0.56 kg CO_2-eq, -4.44 MJ non-renewable energy demand, -16.19 MJ total cumulative energy demand, and -0.058 kg N-eq eutrophication per kilogram of collected textile material. As processing losses increased, these benefits declined because progressively less virgin textile production was displaced by recycled material.

Climate change and non-renewable energy demand were the most sensitive impact categories (Figure 18). Net climate impacts increased from -0.56 kg CO_2-eq/kg at the baseline to +0.53 kg CO_2-eq/kg at the 90% loss scenario. Similarly, non-renewable energy demand increased from -4.44 MJ/kg to +5.24 MJ/kg. For both categories, the transition from a net environmental benefit to a net environmental burden occurred between the 50% and 60% loss scenarios. At 50% losses, textile recycling remained marginally beneficial for climate change (-0.10 kg CO_2-eq/kg) and non-renewable energy demand (-0.29 MJ/kg), whereas at 60% losses both indicators became net burdens (+0.06 kg CO_2-eq/kg and +1.09 MJ/kg, respectively).

Total cumulative energy demand was less sensitive, remaining net beneficial through the 70% loss scenario and becoming positive only between the 70% and 80% loss scenarios (Figure 18). Eutrophication impacts remained net beneficial across the entire range

evaluated, although the magnitude of the benefit decreased steadily as processing losses increased.

These results indicate that the environmental benefits of textile recycling are strongly dependent on material recovery efficiency. Specifically, recovery of approximately one-half or more of collected textile material as marketable recycled products is necessary to maintain net climate change and non-renewable energy benefits. Below this threshold, the environmental burdens associated with collection, transportation, sorting, and processing are no longer offset by avoided virgin textile production.

Impact categories	Unit	20% material processing loss	30% material processing loss	40% material processing loss	50% material processing loss	60% material processing loss	70% material processing loss	80% material processing loss	90% material processing loss
Climate change	kg CO2-Eq	-0.6	-0.4	-0.3	-0.1	0.1	0.2	0.4	0.5
Energy resources: non-renewable	MJ-Eq	-4.4	-3.1	-1.7	-0.3	1.1	2.5	3.9	5.2
Total CED	MJ-Eq	-16.2	-13.3	-10.4	-7.5	-4.7	-1.8	1.1	4.0
Eutrophication no LT	kg N-Eq	-0.1	-0.1	0.0	0.0	0.0	0.0	0.0	0.0

Figure 18 - Impact of processing losses during the textile recycling process on environmental impacts

The sensitivity analysis demonstrates that the baseline conclusion that textile recycling provides environmental benefits is robust to moderate changes in processing efficiency. Net benefits were maintained under loss rates up to approximately 50%, representing a substantial reduction in recovery performance relative to the baseline assumption. The higher-loss scenarios (60%–90%) should be interpreted as bounding analyses intended to identify performance thresholds rather than representative operating conditions. Commercial textile recycling systems generally target recovery rates substantially above these levels because economic viability depends on generating sufficient quantities of marketable recycled products.

From a waste management perspective, these findings highlight the importance of collection quality, contamination reduction, effective sorting technologies, and stable end markets for recycled textiles. Maintaining material recovery rates above approximately 50% appears critical to preserving the climate and energy benefits associated with textile recycling. As a result, programs that improve the quantity and quality of recyclable textile feedstocks may be as important to environmental performance as increases in collection volumes alone.

5.6.2.4 Interpretation of Sensitivity Results

Among the textile recycling sensitivity analyses evaluated, processing losses were identified as a potentially significant contributor to result uncertainty. However, the analysis also demonstrates a clear and interpretable performance threshold. While the

magnitude of environmental benefits declines as processing losses increase, the direction of the results remains unchanged under a wide range of plausible recovery efficiencies. Consequently, the study's overall conclusion that textile recycling can provide environmental benefits relative to disposal pathways is considered robust, provided that material recovery efficiencies remain above approximately 50% of collected textile inputs. This threshold represents a useful operational benchmark for evaluating the environmental performance of existing and future textile recycling systems.

5.6.2.5 Structural Interpretation and Improvement Opportunities

Contribution analysis confirms that collection logistics and long-distance transport are major drivers of gross impacts. The 1,835 km average transport distance to out-of-state recyclers significantly contributes to scenario burdens. Accordingly, environmental performance is strongly influenced by supply chain configuration.

Potential structural improvements include:

- Reducing transport distances by identifying recycling operators nearer to Iowa or encouraging regional recycling capacity development.
- Optimizing collection schemes to reduce passenger vehicle transport.
- Improving mechanical recycling efficiency to increase yield and effective substitution.
- Developing alternative outlets for mixed organic-synthetic textiles currently discarded in the modeled system (e.g., industrial wiping cloths, insulation, padding, or other downcycled applications).

Under current assumptions, mixed blended textiles that cannot be mechanically separated are landfilled, contributing to gross burdens. Expanding secondary markets for these materials could materially improve scenario performance.

5.6.2.6 Overall Interpretation

Under baseline assumptions (100% substitution, and 20% material processing losses) , centralized residential textile recycling produces substantial eutrophication benefits and moderate air emissions impact and energy reductions at the statewide scale. Eutrophication impacts improve markedly due to avoided cotton production, increasing the statewide net credit from approximately –21,000 kg N-eq/year to – 604,850 kg N-eq/year. Reductions of approximately 9,700 t CO_2-eq/year arise from both avoided landfill methane and displaced virgin fiber production. Non-renewable and total energy demand also decline, though partially offset by long-distance transport.

The substitution sensitivity analysis demonstrates that these results are contingent on the assumed displacement rate of virgin fibers. At a 40% substitution factor, avoided production credits decline sufficiently that air emissions impact and energy results reverse sign at the per-ton level, while eutrophication remains beneficial but reduced in

magnitude. This indicates that textile recycling operates with narrower margins between gross burdens and avoided credits than aluminum systems.

Performance is therefore sensitive to substitution rates, processing losses, and logistics configuration. Collection and long-distance transport represent a significant share of gross impacts, and improvements in regional processing capacity, transport optimization, and recovery of currently discarded mixed-fiber textiles could materially strengthen outcomes.

Overall, textile recycling presents a potentially beneficial strategy—particularly for eutrophication—but air emissions impact and energy benefits are conditionally robust and dependent on achieving high functional substitution and efficient system design.

5.6.3 Scenario 3 – 10% Reduction in Fugitive Methane Emissions

This scenario models a 10% reduction in fugitive methane emissions across all landfill processes, consistent with improvements achievable through enhanced gas capture and operational management.

Statewide air emissions impacts decrease by approximately 47,000 t CO_2-eq/year, corresponding to 9.7% of baseline landfill-related impacts. Because methane emissions dominate the profile of degradable landfill streams, benefits scale directly with the baseline methane contribution. Food Waste landfill accounts for approximately 90% of total air emissions impact reduction due to both its methane generation potential and higher statewide tonnage.

Energy indicators (non-renewable CED and Total CED) exhibit negligible change, as methane emissions primarily influence air emissions impact characterization rather than primary energy accounting. Similarly, eutrophication impacts show minimal variation, reflecting the limited role of methane emissions in eutrophication characterization under the selected LCIA method.

This scenario therefore produces a targeted benefit without materially affecting energy or eutrophication outcomes.

5.6.4 Comparative Scenario Interpretation

When evaluated collectively, the three scenarios demonstrate distinct structural characteristics:

- Increased aluminum deposit recycling produces the largest energy savings and substantial air emissions impact benefits, with improvements consistent across all impact categories.
- Textile diversion produces moderate air emissions impact and energy benefits, with greater sensitivity to recovery efficiency and substitution assumptions.

- Methane reduction produces a direct and predictable reduction in air emissions impacts, proportional to landfill methane's contribution, without cross-category effects.

No modeled scenario resulted in burden shifting across impact categories under baseline assumptions. At the statewide scale, expansion of aluminum recycling produces the largest reduction in air emissions impacts and also delivers the greatest decreases in non-renewable energy demand and eutrophication, driven primarily by avoided primary aluminum production. Textile recycling produces the largest eutrophication benefits.

5.7 Limitations of Sensitivity Approach

Sensitivity analyses were conducted using a one-at-a-time deterministic approach. Interaction effects between parameters (e.g., simultaneous variation in methane and aluminum recovery rates) were not explicitly modeled.

While deterministic sensitivity identifies high-leverage parameters, it does not provide probabilistic uncertainty distributions.

Future work could incorporate Monte Carlo simulation for methane and aluminum recovery parameters to quantify joint uncertainty effects if required.

5.8 Robustness of Conclusions

Sensitivity analyses were conducted to determine whether plausible variation in high-leverage parameters or structural modeling assumptions could alter comparative pathway rankings, reverse net benefit conclusions, or materially affect statewide interpretation. Results presented in Section 5.4 demonstrate that robustness varies by impact category and by the type of uncertainty evaluated.

5.8.1 Air Emissions Impacts

Statewide air emissions impact outcomes are structurally dominated by fugitive methane emissions from landfill processes. As shown in Section 5.4.7, methane sensitivity produced statewide shifts of approximately +19% under the high-emission case and −22% under the low-emission case. All other tested parameters, including transportation allocation, curbside participation, aluminum recovery, processing losses, textile recovery efficiency, and electricity displacement assumptions, produced comparatively smaller statewide variation (generally within ±4% for transportation-related parameters).

Importantly, even under high- and low-methane cases, the relative ranking of primary waste management pathways did not reverse. Recycling pathways that demonstrate net air emissions impact benefits in the baseline case remain favorable within tested bounds,

and landfill pathways for degradable materials remain strongly influenced by methane generation potential.

Accordingly, statewide conclusions are considered structurally robust, with methane management performance representing the principal decision-sensitive parameter.

5.8.2 Non-Renewable Energy and Total CED

Energy indicators exhibit greater sensitivity to residential collection structure and travel attribution assumptions than air emissions impacts. Variation in the allocation of residential drop-off vehicle travel between 37.5% and 62.5% attribution to waste management activities produces statewide changes of approximately ±32% for CED Non-renew. and ±11% for Total CED. These outcomes reflect the relatively high fossil energy intensity of passenger-vehicle travel compared with the mass of material transported in residential self-haul systems.

The baseline model allocates 50% of residential drop-off vehicle travel to waste management activities, reflecting the assumption that many drop-off trips occur in combination with other household errands. Consistent with the attributional modeling framework applied in this study, only the fraction of shared vehicle travel attributable to waste management is assigned to the waste system.

Additional behavioral parameters, including curbside participation rates, produce moderate statewide variation in energy indicators (approximately ±28–30% for CED Non-renewable), reflecting changes in transportation intensity associated with shifts between curbside collection and self-haul behavior. In contrast, aluminum processing losses and recovery rate assumptions produce comparatively small statewide changes in energy results.

Despite these changes in magnitude, no sensitivity scenario alters the relative ranking of major waste management pathways in energy categories. Recycling pathways remain the lowest-energy options, landfill systems remain among the higher-energy pathways for several materials, and no sensitivity case produces reversals between landfill and recycling options or changes the sign of pathway results.

Energy conclusions are therefore robust in comparative terms, although absolute magnitudes remain sensitive to assumptions regarding residential travel attribution and collection behavior.

5.8.3 Eutrophication

Eutrophication outcomes exhibit moderate sensitivity to transportation-related parameters and avoided production assumptions. Variation in the allocation of residential drop-off travel between 37.5% and 62.5% attribution to waste management activities produces statewide changes of approximately ±19% in eutrophication impacts. This

sensitivity arises primarily from vehicle-related emissions associated with residential self-haul systems.

Landfill methane variation produces negligible statewide eutrophication change, consistent with the characterization methodology applied and the limited role of methane emissions in nutrient enrichment pathways.

Scenario analysis of residential textile recycling highlights the importance of avoided upstream agricultural production in eutrophication results. Under Scenario 2, diversion of 15% of residential textiles to centralized mechanical recycling substantially increases the statewide eutrophication credit. In the baseline waste management system, the combined management of food waste, plastic films, aluminum, textiles, and leather produces a net credit of approximately –21,000 kg N-eq per year. With the addition of textile recycling, the statewide eutrophication credit could increase to approximately – 604,850 kg N-eq per year, reflecting avoided fertilizer use associated with virgin cotton production. Cotton cultivation is characterized by high nutrient emissions, exceeding 180 kg N-eq per ton of fiber, and displacement of these upstream burdens dominates the scenario outcome.

Sensitivity analysis of the substitution assumption demonstrates that this benefit depends strongly on the extent to which recycled fibers displace virgin production. Under a reduced substitution rate of 40%, eutrophication benefits remain negative but decrease substantially in magnitude. In contrast to aluminum recycling systems—where avoided burdens consistently exceed processing impacts—textile recycling exhibits narrower margins between gross burdens and avoided production credits, making scenario outcomes more sensitive to substitution assumptions.

Despite these variations in magnitude, no sensitivity scenario alters the comparative interpretation of preferred waste management pathways. Recycling pathways continue to generate eutrophication benefits relative to landfill disposal, and no scenario produces reversals between landfill and recycling options.

Overall, eutrophication conclusions are stable in relative terms but sensitive in absolute magnitude to assumptions regarding transportation behavior and the degree of substitution achieved by recycled textile fibers.

5.8.4 Structural Versus Parameter Uncertainty

Two distinct categories of uncertainty were identified:

1. Physical emission parameter uncertainty, primarily landfill methane emissions, which governs statewide air emissions impact results.
2. Structural attribution assumptions, particularly residential travel allocation, which govern statewide energy and eutrophication magnitude.

The former affects environmental intensity of treatment processes; the latter affects how shared household activities are attributed within an attributional framework.

No tested parameter combination resulted in cross-category burden shifting sufficient to alter the overall interpretive conclusions of the study. That is, no scenario resulted in a pathway becoming environmentally preferable in one category while simultaneously becoming comparatively unfavorable in another in a manner that would change decision-relevant conclusions.

5.8.5 Overall Interpretation

Within the tested parameter ranges and structural assumptions:

- Statewide air emissions impact conclusions are primarily governed by methane control performance and remain robust across other tested sensitivities.
- Statewide energy and eutrophication results are structurally influenced by residential collection configuration and travel attribution assumptions but remain stable in comparative ranking.
- Recycling pathways that demonstrate net environmental benefits in the baseline case retain their directional advantage within tested bounds.
- No sensitivity case reversed the direction of statewide net impacts in any category.

Accordingly, the principal conclusions of this study are considered robust within the defined modeling framework, with the explicit caveat that energy-related outcomes for residential drop-off systems depend on documented travel allocation assumptions.

5.9 Overall Data Quality Evaluation

The overall quality of data used in the foreground system is judged to be sufficient. The LCA model was constructed using the Ecoinvent 3.11 background datasets with extensive Iowa-specific foreground customization, wherever possible, with data drawn from SCS Engineers operational and site surveys, IA DNR-regulated facility tonnage data, GIS-derived transport distances, EPA defaults, methodologies, tools and modeling resources, as well as IPCC emissions factors and the WasteTools model. Data quality was assessed across six dimensions, including reliability, temporal correlation, geographic correlation, technological correlation, completeness, and process review, using an EPA template-derived, study-specific pedigree matrix that aligns with the data quality requirements of ISO 14040/14044 Subsection 4.2.3.6. Target scores of <3 were established for 5 of 6 dimensions, with lower ratings for some Food Waste landfill metrics due to lower quality data on fugitive methane emissions related to food waste. Ratings were assigned at the pathway-driver level, which targets the dominant parameters identified through contribution analysis, rather than scoring individual background dataset providers. A total of 33 pathway-driver rows were scored across 17 pathway-specific treatment and process bases, 10 shared assumptions (e.g., transport, substitution credits, leachate, and other

operational parameters). Full data quality scoring definitions, pathway model maps, and the scored data quality matrix are provided in section D of the Appendix.

The following observations summarize the key findings and limitations of the data quality assessment:

- 4 of 6 data quality dimensions, namely, reliability, technological correlation, completeness, and process review, meet the target threshold of ≤3 for 100% of scored rows. No data quality dimension received a score of 5 (unknown or undocumented) for any pathway driver. Scores of 4 are limited to six instances, all concentrated in the food waste landfill pathway and geographic or temporal dimensions for textile and leather waste streams.

- Temporal correlation is strong across the study, with 97% of scored rows meeting the target threshold of ≤3 years. This reflects the use of 2020–2025 primary data from SCS surveys, Iowa DNR reports, and recent Ecoinvent releases. The single row exceeding the temporal target is the textile fiber composition split assumption, which draws on an older European reference study.

- Geographic correlation is the dimension with the most exceedances, though 91% of scored rows still meet the target threshold of ≤3. The three rows scoring 4 are the food waste landfill treatment, the natural textile landfill treatment basis (Doka WasteTools, EU origin), the leather inert waste proxy (Ecoinvent, non-U.S.), and the textile fiber composition split (EU study). These scores reflect the absence of U.S.- or Iowa-specific LCI datasets for textile and leather end-of-life treatment, which represents the most significant data gap in the study.

- Geographic representativeness is otherwise strong: 70% of all rows score 1 or 2, reflecting the degree of Iowa-specific customization applied across the model. This includes GIS-derived collection and interfacility transport distances, Iowa precipitation-based leachate parameterization, Iowa-specific fugitive methane emission factors derived from state agency landfill gas recovery data, and facility-specific operational data from the Ames RRP.

- The fugitive methane emission factor for landfilled food waste is the single most influential parameter in the study, accounting for over 90% of food waste greenhouse gas impacts. Since we did not have access to Iowa-specific data on emissions from food waste, proxies needed to be used, which impacted the Reliability, Geographic, and Completeness scores (Appendix E). This parameter was derived using Iowa-specific inputs to the IPCC Waste Model, including state-level landfill gas recovery data, and is scored separately from the Doka-based treatment process. High-end (0.047 kg CH_4/kg) and low-end (0.027 kg CH_4/kg) bounds were developed and evaluated through sensitivity analysis (Section 5). The resulting range is consistent with published EPA estimates.

- Substitution credits for avoided primary production, including 1:1 displacement of North American primary aluminum and displacement of virgin cotton fiber and PET granulate in the textile recycling scenario, represent the upper bound of plausible avoided burden credits. These assumptions are consistent with ISO 14044 system expansion practice and are standard in waste management LCA, but they do not reflect short-term market dynamics, quality-adjusted substitution ratios, or future system evolution. The substitution credit rows received the highest average scores across the matrix (i.e., mean reliability and geographic scores of 3), reflecting their reliance on global or regional market averages rather than Iowa-specific data.

- Electricity displacement credits for both landfill gas recovery and incineration pathways assume displacement of average US-MRO grid mix electricity. LFG recovery credits are further adjusted for Iowa-specific conditions, including a 33% flaring fraction and Iowa-derived recovery factors. These assumptions influence the magnitude of credits in energy-recovering pathways, but do not independently determine comparative rankings across waste management alternatives, as confirmed by sensitivity analysis.

- Leachate generation, transport, and treatment is modeled as a shared assumption across all landfill pathways. Leachate volume is estimated using an Iowa precipitation-derived regression, transport distances are GIS-based using Ecoinvent transport processes, and treatment uses Ecoinvent WWTP processes.

In conclusion, we assert that the ISO-aligned data quality assessment confirms that the study meets target data quality goals for most scored pathway drivers. The four exceedances are concentrated in textile and leather waste streams, where the absence of U.S.- or Iowa-specific LCI datasets necessitated non-U.S. proxy data. These limitations are documented, data gaps or uncertainty have been evaluated through sensitivity analysis, and these do not affect the robustness of principal comparative findings. The complete data quality pedigree matrix, scoring definitions, and pathway architecture are provided in Appendix D.

1 Abbreviations

Abbreviation	Term Defined
Al	Aluminum/Aluminium
C&D	Construction and Demolition
CED	Cumulative Energy Demand
CFF	Circular Footprint Formula
CH	Switzerland (in Ecoinvent)
CO2e	CO2 Equivalent
CRI	Container Recycling Institute
DOE	Department of Energy
EF	Environmental Footprint 3.1
ENR	Non-renewable CED
Etot	Total CED
Eu	Eutrophication
EuF	Freshwater Eutrophication
EuM	Marine Eutrophication
EURO5	European Class 5 Emissions Standards
FAE	Fly Ash Incineration
FOG	Food, Oil, and Grease
FU	Functional Unit
FW	Food Waste
GHGRP	Greenhouse Gas Reporting Program
GIS	Geographic Information System
GWP	Global Warming Potential
HDPE/LDPE	High/Low Density Polyethylene
HHV	High heating value
ICI	Industrial, commercial, and institutional
Inc	Incineration
IPCC	Intergovernmental Panel on Climate Change
ISO	International Standards Organization
LFG	Landfill Gas
LHV	Low Heating Value
MCF	Methane Correction Factor
mmscfd	million standard cubic feet per day
MPG	Miles Per Gallon
MRF	Materials Recovery Facility
MRO	Midwestern Reliability Organization
MSW	Municipal Solid Waste
N-Equivalent	Nitrogen-equivalent
NGCC	Natural Gas Combined Cycle
No LT	Excluding long-term emissions
NWS	National Weather Service
OOS	Out-of-State

PEF	Product Environmental Footprint
PET	Polyethylene Terephthalate
PPE	Polypropylene
REC	Recycling
Res.	Residential
RRP	Resource Recovery Plant
SLF	Sanitary Landfill
SMM	Sustainable Materials Management
TRACI	Tool for the Reduction and Assessment of Chemical and Other Impacts
UBC	Used Beverage Container
USLCI	United States Life Cycle Inventory
w%	% by weight
WARM	Waste Reduction Model
WWTP	Wastewater Treatment Plant

1

2

Works Cited:

Ballou, R. H., Rahardja, H., & Sakai, N. (2002). Selected country circuity factors for road travel distance estimation. Transportation Research Part A: Policy and Practice, 36(9), 843-848.

Brown, Martin J., & Canepa, Peter. Technical Overview of the Waste Impact Calculator. Oregon Department of Environmental Quality, 11 April 2023.

Circular Thrift and SWACO. 2025 Columbus Textile Collection Pilot Case Study. 2025.

Ciroth, A. (2012). *Refining the pedigree matrix approach in ecoinvent*. GreenDelta GmbH. https://www.greendelta.com/wp-content/uploads/2017/03/Pedigree_report_final_May2012.pdf

City of Ames. (2024). 2024 Resource Recovery Plant annual report. City of Ames, Iowa. https://www.cityofames.org/files/content/city/v/7/my-government/departments/resource-recovery-system/general-information-reports-analysis/2024-rrp-annual-report.pdf

Container Recycling Institute. (2024, May). 2022 recovery rate report for deposit beverage containers: Created for the Iowa Department of Natural Resources. Prepared for the Iowa Department of Natural Resources.

Container Recycling Institute. (2025). Ten states' bottle bill redemption rates. https://www.bottlebill.org/images/Allstates/10_states_Redemption_Rates_080125.pdf

Doka, G. Guidance on How to Consider Waste Disposal in Inventories of Waste Producing Activities, Zurich, May 2023.

Doka, G. (2023) WasteTools2023, https://www.doka.ch/WasteTools2023.zip

Ecoinvent Association. Ecoinvent database version 3.11. Zürich, Switzerland: Ecoinvent Association. 2024.

Ecoinvent Association. (2024). ecoinvent database version 3.11. Zürich, Switzerland: ecoinvent Association. Retrieved August 2025, from https://ecoinvent.org

Enking, J., Becker, A., Schüe, G., Gausmann, M., et. al. (2025). Recycling processes of polyester-containing textile waste – A review. Resources, Conservation & Recycling, 219, Article 108256. https://doi.org/10.1016/j.resconrec.2025.108256

Fathima, N., Rao, R., & Nair, B. U. (2012). Tannery solid waste to treat toxic liquid wastes: A new holistic paradigm. Environmental Engineering Science, 29(6), 363-372.

Frischknecht, R., Wyss, F., Büsser Knöpfel, S., Lützkendorf, T., & Balouktsi, M. (2015). Cumulative energy demand in LCA: The energy harvested approach. The International

Journal of Life Cycle Assessment, 20(7), 957–969. https://doi.org/10.1007/s11367-015-0897-4

Haupt, Melanie, et al., “Life Cycle Inventories of Waste Management Processes,” *Data in Brief* 19 (August 2018): 1441–57, https://doi.org/10.1016/j.dib.2018.05.067;

Hemati, S., Udayakumar, S., Wesley, C., Biswal, S., Nur-A-Tomal, M. S., Sarmadi, N., & Sahajwalla, V. (2023). Thermal transformation of secondary resources of carbon-rich wastes into valuable industrial applications. Journal of Composites Science, 7(1), 8.

Iowa Department of Natural Resources. (2025). *Solid waste program data portal*. State of Iowa. https://programs.iowadnr.gov/solidwaste/

IPCC. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse gas Inventories. Volume 5: Waste. 2019.

Jiang, H., Liu, J., & Han, W. (2016). The status and developments of leather solid waste treatment: A mini-review. *Waste management & research : the journal of the International Solid Wastes and Public Cleansing Association, ISWA*, *34*(5), 399–408. https://doi.org/10.1177/0734242X16633772

Krause, M., Kenny, S., Stephenson, J., & Singleton, A. (2023). Quantifying methane emissions from landfilled food waste (EPA/600/R-23/064). U.S. Environmental Protection Agency, Office of Research and Development. https://www.epa.gov/system/files/documents/2023-10/food-waste-landfill-methane-10-8-23-final_508-compliant.pdf

Moazzem, S., Wang, L., Daver, F., & Crossin, E. (2021). Environmental impact of discarded apparel landfilling and recycling. Resources, Conservation and Recycling, 166, Article 105338. https://doi.org/10.1016/j.resconrec.2020.105338

Morell-Delgado, G., et. al., (2024). Revealing the management of municipal textile waste and citizen practices: The case of Catalonia. *Science of the Total Environment, 843*, Article 168093. https://doi.org/10.1016/j.scitotenv.2023.168093

National Weather Service, National Oceanic and Atmospheric Administration (NOAA). (2025). *Iowa climate Normals: Precipitation*. National Weather Service Des Moines, IA. Retrieved September 2025, from https://www.weather.gov/dmx/climatenormals

Navarro, D., Wu, J., Lin, W., Fullana-i-Palmer, P., & Puig, R. (2020). Life cycle assessment and leather production. *Journal of Leather Science and Engineering, 2*, Article 26. https://doi.org/10.1186/s42825-020-00035-y

New Jersey Department of Environmental Protection, Division of Sustainable Waste Management, Bureau of Planning and Licensing. (2022, August 5). Municipal solid waste quantification and characterization of Burlington, Mercer & Passaic Counties.

New York City Department of Sanitation. (2016). *Commercial waste zones: A plan to reform, reroute, and revitalize private carting in New York City*. City of New York. https://www.nyc.gov/assets/dsny/docs/about/commercial-waste-zones-plan.pdf

Popiţa, G. E., Manciula, D., Popovici, A., & Roşu, C. (2020). Conductometric Tests and Total Chromium Leachability in Aqueous Solution, From Tanned Leather Waste. Ambientum, 59.

SCS Engineers. (2017). Rural Iowa recycling hub & spoke final report. Prepared for the Iowa Department of Natural Resources.

SCS Engineers. (2022). Phase II: Sustainable materials management: Vision for Iowa. Prepared for the Iowa Department of Natural Resources.

SCS Engineers, for Iowa Department of Natural Resources. (2022). *2022 Iowa Statewide Material Characterization Study*, no. 27222148.00 (2022): 1–195;

SCS Engineers. (June 2024). Iowa recycling facility study. Prepared for the Iowa Department of Natural Resources.

SCS Engineers. (June 2024). Landfill material analysis report. Prepared for the Iowa Department of Natural Resources.

SCS Engineers. (unpublished). Draft food waste report. Prepared for the Iowa Department of Natural Resources.

U.S. Department of Energy. (2024). Alternative Fuels Data Center – Vehicle registration counts by state.

U.S. Environmental Protection Agency. (2016). *Guidance on data quality assessment for life cycle inventory data* (EPA/600/R-16/096). National Risk Management Research Laboratory, Office of Research and Development. URL to EPA website

U.S. Environmental Protection Agency. (2020). Advancing sustainable materials management: 2018 fact sheet—Assessing trends in materials generation and management in the United States (EPA 530-F-20-009).

U.S. Environmental Protection Agency. (2023). Food waste management life cycle inventory tool (v1.0.0). Data.gov. https://catalog.data.gov/dataset/food-waste-management-life-cycle-inventory-tool-v1-0-0 (Updated Sept. 2024.)

U.S. Environmental Protection Agency. (2023). Waste Reduction Model (WARM), version 16 – Excel User's Guide (EPA 530-R-23-024). Office of Resource Conservation and Recovery, Washington, DC. Released December 2023

US EPA. From Field to Bin: The Environmental Impacts of U.S. Food Waste Management Pathways. (2023). U.S. Environmental Protection Agency Office of Research and Development EPA/600/R-23/065. October 2023.

U.S. Environmental Protection Agency. (2024). *Avoided landfilled food waste methane emissions calculator* (Excel tool). U.S. EPA Office of Research and Development. https://www.epa.gov/system/files/documents/2024-12/avoided-landfilled-food-waste-methane-emissions-calculator.xlsx

U.S. Environmental Protection Agency. (2024). *Quantifying methane emissions from landfilled food waste (Landing Page for Excel Tool)*. U.S. EPA. https://www.epa.gov/land-research/quantifying-methane-emissions-landfilled-food-waste

U.S. Environmental Protection Agency & RTI (MSW DST). (2000). Data and Data Input Requirements for the Municipal Solid Waste Management Decision Support Tool. PDF: https://mswdst.rti.org/docs/Inputs_Document_OCR.pdf.

Van Rensburg, M. L., Nkomo, S. L., & Mkhize, N. M. (2020). Life cycle and End-of-Life management options in the footwear industry: A review. *Waste management & research : the journal of the International Solid Wastes and Public Cleansing Association, ISWA*, *38*(6), 599–613. https://doi.org/10.1177/0734242X20908938

Vico, A., Maestre-Lopez, M. I., Arán-Ais, F., Orgilés-Calpena, E., Bertazzo, M., & Marhuenda-Egea, F. C. (2024). Assessment of the Biodegradability and Compostability of Finished Leathers: Analysis Using Spectroscopy and Thermal Methods. Polymers, 16(13), 1908. https://doi.org/10.3390/polym16131908

Appendix A – Material Category and Component Definitions

Categories in **bold** included in this study

#	Physical Sort Categories	Material Description
1	Compostable Paper	Low grade paper that could be composted; examples include uncoated paper towels, paper plates, napkins, tissues, and food-contaminated recyclable paper.
2	High Grade Office Paper	Paper free of ground wood fibers; includes office printing and writing papers such as white ledger, colored ledger, envelopes, and computer printout paper.
3	Mixed Recyclable Paper	Other recyclable paper; includes folders, envelopes, index cards, notebook paper, junk mail, chipboard, and uncoated paperboard.
4	Newsprint	Paper chiefly used for printing newspapers.
5	Magazines/Catalogs	Glossy coated paper such as magazines, catalogs, brochures, and pamphlets.
6	OCC and Kraft Paper	Corrugated boxes or paper bags made from Kraft paper; includes shipping boxes and cartons.
7	Aseptic/Gable Top Containers	Containers made from paper, polyethylene, and aluminum layers such as juice boxes and milk cartons.
8	Polycoated Paper	Paper coated with plastic; includes paper cups, plates, frozen food containers.
9	Other Non-Recyclable Paper	Paper combined with other materials such as plastic, foil, or glue.
10	#1 PET IA Deposit Beverage Containers	Plastic beverage containers subject to Iowa deposit law.
11	#1 PET Beverage Containers	Clear or colored PET beverage bottles not under deposit.
12	#1 PET Other Non-Beverage Containers	PET jars, clamshell containers, and food packaging.
13	#2 HDPE Natural Containers	Cloudy plastic containers allowing light through; marked #2.
14	#2 HDPE Colored Containers	Colored HDPE containers blocking light.
15	#3 PVC	Rigid and flexible PVC products such as pipes, siding, packaging.

16	#4 LDPE	Low-density polyethylene used for bags and squeezable bottles.
17	#5 PP	Polypropylene tubs, cups, clamshells, and trays.
18	#6 PS	Non-foam rigid polystyrene containers.
19	#7 Other/Unknown Plastic	Plastic items not fitting other categories.
20	Expanded Polystyrene	Foam products such as cups, plates, and packaging.
21	**Retail Shopping Bags**	**Plastic shopping bags provided at retail.**
22	**Other Plastic Film**	**Garbage bags, sandwich bags, wrappers, and film.**
23	Other Plastic Products	Composite plastic items such as toys, furniture, hoses.
24	**Aluminum IA Deposit Beverage Containers**	**Aluminum beverage containers under deposit law.**
25	**Aluminum Beverage Containers**	**Aluminum beverage containers not under deposit.**
26	Ferrous Food and Beverage Containers	Steel food and beverage containers.
27	**Aluminum Foil and Foil Containers**	**Aluminum foil and trays.**
28	**Other Aluminum Containers**	**Other aluminum items such as cans and foil.**
29	Other Ferrous Scrap Metals	Magnetic iron or steel items.
30	Other Non-Ferrous Scrap Metals	Non-magnetic metals such as copper or brass.
31	Glass IA Deposit Containers	Glass beverage containers under deposit law.
32	Clear Glass Containers	Clear glass bottles and jars.
33	Green Glass Containers	Green glass bottles and jars.
34	Brown/Other Colored Glass Containers	Colored glass containers.
35	Other Mixed Glass	Glass combined with other materials.
36	Wood – Untreated	Wood without paint, stain, or preservatives.
37	Wood – Treated	Wood with adhesives, paint, or preservatives.
38	Plastic Lumber	Wood-like products made from recycled plastic.

39	Insulation	Materials used to reduce heat loss such as fiberglass and foam.
40	Asphalt Paving	Asphalt materials used for paving.
41	Soil, Rock, Sand	Soil, clay, and granular materials.
42	Brick, Ceramics, Porcelain	Masonry materials and fixtures.
43	Concrete	Concrete and cinder blocks.
44	Asphalt Roofing	Roofing materials made with asphalt.
45	Drywall/Gypsum Board	Interior wall coverings made of gypsum.
46	Carpet and Carpet Padding	Flooring fibers and padding.
47	Mattresses	Fabric cases filled with resilient material.
48	Furniture	Tables, chairs, and other furnishings.
49	Yard Waste	Grass, leaves, brush, and tree debris.
50	**Food Waste - Loose**	**Unpackaged food waste.**
51	**Food Waste - Packaged**	**Packaged food waste including liquids.**
52	Television – LCD	Non-CRT televisions.
53	Television – CRT	Cathode-ray tube televisions.
54	Television Peripherals	Devices connected to televisions.
55	Computer Monitor – LCD	Non-CRT computer monitors.
56	Computer Monitor – CRT	CRT computer monitors.
57	Computers – Desktop and Laptop	Computer CPUs and laptops.
58	Computer Peripherals	Devices such as keyboards and mice.
59	Computer Printers	Printers and photocopiers.
60	Audio Equipment	Speakers and audio devices.
61	Electronic Gaming Equipment	Video game consoles.
62	Cell Phone and Chargers	Cell phones and chargers.
63	Other Plug-in Electronics	Miscellaneous electronics.

64	Other Smart Devices – with chip	Smart electronic devices.
65	Other Electrical and Household Appliances	Small appliances and electronics.
66	**Textiles and Leather**	**Clothing, fabrics, and leather items.**
67	Rubber	Rubber products such as hoses and tires.
68	Automotive Products	Products with vehicle fluids.
69	Household Cleaners	Toxic or corrosive cleaners.
70	Lead Acid Batteries	Vehicle batteries.
71	Mercury Containing Products	Items with mercury such as thermometers.
72	Lithium Batteries	Rechargeable lithium batteries.
73	Other Batteries	Alkaline and household batteries.
74	Paints and Solvents	Products containing paint or solvents.
75	Pesticides	Products used to control pests.
76	Sharps	Used needles and medical sharps.
77	Prescription Medications	Medications requiring prescription.
78	Diapers	Adult and infant diapers.
79	Other Organics	Organic materials not otherwise classified.
80	Other Inorganics	Inorganic materials not otherwise classified.
81	Other Consumer Products	Items not elsewhere classified.
82	Other C&D	Mixed construction and demolition materials.
83	Other HHMs	Hazardous materials not elsewhere classified.
84	Fines	Small unsortable fragments under 1/2 inch.

APPENDIX B. Data Assumptions Table

Appendix C. Using WasteTools to create datasets for Ecoinvent

This tutorial presentation can be accessed from here: URL link to DropBox

WasteTools2023

What is WasteTools2023?

- WasteTools2023 is a tool built by Gabor Doka which was used to build most (all?) of the waste treatment processes within ecoinvent

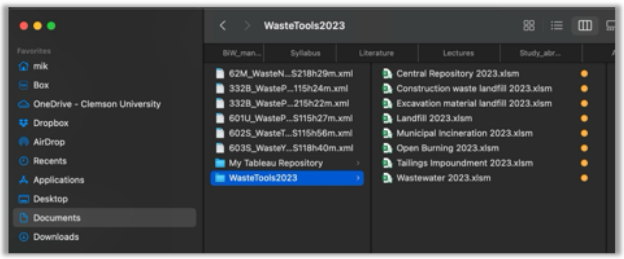
How to access WasteTools2023?

- WasteTools2023 can be accessed directly from this link
- The instruction manual can be accessed here
- Alternatively, you can access Gabor Doka's publications page to find more details about the model under "***Third update location- and waste-specific life cycle inventories of waste treatment***"

- NB: This tutorial assumes that you have openLCA and an ecoinvent 3.10 or higher already installed
  - If you haven't, see openLCA_tutorial_prep_work

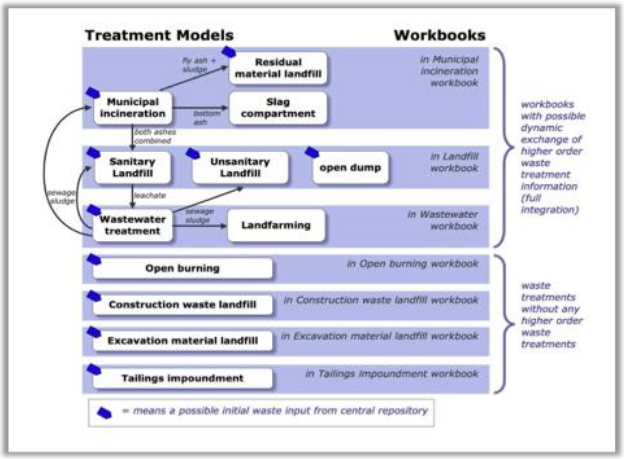
WasteTools2023

- After downloading and extracting the zip file you'll see the following files
- It is recommended to make a copy of the whole folder for working with and keep a backup copy of the original files.



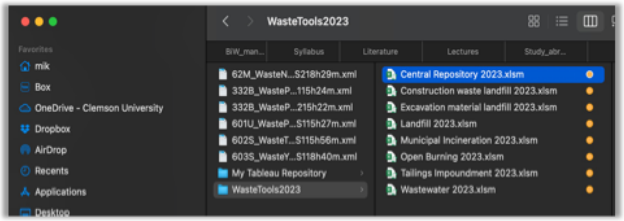
WasteTools2023

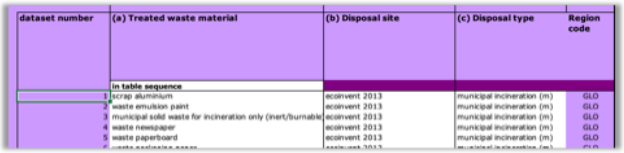
- The tool works by calculating between excel files, as in the figure
- As such, these workbooks need to be accessible when you run the model, normally by opening them

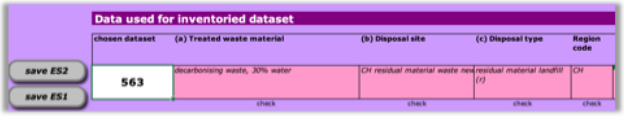


Using WasteTools2023

- Open the workbook "Central Repository 2023.xlsm"
- You must "Enable Macros", but "Don't Update" any links
- Open the "DS info" sheet
- Processes consist of (a) treated waste material, (b) disposal site, (c) disposal type, and region
- Starting in cell C22 are processes that have already been built with the tool (i.e. exist within ecoinvent)
- In order to run the model enter the ID number from column B for any of the processes are interested in (1-598) into cell B12 and then hit either "save ES2" (ecospold2) or "save ES1" (ecospold1).
- Since openLCA can only handle ES1, we'll stick with those.

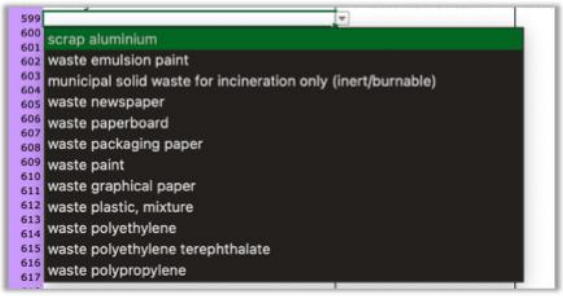






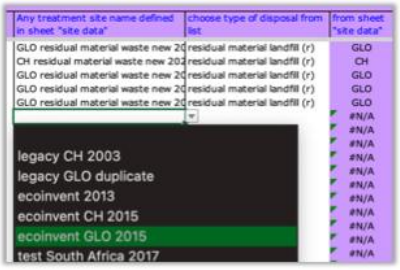
Using WasteTools2023

- The steps to build a new process are:
  - Choosing a waste material
  - Choosing a geographic location
  - Choosing a disposal/treatment type
- Let's go through each in turn
- Go to the an empty (white) cell in column C, probably around C620
- Choose a waste stream from the dropdown menu
- For this tutorial we'll choose "biowaste, 65% water"



Using WasteTools2023

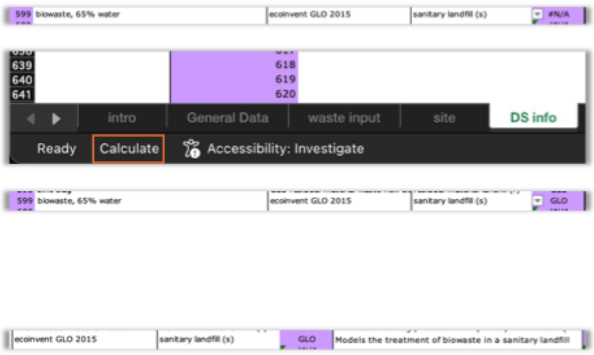
- In column D we need to choose a location from the dropdown menu
- Let's choose "ecoinvent GLO 2015"
- For disposal type in column E we'll choose "sanitary landfill (s)"
- **Caution**: the tool will calculate any combination of waste stream, location, and disposal type, *regardless* if this is realistic or makes sense in the real world





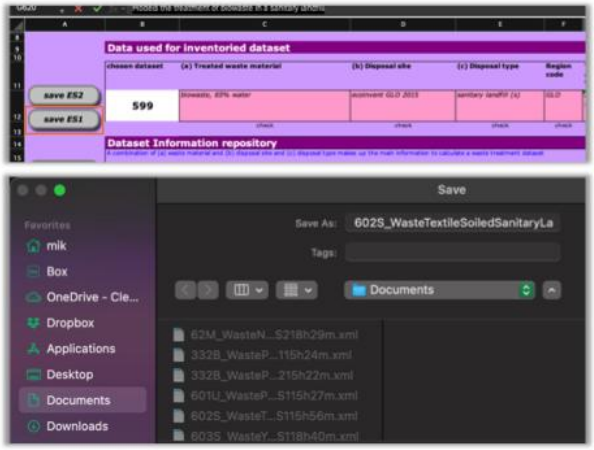
Using WasteTools2023

- Some additional work is required to set up the model
- In column F, if you see “N/A” you should hit “Calculate”
- You should now see “GLO” in that cell
- In column G we should enter a description of the process, “Models the treatment of biowaste in a sanitary landfill”



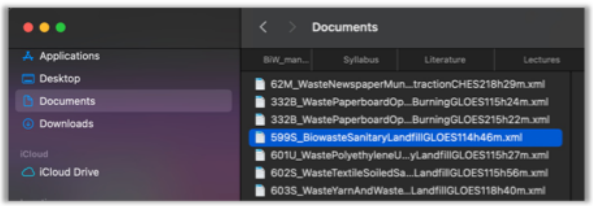
Using WasteTools2023

- We can now run the model
- Enter the correct ID number in cell B12 and then hit “Save ES1”
- Open any files that are requested
- Choose where to save the file
- It is better to choose a location with a short file path, e.g. “Documents” to avoid errors
- You may need to “Grant access” to the file location



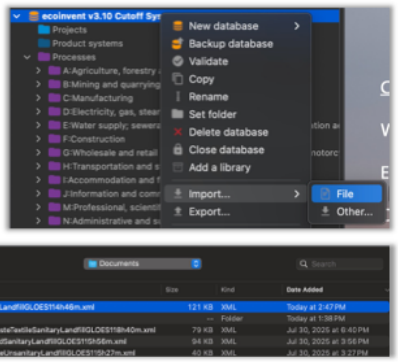
Using WasteTools2023

- The model should take around 2 minutes to run
- When finished, you'll see an xml file in the save location



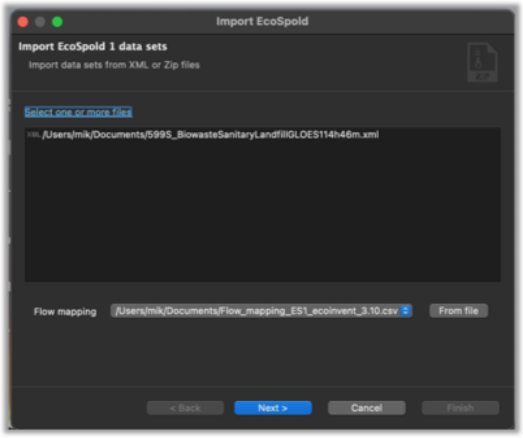
Importing into openLCA

- Open openLCA and the ecoinvent database you want to import the ES1 file into
- To import, right click on the database name, choose "Import..." and then "File"
- Navigate to the file and then hit "Open"



Importing into openLCA

- When we import files, we have an option to map flows in the ES1 file from WasteTools to flows in the ecoinvent database
- You can download a flow mapping here which you'll need to save as a csv file
- In the dialog box under "Flow mapping" choose "From file" and navigate to where you saved the csv
- Hit "Next >" and in the next screen hit "Finish"

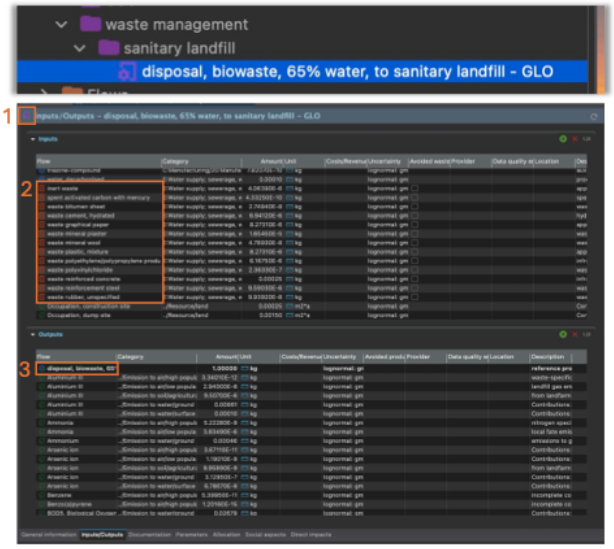


Checking the process

- openLCA will put the new process into a folder called "waste management > sanitary landfill"
- Open the process and go to the "Inputs/Outputs" tab
- There are three problems with the process:

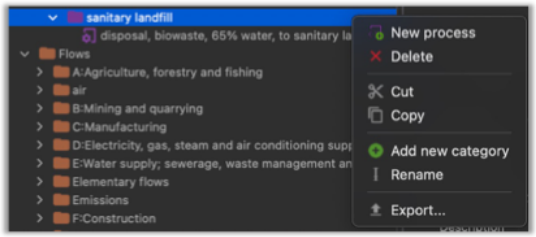
1. the process is a production process (has a cog wheel) rather than a waste treatment process (trash can)
2. some waste outputs from the process are modeled as inputs
3. the quantitative reference flow is a product output and not a waste flow input

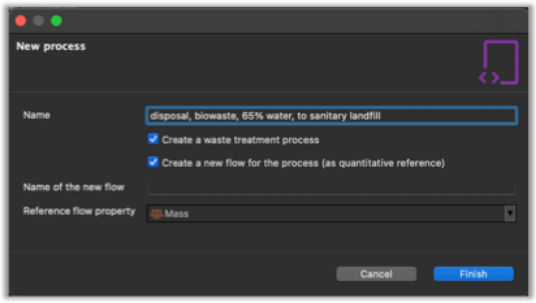
- Let's see how to address all these issues



Editing the process

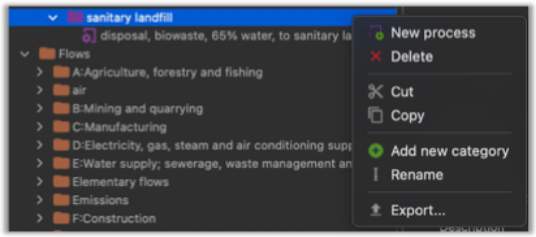
- To solve issues 1 and 3, we can create a new waste flow and treatment process simultaneously
- Right-click the “sanitary landfill” folder and choose “New process”
- In the dialog box, name the process “disposal, biowaste, 65% water, to sanitary landfill”, check both boxes, and choose “Mass” as the Reference flow property
- Hit “Finish”

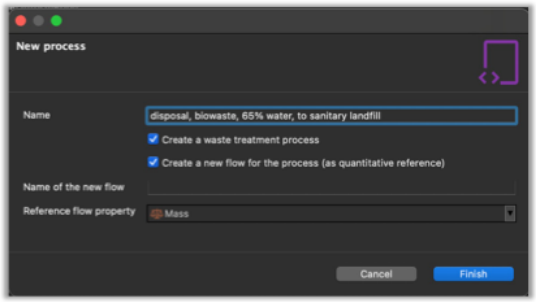




Editing the process

- We now need to copy all of the data into the new process
- In the “General information” tab of the new process, copy and paste information for the “Description” under “General information”, “Geography”, and “Technology”
- Adjust the start and end dates to match
- Choose “GLO” for the “Location”





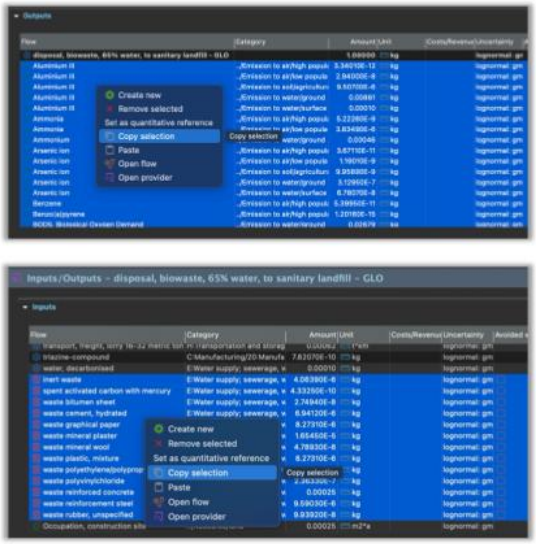
Editing the process

- Open the “Inputs/Outputs” tab
- Select all the inputs from the old process into the inputs table for the new process, with the following exceptions:
  - All waste flows (trash can symbols)
  - “Organic carbon, placed in landfill”
  - “Waste mass, total, placed in landfill”
- Right-click and choose “Copy selection”
- In the new process inputs, right-click and choose “Paste”



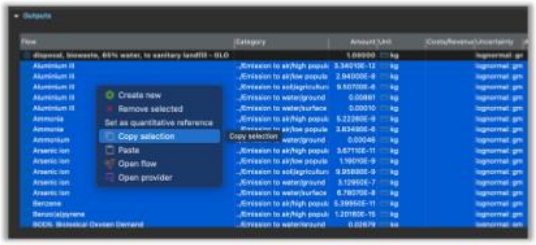
Editing the process

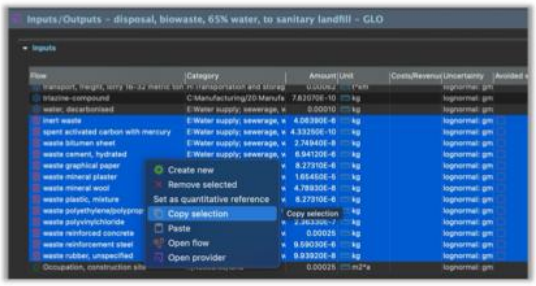
- In the old process, select all of the outputs except the “disposal, biowaste…”, right-click and choose “Copy selection”
- Paste these into the “Outputs” of the new process
- Do the same for the waste flows (trash can symbols), “Organic carbon, placed in landfill”, and “Waste mass, total, placed in landfill” flows from the old process



Double-checking the process

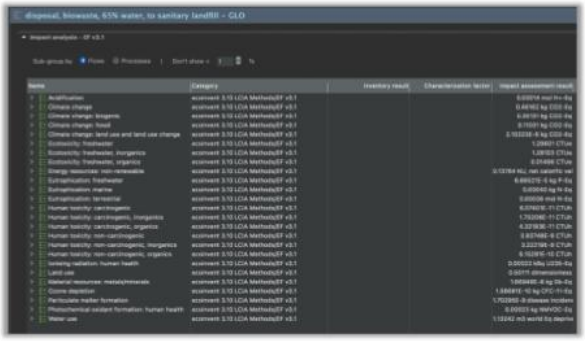
- A way to verify that the steps have been followed is to run an LCIA on the new process
- Go to the “General information” tab and click on “Direct calculation”
- On the info screen click “Run calculation”
- In the dialog, choose “EF v3.1” as the “Impact assessment method”, then hit “Finish”





Double-checking the process

- In the results, go to the “Impact analysis” tab
- The results should (more or less) match the results in the image on the right



APPENDIX D. Data quality scoring for important model drivers and processes.

Table D1 – Modeled Pathways

Waste Stream	Collection Model	Processing Model	Treatment Model	Substitution Model	Customizations Applied? (Y/N)	Key Data Sources	Notes
Aluminum							
Aluminum Recycling AL_REC	Curbside-MRF (45% of residential); Drop-off-MRF (55% residential, personal vehicle 27 km RT); Redemption centers (deposit containers, personal vehicle 16 km RT, 50% trip allocation); ICI contracted hauling	MRF sorting and baling (8% rejection rate per SCS 2024 survey, n=17 facilities); UBC sorting via ecoinvent "treatment of used beverage cans, for recycling, unsorted, sorting"; 0.9% additional remelter loss (ecoinvent default)	Secondary aluminum recycling (Ecoinvent remelting of post-consumer scrap default + UBC sorting process dataset). Net yield: 0.912 metric tons recycled ingot per metric ton collected.	System expansion: 1:1 displacement of "market for aluminium, primary, ingot — IAI Area, North America" (ecoinvent 3.11). Credit = 0.912 t primary Al avoided per t waste entering recycling.	Y. Iowa parameterization: 8% MRF rejection rate (SCS 2024). Other parameters customized: OOS transport to remelters (560 km avg, truck only, based on 3 facilities in IL/IN/MN); residual routing to nearest LF (15.1 km); deposit vs. mixed collection distances (GIS-derived).	Ecoinvent 3.11 (sorting, remelting, primary Al market); SCS 2024 Recycling Facility Survey (rejection rates, facility locations); Iowa DNR Solid Waste Program (facility shapefiles); CRI 2022 Recovery Rate Report (deposit container rates); EPA WARM v16 (collection distance default); Google Maps (remelter distances)	Background providers (diesel, electricity, steel) selected for Iowa/US representativeness per ecoinvent cutoff system model; not independently scored in DQR.
Aluminum Landfill AL_LF	Curbside-LF (67% residential, 21-ton lorry, 32.2 km); Drop-off-LF (33% residential, personal vehicle 23 km one-way, 50% trip allocation); ICI contracted hauling (32.2 km, waste-hauling truck)	None. Aluminum waste bound for landfill undergoes no preprocessing.	Adapted Ecoinvent (treatment of waste aluminium, sanitary landfill (CH)) No methane generation; no LFG energy credit. Aluminum modeled as effectively inert in landfill.	None (no LFG energy credit)	Y. Provider selection for US/Iowa representativeness; leachate volume adjusted for Iowa precipitation (NWS regression); 32% of leachate trucked to WWTP (33.65 km, SCS internal study); interfacility transport TS→LF (38.2 km, >32t lorry, GIS-derived)	Ecoinvent 3.11 (landfill process, WWTP treatment); EPA WARM v16 (collection distance); NWS Iowa climate normals (precipitation); Iowa DNR Solid Waste Program (facility shapefiles, tonnage data); SCS internal leachate handling study.	Aluminum treated as inert in landfill per Ecoinvent characterization. No degradation, no gas generation. Leachate and transport are the primary customization layers.

Textiles							
Textiles Landfill TL_LF	Curbside-LF (67% residential, 21-ton lorry, 32.2 km); Drop-off-LF (33% residential, personal vehicle 23 km one-way, 50% trip allocation); ICI contracted hauling (32.2 km)	None. No preprocessing of textile waste bound for landfill.	Disaggregated by fiber type: (1) Natural fibers (59% of textiles: cotton, wool, viscose): Doka WasteTools 2023 "disposal, waste yarn and waste textile, to sanitary landfill" with parameter-ization, imported to OpenLCA. LFG electricity credit adjusted for 33% IA flaring share. (2) Synthetic fibers + fasteners (41%): Ecoinvent "treatment of waste plastic, mixture, sanitary landfill	LFG electricity credit for natural textile fraction only (adjusted for 33% IA flaring), displacing US-MRO grid average electricity. No credit for synthetic fraction	Y. Iowa-specific: Doka WasteTools parameterization for Iowa landfill conditions; fiber composition split from SCS IA Recycling Facility Study; LFG flaring adjustment (33%); transport distances (GIS-derived); leachate (Iowa precipitation regression) into Ecoinvent processes.	Doka WasteTools 2023; ecoinvent 3.11; SCS 2024 IA Recycling Facility Study (fiber composition); Iowa DNR (facility data); NWS (precipitation); EPA WARM v16 (collection distances); and cited studies from Europe.	No established Ecoinvent dataset exists for sanitary landfill treatment of textiles. Doka WasteTools 2023 was used to construct the natural fiber LCI, consistent with the methodological foundations of ecoinvent's waste treatment datasets. Synthetic textiles proxied to waste plastic mixture.
Textiles Incineration TL_INC	Same as TL_LF collection to Ames RRP: Curbside (97%, waste-hauling truck, 32.2 km); Drop-off (3%, personal vehicle, 23 km one-way).	None. Textile waste delivered to Ames RRP undergoes no external preprocessing. 47.5% of all delivered MSW is rejected as unburnables at plant gate and landfilled (48.3 km to nearest LF, >32t truck).	(1) Natural/organic textiles: ecoinvent "treatment of waste textile, soiled, municipal incineration FAE" with parameter-ization. (2) Synthetic textiles: ecoinvent "treatment of waste polyethylene, municipal incineration FAE" with contextual parameterization.	Electricity credit: Ecoinvent per-kg electricity output value for each waste type, displacing US-MRO grid average. Ecoinvent proxy used because Ames RRP electricity output cannot be disaggregated by waste type.	Y. Iowa region-specific: MRO grid displacement; 47.5% unburnables allocation (Ames RRP 2024 Annual Report); ash transport to landfill (48.3 km, Iowa GIS); Iowa-specific providers. Default ecoinvent processes for residues and spent activated carbon due to data gaps.	Ecoinvent 3.11; Ames RRP 2024 Annual Report (unburnables fraction, operational data); Iowa transport assumptions (GIS); SCS 2024 IA Recycling Facility Study (fiber composition for organic/synthetic split).	Leather waste (~54 t/yr incinerated) excluded per de minimis criterion. Electricity credit uses ecoinvent per-kg proxy; facility-level disaggregation not possible.

Leather Landfill LE_LF	Same as TL_LF: Curbside-LF and Drop-off-LF. Leather collected as part of combined textiles/leather waste stream.	None. Leather waste bound for landfill undergoes no preprocessing.	Ecoinvent "treatment of inert waste, sanitary landfill" with Iowa-specific parameterization. Leather modeled as inert based on literature review: tanned leather exhibits minimal degradation in standard landfill conditions.	None.	Y. Iowa-specific: transport distances (GIS); leachate (Iowa precipitation regression). Other parameters and providers customized for U.S. or region. Leather mass fraction: 6% of residential textiles/leather, 3% of ICI (proxy from NJ DEP 2022 waste characterization study, applied to Iowa).	Ecoinvent 3.11; NJ DEP 2022 (leather fraction proxy); Hemati et al. 2023, Popita et al. 2020, Fathima et al. 2012 (inert classification basis); Iowa DNR; NWS	No established LCI datasets exist for leather waste in sanitary landfill. Inert waste proxy selected based on literature indicating minimal degradation of chrome-tanned leather. NJ DEP data used for mass fraction due to absence of Iowa-specific leather characterization.
Food Waste							
Food Waste Landfill FW_LF	Curbside-LF (67% residential, 21-ton lorry, 32.2 km); Drop-off-LF (33% residential, personal vehicle 23 km one-way, 50% trip allocation); ICI contracted hauling (32.2 km). Composting and WWTP co-digestion excluded (<0.1% of food waste in 2024).	None. Food waste for landfills has no preprocessing.	Doka WasteTools 2023 export for “biowaste, 65% water” sanitary landfill (proxy for food waste), imported to openLCA; Custom IA-CH4 model	Electricity offset against US-MRO grid based on calculated LFG-to-energy recovery (adjusted for flaring)	Y (EPA Warm, IA climate, transport, MRO, and CH4 decay); Iowa-specific fugitive CH_4 EF (baseline + high/low); Iowa recovery factor logic; flaring adjustment (~33%); electricity conversion parameters (LHV, efficiency, density)	Doka WasteTools 2023; IPCC Waste Model (2019 Refinement); EPA Food Waste LCI Tool v1.0.0 (2024); EPA “From Field to Bin” meta-analysis; EPA WARM v16 (moisture check); Iowa landfill recovery from SCS data	Doka "biowaste, 65% water" used as proxy for food waste (vs. 72% moisture in EPA WARM). Fugitive CH_4 EF is the dominant impact driver (>90% of food waste GWP). Iowa-specific CH_4 EF derived from IPCC Waste Model with IA recovery data: baseline 0.036, low 0.027, high 0.047 kg CH_4/kg. LFG electricity credit: mean 60.5 kWh/t. See §3.3.4 for full derivation.

Food Waste Incineration FW_IN	Same as TL_INC collection to Ames RRP: Curbside (97%, waste-hauling truck, 32.2 km); Drop-off (3%, personal vehicle, 23 km one-way); contracted ICI waste hauling.	None. Food waste for incineration undergoes no preprocessing.	Modified ecoinvent treatment of biowaste, municipal incineration FAE with Iowa-specific parameterization + added transport + operational additions	Electricity offset against US-MRO grid (uses ecoinvent per-kg electricity output proxy)	Y (transport distances; MRO); Added delivery transport; allocation of 47.5% unburnables handling; ash + residue routing (regionalized ash transport; default residues/activated carbon)	Ecoinvent 3.11; Ames RRP assumptions; Iowa transport assumptions; internal provider appendix	47.5% of delivered MSW rejected as unburnables, allocated per ton incinerated. Iowa-specific ash transport (48.3 km). Ecoinvent defaults for residues and activated carbon treatment. Electricity credit uses ecoinvent per-kg proxy.
Plastic Films							
Plastic Films Landfill PF_LF	Curbside-LF (67% residential, 21-ton lorry, 32.2 km); Drop-off-LF (33% residential, personal vehicle 23 km one-way, 50% trip allocation).	None. Plastic film waste undergoes no preprocessing prior to landfill.	Ecoinvent "treatment of waste polyethylene, sanitary landfill" (CH) with parameterized providers. Waste PE used as represent-ative proxy for all plastic films (LDPE, HDPE, PPE, multi-layer) based on similar end-of-life behavior.	Minimal LFG electricity credit from Ecoinvent default, reduced by 1/3 for Iowa flaring share. Displaces US-MRO grid.	Y. U.S. provider selection; waste PE proxy for mixed films; LFG credit adjusted for 33% IA flaring; leachate (Iowa precipitation regression); transport distances (GIS-derived).	Ecoinvent 3.11 (waste PE landfill process); NWS Iowa climate normals (precipitation); EPA WARM v16 (collection distance); Iowa DNR (facility data, flaring share); SCS data	Waste PE proxy selected to represent heterogeneous plastic film waste stream. All film types exhibit similar end-of-life behavior in landfill.
Plastic Films Incineration PF_INC	Curbside-INC: 97%, waste-hauling truck, 32.2 km; Drop-off-INC: 3%, personal vehicle, 23 km one-way (50% trip allocation.)	None at facility level. 47.5% of delivered MSW rejected as unburnables at Ames RRP plant gate.	Modified ecoinvent treatment of waste polyethylene, municipal incineration FAE with Iowa-specific parameterization + transport + unburnables + ash/residue handling	Electricity offset against US-MRO grid (uses ecoinvent per-kg electricity output proxy)	Y. Iowa region-specific: MRO grid displacement; 47.5% unburnables allocation (Ames RRP 2024); ash transport (48.3 km).	Ecoinvent 3.11; Ames RRP assumptions; Iowa transport assumptions; internal provider appendix	Highest per-ton air emissions intensity in study due to fossil carbon in PE. Energy recovery credits significant (~8.5 GJ/t non-renewable CED offset) but insufficient to offset fossil CO_2 emissions.

Table D2 - Data Quality Scoring Results

Path-way	Pathway Driver/Process Basis	Basis Type	Reliability	Temporal	Geographic	Techno-logical	Complete-ness	Process Review
AL_LF	Aluminum landfill treatment proxy basis as waste aluminum, sanitary landfill with IA modifications covering UBC and mixed aluminum via composite assumption + SCS data	Customized process/proxy + SCS Observation	3	2	2	2	3	2
AL_REC	Out-of-State Transport Assumption by 32 ton lorry (560 km)	Customized transport assumption	2	2	2	2	3	2
AL_REC	Sorting and remelting treatment basis for aluminum recycling (ecoinvent UBC + remelter basis)	Custom process + Ecoinvent basis	3	2	2	2	3	2
AL_REC	Aluminum recycling rejection rate in IA (8%) & Deposit UBC & non-deposit mixed aluminum	Observed/inferred from SCS studies	2	1	1	1	3	2
AL_REC	1:1 Substitution of North American primary aluminum production	Substitution credit	3	3	3	3	3	3
TL_LF	Natural textile landfill treatment basis with waste-yarn, sanitary landfill	Customized process/Doka	3	2	4	3	3	2
TL_LF	Synthetic textile landfill treatment basis with mixed plastic waste, sanitary landfill	Custom process/ Ecoinvent proxy	3	2	3	3	3	2
LE_LF	Leather mass fraction assumption (6% residential; 3% ICI)	Proxy data from NJ	3	2	3	2	3	2
LE_LF	Leather landfill treatment basis using inert waste proxy at sanitary landfill	Custom process + Ecoinvent basis	3	2	4	3	2	2
TL_INC	Synthetic textile incineration treatment basis with waste polyethylene proxy, muncipal incineration	Custom process/ecoinvent proxy	3	2	2	3	3	2

TL_INC	Organic waste textile incineration treatment basis using soiled, municipal incineration FAE	Custom process + Ecoinvent basis	3	2	2	2	3	2
TL_INC	Ash and residue handling basis with bottom ash and residues routing (organic and synthetic textile fractions)	Custom process + Ecoinvent basis	3	2	3	3	3	2
FW_LF	Food waste landfill treatment basis as biowaste 65% water, sanitary landfill with IA moisture and CH4 factor	Customized process element	2	1	1	2	3	2
FW_LF	Landfill Methane Emissions Factor based on IA contextualized parameter inputs (CH4 generation, decay, temperature, and moisture)	Customized process element	4	2	1	4	4	3
FW_INC	Food waste incineration treatment basis as biowaste, municipal incineration FAE with IA parameterization.	Custom process + Ecoinvent basis	3	2	2	2	2	2
PF_LF	Plastic film landfill treatment basis with waste polyethylene, sanitary landfill	Custom process/ EcoInvent proxy	3	2	2	3	3	2
PF_INC	Plastic film incineration treatment basis with waste polyethylene, municipal incineration FAE (Driven by Fossil carbon emissions from PE combustion)	Custom process/ EcoInvent proxy	3	2	2	3	3	2

Table D3 - Shared Assumptions Table

Path-way	Pathway Driver/Process Basis	Basis Type	Reliability	Temporal	Geographic	Techno-logical	Complete-ness	Process Review
PF_INC; FW_INC; TL_INC	Incineration Electricity credit / US-MRO substitution basis	Substitution credit	3	1	1	1	2	2
All	Municipal waste collection assumption (21 ton lorry)	Shared transport assumption	3	2	3	2	3	2
PF_LF; TL_LF; LE_LF	In-state interfacility transport assumption (MRF to LF) via 16-32 ton lorry	Shared transport assumption	2	2	2	2	3	2
PF_LF; TL_LF; FW_LF	LFG electricity credit / US-MRO substitution basis	Substitution credit	3	2	2	2	2	2
All	Residential Drop-Off Transportation Assumption (Euro 5 passenger car transport = 24 mpg)	Shared transport assumption	2	1	1	2	2	2
FW_INC; TL_INC; PF_INC	In-state transport assumption (INC to LF) via 16-32 ton lorry from Ames RRP + Ames RRP unburnables allocation (47.5%)	Observed/inferred from SCS studies	2	1	1	2	2	2
TL_LF; LE_LF	Textile fiber composition split assumption (natural 59%/synthetic 41%)	EU study	3	4	4	3	3	2
All LF	LFG recovery factor and flaring adjustment (~33%) as a custom calculation basis	Observed/inferred from SCS studies	2	1	1	2	3	2
All LF	Leachate Management: Leachate generation, transport, and treatment basis (Iowa precipitation-derived volume, GIS	Customized process element + Ecoinvent basis	2	2	1	2	2	2

	transport distances, ecoinvent WWTP treatment)							
FW_INC; PF_INC; TL_INC	Ash and residue handling basis with bottom ash and residues routing	Custom process/ecoinvent proxy	3	2	3	3	3	2

Appendix E – Additional TRACI 2.1 Indicators. Baseline Results, net impact after credits.

Table F1. Impact indicators per functional unit, ton of waste treated, ICI

Impact categories	Unit	ICI Aluminum Waste - Landfill	ICI Deposit Cans, Aluminum - Recycling	ICI Food Waste - Incineration	ICI Food Waste - Landfill	ICI Leather Waste - Landfill	ICI Mixed Aluminum - Recycling	ICI Plastic Films - Landfill	ICI Plastic Films- Incineration	ICI Textile Waste - Incineration	ICI Textile Waste - Landfill
Acidification no LT	kg SO2-Eq	0.38089	-66.1255	0.19996	0.27019	0.27863	-65.9762	0.29807	-2.14535	0.10384	0.26464
Ecotoxicity: freshwater no LT	CTUe	32.04416	-1761.18	44.23372	51.00431	25.90126	-1796.57	44.07209	2755.645	1103.816	28.01888
Human toxicity: carcinogenic no LT	CTUh	1.03E-07	-9.77E-05	1.07E-06	1.54E-08	6.36E-08	-9.80E-05	5.85E-08	-4.04E-06	-2.80E-06	1.03E-08
Human toxicity: non-carcinogenic no LT	CTUh	8.29E-06	-2.67E-05	1.61E-05	2.24E-05	3.42E-06	-2.72E-05	4.46E-06	-5.81E-05	-3.90E-05	8.22E-06
Ozone depletion no LT	kg CFC-11-Eq	1.17E-06	-7.26E-05	9.34E-07	8.93E-07	1.00E-06	-7.18E-05	1.07E-06	-8.87E-07	3.98E-07	9.56E-07
Particulate matter formation no LT	kg PM2.5-Eq	0.04408	-9.85494	-0.2827	-0.14899	0.03796	-9.83515	0.03005	-4.02677	-2.45794	-0.00268
Photochemical oxidant formation no LT	kg O3-Eq	9.96102	-605.732	11.04534	9.12176	8.87916	-600.352	9.61901	-5.45204	42.15124	9.20827

Table F2. Impact indicators per functional unit, ton of waste treated, Residential

Impact categories	Unit	Residential Aluminum Waste - Landfill	Residential Deposit Cans, Aluminum - Recycling	Residential Food Waste - Incineration	Residential Food Waste - Landfill	Residential Leather Waste - Landfill	Residential Mixed Aluminum - Recycling	Residential Plastic Films - Incineration	Residential Plastic Films - Landfill	Residential Textile Waste - Incineration	Residential Textile Waste - Landfill
Acidification no LT	kg SO2-Eq	0.49142	-63.7291	0.21646	0.38071	0.38916	-65.8911	-2.12885	0.4086	0.12034	0.37516
Ecotoxicity: freshwater no LT	CTUe	154.7619	-71.8095	55.59877	173.7221	148.619	-1657.52	2767.01	166.7899	1115.182	150.7366
Human toxicity: carcinogenic no LT	CTUh	3.73E-07	-9.40E-05	1.09E-06	2.85E-07	3.33E-07	-9.77E-05	-4.01E-06	3.28E-07	-2.77E-06	2.80E-07
Human toxicity: non-carcinogenic no LT	CTUh	1.22E-05	3.44E-05	1.66E-05	2.62E-05	7.29E-06	-2.29E-05	-5.76E-05	8.33E-06	-3.86E-05	1.21E-05
Ozone depletion no LT	kg CFC-11-Eq	2.16E-06	-5.63E-05	1.04E-06	1.88E-06	1.99E-06	-7.10E-05	-7.76E-07	2.05E-06	5.09E-07	1.94E-06
Particulate matter formation no LT	kg PM2.5-Eq	0.06342	-9.47778	-0.2801	-0.12966	0.0573	-9.81804	-4.02418	0.04938	-2.45535	0.01665
Photochemical oxidant formation no LT	kg O3-Eq	11.11578	-559.607	11.36515	10.27652	10.03392	-600.42	-5.13223	10.77377	42.47105	10.36303