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A Comparative Analysis of Waste-as-a-Feedstock Accounting Methods in Life Cycle Assessments

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Abstract

Global waste generation is a ubiquitous challenge, driving a paradigm shift towards viewing waste as a valuable resource for a circular economy across diverse sectors. While innovative waste-to-resource pathways are crucial, rigorous Life Cycle Assessment (LCA) is essential to ensure the pathways are an important part of current practices. However, LCA application to waste valorization varies, leading to incomparable results due to differing methodological choices. This paper examines three key nuances in waste-as-resource LCAs: the zero-burden assumption, the biogenic carbon neutrality assumption, and the benchmark assumption for emissions avoidance. Using a waste gasification to hydrogen case study, we demonstrate how these methodological decisions impact LCA outcomes. Our findings reveal that waste composition significantly influences the results and highlight challenges associated with biogenic carbon accounting under various system boundary assumptions. Emissions avoidance accounting requires multi-functional unit perspectives and robust benchmark selection. This paper clarifies these accounting approaches, empirically illustrates their influence, and discusses broad implications for accurate sustainability assessment, emphasizing the critical role of transparent LCA choices for effective policy and investment in circular economy solutions.



Academic Editor: George E. Marnellos

Received: 20 June 2025

Revised: 21 August 2025

Accepted: 2 September 2025

Published: 24 September 2025

Citation: Davis, T.W.; Mahmud, R.; McNaul, S.; Jamieson, M.; Lewis, E. A Comparative Analysis of Waste-as-a-Feedstock Accounting Methods in Life Cycle Assessments. *Hydrogen* **2025**, *6*, 74. <https://doi.org/10.3390/hydrogen6040074>

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Keywords: biogenic carbon; circular economy; hydrogen production; life cycle assessment; municipal solid waste; waste as a resource

1. Introduction

Solid waste generation presents a ubiquitous and growing challenge that carries significant global economic, environmental, and social implications [1–5]. The sustainability of the prevailing linear model of waste management is being called to question and, with it, a fundamental paradigm shift that views waste not as an unavoidable burden but as a valuable resource for a circular economy [6–8]. These waste streams originate across various provenances (e.g., municipal solid waste [MSW], agricultural and forestry residues, and industrial and specialty wastes), and their reuse as a feedstock spans several sectors, such as in commercial materials made from the recycling of plastics, metals, critical minerals, papers, glasses, textiles, and rubbers [9]; in construction materials such as aggregates, fill, engineered wood, and insulation [10]; in energy products from direct combustion, gasification, and liquid/solid fuel extraction [11]; in agriculture soil amendments such as compost, digestates, and biochar [12]; and in manufacturing industrial chemicals such as

syngas, naphtha, and oils [13]. The development of these innovative waste-to-resource pathways is critical to realizing the goals of a circular economy. However, the advancement of resource optimization practices requires more than just waste diversion; it demands a rigorous, holistic assessment to quantify the net gains or potential tradeoffs associated with each waste-as-a-feedstock pathway. This is why the comprehensive, multi-dimensional Life Cycle Assessment (LCA) method is indispensable [14].

While established LCA frameworks provide a foundation for this type of analysis [15,16], their operationalization in the context of waste valorization pathways often involves varying interpretations and methodological choices leading to large and often incomparable results [17–19]. This is compounded with the well-documented limitations and shortcomings of waste management LCA in the literature [20,21]. This paper addresses three nuanced approaches in waste-as-resource LCAs: the treatment of waste as a burden-free resource (the zero-burden assumption), the exclusion of biogenic carbon in impact assessments (the neutrality assumption), and the application of differing baseline scenarios in emissions avoidance calculations (the benchmark assumption). Such choices directly impact the environmental profiles and perceived sustainability benefits of utilizing waste as a feedstock. While other studies classify and discuss deficiencies in LCAs, this paper aims to provide a comprehensive description of these approaches, demonstrate the impact of these choices through an example using the gasification of MSW to hydrogen production, and discuss the results in the context of the case study and provide a synthesis of the broader implications of these accounting methods towards the assessment of circular pathways.

2. Materials and Methods

2.1. Life Cycle Assessment Framework

2.1.1. Principles of LCA

Environmental LCA is a standardized quantitative method that is used to evaluate the potential environmental impacts of product, process, or service throughout its entire life cycle (i.e., from raw material extraction through manufacturing, use and end of life treatment stages). There is a structured approach to LCA that aims to provide a comprehensive and systematic assessment that identifies the shifting of burdens between life cycle stages or impact categories.

Standards of Practice. LCA is governed by the International Organization for Standardization (ISO) 14040 and 14044 [15,16]. Together, these standards provide a foundational framework for conducting LCA studies. Critical to these frameworks are the definitions they provide of the general LCA phases (e.g., goal and scope, inventory analysis, impact assessment, and interpretation) and core principles (e.g., transparency and consistency). However, and with due respect, these guidelines inherently allow for a certain degree of interpretational flexibility; a single set of strict rules for all possible LCA studies is likely impossible. The flexibility provided by these guidelines accommodates the vast diversity of products, processes, and study objectives that may fall under the umbrella of LCA. Simultaneously, the flexible guidelines also present a challenge with ensuring comparability between LCA studies, particularly with complex systems such as waste-to-resource pathways. A few of these methodological choices are highlighted in this paper, including the choices of system boundary, impact assessment method and functional unit perspective.

System Boundaries. In LCA studies, the system boundary defines the interface between the product system and the environment. It also determines what unit processes are to be included (and excluded) from the analysis. As defined by the ISO 14040/44 [15,16], LCA studies are designed to assess the environmental impacts from raw material extraction through end-of-life treatment. With waste-to-resource pathways, the question is over the

definition of what constitutes a raw material (i.e., Is waste a raw material?). This creates a special topic on how to define the upstream boundary for waste as a feedstock.

The simplest accounting method for waste is to treat it as an end-of-life material that is free from the environmental burdens associated with the original products of which it is comprised and is regarded as a new resource in the waste-as-resource product system (i.e., the “zero-burden” assumption). This concept of waste as a free resource is the generally accepted practice for waste-to-resource LCAs, assuming that the proposed feedstock meets the definition requirements of waste [14]. The system boundary for traditional LCAs of waste management begins with the collection and transportation of the waste—occasionally overlooking these early steps of the life cycle due to the assumption (and general acceptance) that collection and transportation are insignificant contributors to the overall impacts of the waste chain [22]—and continues through preparation, product and co-product generation, material recycling, and final disposition of unusable waste. Contrary to this belief is the perspective that waste carries some proportion of the environmental burdens associated with the life of the products before they became a waste and that this proportion of environmental burden should be allocated to the waste material as it enters the new waste-as-a-resource product system. Similarly, if there are environmentally desirable attributes of that waste (e.g., biogenic carbon content), there should be a determination to assign those attributes to the original product, to the waste, or both. This expands the system boundary to include upstream processes such as the raw material extraction of products before they enter the waste stream. Diverse and complex waste compositions and data availability often hamper this type of expanded system boundary, which can lead to simplifications and assumptions that have an impact of LCA results.

Impact Assessment Methods. The impact assessment of any product or system relies on the application of characterization factors to convert emissions (i.e., quantity and respective unit of measure) to potential impacts within one or more impact categories based on the definitions provided in one or more of the numerous existing impact assessment methods [15]. There is no standard impact assessment method defined by ISO 14040/44, and the conversion of emissions to impacts may be treated quite differently depending on what method is chosen. This becomes important when considering how certain complex emissions are handled, such as biogenic carbon (i.e., carbon that originated from a non-fossil biogenic source), and whether this class of emissions should have a neutral or negative impact on the environment (i.e., the “neutrality” assumption).

Functional Units. To comply with ISO standards, LCA product systems are required to perform (and be aligned to) a specified function. The function must be quantitatively measurable, resulting in a functional unit that is used to standardized impacts (e.g., per kilogram of new product produced or per ton of waste managed).

Functional units for waste-to-resource pathways come in two main varieties: input-based and output-based [14,20]. The choice between these two has implications for the information and utility provided by the LCA results. Input-based functional units (e.g., per ton of waste treated) facilitate comparisons between different waste treatment pathways, while output-based functional units (e.g., per kilogram of new product produced) highlight the useful services of the waste-to-resource system.

There is no standard or guideline from which to choose the functional unit perspective, and the choice alters the results of impact assessments for comparative LCAs [23], which should ultimately align with the scope of the overall LCA and its intended application. While the choice of functional unit is critical to an LCA study, its perspective (e.g., input-based versus output-based) merely changes how the results (e.g., from an impact assessment) are compared to other, similar systems.

2.1.2. The Zero-Burden Assumption

It has long been the understanding that waste products are treated differently from elementary, product, and co-product flows [15]. Waste as a feedstock is an interesting concept from a life cycle perspective, particularly when determining where the “cradle” in “cradle-to-grave” begins. The zero-burden assumption serves as the de facto treatment of waste as a feedstock, meaning that all activities (including all direct and indirect emissions) pertaining to a product before it was designated as a waste are excluded from the system boundary [24]. The benefits of the zero-burden assumption are that it is easy to account and that it provides comparability with other end-of-life pathways. This accounting also tends to favor recycling pathways, particularly when compared against virgin material alternatives [25]. While this assumption permits the comparison of different waste management strategies, it is criticized for being an oversimplification of reality and failing to assess waste prevention strategies, where quantities of waste generation are subject to change [26].

There is a growing interest in and recognition of the need to reassess the zero-burden assumption and perform LCAs with waste as a feedstock that include a full spectrum of upstream impacts. While outside today’s paradigm, a new design would be necessary should the future of waste-as-a-feedstock shift closer to a perspective of “extended producer responsibility”, where manufacturers take on greater accountability for their products throughout the products’ life cycles [27]. The shared burden perspective identifies waste as a transformation of its original products, and, as such, it carries a proportion of its environmental burdens of its past life when it becomes a waste-based feedstock. To accommodate the upstream life cycle burdens of waste requires significantly more effort to account for all materials entering a waste stream, which, for complex waste compositions, may be technically and economically infeasible, thus reliant on estimation and prediction models [28].

2.1.3. The Neutrality Assumption

While carbon dioxide (CO₂) in the atmosphere behaves exactly the same regardless of the source that created it, impact assessment methods may characterize CO₂ emissions differently depending on the origin of the feedstock [1]; this is mainly to distinguish anthropogenic (i.e., human caused) sources as either biogenic (i.e., related to the natural carbon cycle) or non-biogenic (e.g., burning of fossil fuels or peat). Biogenic carbon accounting is a method for tracking carbon associated with biomass. It differentiates from fossil carbon accounting due to the vast difference in timing of the carbon cycles. Biogenic carbon can take years or even decades to move into or out of the current biomass system, whereas fossil carbon takes millions of years for formation and has no real connection with the current biosphere. The core approach to biogenic carbon accounting is the carbon neutrality method whereby carbon emissions from biomass (e.g., combustion, oxidation, and decay) are assumed balanced by the carbon uptake of the biogenic system, and, producing no net change in the overall carbon balance, biogenic carbon emissions are therefore ignored in LCA studies (i.e., a 0/0 approach) [29], whereas non-biogenic carbon emissions (e.g., fossil carbon) are categorized with +1 units of carbon dioxide equivalence (CO_{2e}). To better account for the carbon neutrality, a −1/+1 accounting method is proposed whereby biogenic carbon causes a carbon sink when entering a product system (i.e., −1 CO_{2e} units) and exits the system with the same +1 CO_{2e} units as other carbon emissions [29].

This has led to the “neutrality assumption” for biogenic carbon within energy pathways. This neutrality assumption prompts the following two considerations for biogenic carbon accounting: (1) biogenic carbon emissions impose no added stress to global warming potential (i.e., they are a part of the natural carbon cycle), and, as such, their characterization factor for global warming potential is zero, and (2) the sequestration of biogenic carbon

emissions has a net-negative impact on global warming potential (i.e., a net reduction of natural CO₂).

To accommodate the neutrality assumption, Wiloso et al. [30] proposed an addendum that the biomass used must be succeeded with fast-growing new plantings (as demonstrated by time series analysis by Schivley et al. [31]). In the context of biogenic carbon, not from trees but from waste (under the zero-burden assumption), it may be argued that the biogenic “source” is constantly replenished, assuming steady-state waste generation rates. Thereby waste-as-a-feedstock would not necessarily be ruled out by this type of accounting.

Looking at how biogenic carbon accounting is handled in other respects, in biomass-to-energy pathways, carbon emissions may be categorized as process, combustion, or avoided CO₂ [32]. Process CO₂ is the carbon emitted during the conversion of biomass to a fuel or chemical, which may be either biogenic or non-biogenic carbon, depending on the fuel source(s) associated with generating the energy used during the conversion. Combustion CO₂ is the carbon emitted from the actual burning of biomass to produce energy (e.g., heat or electricity) and is considered biogenic carbon. Avoided CO₂ is the carbon not emitted due to the replacement of the business-as-usual practice with the proposed product or system (e.g., biomass fuels displacing fossil-based fuels). This last method of avoided emissions accounting depends on the benchmark process. The avoided emissions are not necessarily biogenic or non-biogenic; therefore, the consideration and accounting of avoided emissions will be handled separately, as discussed in the next section.

The challenge, then, of biogenic accounting lies in correctly labelling feedstock sources as either biogenic or non-biogenic. This may be an impossible task for some waste streams. For example, the U.S. Energy Information Administration (EIA) found that insufficient data were available to directly allocate biogenic and non-biogenic carbon to MSW streams and, therefore, proposed a simple categorization scheme whereby individual waste streams are labeled as either sources of biogenic carbon (e.g., paper, textiles, food scraps, and wood) or non-biogenic carbon (e.g., rubber and plastics) and are weighted based on their relative heating capacity (i.e., energy allocation method) [33].

2.1.4. The Benchmark Assumption

Avoided emissions are emissions that would have been emitted but were prevented due to the implementation of an alternative pathway (i.e., a waste-to-resource pathway). Examples of avoided emissions include switching from fossil to lower-carbon fuels and capturing and storing fossil-derived emissions in geologic formations [32]. One illustration of avoided emissions is the highly negative carbon intensity given to dairy farmers in California, the United States, who install anaerobic digesters to capture and clean biogas from cow manure for use in transportation and electricity generation [34].

In LCA accounting, avoided emissions depend on a benchmark, such as the business-as-usual process (i.e., the assumed or understood occurrence of what would take the place of the proposed product or system) or a counterfactual process (e.g., a contrary occurrence to business as usual). An LCA is conducted with both the proposed product system and the benchmark process(es), and the resulting impacts are then compared. The comparative nature of emission avoidance (e.g., via the displacement of existing products or processes) suggests that the LCA type is no longer attributional (i.e., dealing with the total burdens of a system or product) but is rather consequential (i.e., dealing with impact benefits of a proposed alternative, as described in A.2b in [15]). Fruergaard and Astrup (2011) employed such a consequential LCA of coal and refuse-derived fuel co-combustion, which includes, in their system boundary, the avoided emissions associated with displaced fossil-based fuels [35]. The exception to this occurs when an attributional LCA employs co-product management using system expansion (e.g., with substitution and subtraction to maintain

a single function) to avoid allocation [36]. A benchmark comparison, therefore, acts as a hybrid between attributional and consequential LCA (i.e., interested in both the full impacts associated with both systems and the differences between them). Either way, the LCA requires that “benchmark” systems be modeled and their respective impacts be added to, subtracted from, or compared against the proposed alternative.

The debate is not over the LCA of proposed or benchmark systems but, rather, the selection and justification of benchmarks used for comparison. Benchmark selection (i.e., business-as-usual or counterfactual processes) is highly contested [32] and varies depending on the policy under question or the attributes of the product or process (e.g., cost of coal or energy under peak demand). It is further contested that waste-to-resource pathways are likely to offset or replace any traditional processes, thus invalidating any benchmark comparison on principle, and it is suggested that their true purpose is more aligned with reducing the overall environmental burdens of the waste stream (e.g., see the discussion on conventional fuel displacement in [37]).

2.2. Illustrative Example: MSW Gasification to Hydrogen

2.2.1. System Description

A gasification pathway of MSW to hydrogen (H_2) was simulated to illustrate the implications of the three LCA accounting methods. There are many waste-to-energy pathways for MSW, including incineration, hydrothermal carbonization, pyrolysis, gasification, and biological methods such as anaerobic digestion that lead to potential products such as solid refuse fuel/refuse-derived fuel, sustainable aviation fuel, heat and power, and renewable chemicals (e.g., methanol) [14,38]. One pathway is the gasification of MSW to produce H_2 , a carbon-free energy carrier and feedstock found in several markets (e.g., ammonia production, food and drug hydrogenation, petrochemical refining, and glass manufacturing) and emerging as a potential option for decarbonizing three fossil-heavy sectors, namely transportation, electricity generation, and manufacturing [39–41]. The gasification of MSW involves converting what is predominantly biomass (e.g., wastepaper, cardboard, wood, and other mixed organics) to syngas, introducing virtually no new carbon into the atmosphere [42]. Syngas is a mixture of H_2 and carbon monoxide (CO) and is a precursor in the production of chemicals and fuels, including H_2 production.

The proposed system is a simulated two-stage, fluidized-bed, oxygen-blown gasifier system. The gasification plant is assumed to be co-located with the landfill’s material recovery facility (MRF), thereby making the transportation distance between the MRF and gasification facility negligible. MSW sorting is handled by the MRF, and further processing (e.g., shredding and drying) is handled within the gasification facility. The gasification system includes carbon capture and storage (CCS) technology, and captured CO_2 is transported and stored in a saline aquifer.

The business-as-usual pathway for H_2 production via gasification was assumed to be steam methane reforming (SMR) of natural gas [41]. Due to the costs of H_2 production and the incentives for clean H_2 (e.g., U.S. Tax Code §45 V, credit for production of clean hydrogen [43]), the business-as-usual production pathway is likely not the competitor of future MSW gasification technologies. Therefore, along with SMR, the results are compared to a second “green” benchmark technology using proton exchange membrane (PEM) electrolysis with 100% wind-generated electricity.

Landfilling was chosen as the avoided business-as-usual pathway for MSW. As a ubiquitous solid waste, the majority of MSW is managed long-term by landfills [3]—large excavations of land where waste is buried and undergoes continuous monitoring, maintenance, and upkeep [1]—which have their own environmental challenges (e.g., methane, dioxin, and leachate emissions into the air, soil, and water) [44]. Landfills produce a

“massive amount” of methane that accounts for 14% of global emissions [45], highlighting landfill gas (i.e., biogas) as an underutilized, large energy resource [46]. As a greenhouse gas and volatile organic compounds mitigation strategy, most landfills today employ methane-capturing to some extent, the fate of which is often flaring (i.e., emitted as CO₂, a chemical with a lower global warming potential) or use in electricity generation and methane-derived fuels [32]. Other mitigation strategies (e.g., recycling and composting) are used to divert and reduce the amount of material that ends up in landfills [47].

2.2.2. Life Cycle Modeling Software and Impact Assessment Method

Modeling Software. The free and open-source LCA modeling platform, openLCA version 2.2.0, developed by GreenDelta, was used to model the different scenarios in this case study. A unit process was developed for the base case, which was used as the foundation for the two accounting cases (i.e., biogenic carbon and extended burden). The U.S. Life Cycle Inventory Database provided many of the background flows and processes and was incorporated into openLCA through the electricity baseline library [48]. A copy of the openLCA database and a reference workbook are available (see data availability section).

Impact Assessment Method. While impact assessments across a broad set of categories that help to ensure tradeoffs between environmental impacts should be considered when proposing or developing a new system or product, the scope of the works may lead to the narrowing of impact categories to just one or two. In this example, the goal is to assess the net greenhouse gas emissions using global warming potential, which normalizes greenhouse gas emissions to their CO₂ equivalent. By focusing on global warming potential (GWP) as the sole impact category, the climate change mitigation aspect of the waste-to-resource pathways is isolated, helping to highlight any differences in emissions profiles between the various accounting methods.

In the United States, TRACI 2.2 is the standard impact assessment method, developed by the U.S. Environmental Protection Agency (EPA) [49]. TRACI’s GWP characterization factors align with the latest international substance hierarchy published by the Intergovernmental Panel on Climate Change (IPCC) [50]; however, other time horizons, climate feedback, and amendments to the characterization factors may be investigated, as published by the IPCC (e.g., as in Assessment Reports 4, 5, and 6). In this study, the characterization factor is based on the 100-year GWP from the IPCC’s Assessment Report 6 (AR6) [51]. For convenience and comparison, Table 1 provides the published 100-year GWP characterization factors for select greenhouse gases as found in Assessment Reports 4, 5, and 6 (AR4, AR5, and AR6, respectively).

Table 1. Atmospheric chemical emission characterization factors for GWP.

Greenhouse Gas Emission (to Atmosphere)	Characterization Factor (100-Year)		
	AR4 2007	AR5 2014	AR6 2021
Carbon dioxide	1	1	1
Methane	25	36	30
Nitrous oxide	298	265	273
Sulfur hexafluoride	22,800	26,087	25,200

2.2.3. Case Model Descriptions

The following five cases were modeled:

1. Base case (i.e., the zero-burden assumption)
2. Biogenic (i.e., neutrality assumption)

3. Burdened (i.e., shared burden with paper, wood, and plastic production)
4. Business-as-usual benchmark (i.e., SMR with electricity generation; paper, wood, and plastic production and MSW landfilling)
5. Counterfactual benchmark (i.e., PEM with electricity generation; paper, wood, and plastic production and MSW landfilling)

Base Case. The proposed system is a two-stage, fluidized-bed, oxygen-blown gasifier system with a steady state feedrate of 181,441 kg/day of sorted, shredded, and dried MSW (comprised of 80% paper, 10% plastic, and 10% wood) and a production rate of 9693 kg of H₂ and 15.8 MWh of grid electricity daily (i.e., 1.63 kWh of exported electricity per kg of H₂ produced). The water consumption for the gasification system is 620,476 kg/day, half of which is labeled raw water (taken from the environment) and the other half of which is labeled as treated water (drawn from municipal drinking water source). The gasifier system emits 32,380 kg/day of CO₂ from its stack while capturing 228,613 kg/day as CO₂ product for sequestration, with a carbon capture rate of 87.6% (i.e., 62,380 kg/day carbon [C] captured per 71,949 kg/day C into the process, with 718 kg/day C in the ash); note that, to simplify the modeling and include the environmental impacts of the CCS technology, the CO₂ captured product is listed as an input (i.e., internal flow). The gasifier system also uses small amounts of sodium hydroxide and sulfuric acid for gas cleaning and scrubbing to remove contaminants from the syngas. The unit process scales input and output flows on a per-kilogram-of-H₂-product basis (>99.9% pure by volume at 6.38 MPa [925 psig]). The input and output flows are provided in Table 2, along with the names of the process providers.

Table 2. Input and output flow table for MSW gasification; note that “carbon dioxide captured product”, while technically a product, is listed as input in openLCA to analyze its emissions during impact analysis.

Input Flow	Amount	Unit (per kg H ₂)	Provider
Processed (sorted + dried) MSW	18.7	kg	None (included with gasifier)
Sodium hydroxide	0.34	kg	Sodium hydroxide; chlor-alkali average, membrane cell; at plant; 50% solution state [52]
Sulfuric acid	6.0×10^{-3}	kg	Sulfuric acid, at plant—RNA [53]
Water, raw	32.0	kg	Not modeled (elementary flow)
Water, treated by cation exchange	32.0	kg	Cation exchange [54]
Carbon dioxide, captured product	23.6	kg	Saline aquifer transport and storage—U.S. [55]
Output Flow	Amount	Unit (per kg H ₂)	Comment
Hydrogen (product)	1	kg	>99.9 vol%, 6.38 MPa
Ash	1.93	kg	Ash has 3.8 wt% carbon content
Electricity (co-product)	1.63	kWh	345 kV, alternating current (AC)
Carbon dioxide	3.34	kg	Emission to air
Water	13.7	kg	Emission to water

The additional electricity produced by the steady-state operation of the MSW-to-H₂ gasification system is assumed to go back to the electricity grid. Co-product management followed ISO guidance [16] and was allocated based on an underlying physical relationship, as electricity and H₂ may both be represented in units of energy. A higher heating value of 38.7 kWh per kg of H₂ was used to convert H₂ from mass to energy. The ratio of energy output (i.e., 1.63 kWh of electricity to 38.7 kWh of H₂) resulted in an allocation of 4% to

electricity product and 96% to H₂ product. An attributional LCA is conducted for a cradle-to-gate analysis that includes electricity generation, feedstock processing, gasification, and CCS (see Figure 1).

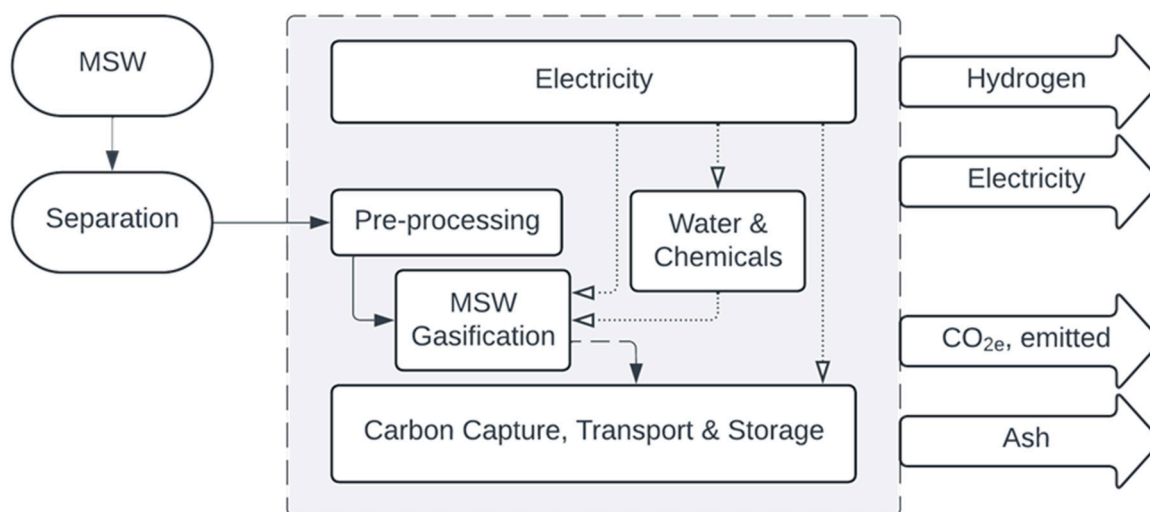


Figure 1. System boundary for the base case (gray box); MSW feedstock and separation, from a material recovery facility, are outside the system boundary; electricity generation; feedstock pre-processing and gasification; carbon capture, transport, and storage to a saline aquifer; and the chemicals and water used for gasification are modeled; co-products include H₂ and excess electricity. The solid line with solid arrow represents the feedstock flow through separation, pre-processing and gasification. The dotted line with empty arrow represents the energy and material flows (i.e., water, and chemicals) found within the system boundary. The dashed line with a solid arrow is the captured carbon from feedstock gasification. The following figures are the same.

Data providers. The CCS technology is modeled based on the gate-to-gate LCA by the National Energy Technology Laboratory (NETL) on saline aquifer sequestration of CO₂ [55,56]. The treated water supply chain is modeled by a cation exchange process based on a unit process for drinking water [54]. The supply chains for sodium hydroxide and sulfuric acid were obtained from the Federal LCA Commons from a repository provided by the National Renewable Energy Laboratory.

Biogenic Case. In this study, the feedstock comprises three primary MSW constituents: paper, wood, and plastic. The former two (i.e., paper and wood) are naturally derived from biomass (e.g., carbon absorbed within trees). According to EPA, wood in MSW comprises mainly pallets, furniture, and other durable goods [57]. Both paper and wood feedstocks are biogenic carbon sources.

Plastic, on the other hand, comprises carbon-based polymers formed by various chemical reactions on fossil-based feedstocks; therefore, plastic is a non-biogenic carbon source. Using the EIA method, the MSW feedstock may be separated into its potential biogenic carbon and non-biogenic carbon stocks using the energy allocation method. The total energy concentration, E_{total} , may be calculated as shown in Equation (1) based on the total weight of the MSW stream, W , multiplied by the waste stream fractions, F , for paper, wood and plastic multiplied by their respective heating values, H , found in [33] (Table 4), also shown in Equations (2) and (3).

$$E_{total} = \sum_{i=paper, wood, plastic} W \times F_i \times H_i \quad (1)$$

For this purpose, low-density polyethylene (LDPE) is the assumed plastic waste (i.e., high generation rate and low recycling rate), and mixed paper is the assumed paper

waste (due to the high recovery rate of newsprint and cardboard). For a base scenario of MSW comprised of 80% paper, 10% wood, and 10% plastic (by weight), the percentage of biogenic energy and non-biogenic energy is given by their respective fractions:

$$E_{bio} = \frac{0.8 \times 6.7 + 0.1 \times 10}{0.8 \times 6.7 + 0.1 \times 10 + 0.1 \times 24.1} = 72.5\% \quad (2)$$

$$E_{non-bio} = \frac{0.1 \times 24.1}{0.8 \times 6.7 + 0.1 \times 10 + 0.1 \times 24.1} = 27.5\% \quad (3)$$

In order to track the biogenic and non-biogenic carbon through the life cycle, a simplified carbon budget is used to account for the carbon, where the total carbon of the system, C_{total} , is provided by the feedstock (i.e., $C_{total} = C_{paper} + C_{wood} + C_{plastic}$), as shown in Figure 2.

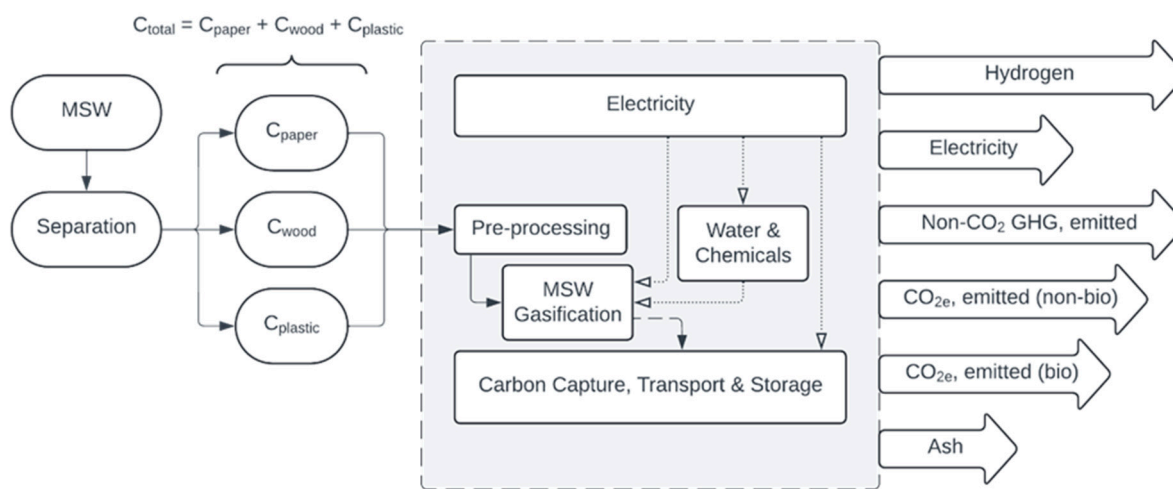


Figure 2. Biogenic carbon accounting for MSW feedstock under the base scenario.

The carbon within each feedstock material is assumed to be proportional to the energy fraction based on the EIA allocation method described above. Note that this accounting method does not include embodied carbon (i.e., the carbon accounting that includes all carbon emitted through the direct and indirect processes for the cradle-to-gate of a product or service [58,59]). Other carbon emissions from upstream processes (e.g., electricity generation) are not considered. It is assumed that the three emission compartments (i.e., CO₂ captured, CO₂ emitted, and ash) share equal amounts of both biogenic and non-biogenic carbon derived from the feedstocks.

The biogenic proportion of the CO₂ captured product is an amount of sequestered carbon taken from the natural carbon cycle and acts as a carbon sink, whereas the sequestered, non-biogenic carbon is accounted as a reduction in anthropogenic emissions that otherwise would have been emitted. Therefore, only the biogenic portion of the captured and stored CO₂ product contributes to carbon sequestration.

Data providers: Most landfill composition data are industry secrets; therefore, this study reviewed the publicly available data provided for Austin, Texas, one of 13 U.S. benchmark cities participating in the Zero Waste Goal [47] (Table 5-4, p. 23), along with a composition analysis of MSW in Texas, prepared by Burns McDonnell for the Texas Commission on Environmental Quality [60] (Table 4-2, p. 4-3). These studies showed MSW compositions that included large proportion of organics (31%), paper (28%), and plastics (15%). When sorting is applied, the result is assumed to be a feedstock including 80% paper, 10% plastic, and 10% wood waste. Allowing that MSW composition is reflective of the economic activity, consumption habits, and growth and density of the local population [61],

differences and uncertainties in reported composition mixes of MSW is examined using a sensitivity analysis.

Burdened Case. Because it is not feasible to account for each item in MSW that contributes to mixed paper, waste plastic, or wood, surrogate products are used instead to represent these items. A surrogate (or proxy) is commonly used to bridge data gaps in LCA either through their direct application or via scaling, averaging, or extrapolating existing data to sufficiently represent the unknown dataset [62]. The application of an existing unit process, such as paper, provides the direct application for the cradle-to-grave life cycle inventory for the upstream burden of a MSW material (e.g., mixed paper). The proxy materials used in this study are given in Table 3. To account for these upstream emissions, the base scenario's system boundary is extended to include raw material acquisition, manufacturing, and disposal (e.g., collection and transportation to a separation facility), as shown in Figure 3.

Table 3. Proxy products used for upstream burdening of MSW feedstocks.

MSW Feedstock Material	Proxy Product	Data Source
Paper	Magazine/third-class mail	WARM 2023
Plastic	LDPE resin	WARM 2023
Wood	Dimensional lumber	WARM 2023

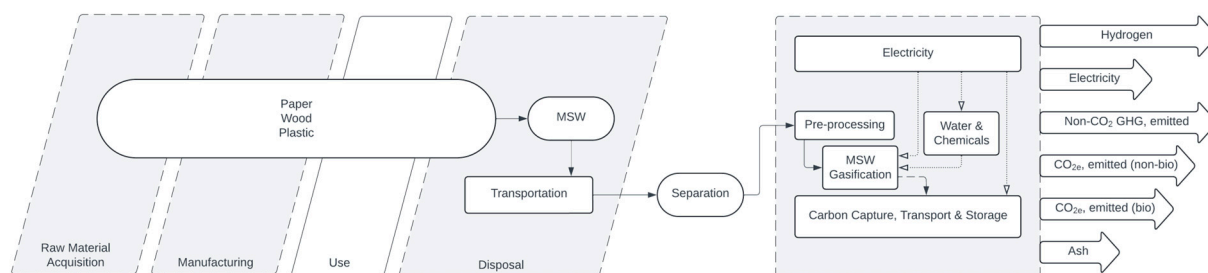


Figure 3. Extended system boundary for burdened scenario (gray boxes), including the raw material acquisition, manufacturing, and disposal of representational products for the three main MSW feedstock materials.

Data providers. The MSW proxy processes for the MSW feedstock were taken from EPA's Waste Reduction Model (WARM). Based on the Greenhouse gases, Regulated Emission, and Energy use in Technologies (GREET) model [63], magazines, office paper, and phone books are assumed to be 100% landfilled, which makes them prime candidates for gasification. The GREET model data represent U.S. average MSW and, as a result, do not include the cradle-to-gate emissions associated with the waste products. EPA's WARM notes that the production emission factors for raw material acquisition and manufacturing of magazines, office paper, and phone books are 8.86, 8.23, and 6.17 metric tons of CO_{2e} per short ton of product produced, respectively [64] (Exhibit 6-1). For this study, paper waste is proxied based on the WARM 2023 production emission factor for magazines/third-class mail. The WARM v16 [65] result is on the same order of magnitude but about four times higher compared to an estimate for the life cycle carbon footprint of the National Geographic magazine, which was reported by [66] to be between 2.36 and 2.38 metric tons of CO_{2e} per metric ton of magazine produced (2.14–2.16 metric tons of CO_{2e} per short ton of magazine produced).

LDPE, associated with single-use shopping bags and shrink wrap, was selected over polyethylene terephthalate (PET), commonly associated with plastic water and soda bottles, given that PET is highly recyclable (and therefore less likely of a candidate for gasification)

and LDPE has a high generation and relatively low recycling rate [67]. Other low-recyclable plastics (e.g., polyvinyl chloride and polystyrene) have lower generation rates, while polypropylene (PP) was another possible contender for a proxy material. LDPE was ultimately chosen over PP, given that various studies have demonstrated successful H₂ gasification using a combination of biomass and polyethylene (see [68]). WARM v16 reports the production emission for LDPE to be 1.80 metric tons of CO_{2e} per short ton produced [64] (Exhibit 6-1). Compare this to the 2.20 metric tons of CO_{2e} per metric ton of LDPE resin produced (2.00 metric tons of CO_{2e} per short ton), as presented by [69].

Wood pallets were chosen as the proxy material for the wood MSW stream. The average U.S. pallet, based on [70], was found to have a cradle-to-grave impact of 10.4 kg CO_{2e} per 45.4 metric tons (100,000 lbs.) of pallet loads of product delivered or 0.26 metric tons of CO_{2e} per metric ton of pallet (using the national average of 2.13 pallets to transport 45.4 metric tons of product and average moist mass of 18.57 kg per pallet). This is about a factor of ten less when compared to the WARM v16 production emission factors for dimensional lumber and medium-density fiberboard—2.34 and 3.42 metric tons of CO_{2e} per metric ton of product [64] (Exhibit 6-1), respectively—suggesting a value closer to the raw material acquisition and manufacturing emission factor of dimensional lumber (i.e., 0.19 metric tons CO_{2e} per metric ton) that excludes forest carbon storage [71] (Exhibit 12-5). For consistency with system boundaries amongst proxy products, the dimensional lumber emission factor from WARM is used in this case study, which was found to be a “very good” proxy for clean wood and pallets [72] (Appendix A).

To remove the impacts of transportation and landfilling (because the MSW in this scenario is re-routed to the gasification site), the amount of 0.02 metric tons of CO_{2e} is subtracted from each proxy product’s emission factor [64] (Exhibit 5-5).

Business-as-Usual Benchmark Case. The SMR unit process converts 3.75 kg of natural gas, 30.56 kg of water, and 2.03 kWh of electricity into 1 kg of H₂ product and 9.56 kg of sequestered carbon. Additional emissions include CO₂, nitrogen, and water. The main inputs and outputs are shown in Table 4.

Table 4. The input and output flows for the SMR unit process; note that “carbon dioxide captured product”, while technically a product, is listed as input in openLCA to analyze its emissions during impact analysis.

Input Flow	Amount	Unit (per kg H ₂)	Provider
Natural gas, delivered	3.75	kg	Natural gas, delivery—U.S. [73]
Electricity, AC, 120 V	2.03	kWh	Electricity; at user; consumption mix—ERCOT—2030 [74,75]
Water, raw	29	kg	Not modeled (elementary flow)
Water, treated by cation exchange	1.56	kg	Cation exchange [54]
Carbon dioxide, captured product	9.56	kg	Saline aquifer transport and storage—U.S. [73]
Output Flow	Amount	Unit (per kg H ₂)	Comment
Hydrogen (product)	1	kg	>99.9%, 6.38 MPa
Carbon dioxide	0.38	kg	Emission to air
Nitrogen	22.3	kg	Emission to air
Nitrogen oxides	1.9×10^{-4}	kg	Emission to air
Water	6.5	kg	Emission to water
Water	0.56	kg	Emission to air

The SMR unit process was designed with a multi-functioning functional unit via system expansion. For comparability, the business-as-usual case produces all the products from the base case (i.e., electricity and H₂) and the products that ultimately become the feedstock (i.e., the paper, wood, and plastic).

Landfilling was chosen as the avoided business-as-usual pathway for MSW. To represent this life cycle stage, a landfilling unit process was created. Paper and wood products, once disposed, produce CO₂ and methane emissions. It is assumed that the CO₂ emitted from landfills within the United States is sourced from sustainable wood harvesting; therefore, it is labeled as biogenic and does not contribute to GWP (i.e., the neutrality assumption). This does not apply to methane emissions. Because not all carbon within paper and wood is decomposed, landfills also serve as permanent carbon sinks. As most landfills in the United States employ methane capture technologies, a fraction of landfill methane emissions from paper and wood MSW are routed to energy production and the remainder vented into the atmosphere.

Paper, wood, and plastic production are modeled identically with the extended burden case. Figure 4 shows the processes for a business-as-usual benchmark case (i.e., Figure 4a–c).

Data providers. The SMR unit process is provided by NETL with a functional unit of 1 kg H₂ that includes carbon capture technology [76]. The natural gas fuel supply chain is modeled using a product system that encompasses the life cycle of natural gas from raw material extraction (i.e., well drilling) to processing and distribution [73]. The same CO₂ transport and storage unit process from the base case was used in the SMR benchmark process. The electricity generation, distribution, and user consumption data are modeled for the Electricity Reliability Council of Texas (ERCOT) balancing authority using a 2020 updated life cycle inventory of the electricity baseline provided by NETL. To account for the future development of the proposed system, the generation mix was modified to reflect the 2030 forecast mix based on the 2023 reference case published by the EIA's Annual Energy Outlook for ERCOT. Data were extracted from Table 54, "Electric Power Projections by Electricity Market Module Region", and Table 56, "Renewable Energy Generation by Fuel", using the provided application program interface [75]. The mix percentages used are given in Table 5. To accommodate the co-production of electricity from the MSW gasification facility, the same ERCOT 2030 electricity generation mix process is used in the expanded system for the business-as-usual process for electricity production.

Also in the expanded system, the paper, wood, and plastic production data were modeled identically to the extended burden case. To determine the environmental impacts of the MSW in landfills, the CO_{2e} emissions are based on reported values from EPA's WARM v16. For plastic, the emission factor is 0.022 kg of CO_{2e} per kilogram of waste, which reflects the diesel emissions from transportation and landfilling of waste [64] (Exhibit 5-5) [77] (Exhibit 5-19). The net landfilling emission factors for paper and wood are −0.48 and −1.03 kg CO_{2e} per kilogram of waste, respectively, based on proxy products and acting as carbon sinks despite methane production (see "Magazines/Third-Class Mail", Exhibit 3-26 [77] and "Dimensional Lumber", Exhibit 12-15 [71]).

Counterfactual Benchmark Case. The PEM unit process converts 172.1 kg of water and 55 kWh of electricity into 1 kg of H₂ product and discharges 160 kg of water back into the environment. There is no CO₂ product to be captured, transported, and stored in the 100% wind-powered PEM electrolysis pathway, and the water treatment requirements for PEM are more stringent (i.e., reverse osmosis compared to cation exchange for SMR). The main input and output flows are provided in Table 6.

The PEM unit process was designed with a multi-functioning functional unit via system expansion and uses the same processes for electricity generation; MSW landfilling;

and paper, plastic, and wood production, as used in the business-as-usual case. Figure 4 shows the processes for the counterfactual benchmark case (i.e., Figure 4a,b,d).

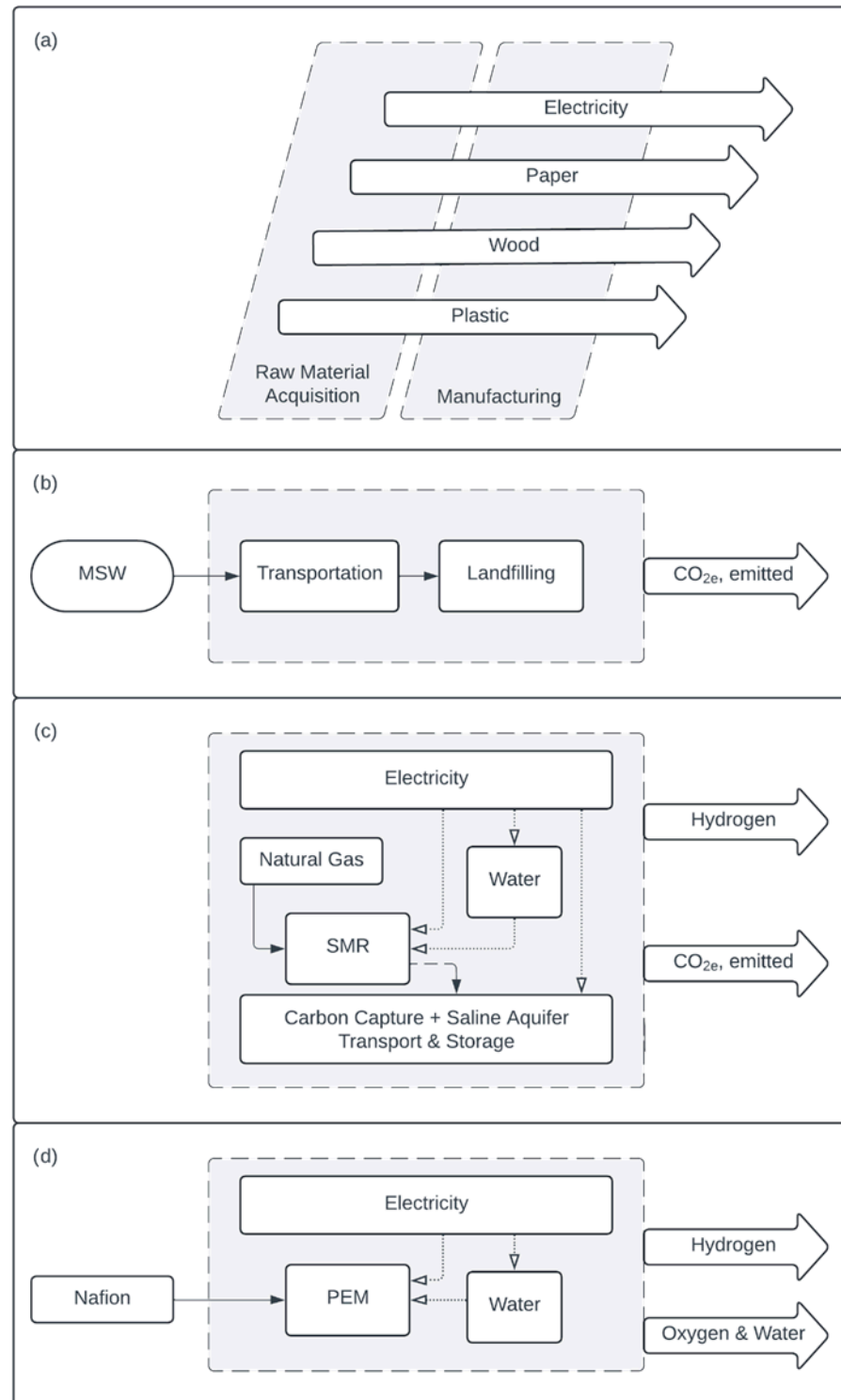


Figure 4. The benchmark system processes for (a) paper, wood, plastic, and electricity production; (b) MSW management via landfiling; (c) H_2 production via SMR of natural gas with carbon capture, transport, and storage; and (d) the "green" alternative H_2 production via PEM electrolysis.

Data providers. The PEM electrolysis unit process is modeled after [78]. The reverse osmosis unit process is based on an EPA drinking water treatment model [54]. The electricity consumption unit process for PEM is identical to that used for the SMR unit process, except

that the mix reflects 100% wind-based generation. No data were provided for Nafion™ membrane production; however, an examination of the GWP impact associated with the production of tetrafluoroethylene (TFE)—a proxy for Nafion—revealed that this process was likely insignificant for impact assessment (i.e., 1.4×10^{-3} kg CO_{2e}/kg H₂) and therefore was cut off from the upstream analysis.

Table 5. Electricity forecast mix for ERCOT 2030 generation.

Fuel Category	Mix Percentage (Relative to Projected Future Generation of All Fuels)
Natural gas	30.9
Wind	27.6
Solar photovoltaic	25.9
Nuclear	9.73
Coal	5.57
Hydroelectric	0.25
Biomass	0.03
Petroleum	0.02

Table 6. The input and output flows for the PEM unit process.

Input Flow	Amount	Unit (per kg H ₂)	Provider
Electricity, AC, 120 V	55.0	kWh	Electricity; at user; consumption mix—ERCOT—2030—wind [74,75]
Water, raw	163.2	kg	Not modeled (elementary flow)
Water, treated by reverse osmosis	8.9	kg	Reverse osmosis [54]
Nafion membrane	1.4×10^{-5}	kg	Not modeled (cutoff)
Output Flow	Amount	Unit (per kg H ₂)	Comment
Hydrogen (product)	1	kg	>99.9%, 435 psig, 60 °C
Water	160.0	kg	Emission to water

3. Results

3.1. Sensitivity to Feedstock Composition and Functional Unit Perspective

A sensitivity analysis was performed on the four cases using an output-based functional unit (i.e., 1 kg of H₂ produced) by increasing the percentage of wood and plastic in the MSW composition from the base case. Two alternative scenarios were examined: (1) 70% paper, 15% wood, and 15% plastic and (2) 60% paper, 20% wood, and 20% plastic.

The total GWP of these cases and scenarios (in kg of CO_{2e} per kg of H₂ produced) is provided in Table 7. A secondary total column (Total *) is computed for comparison purposes to demonstrate cases without biogenic carbon accounting (i.e., with non-zero biogenic carbon emissions and without biogenic carbon sequestration), without electricity co-production, and without landfilling impacts.

Table 7. The GWP (in kg CO_{2e} per kg of H₂ produced) for three scenarios and five cases based on IPCC’s AR6 100-year characterization factors. The benchmark case includes two pathways: BU (business-as-usual, SMR) and CF (counterfactual, PEM), both with paper, wood, and plastic production and landfilling and electricity generation. The fourth scenario examines the 80:10:10 scenario from an input functional unit perspective. The scenario percentages are for the MSW feedstock composition (i.e., paper:wood:plastic).

Case	Paper Product	Wood Product	Plastic Product	NaOH Product	H ₂ SO ₄ Product	Elec. Use	Natural Gas Use	Water Use	CC&T	Bio. C Storage	MSW Gasification (bio)	MSW Gasification (fossil)	SMR/MSW Gasification	Elec. Product	Landfill	Total *	Total
80:10:10 Output Scenario																	
Base				0.26	2.1×10^{-3}			0.02	0.33				3.21				3.82
Biogenic				0.26	2.1×10^{-3}			0.02	0.33	−16.4	2.32	0.88					−14.9
Burdened	142	4.20	3.58	0.26	2.1×10^{-3}			0.02	0.33	−17.1	2.32	0.88		0.16		154	135
BU (SMR)	142	4.20	3.58			0.61	3.06	1.0×10^{-3}	0.14				0.38	0.47	−9.10	154	146
CF (PEM)	142	4.20	3.58			1.49		4.2×10^{-3}						0.47	−9.10	152	142
70:15:15 Output Scenario																	
Biogenic				0.26	2.1×10^{-3}			0.02	0.33	−14.3	2.02	1.18				3.82	−12.5
Burdened	124	6.31	5.37	0.26	2.1×10^{-3}			0.02	0.33	−14.9	2.02	1.18		0.16		140	123
BU (SMR)	124	6.31	5.37			0.61	3.06	1.0×10^{-3}	0.14				0.38	0.47	−9.14	140	132
CF (PEM)	124	6.31	5.37			1.49		4.2×10^{-3}						0.47	−9.14	138	129
60:20:20 Output Scenario																	
Biogenic				0.26	2.1×10^{-3}			0.02	0.33	−12.6	1.78	1.43				3.82	−10.5
Burdened	107	8.41	7.16	0.26	2.1×10^{-3}			0.02	0.33	−13.1	1.78	1.43		0.16		126	111
BU (SMR)	107	8.41	7.16			0.61	3.06	1.0×10^{-3}	0.14				0.38	0.47	−9.18	126	118
CF (PEM)	107	8.41	7.16			1.49		4.2×10^{-3}						0.47	−9.18	124	115
80:10:10 Input Scenario (kg CO _{2e} /kg MSW managed)																	
Base				0.01	1.2×10^{-4}			1.1×10^{-3}	0.02				0.18				0.21
Biogenic				0.01				1.1×10^{-3}	0.02	−0.91	0.13	0.05				0.21	−0.83
Burdened	7.92	0.23	0.20	0.01				1.1×10^{-3}	0.02	−0.91	0.13	0.05		0.01		8.57	7.53
BU (SMR)	7.92	0.23	0.20			0.03	0.16	5.4×10^{-5}	7.5×10^{-3}				0.02	0.03	−0.49	8.58	8.12
CF (PEM)	7.92	0.23	0.20			0.08		2.3×10^{-4}						0.03	−0.49	8.43	7.97

* Totals without biogenic carbon accounting (e.g., in biogenic and burdened cases), without electricity production (e.g., in burdened and benchmark cases), and without landfilling (e.g., in benchmark case) impacts.

The landfill column includes the transportation, landfilling, methane emission (and capture), and carbon storage of the three proxy products (i.e., paper, wood, and plastic). Landfill impacts are only shown for benchmark processes. Negative emissions indicate that landfilling, as an end-of-life process, acts as a carbon sink.

The electricity product column reflects the impacts associated with the excess electricity produced by the MSW gasifier. For comparison purposes, the burdened case includes the impacts of producing excess electricity, which reflects ~4% of the impacts based on the selected method of physical allocation. The same amount of electricity is produced using the regional electricity generation mix in the benchmark case.

In the base and benchmark cases, MSW gasification and SMR emissions are unlabeled; however, in the biogenic carbon and extended burden accounting cases, the MSW gasifier emissions are separated into columns for biogenic and fossil-based contributions, hence three columns for communicating the MSW gasification/SMR emissions. The PEM electrolyzer has no direct emissions.

Carbon capture and transport (CC&T) impacts account for emissions associated with CCS technology and piping the CO₂ to a saline aquifer. The biogenic carbon storage column (Bio. C Storage) is separated for the biogenic carbon and extended burdened cases to indicate the impacts associated with biogenic carbon sequestration. Due to the physical allocation of impacts to the H₂ product, a smaller impact of sequestration is associated with the biogenic carbon case compared to the extended burden case, which includes sequestering biogenic carbon associated with both the H₂ and electricity products.

The water use column reflects the upstream impacts associated with treated water, both in terms of quantity and purity (e.g., cation exchange versus reverse osmosis).

The natural gas use column reflects the contributions associated with the SMR feedstock. Note that natural gas is found in other background processes (e.g., in sodium hydroxide and sulfuric acid production and in concrete production for reverse osmosis); however, these contributions are included in their respective process categories.

The electricity use column represents the indirect emissions associated with electricity generation and distribution based on the grid demand of the H₂ production process. Note that electricity contributions to other upstream processes (e.g., water treatment) are included in their respective process categories.

The sodium hydroxide and sulfuric acid product columns represent the raw material acquisition and manufacturing of these chemicals for MSW gasification.

The burdened case includes life cycle impacts associated with the raw material acquisition and manufacturing of three proxy products for paper, wood, and plastic (i.e., magazines/third-class mail, dimensional lumber, and LDPE plastic). The benchmark processes also include the production of paper, wood, and plastic.

The first scenario (i.e., 80:10:10) was reproduced with the functional unit changed from 1 kg of H₂ produced (output-based) to 1 kg of MSW managed (input-based). The definitions of the categories are the same as in the output scenario. Note that there is no physical allocation of impacts when considering MSW managed as the functional unit; therefore, the biogenic carbon sequestered is the same in both the biogenic carbon and extended burden cases. The results of this scenario are also found in Table 7.

3.2. Biogenic Carbon Sensitivity to Feedstock Composition

The total GWP for the base case is 3.82 kg CO_{2e} per kg of H₂ produced (Table 7). When biogenic carbon accounting is applied, 90% of the feedstock is organic (i.e., paper or wood) and contributes to biogenic CO₂. The two scenarios reduce the overall biogenic contribution to 85% (70:15:15 scenario) and 80% (60:20:20 scenario). As the contribution of organic feedstock is reduced (e.g., from 90% in the 80:10:10 scenario to 80% in the

60:20:20 scenario), so too is the proportion of the biogenic carbon emission reduced, which is reflected in the decreasing green bar in Figure 5. The hashed green bars indicate that, under biogenic carbon neutrality, a significant amount of carbon is sequestered (ranging from -12.6 kg CO_{2e} to -16.4 kg CO_{2e}).

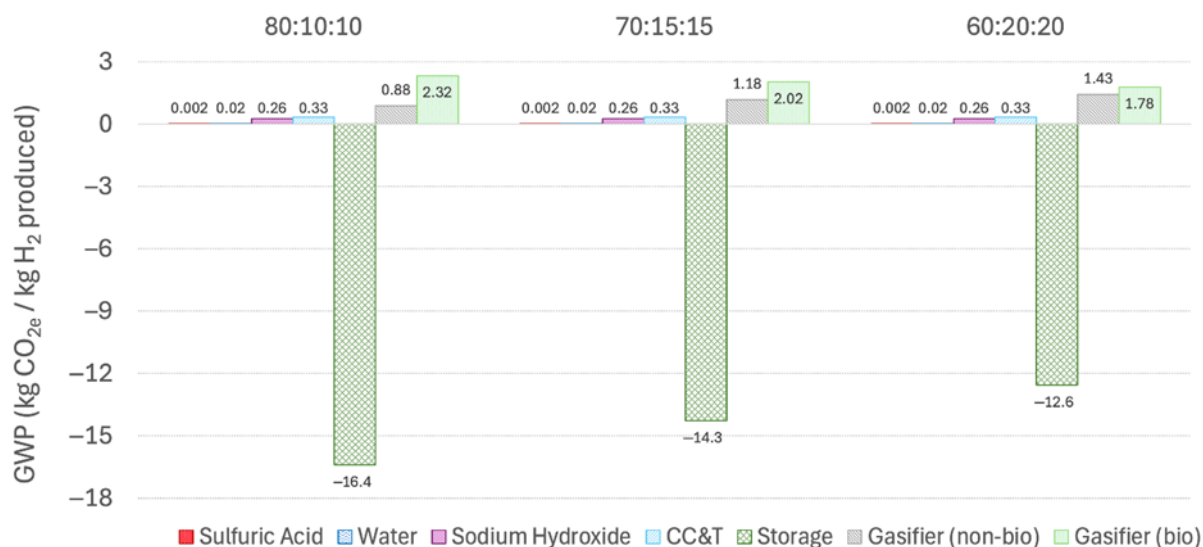


Figure 5. Contributions towards GWP for MSW gasification including sulfuric acid, water use, sodium hydroxide, CC&T, biogenic carbon storage, and gasifier emissions (non-biogenic and biogenic) across three MSW feedstock scenarios: 80% paper, 10% wood, and 10% plastic (80:10:10); 70% paper, 15% wood, and 15% plastic (70:15:15); and 60% paper, 20% wood, and 20% plastic (60:20:20).

3.3. Benchmark Sensitivity to Paper Under Extended Burden

Under the 80:10:10 scenario, the paper production and paper landfilling emission factors were analyzed under high and low values using newspaper and textbooks as proxies. The emission factors for newspapers and textbooks are used to calculate the total GWP impacts of the extended burden case with biogenic carbon accounting and the two benchmarks, SMR and PEM. The results are shown in Figure 6.

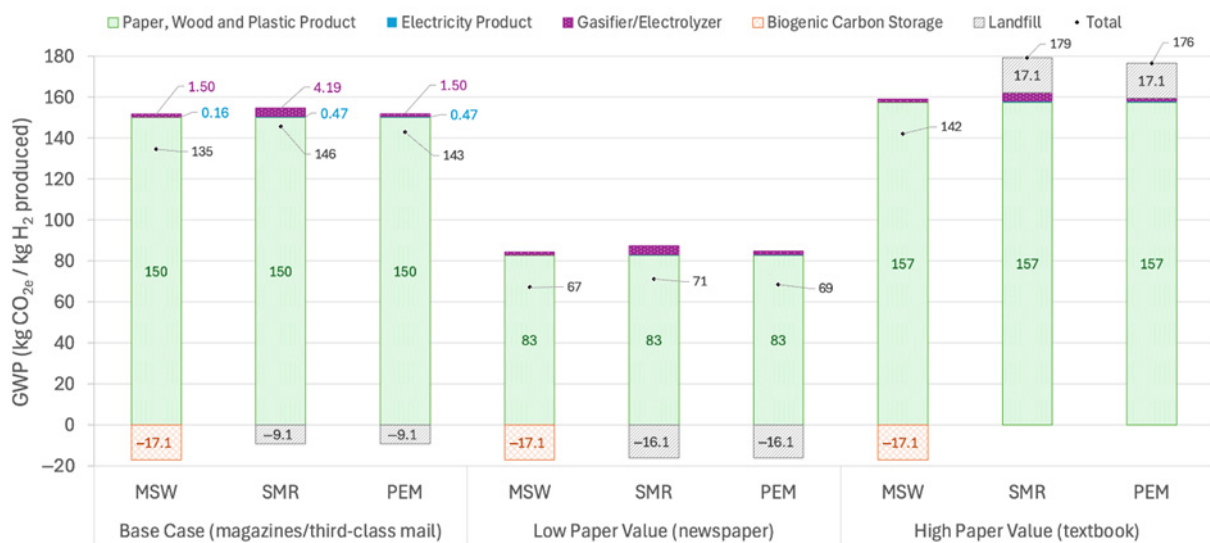


Figure 6. The GWP sensitivity to paper production and paper landfilling emissions under the base case (magazines/third-class mail), low paper (newspaper), and high paper (textbook) emission values; GWP contributions are broken down by raw material acquisition and manufacturing of paper, wood, and plastic products; electricity production; gasification or electrolysis (e.g., for MSW gasifier, SMR, and PEM electrolyzer); biogenic carbon storage; and landfilling.

The MSW gasification, SMR, and PEM electrolyzer GWP contributions (shaded purple regions) are consistent across the three comparisons in Figure 6 (1.50, 4.19, and 1.50 kg CO_{2e} per kg H₂ produced, respectively), and the same is for the contributions of electricity production (0.16, 0.47, and 0.47 kg CO_{2e} per kg H₂ produced, respectively). The choice of paper influences both the product emissions (83–157 kg CO_{2e}/kg H₂) and the landfilling emissions (−16.1–17.1 kg CO_{2e}/kg H₂). Constant biogenic carbon storage is assumed regardless of paper selection, as reflected in the 80:10:10 scenario for the burdened case in Table 7, because the higher heating value of paper is assumed unchanged.

For the lower paper value, the avoided emissions are between 2 and 6%, making the choice nearly identical between the MSW base case and the PEM. Under the high paper value, the landfilling emission impacts of the benchmarks are nearly equal to the biogenic carbon storage of the MSW base case. For the high paper value, with 20–21% avoided emissions, the decision between the MSW base case and either SMR or PEM becomes clearer, demonstrating the importance of the proxy process selection and its impact on various life cycle stages.

4. Discussion

4.1. Biogenic Carbon Accounting

Figure 5 shows the emissions associated with the proposed system by contribution. The green solid and green hashed bars represent the impacts of biogenic carbon accounting. The solid green bar is the emissions associated with biogenic carbon, which, under the neutrality assumption, may have zero GWP impact. The green hashed bar represents the contribution of biogenic carbon captured and stored in a saline aquifer (i.e., carbon sequestration from the natural carbon cycle). Zeroing the biogenic carbon emissions and allowing for biogenic carbon sequestration, in Figure 5, the GWP impacts of MSW gasification across the three scenarios, starting with the highest contribution of biogenic carbon, are −14.9, −12.5, and −10.5 kg CO_{2e} per kg H₂ produced or GWP reductions of 490, 427, and 376% from the unallocated base case, respectively.

The contributions of sequestering biogenic carbon in a saline aquifer are enough to make biogenic carbon accounting net negative; however, this does not account for the carbon that is put into the products that become the gasifier feedstock, making this accounting method an overestimate of reality (i.e., implementing a 0/−1 approach that differs from both the 0/0 and −1/+1 carbon neutrality approaches). In an attributional sense, waste-to-energy pathways provide an alternative end of life for their products. The extended burden accounting looks to correct the overestimation and avoid possible double counting (or miscounting) in the LCAs of original products.

4.2. Extended Burden Accounting

For MSW gasification to H₂, there are three potential emission sources that can be avoided. The first is the fuel source. H₂ is predominantly manufactured using fossil-derived fuels (i.e., natural gas). Therefore, use of a less carbon-intensive fuel (i.e., MSW in place of natural gas) is a candidate for emission avoidance. A second candidate lies in the excess electricity produced during MSW gasification, which could offset fossil-based electricity generation, thereby reducing overall emissions. Lastly is the alternative fate of the biomass in MSW (i.e., waste diversion from landfills). Anaerobic decomposition of organic matter in landfills is responsible for generating biogas (i.e., methane), which has a high GWP and a business-as-usual fate of being vented, flared, converted into electricity, or converted into fuel [79]. Because most landfills routinely capture landfill gas, it may not be valid to account for all biogas generated from anaerobic decomposition of organics in MSW as avoided emissions [32].

The extended burden case uses proxy processes to represent the life cycle impacts of the raw material acquisition and manufacturing of the waste products used as feedstock. Using emission factors relative to manufacturing from virgin material provides an upper bound estimate of the potential impacts associated with these products. Given the large percentage of paper in all three scenarios, it is unsurprising that paper has the highest impacts among the three feedstocks. What is surprising is the relative magnitude of the upstream impacts when compared to the gasifier, which ranged from 122 to 150 kg CO_{2e}/kg H₂ produced (3200–3900% increase in GWP impacts compared to the base case with no biogenic carbon accounting or 2800–3400% increase with biogenic carbon accounting). This is caused by a combination of paper being the primary MSW feedstock and paper having a substantially higher emission factor (i.e., 9.9 kg CO_{2e} per kg of paper compared to 2.3 kg CO_{2e} per kg of wood and 2.0 kg CO_{2e} per kg of plastic).

Paper production, which is the main driver of GWP impacts (as compared to the relatively smaller impacts of wood and plastic production), varies in emission factor in WARM v16 based on the proxy product selection. Magazines and third-class mail represent the maximum emission for production from virgin material. The emission factor may be as low as 5.2 kg CO_{2e} per kg of mixed recycled newspaper and as high as 10.4 kg CO_{2e} per kg of textbooks from 100% virgin material [64] (Exhibit 6-1). Varying the emission factor of paper between these two extremes in the 80:10:10 scenario produces overall impacts that range from 67.3 to 142 kg CO_{2e} per kg of H₂ produced, assuming the neutrality and sequestration of biogenic carbon, as compared to 135 kg CO_{2e} per kg H₂ produced using magazines and third-class mail from 100% virgin material (as shown in Table 7). The choice of a paper feedstock proxy has a considerable influence on the impacts (i.e., 50–105% of GWP).

It is noteworthy that the impacts of the non-biogenic accounting of the extended burden case without electricity production are nearly identical to the SMR pathway without landfilling or electrical production (i.e., from the Total * column in Table 7) and only about 2 kg CO_{2e}/kg H₂ product greater than the PEM counterfactual with the same accounting.

4.3. Emission Avoidance Accounting

The proposed system of waste management to produce H₂ provides the opportunity to evaluate its performance against the benchmark processes for MSW management (e.g., landfilling), electricity generation, and H₂ production (i.e., SMR and PEM technology).

The CO₂ capture and transport impacts are the same for the base, biogenic, and extended burdened cases (as shown in Table 7). The SMR technology in the business-as-usual case produces less CO₂ product (9.56 kg compared to 23.56 kg in the base case); therefore, the GWP impact is less for this stage in the business-as-usual case. The PEM electrolyzer produces no CO₂; hence, there is no capture in the counterfactual case.

Biogenic carbon storage varies depending on the case. For the biogenic case, the impacts of sequestering CO₂ in a saline aquifer are allocated based on two separate factors: the biogenic carbon allocation factor (approximately 72%) and the H₂ product allocation factor (approximately 96%). In the extended burden case, with both H₂ and electricity products, the impacts of CO₂ sequestration are about 4% lower than in the biogenic carbon accounting cases across the three scenarios. As there is no biogenic carbon associated with the natural gas feedstock for SMR, there is no negative impact associated with CO₂ storage in the SMR business-as-usual process.

When examining just the H₂ production technologies, the GWP for SMR (with upstream natural gas) is higher than the proposed system (i.e., 4.19 kg CO_{2e}/kg H₂ compared to 3.82 kg CO_{2e}/kg H₂). The PEM electrolyzer, running on 100% wind, is less than the proposed system, with 1.50 kg CO_{2e}/kg H₂. If the upstream natural gas contribution is

omitted from the comparison, the “unburdened” SMR technology becomes substantially less than the proposed system (i.e., 1.13 kg CO_{2e}/kg H₂), which demonstrates the significance of the natural gas raw material acquisition, manufacturing, and distribution life cycle phases to the overall GWP of SMR (73%). Similar conclusions can be made from an input perspective where SMR and PEM have GWP impacts of 0.22 and 0.08 kg CO_{2e} per kg of MSW managed, respectively, as compared to 0.21 kg CO_{2e} per kg of MSW managed for the proposed system. The percentage differences are the same but on a much smaller scale.

When including electricity generation, the physically allocated burden of emissions to the electricity product from the MSW gasifier is much lower than that allocated to H₂ (about 4 and 96% to electricity and H₂ products, respectively), which is reflected in the impact results in Table 7 for the electricity product, 0.16 kg CO_{2e}, compared to 3.82 kg CO_{2e}/kg H₂ produced. For the benchmarks, the additional electricity generation is about three times as impactful compared to the gasification facility (i.e., 0.47 kg CO_{2e}/kg H₂), which reflects the burden of approximately 36% fossil-based electricity generation within the study region. This added impact is overshadowed by the much higher contribution by the paper, wood, and plastic production processes.

Examining the waste management pathway, landfilling produces a net carbon sink, given the large quantity of carbon that is not decomposed in wood and paper. In fact, as the percentage of wood increases, the landfilling impacts decrease (i.e., −9.10, −9.14, and −9.18 kg CO_{2e}/kg H₂ for 10, 15, and 20% composition of wood, respectively). When compared to the benchmarks, the total GWP impacts of the burdened case are always equal to or greater than those for SMR and PEM unless biogenic carbon accounting is also considered. With biogenic carbon accounting, the burden case varies 3–8% lower than the total impacts of SMR or PEM depending on the scenario. This percentage decrease in total emissions from the benchmarks demonstrates emissions avoidance by choosing the proposed system (i.e., MSW gasification) over either the business-as-usual (i.e., SMR) or counterfactual (e.g., PEM) system.

In emissions avoidance accounting, the MSW gasification pathway may be credited for the offset emissions and is calculated by taking the difference between the impacts of the proposed case (i.e., the MSW gasification pathway) and the impacts of the benchmark case (e.g., business as usual or counterfactual). The impacts of paper, wood, and plastic production are identical in both the proposed and benchmark cases and are therefore balanced in the comparison. In this case, the emissions avoided for the SMR pathway vary from 7 to 11 kg of CO_{2e}/kg H₂ produced, depending on the scenario, and from 4 to 7 kg of CO_{2e}/kg H₂ produced for the PEM counterfactual scenario. It is noted that the relative GWP from avoided emissions (i.e., from −11 to −4 kg CO_{2e}/kg H₂ produced) falls between the absolute magnitudes of the biogenic and base cases (i.e., from −13.9 to 3.82 kg CO_{2e}/kg H₂ produced) and may represent a more accurate assessment of the benefits associated with waste-to-energy pathways. Reporting of avoided emissions is recommended to accompany the total emissions; for example, state “the proposed MSW gasification impact is 135 kg CO_{2e}/kg H₂ produced with 11 kg CO_{2e}/kg H₂ of avoided emissions” rather than strictly reporting avoided emissions as a credit (e.g., 124 kg CO_{2e}/kg H₂ produced). Similarly, for the PEM comparison system, the MSW gasifier pathway’s GWP is 135 kg CO_{2e}/kg H₂ with 8 kg CO_{2e}/kg H₂ of avoided emissions.

However, these results should be taken cautiously. Considering other proxy materials from WARM v16, the emission factor for landfilling paper can fluctuate from as low as −0.95 kg CO_{2e} per kg of newspaper to as high as 1.26 kg CO_{2e} per kg of textbook paper [77] (Exhibit 3-26). Varying the landfill emission factor of paper between these two extremes for the 80:10:10 scenario produces overall landfilling GWP impacts that vary from −16.1 to +17.1 kg CO_{2e} per kg of H₂. Given the influence of paper proxy on both burdened impacts

as well as the landfill impacts, a sensitivity analysis was performed on the choice of paper to see whether there is a possibility for emissions avoidance.

4.4. Functional Unit Perspective

For this case study, the relationship between MSW feedstock and MSW gasification product was 18.7 kg of MSW to 1 kg of H₂ produced. This relationship was used for scaling the unit processes between functional unit perspectives. For both the SMR and PEM, the target functional unit was set to 0.05 kg of H₂ (i.e., the amount of H₂ produced by 1 kg of MSW feedstock).

As shown in the fourth scenario in Table 7, the functional unit perspective is flipped to input-based and described per kg of MSW managed. The percent difference in biogenic carbon emissions (assuming the neutrality) is the same as that for output-based (i.e., 490% reduced for the 80:10:10 scenario), as is the percent increase in emissions for the extended burden case (i.e., a 3900% increase for the 80:10:10 scenario without biogenic carbon accounting). There is a slight difference when comparing the biogenic carbon accounting in the extended burden case across functional unit perspectives. This is likely due to the difference in allocation factors used in the output-based functional unit perspective. Despite this small discrepancy, these results provide a level of confidence for decision-making between cases and scenarios, regardless of the functional unit perspective.

4.5. Assumptions and Limitations

The modeling framework used for this case study has the following limitations:

The base case process data are not influenced by changes in the MSW feedstock composition (i.e., more plastic/less paper does not impact gasifier efficiency, CO₂ product, or H₂ product), which may not reflect reality. The gasifier efficiency and H₂ yield will likely be influenced by the heating value and material composition of the MSW feedstock. The carbon content of the feedstock influences the carbon balance of the process and would likely impact the amount of CO₂ emitted and captured. Furthermore, the pre-processing requirements (e.g., for drying and comminution) will change depending on the properties and percentages of the feedstock. For simplicity, the gasification system process data are assumed constant for the changing MSW composition.

All flows were converted from their raw rates (e.g., kg/h) to daily totals assuming 24-h operation. This is likely an overestimation of the facility's operation time.

The allocation factors used to separate impacts for the MSW gasification output-based scenarios assume the higher heating value of H₂ is 38.7 kWh per kg of H₂.

The saline aquifer storage and transportation process is based on a 550 MWh coal power plant at 85% capacity with a 98% capture rate, which is different from the proposed system and SMR cases. Because the same process was used in all four scenarios, this will likely not cause an issue with this case study but may be a cause of concern for comparison with other studies.

The extended burden proxy processes (i.e., paper, wood, and plastic) do not have biogenic carbon categorization in their raw material acquisition and manufacturing; therefore, biogenic carbon only shows up in the waste management phase of the LCA for this case.

The extended burden proxy processes assume 100% virgin inputs. For paper (i.e., magazines and third-class mail), this was reasonable against the 96% virgin material reported in WARM's current mix of recycled inputs used for magazines and third-class mail. WARM has no recycling pathway for LDPE (i.e., the plastic proxy process), and dimensional lumber (i.e., the wood proxy process) cannot be manufactured from recycled material.

There are temporal differences in emission releases between the proposed system and the landfill gas (the business-as-usual waste management strategy), which were not considered in this study.

The carbon capture rate is assumed fixed for all cases that include CCS.

The PEM electrolyzer utilizes a small amount of Nafion membrane (i.e., 1.4×10^{-5} kg). The raw material acquisition and manufacturing processes for this material were omitted from the system boundary due to the proprietary nature of the product. This was justified through an impact assessment of TFE, which, if used as a direct proxy, reveals a small overall contribution to GWP (i.e., 1.4×10^{-3} kg CO_{2e} per kg H₂ produced). The TFE unit process is also proprietary; hence, it was left out of this study.

Under the benchmark comparisons, there are no LCA emissions associated with the production of capital or infrastructure associated with the proposed system.

The MSW gasification facility's excess electricity is generated at a higher voltage than that produced by a power plant (i.e., 345 kV compared to 2.3–7.7 kV); however, the electricity from the gasification facility is assumed to be an equivalent replacement, hence the selection of the electricity generation business-as-usual process.

5. Conclusions

This study presents three nuanced accounting methods that are encountered in waste-as-a-resource LCA studies and highlights the challenges and benefits of choices under each case using an example of MSW gasification to H₂ as a backdrop for the discussion. Due to the uncertainty associated with MSW feedstock compositions, a sensitivity analysis was completed using proxy processes that demonstrated the influence of waste composition on extended burdens, avoided emissions, and biogenic carbon sequestration.

The key takeaways from the case study include the following:

The choice of waste composition and its proxies is an important consideration. In the biogenic case, the composition influences the amount of biogenic carbon considered (e.g., the selection of wood and paper and their relative percentages compared to plastic). In the extended burden case, the proxy production processes can be several magnitudes larger with respect to their GWP impacts. In the avoided emissions accounting case, the proxy processes influence both the production and end-of-life emissions.

Biogenic carbon accounting under the zero-burden assumption provides an optimistic estimate of GWP by crediting the sequestration of biogenic carbon and ignoring its uptake. In this study, the biogenic carbon accounting case was the only one to produce net-negative GWP estimates, which may indicate the growing interest in biogenic carbon accounting and aligns with the perception of zero-burden accounting as promotional towards circular economies.

The choice between input- or output-based functional units does not appear to influence the comparative results for decision-making; however, both perspectives provide their own insights depending on the LCA scope and intended application. Reporting on both perspectives may lead to an increase in the value of future LCA studies on waste management.

Emission avoidance accounting requires a multi-functional unit perspective (i.e., waste management, feedstock use phase, and co-product generation) to be fairly compared with the proposed system, which serves as a waste management practice, an energy provider, and an H₂ source. The choice of benchmark for emissions avoidance depends on the scope of the LCA. Due to the uncertainty and disagreement of business-as-usual practices, it is recommended to include several benchmark processes (e.g., including a counterfactual). Performing a sensitivity study of these variables helps elucidate the uncertainty in the results and assists with any decision-making.

Due to the limited information provided by a single-metric comparison, it may be that the true benefit of waste-to-resource pathways may reside in a non-GWP environmental metric (e.g., fossil fuel resource depletion and land use).

For the LCA practitioner, the choice of biogenic carbon accounting should be balanced against whether carbon sequestration is a consideration, and, if so, the $-1/+1$ accounting method is more appropriate to avoid overcounting the net benefits of biogenic carbon sequestration. The extended burden perspective provides an opportunity for biogenic carbon to enter the life cycle from the biosphere but with the added challenge of waste composition analysis and the cost of significantly higher LCA impacts associated with the upstream activities (i.e., the green bars in Figure 6). Because waste management is often a multi-functional process, comparisons with benchmarks should also be completed on a multi-functional basis, with the added challenge of determining business-as-usual (and potentially counterfactual) processes that can stand against scrutiny.

Author Contributions: Conceptualization, E.L. and M.J.; Methodology and Data Curation, T.W.D., R.M. and S.M.; Writing—Original Draft Preparation, T.W.D. and R.M.; Writing—Review and Editing, E.L., M.J. and S.M. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the U.S. Department of Energy’s Office of Fossil Energy and Carbon Management, conducted under the contract number 89243323CFE000075.

Data Availability Statement: The original data presented in this study are openly available on NETL’s Energy Data eXchange at <https://doi.org/10.18141/2588500> (accessed on 18 September 2025).

Acknowledgments: The authors thank the support of the NETL PCE and EMA teams. Special thanks go to the following for their technical support: M. Henriksen, J. Izar-Tenorio, S. Moni, and S. Carr. Thanks also go to R. Wallace and M. Whiston for project management. During the preparation of this manuscript, the authors used the U.S. Department of Energy’s internal generative AI tool for summarization, phrasing and transitions, organization, and keyword extraction; the results of which were human reviewed and the authors take full responsibility for the content of this publication.

Conflicts of Interest: Authors Tyler W. Davis, Roksana Mahmud, and Shannon McNaul are employed by KeyLogic, LLC—a site support contractor to the United States Department of Energy National Energy Technology Laboratory. The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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