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Part 2

From Field to Bin:

The Environmental Impacts of U.S. Food Waste Management Pathways

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Disclaimer

This document has been reviewed in accordance with U.S. Environmental Protection Agency (EPA) policy and approved for publication.

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List of Abbreviations

AD	anaerobic digestion
ARB	antibiotic-resistant bacteria
ASP	aerated static pile
BOD	biochemical oxygen demand
CHP	combined heat and power
CO_2eq	CO_2 equivalent
COD	chemical oxygen demand
DOC	degradable organic carbon
DOCf	degradable organic carbon fraction
EPA	U.S. Environmental Protection Agency
FOG	fats, oils, and grease
GHG	greenhouse gas
GWP	global warming potential
HTC	hydrothermal carbonization
km	kilometer
LCA	life cycle assessment
LFG	landfill gas
MSW	municipal solid waste
$MTCO_2eq$	metric tons of CO_2 equivalent
Neq	N equivalents
OFMSW	organic fraction of municipal solid waste
PFAS	polyfluoroalkyl substances
RDF	refuse derived fuel
SO_2eq	SO_2 equivalents
SOC	soil organic carbon
SOM	soil organic matter
t	metric ton (1,000 kilograms)
TSS	total suspended solids
USD	U.S. dollars
VOC	volatile organic compound
WARM	Waste Reduction Model (EPA)
WRRF	water resource recovery facility
WWT	wastewater treatment

List of Chemical Symbols

C	carbon
CH_4	methane
CO_2	carbon dioxide
Hg	mercury
K	potassium
N	nitrogen
N_2O	nitrous oxide
NH_3	ammonia
NO_2	nitrogen dioxide
NO_{3-}	nitrate
NO_x	nitrogen oxides
P	phosphorus
PM	particulate matter
$PM_{2.5}$	particulate matter (2.5 micrometers in diameter and smaller)
SO_2	sulfur dioxide
SO_x	sulfur oxides

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Executive Summary

Over one-third of the food produced in the United States is never eaten, wasting the resources used to produce it and creating a myriad of environmental impacts. Wasted food is the single most common material landfilled and incinerated in the United States, comprising 24% and 22% of landfilled and combusted municipal solid waste (MSW), respectively, presenting opportunities for increased prevention and recycling.

The purpose of this report is to investigate the environmental impacts and contributions to a circular economy of eleven common pathways to manage wasted food – from source reduction to composting to landfill. The report presents a new ranking of the wasted food pathways, from most to least environmentally preferable. EPA's new ranking—called the Wasted Food Scale—replaces the Agency's Food Recovery Hierarchy developed in the 1990s. The Wasted Food Scale reflects the latest science as well as technological advances and changes in operational practices in the wasted food pathways since the Food Recovery Hierarchy was developed. Wasted food is generated all along the food supply chain, and thus the audience for this report includes a broad range of stakeholders from farms to food businesses to households to waste managers, as well as policymakers seeking advice on how to reduce the environmental impacts of wasted food.

Methodology

The report investigates eleven common wasted food pathways: source reduction, donation, upcycling (i.e., repurposing wasted food into new food for human consumption), animal feed, anaerobic digestion, composting, controlled combustion (i.e., incineration), land application (i.e., applying raw wasted food to soil), landfilling, sewer/wastewater treatment (i.e., sending wasted food "down the drain"), and unharvested/plowed in (i.e., leaving crops in the field). Wasted food is defined in the report as food meant for human consumption but not ultimately consumed by humans. Food crops grown for other purposes, such as biofuels or animal feed, are outside the scope of the report. Wherever possible, the report focuses specifically on wasted food as the feedstock for the pathway, taking into account the specific properties of food (e.g., moisture level, energy value, and decay rate) and how they affect a pathway's impacts and benefits. Thus, the findings are specific to wasted food and not applicable to other components of MSW. The report does not consider social or economic factors.

The report employs two methodologies to assess the pathways: Life Cycle Assessment (LCA) and Circularity Assessment. First, the report considers environmental impacts of the wasted food pathways through a quantitative and qualitative review of the LCA literature. LCA is widely recognized for its ability to identify issues and assess tradeoffs across the value chain and across environmental impacts. Approximately 250 studies were reviewed in depth, covering global warming potential, soil carbon sequestration, energy demand, acidification, particulate matter formation, human toxicity, ecotoxicity, eutrophication, water consumption, land occupation, and soil health. Second, the report presents a qualitative assessment of each pathway's contribution to a circular economy, focusing on the environmental aspects of circularity. The goal of a circular economy is to decouple economic growth from resource extraction by capturing greater value from existing products and materials. This contrasts with the traditional linear "take-make-use-dispose" economy where waste is seen as something to "send away" or dispose. In the assessment, pathways are evaluated against four circularity themes consistent with the definition of circularity presented in the Save Our Seas Act 2.0: preventing waste and having outputs maintain their highest potential value, stay free of contaminants to enable reuse, and help to regenerate ecosystems through improved soil health and climate resilience.

The two approaches utilized in this report – LCA and circularity assessment – are complementary and together inform the conclusions of this report and the Wasted Food Scale. There is precedent in the literature for combining these methodologies as it can reveal trade-offs between circularity and environmental impacts and among environmental impacts, helping decision-makers avoid shifting problems from one area to another. The LCA review enabled quantitative comparison of some environmental indicators, while the circularity assessment provided insight into other areas of environmental concern not covered by the LCAs (e.g., soil health and resource conservation). Pathway rankings from each methodology are provided in the main body of the report. Key differences between the results of the two assessments arise from how each methodology values the recovery of energy versus the recovery of nutrients, considers emissions and releases during the waste management process, and assesses soil health and ecosystem benefits. Results of the two assessments were merged through a combination of quantitative analysis and qualitative professional judgment by a panel of experts.

Conclusions

EPA presents the new Wasted Food Scale (see Figure ES-1), ranking wasted food pathways from most to least environmentally preferable, based on LCA and circularity assessment. Pathways are grouped into tiers where EPA determined them to have equivalent performance. The Scale emphasizes the importance of prevention and of diverting food waste from the sewer/wastewater treatment, landfill, and controlled combustion (i.e., incineration) pathways. The Wasted Food Scale replaces EPA's Food Recovery Hierarchy.

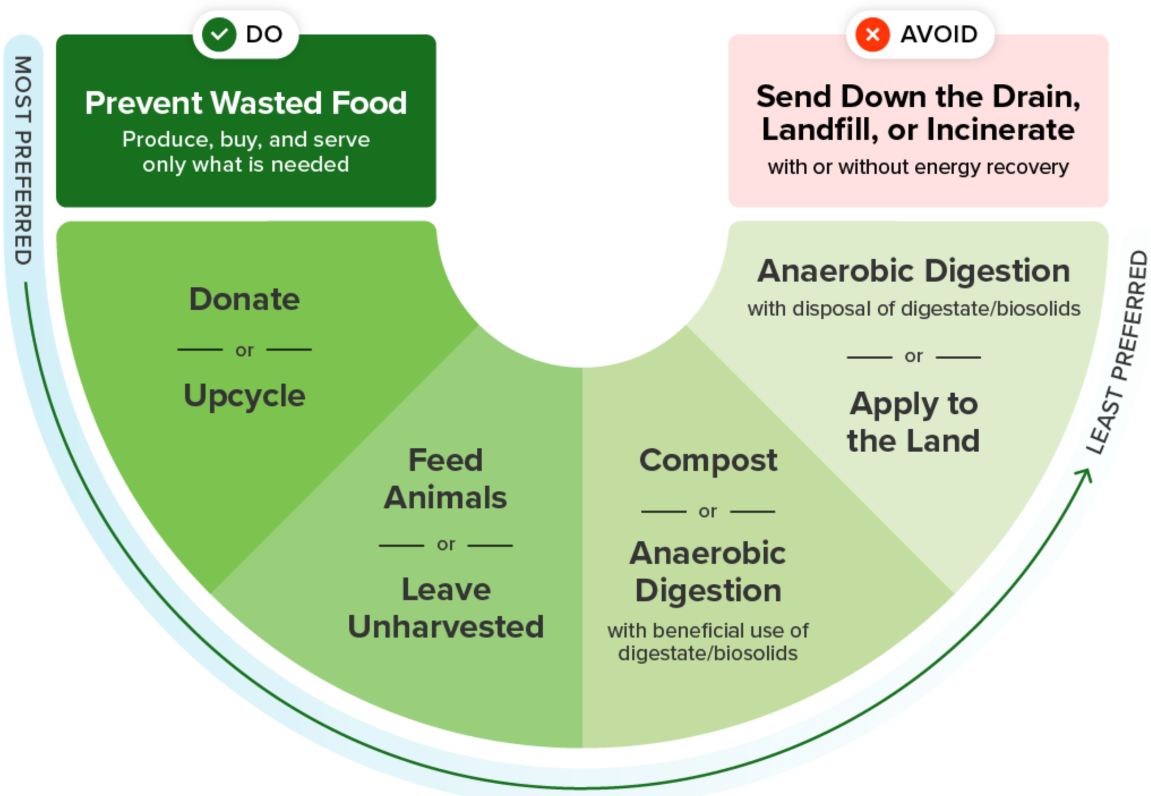

FIGURE ES-1. THE WASTED FOOD SCALE: EPA'S NEW RANKING OF WASTED FOOD PATHWAYS BASED ON LIFE CYCLE ASSESSMENT AND CIRCULARITY ASSESSMENT

In addition to the ranking provided, EPA draws the following conclusions:

Source reduction, donation and upcycling are the most environmentally preferable pathways because they can displace additional food production. Source reduction demonstrates global warming potential benefits an order of magnitude greater than all other pathways by reducing the amount of additional food that must be produced. Source reduction can be achieved at any stage of the supply chain, and source reduction options should be exhausted before other pathways are considered. Donation and upcycling can also displace additional food production, and thus exhibit benefits closest to those of source reduction. However, researchers estimate that up to 40 percent of donated food may be wasted (Alexander and Smaje 2008), reducing benefits of donation relative to source reduction. In addition, unlike source reduction, donation and upcycling require additional energy use (e.g., for transportation, cold storage, or processing), reducing net environmental benefits since increased fossil fuel energy use results in increased particulate matter, acidification, eutrophication, water consumption, and human and eco-toxicity.

The benefits of pathways beyond source reduction, donation, and upcycling are small relative to the environmental impacts of food production; thus, they can do little to offset the environmental impacts of food production. For example, anaerobic digestion can produce more energy per unit of wasted food than the other energy-producing pathways considered in this report (i.e., controlled combustion and landfill), but anaerobic digestion can recover only around 20 percent of the energy that was required to produce each unit of food. Also, unlike some other renewable energy sources (e.g., solar or wind), combustion of biogas at anaerobic digesters generates criteria air pollutant emissions similar to natural gas combustion.

Sewer/wastewater treatment (i.e., sending wasted food "down the drain") and landfilling stand out for their sizeable methane emissions. Wasted food decays rapidly in anaerobic conditions. Sewer transport allows for uncontrolled methane releases, and wasted food at landfills begins to decompose prior to capping and placement of landfill gas capture systems. These emissions far exceed the benefits of energy recovery technologies (i.e., anaerobic digestion at wastewater resource recovery facilities or gas capture systems at landfills) or the carbon sequestration potential of landfills. Wastewater resource recovery facilities with anaerobic digestion have lower global warming potential than those without, but the difference is not big enough to affect the final ranking of the sewer/WWT pathway.

All wasted food pathways other than landfill and sewer/wastewater treatment demonstrate beneficial or near neutral global warming potential. While all pathways (except source reduction) use energy and release carbon dioxide, and some pathways release other greenhouse gases (methane and nitrous oxide in composting and methane in anaerobic digestion), this is generally offset by the benefits of avoided production of energy, food, animal feed and/or fertilizer, avoided fertilizer use, and/or carbon sequestration.

Recycling wasted food into soil amendments offers opportunities to make long-term improvements in soil structure and health and help regenerate ecosystems by recovering nitrogen and carbon and returning them to the soil. The nutrient-rich outputs of composting, anaerobic digestion, and sewer/wastewater treatment (i.e., compost, digestate or biosolids) can help build soil organic matter, which decreases bulk density, increases porosity, water retention, and infiltration, promotes nutrient cycling and retention, and stabilizes soil aggregates, reducing the risk of erosion. These improvements are regenerative and can help to improve the resilience of agricultural systems in the face of climate change. Synthetic fertilizers lack the organic matter content of these amendments and cannot provide similar benefits, other than indirectly through promoting crop growth. In addition, the nitrogen in wasted food can be used to provide nutrients to crops over the short or long-term in lieu of synthetic fertilizers. These organic amendments require less energy to produce than synthetic fertilizers and when applied result in less nitrogen loss through volatilization, leaching, or runoff than synthetic fertilizers, reducing acidification, eutrophication, particulate matter emissions, and global warming potential. However, nutrient-rich outputs from anaerobic digestion and sewer/wastewater treatment may be disposed rather than beneficially used.

As the U.S. becomes less dependent on fossil fuels for energy, the environmental value of producing energy from wasted food will decrease. The LCA results are driven largely by global warming potential and energy demand, due to widely available data and, in the case of energy demand, the effect of energy use on many of the other selected indicators. Energy use by pathways contributes to more than half of the indicators assessed in the LCA chapter of this report, including energy demand, acidification, eutrophication, particulate matter, and human and eco-toxicity performance. Thus, the energy mix assumed (when calculating effect of energy use and/or avoided energy production) has a substantial effect on the outcome of LCA studies. As the U.S. increases the use of lower-impact energy sources in its portfolio, the environmental “cost” of energy use and the environmental “benefit” of avoided energy production will both decrease. This would likely, for instance, lower the environmental performance of controlled combustion and anaerobic digestion but improve the environmental performance of composting. For example, if energy produced by anaerobic digestion was displacing wind power, rather than fossil fuel-derived power, the pathway would be net positive (increase GHG emissions), rather than net negative (reduce GHG emissions), for global warming potential (Slorach et al. 2019a).

CHAPTER 1 INTRODUCTION

Over one-third of the food produced in the United States is never eaten, wasting the resources used to produce it and creating a myriad of environmental impacts. EPA estimates that the U.S. wastes 73 to 152 million metric tons (161 to 335 billion pounds) of food per year, or 223 to 468 kg (492 to 1,032 pounds) of food per person per year, equal to approximately 35% of the U.S. food supply (U.S. EPA 2021c).

Recognizing the critical importance of reducing wasted food, in September 2015, the United States announced the 2030 *Food Loss and Waste Reduction Goal* to halve per person wasted food at the retail and consumer level and reduce on-farm losses by the year 2030 (U.S. EPA 2020d). This domestic goal is aligned with the *Sustainable Development Goal Target 12.3* of the 2030 Agenda for Sustainable Development (UN General Assembly 2015). In addition, in 2020, the U.S. EPA set a *National Recycling Goal* to increase the national recycling rate to 50% by 2030 (U.S. EPA 2022c). Currently, wasted food is the single most common material landfilled and incinerated in the United States, comprising 24% and 22% of landfilled and combusted municipal solid waste (MSW) (U.S. EPA 2020a), respectively, presenting opportunities for increased reuse and recycling.

This report is the second in a pair of EPA reports on the environmental impacts of wasted food. The first report — *From Farm to Kitchen: The Environmental Impacts of U.S. Food Waste* (U.S. EPA 2021c) — described wasted food in the United States and the environmental footprint associated with producing, processing, and distributing food that is ultimately wasted, including land use, water consumption, pesticide and fertilizer use, and climate change. This second report investigates the environmental impacts associated with the management of wasted food.

1.1 Purpose

The purpose of this report is to inform public- and private-sector policy- and decision-making about which U.S. wasted food pathways to encourage, incentivize, or invest in based on environmental impact and contributions toward a circular economy. The report considers eleven common pathways: source reduction, donation, upcycling, animal feed, anaerobic digestion, composting, controlled combustion (i.e., incineration), land application, landfills, sewer/wastewater treatment, and unharvested/plowed in.

The report presents a new ranking of the wasted food pathways, from most to least environmentally preferable. EPA’s new ranking–called the Wasted Food Scale--replaces the agency’s Food Recovery Hierarchy developed in the 1990s.

1.2 Scope

The scope of this report is to document and compare the available literature on the environmental impacts and benefits of wasted food pathways, beginning the moment food is wasted. The report also considers each pathway’s potential contribution to a circular economy, focusing on the environmental aspects of circularity. The report does not address social and economic factors related to the pathways. Where possible, the report evaluates the specific properties of wasted food and how they affect a pathway’s environmental performance.

1.3 Definitions

In this report, the term “wasted food” is defined as food grown for human consumption that is not used for its intended purpose and is managed in a variety of ways, such as donation to feed people, creation of animal feed, composting, anaerobic digestion, or disposal in landfills or controlled combustion facilities (U.S. EPA 2023a). Wasted food can be generated at any stage of the supply chain, from farm to consumer. Examples include unharvested crops; by-products from food and beverage processing facilities; unsold food from retail stores; or plate waste, uneaten prepared food, or kitchen trimmings from restaurants, cafeterias, and households. Wasted food includes parts of food deemed edible and those deemed inedible, such as shells, bones, pits, or peels. Food crops grown for other purposes, such as biofuels or animal feed, is excluded from the definition.

The term “wasted food pathways” refers to all eleven pathways discussed in the report, including source reduction, donation, upcycling, animal feed, anaerobic digestion, composting, controlled combustion, land application, landfills, sewer/wastewater treatment, and unharvested/plowed in.

This report uses the definitions of each wasted food pathway provided in Table 1-1. These pathway definitions are largely consistent with both the U.S. EPA 2019 Wasted Food Report (U.S. EPA 2023a) and the Food Loss and Waste Protocol (Hanson et al. 2016), a collaborative effort among several organizations[1] that establishes guidelines for the accounting, reporting, uncertainty and tracking of wasted food over time. Key differences are that this report adds the source reduction and donation pathways and excludes the bio-based materials/biochemical processing pathway since many of these technologies are not yet at commercial scale. The upcycling pathway discussed in this report is a subset of the bio-based materials/biochemical processing pathway in the Food Loss and Waste Protocol. Upcycling, as defined in this report, produces food for human consumption, whereas bio-based materials/biochemical processing may also produce non-consumable products such as cleaning products or chemicals. The terms “controlled combustion” and “incineration” are used interchangeably in the report, as are the terms “down the drain” and “sewer/wastewater treatment.”

Additional pathways to manage wasted food exist but were excluded from the analysis in this report because they are relatively novel and still being developed or data were unavailable. There is an emerging group of processes that includes pyrolysis, torrefaction, gasification, and hydrothermal liquefaction. Appendix E provides brief descriptions and summarizes available life cycle assessment data on these processes. Appendix E also includes data on rendering, which is excluded from the main analysis due to limited data availability. Rendering is the well-established process of cooking and separating animal waste products and used cooking oil into purified fats and ground protein to produce biofuels, animal feed ingredients, fertilizer, and commercial and industrial products (NARA 2019).

TABLE 1-1. DEFINITIONS OF WASTED FOOD PATHWAYS

Source Reduction	Reducing the generation of wasted food.
Donation	Redistributing wasted food for human consumption.
Upcycling	Remaking wasted food into new food products for human consumption.
Anaerobic Digestion	Breaking down wasted food via microorganisms in the absence of oxygen to generate biogas and nutrient-rich matter (i.e., digestate or biosolids). May occur at a stand-alone digester, an on-farm digester, or a digester at a water resource recovery facility (WRRF)[2]. This pathway can include co-digestion, which is the simultaneous anaerobic digestion of wasted food and other organic material in one digester. In this pathway, wasted food is transported to digester by vehicle or pipe. (Digestion of wasted food traveling to a digester by sewer is classified as the sewer/wastewater treatment pathway.)
Animal Feed	Diverting wasted food originally intended for human consumption or byproducts of food manufacturing (e.g., corn husks or fry oils) to animals, either directly or after processing.
Composting	Breaking down wasted food via microorganisms in an oxygen-rich environment to produce organic material (i.e., compost) that can be used as a soil amendment.
Controlled Combustion	Combustion (i.e., incineration) of wasted food in a controlled manner. This report examines only controlled combustion that recovers energy.
Land Application	Spreading, spraying, injecting, or incorporating wasted food from food manufacturing onto or below the surface of the land to enhance soil health.
Landfill	Sending wasted food to an area of land or an excavated site that is specifically designed and built to receive wastes. This report examines only landfills that capture landfill gas to recover energy.
Sewer/ Wastewater Treatment (WWT)	Sending wasted food down the drain and through the sewer (with or without prior treatment) to WRRFs with or without anaerobic digestion. (Co-digestion of wasted food delivered by truck or pipe directly to anaerobic digesters at WRRFs is covered by the anaerobic digestion pathway.)
Unharvested/ Plowed In	Leaving crops that were ready for harvest in the field or tilling them into the soil.

1.4 Background

This section provides background on the prevalence of each of the wasted food pathways and existing wasted food hierarchies, including the original 1990s EPA Food Recovery Hierarchy (U.S. EPA 2021g).

Prevalence of Wasted Food Pathways in the United States

Figure 1-1 presents the prevalence of each wasted food pathway in the U.S. in 2019 (U.S. EPA 2023a). The figure excludes the quantity of wasted food that is avoided through source reduction, the quantity upcycled into new food products, and the quantity of on-farm wasted food that is unharvested and plowed back into the soil.

Currently 60% of wasted food generated by retail, food service, and households is sent to landfill. Controlled combustion (i.e., incineration) is the next most common pathway. In contrast, the predominant wasted food pathway in the food manufacturing sector is anaerobic digestion, which receives 43% of the sector's wasted food. Animal feed (34%) and land application (13%) are also common pathways for the food and beverage manufacturing industry, with only two percent of the sector's wasted food destined for landfills.

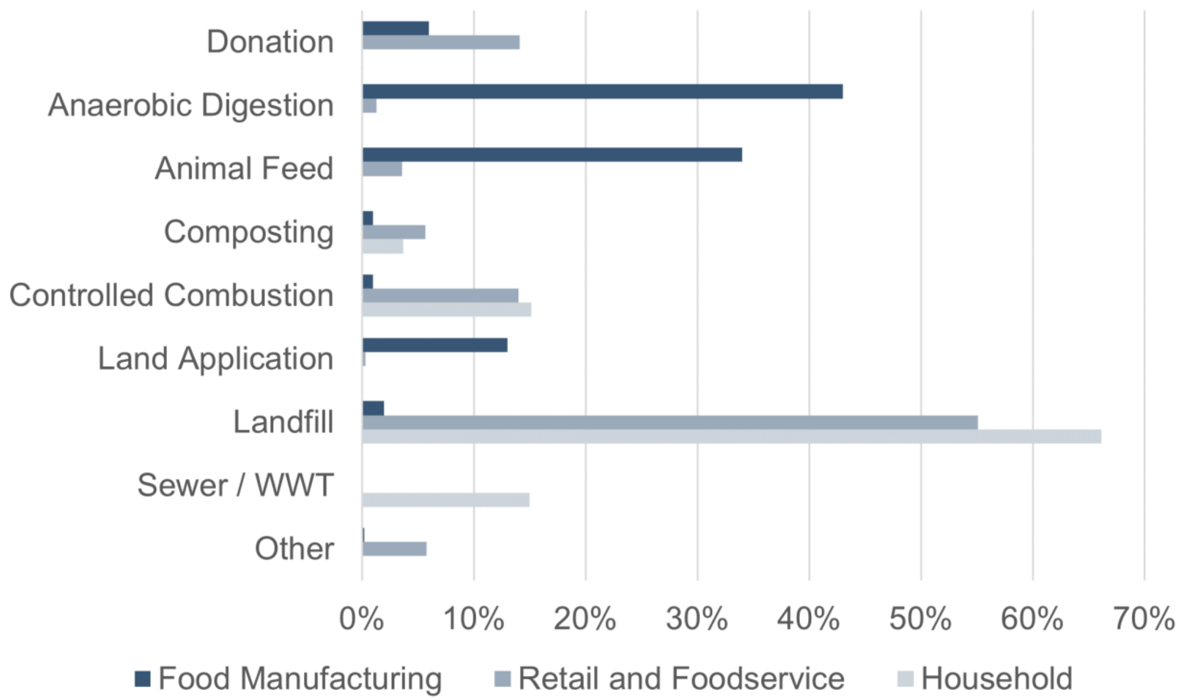

FIGURE 1-1. PERCENTAGE OF WASTED FOOD MANAGED BY EACH PATHWAY, BY SECTOR IN WHICH THE WASTED FOOD IS GENERATED

Each sector sums to 100%.
Data Source: U.S. EPA (2023a)

Wasted Food Hierarchies

Wasted food frameworks provide high-level guidance on social, economic, and environmental preferences for managing wasted food. They have been developed in academic settings (e.g., Lombardi and Costantino 2021) and by non-profit organizations and governments—for example, the United Kingdom's Food and Drink Waste Hierarchy (DEFRA 2021) and the U.S. EPA Food Recovery Hierarchy (Figure 1-2) (U.S. EPA 2021g). The available frameworks differ in the number and priority order of wasted food pathways, based upon the aim, perspective, and methodology of the researchers and/or stakeholders (Moshtaghian et al. 2021).

The EPA Food Recovery Hierarchy is widely cited in the United States; however, it was established in the 1990s so might not reflect technological advances and changes in operational practices from recent decades. Also, it does not consider all the wasted food pathways discussed in this report. This report synthesizes the latest science on the life cycle environmental impacts and potential contributions to a circular economy of eleven common pathways for managing wasted food in the U.S. and presents a new ranking of pathways (called the Wasted Food Scale) on this basis. The original EPA Food Recovery Hierarchy is not, nor should the results of this report (the Wasted Food Scale) be interpreted as, a set of strict criteria. Rather, the report aims to evaluate the many complex factors that influence the environmental performance of the wasted food pathways and identify environmental tradeoffs, recognizing that conclusions may vary based on location, climate, available infrastructure, or other factors.

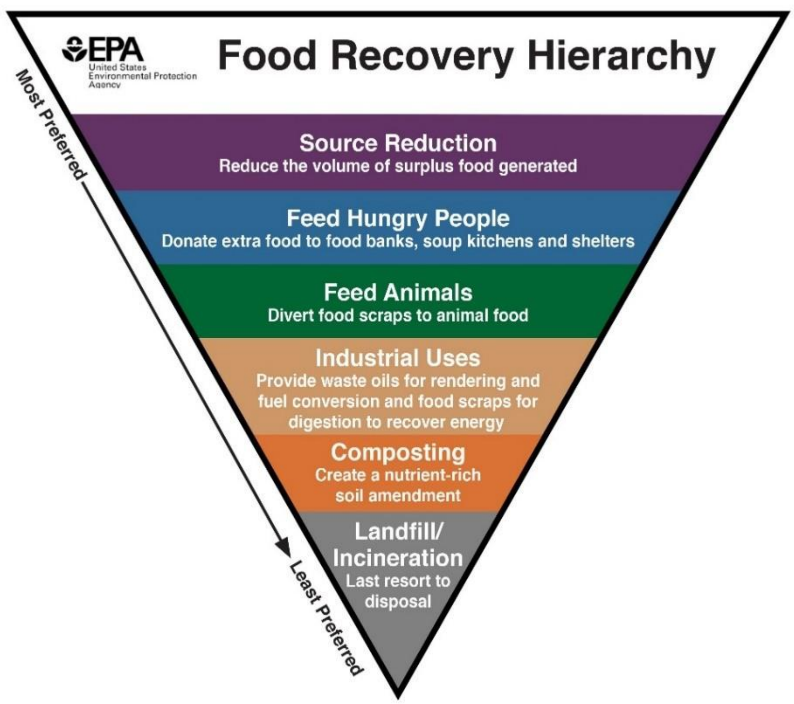

FIGURE 1-2. EPA'S PREVIOUS RANKING OF WASTED FOOD PATHWAYS

Source: U.S. EPA (2021g)

1.5 Approach and Research Methods

This report synthesizes information from existing literature on wasted food. The systematic literature review focused on three classes of studies:

- Those employing a systems perspective or life cycle assessment (LCA) approach for quantifying the collection-to-grave environmental impacts of wasted food pathways.
- Those employing a circularity perspective for comparing the wasted food pathways.
- Select studies addressing key data gaps not addressed by the identified studies.

The literature review protocol used to develop the findings in this report is provided in Appendix A. Subsequent sections briefly introduce the concepts of LCA and circularity assessment and describe the terminology used to communicate the magnitude of quantitative results in this report.

Life Cycle Assessment Approach

LCA is widely recognized for its ability to identify issues and assess tradeoffs across the value chain and across environmental impacts (ISO 2006b). In LCA, information on the inputs and outputs of a defined system or process are assessed to quantify the associated environmental impacts. The breadth of coverage in LCA is intended to minimize the potential to shift environmental burdens between processes, life cycle stages, or impact categories (ISO 2006a).

The management of wasted food requires collection, transport to a processing or treatment facility, processing or treatment, distribution of beneficial products, and disposition of final waste outputs. Each of these steps entails assumptions about system parameters, such as distances traveled, composition of waste and efficiencies of processing.

The basis of comparison in an LCA—the functional unit—can vary widely across studies in a way that sometimes hinders comparisons between them. For LCAs of wasted food pathways, though, the mass of wasted food is typically chosen as the functional unit, and many authors additionally specify the quality (e.g., moisture content and carbon:nitrogen ratio) and management duration (often one year), further facilitating cross-study comparisons. Other modeling choices such as the system boundary, impact allocation, and avoided product crediting can also affect the comparability of studies.

Global warming and eutrophication potential are the most studied environmental impacts in LCA of wasted food pathways. Other impacts such as those related to soil health characteristics are underdeveloped. In this literature review, we searched for quantitative information related to global warming potential, energy demand, acidification, particulate matter formation, human toxicity, ecotoxicity, eutrophication, water consumption, land occupation, and soil carbon sequestration / soil health.

Circularity Assessment Approach

This report also employs a circularity perspective to evaluate the wasted food pathways. Generally, the goal of a circular economy is to decouple economic growth from resource extraction by capturing greater value from existing products and materials. This contrasts with the traditional linear “take-make-use-dispose” economy where waste is seen as something to “send away” or dispose of. The concept draws from schools of thought such as industrial ecology and regenerative design (Geissdoerfer et al. 2017), and many organizations and countries have adopted circularity principles over the last decade, including the United Nations, the World Economic Forum, China, and the European Union (Bocken et al. 2016; Geissdoerfer et al. 2017; Mancini and Raggi 2021).

In 2020, the U.S. enacted the Save Our Seas Act 2.0, directing EPA to develop circular economy strategies for post-consumer materials management. While many definitions for a circular economy can be found in the literature (Geissdoerfer et al. 2017; Kirchherr et al. 2017), in this report we rely primarily on the definition from the Save Our Seas Act 2.0 – where a circular economy is one that “uses a systems-focused approach and involves industrial processes and economic activities that:

1. are restorative or regenerative by design;

2. enable resources used in such processes and activities to maintain their highest values for as long as possible; and
3. aim for the elimination of waste through the superior design of materials, products, and systems (including business models)" (U.S. Congress 2020).

Terminology

Table 1-2 summarizes the terminology used to make quantitative statements about processes or model parameters driving environmental performance in the report. In this table, the net and gross impacts refer to those modeled in a particular study for a particular pathway.

TABLE 1-2. QUANTITATIVE THRESHOLDS USED TO MAKE STATEMENT ABOUT THE MAGNITUDE OF DRIVER CONTRIBUTIONS

Term	Threshold(s)
Impact/Gross Impact	Refers to cumulative environmental impact excluding consideration of avoided product benefits or other environmental credits (e.g., carbon sequestration).
Benefit	Refers to a net negative environmental impact including consideration of avoided product benefits or other environmental credits.
Significant	Used only when cited authors make a claim of statistical significance in analyzing environmental performance. Other terms and comparisons presented in this report should not be interpreted as statistically significant.
Considerable	Contributing greater than 10% of net or gross impact.
	The largest contributor to net or gross impact, even if less than 10%.
Sensitive	A result is sensitive to a particular parameter or practice when its presence or absence leads to a 10% or greater change in impact.
	A result is sensitive to a particular parameter when a change in that parameter's value leads to a disproportionate change in the analysis result (such as when a 10% change in biogas yield leads to a 15% decrease in GWP).
Negligible	Contributing less than 2% of net or gross impact.
Minor	Contributing between 2% and <10% of net or gross impact.
Moderate	Contributing between 10% and 30% of net or gross impact.
Major	Contributing greater than 30% of net or gross impact.

Quality Assurance

In accordance with the project's Quality Assurance Project Plan (QAPP) entitled *Quality Assurance Project Plan for Environmental Impacts of Food Waste* approved by EPA on November 2, 2022, this report synthesizes literature on the environmental impacts and benefits of U.S. food waste pathways, compares the environmental impacts and benefits of each pathway to one another under varying conditions and using a suite of life cycle-based and circularity indicators, and identifies key knowledge gaps where original scientific research may greatly improve our understanding of the issue and lead to effective solutions. The literature search method employed complies with the QAPP and is described in Appendix A. Appendix A also documents example queries and keyword searches used. The collected data sources include peer-reviewed literature, government and NGO reports, national survey and measurement information, and model simulations. The report authors evaluated the collected information for completeness, accuracy, and reasonableness. In addition, the publication date, geographic coverage and funding source were assessed when reviewing data quality. All compiled literature information was also reviewed by a report author who did not conduct the original data search. The literature review, according to the QAPP, did not require specific criteria in terms of age of data, data source, geographic coverage to determine whether the source was included or excluded. All data source attributes were recorded in a transparent fashion and fully referenced. Extracted data values from literature are fully documented in Appendix D. Specific attributes of each study such as the system boundary and inclusion/exclusion of environmental credits were, however, documented to assess the comparability of study results. The scope criteria for these inter-study comparisons are described in Section 3.4.

1.6 Report Overview

The report is organized in the following way:

- Chapter 2 describes each wasted food pathway and identifies the key aspects of each pathway that contribute to environmental impacts or benefits.
- Chapter 3 compares the wasted food pathways through a review of Life Cycle Assessments.
- Chapter 4 compares the wasted food pathways through a circularity assessment.
- Chapter 5 discusses the report's findings and presents a new hierarchy of the wasted food pathways.
- Chapter 6 presents implications and priority research gaps.

CHAPTER 2
WASTED FOOD PATHWAYS

This chapter describes each wasted food pathway and discusses key aspects of each pathway that contribute to its environmental impacts and/or benefits, including its potential contributions to a circular economy.

Each pathway section provides details on system “types,” referring to the specific pathway technology used, and process steps. For example, composting might use windrow or aerated static pile systems; controlled combustion might involve grate furnaces, spreader stoker furnaces, or another technology. While it is beyond the scope of this analysis to comprehensively discuss how different combinations of these practices affect the environmental performance of wasted food pathways, the discussion in this section looks to introduce these variables and to provide context for pursuing more detailed inquires related to local wasted food management decisions and policy questions.

Given the breadth of environmental impact categories considered, the following discussion aims to strike a balance between synthesis (looking for patterns across impact categories) and detail (recognizing that individual impact categories may not fit broader patterns). The final section (Section 2.13) summarizes topics that apply to multiple wasted food pathways, such as waste collection, transportation, and pathway requirements regarding homogeneity and contamination of wasted food.

Many of the pathways described in this chapter do not exist exclusively to process wasted food. This report focuses on the environmental performance of each pathway with respect to wasted food (e.g., the focus is on the behavior of wasted food in a landfill, rather than on the performance of the landfill in aggregate). At the same time, this report recognizes that the addition of wasted food may change the behavior of the overall system (e.g., co-digestion of wasted food may improve the biogas yield of an anaerobic digester).

System Diagram Introduction

Each wasted food pathway covered in this chapter includes a diagram of typical LCA system boundaries for that process. Figure 2-1 provides an example of a generic system diagram. These diagrams illustrate key processes that may be included in specific LCA studies. Not all listed processes in each system diagram are necessary or relevant for inclusion in every LCA study.

Each diagram includes the following elements:

- Wasted food and other possible waste inputs.
- Key processes including wasted food collection, pre-treatment, operation of the main pathway process (e.g., landfill or compost facility) and product processing and valorization. In some cases, icons from Table 1-1, where wasted food pathways are defined, are used to identify the main pathway process.
- Additional inputs required, such as energy, fuel or chemicals.
- Process outputs, which constitute the main beneficial products and co-products that result from operation of a wasted food pathway for resource recovery. Process outputs include compost, animal feed, digestate, biogas and landfill gas.
    - Avoided products. Process outputs are often modeled within the LCA literature as avoiding alternative products that would be utilized in the wider economy if wasted food resource recovery was not pursued. An example of this would be avoiding the production and use of chemical fertilizers in favor of an organic amendments such as compost, digestate or biosolids. Within an LCA study, avoided products generate environmental benefits which are also discussed as environmental credits.

- Arrows are utilized to show the connection between processes and products and may include transportation impacts when within the scope of individual studies. Orange arrows are used to callout select input and output flows relevant to each pathway.
- Disposed waste. The trash can icon is used to show that contaminant flows present in wasted food are sent to traditional waste disposal facilities such as landfills or controlled combustion. It can also denote portion of wasted food ultimately being landfilled (e.g., up to 40 percent of donations may be wasted rather than consumed).
- Outputs (e.g., effluent) and emissions. Sequestered carbon is also noted here.

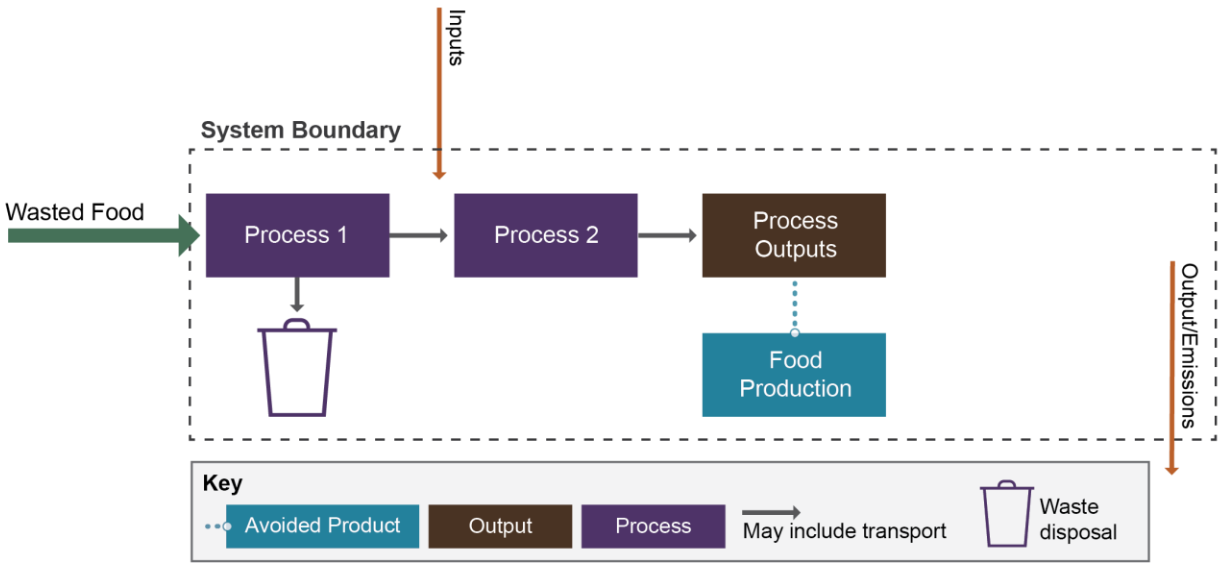

FIGURE 2-1. GENERIC SYSTEM DIAGRAM

2.1 Source Reduction

Source reduction is fundamentally distinct from all the other wasted food pathways, as wasted food is not generated. Source reduction decreases the generation of wasted food by preventing food from becoming waste at any point in the food supply chain. Source reduction strategies can be sector-specific, such as improved harvesting techniques, or applicable across all food sectors. Factors that result in the generation of wasted food may be interlinked across the food supply chain. For example, produce might go unharvested on a farm because it does not meet consumer standards for retail sale. Ensuring food supply more closely meets demand requires coordination across the food supply chain. For instance, farmers plant only what will be harvested and find secondary markets for surplus or “unmarketable” produce, while food retailers manage inventory to avoid overstocking items that will not sell. Improvements to packaging and storage can extend the edible life of food across the supply chain, from cold storage to drying facilities, and packaging that extends shelf life to consumers optimally storing produce. Enhancing the distribution of food products to decrease transit time, improving handling techniques, optimizing product use, and monitoring for quality and safety are just a few of the many actions that can reduce the generation of wasted food across the supply chain (ReFED 2023). Increasing education and supporting behavioral change, especially at the consumer level, are also important steps toward source reduction (Oregon DEQ 2017).

Minimizing waste offers the potential to reduce the amount of food production required to feed a population (Sherwood 2020), thus reducing the environmental impacts associated with production, including energy, water, and nutrient inputs, land use, and GHG emissions. Avoided impacts of food production depend on the type of avoided food product and the management practices used to produce it. Drawing on several resources, EPA estimates a wide spectrum of food production global warming potential (GWP) impacts, ranging from 0.5 to 33 kg CO_2eq per kg of food produced and delivered to consumers (U.S. EPA 2021c).

2.2 Donation

The donation pathway rescues or redistributes wasted food for human consumption. Donated food is "food that is donated to people via food banks or pantries, food distribution services, etc." (ReFED 2021).

Figure 2-2 diagrams the donation pathway. Wasted food intended for donation needs to be collected at the source and transported to the point of distribution. For food donation, there is an extra logistical challenge: food safety protocols must be maintained, and there are a variety of stakeholders in the distribution system who must collaborate to ensure food safety. Collected food may require culling or repackaging. The distribution phase of the donation pathway includes product storage and transportation to individuals and wholesale or retail establishments.

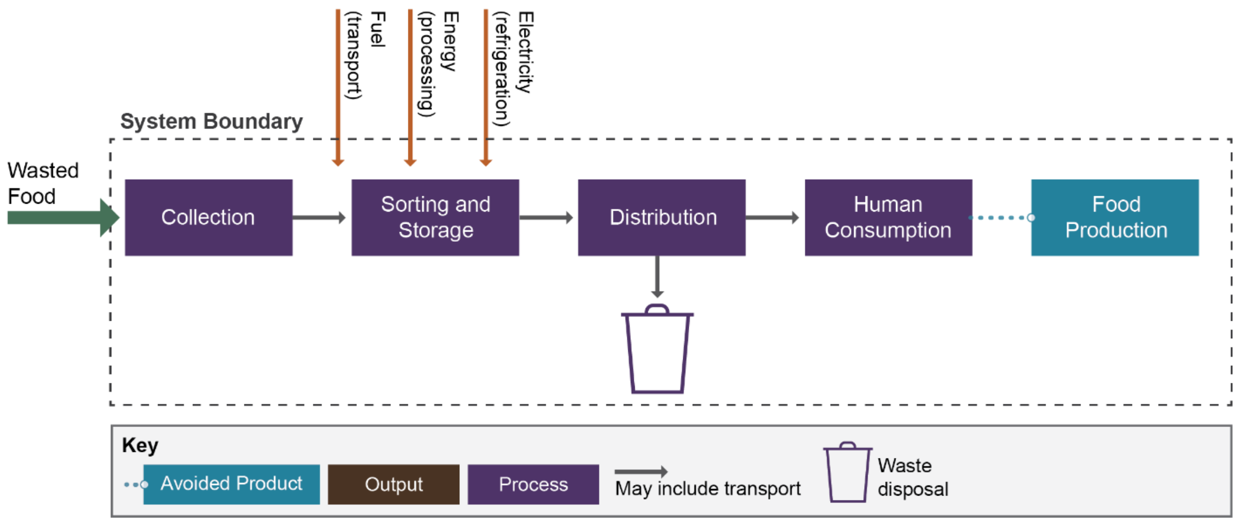

FIGURE 2-2. DIAGRAM OF DONATION

Collection and Distribution

Food donation is not environmentally 'free': there are impacts associated with collection and distribution of food. Gleaning is a specific form of food donation where unharvested or leftover crops are collected directly from the farm field, often by volunteers or organizations in support of food pantries (D. Lee et al. 2017). Food to be rescued is often in dispersed locations (e.g., at retail stores) and therefore can require a significant amount of effort to collect and distribute. When the use of personal vehicles is considered, transportation may be an important contributor to overall environmental impacts, as trips may be numerous or inefficient (Oregon DEQ 2019; Damiani et al. 2021). Despite this, transportation impacts are often an order of magnitude smaller, if not more, than the avoided burden of producing food in the first place (Schneider 2013). In addition, in some cases the impacts of transportation could reasonably be excluded from the donation pathway (e.g., if a person drops off donated food at a collection site located in a grocery store during a trip to buy groceries). Efficient storage and prevention of spoilage are important, too, for this pathway. WRAP (2015) reports that increased use of appropriate packaging and increased refrigeration result in a net GWP benefit, since less food is wasted.

Waste within the Donation Pathway

In addition, the donation pathway is not waste free, and a share of wasted food collected for donation will ultimately be disposed of. Donated food may be wasted in several ways – food banks may not accept a donation if it isn't a type of food they currently need or can use, food banks may accept a donation but be unable to provide it to a customer before it spoils, or customers may take home donated food but not consume it.

Estimates of this secondary waste vary, from 4% to 40%. A study by the Oregon Department of Environmental Quality (Oregon DEQ 2019) assumes loss rates of about 7% at sorting and 7% again at distribution, and model consumer waste (i.e., waste by donation recipients) at 7%, 14%, or 20%, while ReFED reports that only 4.2% of donated food is not consumed (Corona et al. 2020). Some LCA studies assume all recovered food is consumed (e.g., Damiani et al. 2021), others acknowledge uncertainty on this question and perform sensitivity analyses, and others use survey or other data to estimate loss rates. Other studies roll all losses during the donation pathway into a single loss rate, such as 22% (Sundin et al. 2022) or up to 40% (Alexander and Smaje 2008).

Avoided Food Production

The use of donated food by consumers results in an environmental benefit associated with avoiding additional food production. Recovery and donation of low environmental impact foods (e.g., potatoes) has a lower net environmental benefit than recovery and donation of high environmental impact foods (e.g., beef).[3]

In some cases, food donation does not cause less food to be produced – for example, when donated food does not end up being eaten for various reasons, or when recipients of donated food could not have otherwise purchased food. While it can be challenging to assess whether food donation actually affects overall food production, donation can be credited with source reduction benefits, since donation is a redistribution of food that would otherwise not be consumed (Heller 2019).

TABLE 2-1. DONATION: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Transportation emissions and associated impacts, which can vary widely depending on the wasted food source and the type of vehicle used.
	Energy demand of cold storage.
	Secondary wasted food.
Source of Environmental Benefit	Avoided food production.
Option to Reduce Impact	Restrict collection and distribution efforts to food that is likely to be consumed and prioritize collection and distribution of high environmental impact foods. Increase food refrigeration and preservation capacity of facilities collecting food to reduce food loss prior to consumer.

2.3 Upcycling

The upcycling pathway remakes wasted food into new food products for human consumption. Waste streams suitable for upcycling are limited to commercially viable, pre-consumer streams. For example, spent cereals from the beer brewing process may be used to create bread or crackers, and unsellable produce and food processing byproducts (e.g., pits or peels) may be used to make beverages and beverage mixes. Food banks may also upcycle wasted food (e.g., upcycling surplus milk donations into cheese).

The use of upcycling as a means of reducing wasted food is a newer concept (Eriksson and Spångberg 2017), and there are limited data on the prevalence of this pathway. This review found only two studies that comprehensively evaluated this pathway (Eriksson and Spångberg 2017; Eriksson et al. 2021); therefore, a robust evaluation of its environmental impacts and benefits is not yet possible.

Figure 2-3 diagrams the upcycling wasted food management pathway.

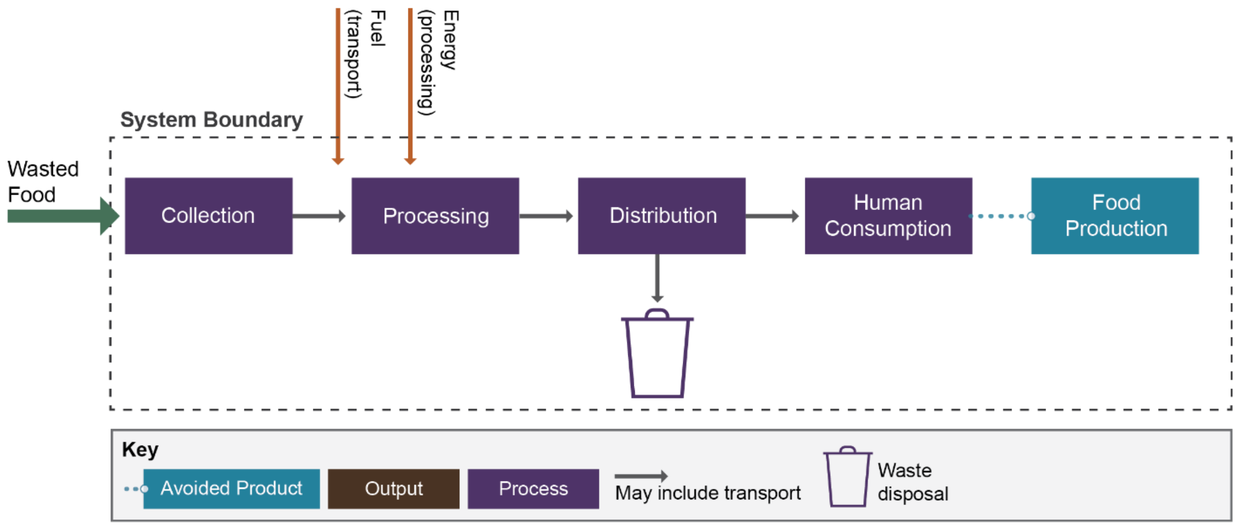

FIGURE 2-3. DIAGRAM OF UPCYCLING

Processing and Distribution

This pathway is unique among the wasted food pathways in that the recovered wasted food is processed, distributed, and consumed as if it were "regular" food. As such, the environmental impacts of processing and distributing this food are identical to those of the foods discussed in the Part 1 report (U.S. EPA 2021c)—with the production impacts removed. Impacts associated with upcycling vary widely depending on the specific product being considered and can be consequential. In the analysis of Eriksson et al. (2021), drying of broccoli leaves added about 0.12 kg CO_2eq per kg of harvested broccoli. When broccoli powder was assumed to displace wheat flour, the avoided benefit of wheat production was not enough to offset the GWP impact associated with drying the broccoli leaves, leading to a net impact. On the other hand, this was not the case when upcycled broccoli displaced broccoli powder or slices, leading to a net GWP benefit.

Avoided Food Production

The capture, processing, and human consumption of wasted food that would otherwise have been disposed of is the defining aspect of the upcycling pathway. Several potential sources of reduced impact or environmental benefit can be associated with the capture of wasted food for human consumption. First, production impacts can be assumed to be zero, because as a former waste product, wasted food enters the management pathway with no environmental burden. In reality, production impacts are not zero, but these impacts are associated with (or allocated to) the non-wasted portion of the product. For example, in the case of broccoli floret production, the impacts are associated with the production of the florets; they are not associated with the waste material (stems and leaves), which can be turned into a new product such as broccoli leaf flour through upcycling.

In addition, by capturing otherwise wasted food, the upcycling pathway captures an environmental benefit by avoiding additional food production. Researchers have applied this benefit in studying the conversion of nonmarketable fruits and vegetables to chutney (Eriksson and Spångberg 2017)[4] and the collection of broccoli leaves for conversion to broccoli powder (Eriksson et al. 2021).[5]

However, the line between upcycling and traditional food production can be difficult to define and justify. If wasted food gains value because it is upcycled, and producers come to rely on that input, then perhaps it ceases to be wasted food, and it would be improper to ignore the environmental impacts associated with its production or to consider avoided food production.[6]

Avoided Waste Management

Upcycling of wasted food that would otherwise have been disposed of can also result in environmental benefits from avoided waste management. However, these credits should not be included in an assessment of net impact/benefit when comparing impacts directly to those of alternative wasted food pathways because this would lead to double counting. This analysis excludes these credits from our comparative assessments. In addition, the upcycling pathway is not waste free, and a share of wasted food collected for upcycling may ultimately be disposed of or managed through another pathway. Estimates of waste from this pathway were not available in the literature.

TABLE 2-2. UPCYCLING: ENVIRONMENTAL DRIVERS

Source of Environmental Benefit	Avoided food production and wasted food management.
Option to Reduce Impact	Displace conventional production of high-impact foods.

2.4 Anaerobic Digestion

Anaerobic digestion (AD) is a process whereby organic materials are microbially degraded in an anaerobic environment (one lacking oxygen), converting the feedstock into biogas and nutrient-rich digestate (or biosolids). Wasted food can be digested alone or co-digested with other organic materials. Digestion may occur at a stand-alone digester, an on-farm digester, or a water resource recovery facility (WRRF) (U.S. EPA 2023b). When wasted food is co-digested with wastewater solids/sewage sludge at a WRRF the end product is considered biosolids (rather than digestate) as it is subject to federal and state regulations for biosolids management and use.[7] AD of wasted food that is collected via the sanitary sewer is not a part of this pathway; it is covered within the sewer/wastewater treatment pathway (see Figure 2-4). AD at WRRFs is included in this pathway only when additional wasted food (beyond that collected by sewer) is source-separated (i.e., separated by the generator) and collected separately from other waste streams and then delivered separately (i.e., not by sewer) to the AD unit.[8]

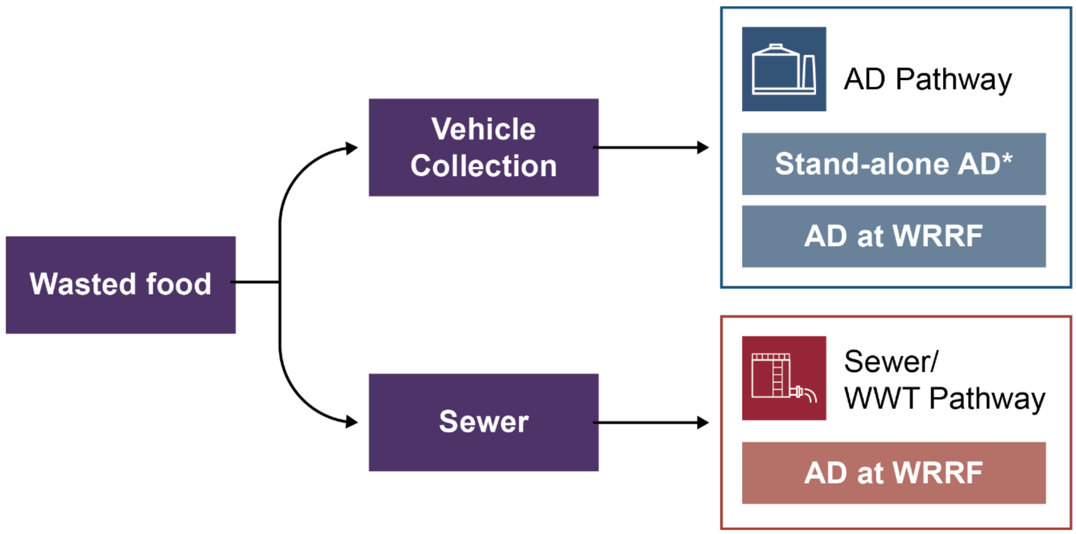

FIGURE 2-4. DIAGRAM DISTINGUISHING ANAEROBIC DIGESTION AND SEWER/WWT PATHWAYS

*Stand-alone facilities may exist only for food waste, or may co-treat other feedstocks (e.g., a digester for manure that receives food waste).

Figure 2-5 diagrams the AD pathway. For this pathway, wasted food is usually collected by vehicle and is subject to one or more pre-treatment steps that include removal of residual contamination, grinding, mixing, and/or moisture adjustment to produce a clean and consistent feedstock for digestion. The digestion process produces biogas (a mixture of gases including methane and carbon dioxide) and digestate (or biosolids, if located at a WRRF).

The biogas can be used to produce one of several energy products, such as heat, electricity, or biofuel.[9] These products can be used to offset other energy sources (e.g., heat from natural gas, electricity from the grid, diesel). The digestate or biosolids may be post-treated and applied to land as fertilizer or disposed of in a landfill or by controlled combustion.

In a voluntary survey of AD facilities processing wasted food in the U.S. (U.S. EPA 2023b), responding AD facilities (stand-alone, WRRFs, and on-farm, n = 106) could choose multiple options from a list of solid digestate or biosolids uses: dewatered and land applied; composted into a reusable/salable product, landfilled, processed into animal bedding, dried into a reusable/salable product (e.g. fertilizer), land applied as is with no dewatering or drying, incinerated, or other. According to the survey data, most AD facilities (63%) use land application, with or without post-treatment (including composting), as one management practice for digestate or biosolids. The majority of reviewed LCA studies assume that digestate is land applied.

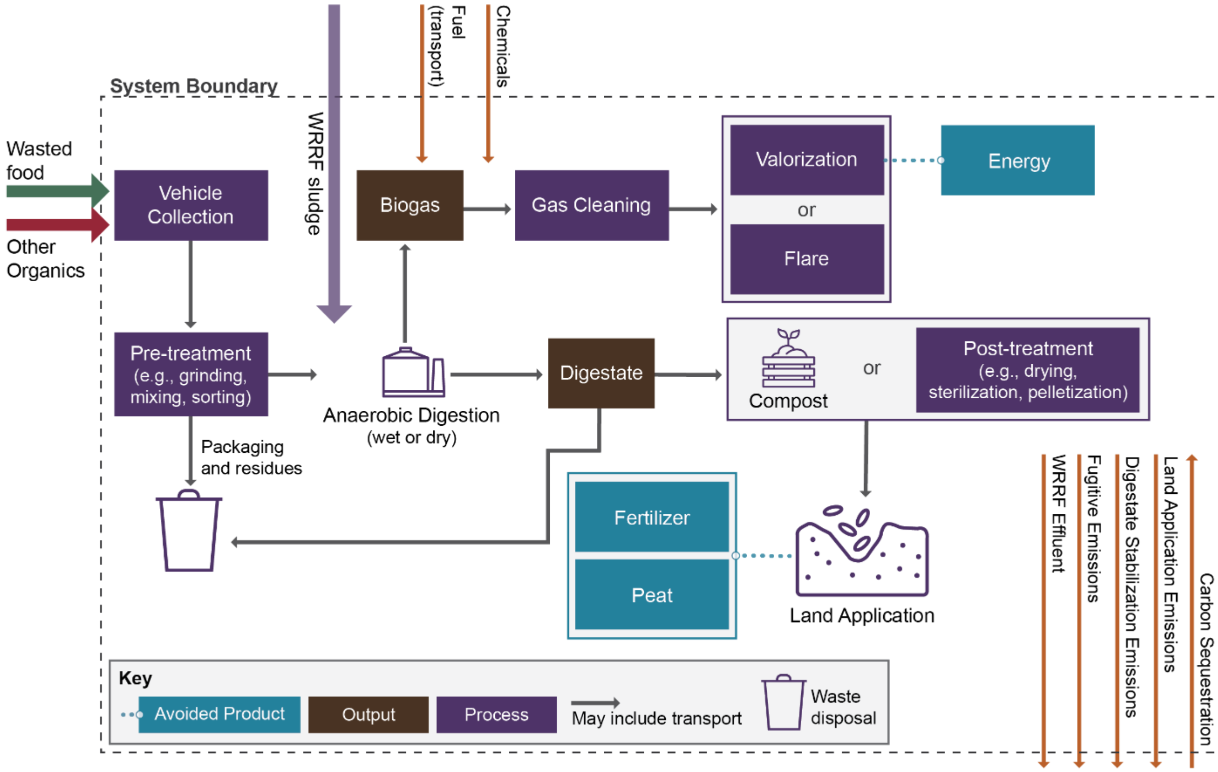

FIGURE 2-5. DIAGRAM OF ANAEROBIC DIGESTION AND CO-DIGESTION OF WASTED FOOD

Anaerobic Digestion Technologies

As noted, there are three types of AD facilities that process wasted food: stand-alone digesters, and co-digestion systems either at WRRFs or on farms (U.S. EPA 2023b). The terms “digestion” and “AD” are used throughout this report to refer to all digester types and feedstock mixtures unless otherwise specified. AD systems at WRRFs that do not receive additional wasted food delivered by vehicle are addressed in the sewer/WWT pathway (Section 2.10).

Stand-alone digesters are often industry-dedicated, located at a food or beverage manufacturing/processing facility and wholly dedicated to processing that facility’s waste. In contrast, multi-source stand-alone digesters (sometimes called “merchant” digesters) accept wasted food and other feedstocks from sources across their region. On-farm digesters are primarily dedicated to digesting manure and agricultural byproducts such as spent grain or crop residuals. Similarly, AD systems at WRRFs are primarily dedicated to digesting wastewater solids. Co-digestion of wasted food with manure or wastewater solids can increase biogas production, generate tipping fee revenue for farms or WRRFs, and provide another option for wasted food management.

Across all digester types, among the AD facilities processing wasted food that responded to the EPA survey, approximately 71% of digesters operate at mesophilic temperatures (U.S. EPA 2023b). Options exist for AD technologies across all types of facilities (e.g., continuously stirred tank reactor, upflow anaerobic sludge blanket), but the relative environmental performance of those different technologies is beyond the scope of this report.

TABLE 2-3. ANAEROBIC DIGESTION TECHNOLOGIES

Location	**Stand-alone**	Primarily built to process wasted food. Includes multi-source wasted food digesters and industry-dedicated digesters.
	On-farm	Primarily used for manure management. These digesters can co-digest wasted food.
	WRRF	Primarily used to digest wastewater solids. Wasted food can be transported to the AD unit by truck and co-digested with the wastewater solids (which may include wasted food transported down the drain through the sewer to the WRRF).
Type	**Wet**	Digesters processing feedstocks with <15% solids.
	Dry	Digesters processing feedstocks with >15% solids.
Temperature	**Mesophilic**	Digester operation at 30–40°C.
	Thermophilic	Digester operation at 50–60°C.
Outputs	**Digestate**	From stand alone or on-farm digesters. Residue from digestion process which can be separated and treated through drying, de-watering, or composting which can be used in animal bedding, land application, other products, or landfilled or incinerated.
	Biosolids	Solid discharge of sewer/WWT treated through various technologies, including AD. An output of AD systems at WRRFs which can be incinerated, landfilled, or land applied with or without post-treatment.
	Energy	Electricity, heat, and biofuels which can be used for heating and powering the facility or sold as electricity or fuel.

Prevalence

The EPA survey of U.S. AD facilities processing wasted food indicates that stand-alone digesters processed over 90% (by weight) of digested wasted food in 2019. This is largely due to stand-alone digesters processing large amounts of beverage processing waste (e.g., from breweries), which is heavier per unit of volume than other types of wasted food such as residential food scraps. If beverage processing waste is excluded, then AD at WRRFs processed the majority of digested wasted food in 2019—about 80% according to EPA survey responses.

There are approximately 70 stand-alone digesters in the U.S. (U.S. EPA 2023b), and there are roughly 16,000 WRRFs in the U.S., of which about 1,270 have AD (American Biogas Council 2018). Of those AD facilities at WRRFs, around 10% are estimated to co-digest wasted food (U.S. EPA 2023b). EPA's AgSTAR program identifies 317 farms in the U.S. with anaerobic digesters, approximately 25% of which are estimated to co-digest wasted food (U.S. EPA 2023b). The American Biogas Council tracks biogas production in the U.S. and identifies over 2,200 biogas projects, including 250 on-farm digesters, 1269 WRRFs with AD, and 66 stand-alone digesters (all other biogas projects are at landfills). Discrepancies in the numbers and types of AD facilities are likely due to differences in tracking and/or data year.

Processing wasted food can add value to an AD system at a farm or WRRF by synergistically increasing biogas production.

Anaerobic Digester Operation

Studies assessing the environmental impact of wasted food management via AD point to operation of the digestion reactor itself as one of the primary sources of impact (Woon et al. 2016; Edwards et al. 2017; Khoshnevisan et al. 2018; Pace et al. 2018). Studies consistently identify energy demand and fugitive emissions from digester operations as impacts; depending on their scope, some studies also include chemical use or infrastructure materials. Of the studies clearly reporting electricity demand for digester operation, three-quarters report values within the range of 10–58 kWh per metric ton of waste processed. Additional heat energy may be required to achieve optimal operating temperatures, but few studies document a specific quantity, sometimes simply noting that residual heat from biogas combustion is used.[10] Among studies that do report heat demand, the values are in most cases at least double the AD electricity demand. Within the reviewed studies, estimates of fugitive methane losses typically vary between 1% and 5% of produced biogas, with some studies assuming negligible losses. However, recent estimates of methane emissions from centralized WRRFs are 1.9 times higher than previous estimates (Moore et al. 2023), suggesting that existing studies may systematically underestimate methane from WRRFs. These recent comprehensive studies (Moore et al. 2023; Song et al. 2023) do not address the contribution of wasted food to WRRF emissions, so further study is needed.

Studies report the potential of wasted food co-digestion to synergistically enhance biogas production at WRRFs, with the potential to increase net energy production (Xu et al. 2015; Edwards et al. 2018).[11] A case study of a Massachusetts WRRF demonstrated positive environmental outcomes when excess AD capacity is utilized for wasted food co-digestion (Morelli et al. 2020).[12] Usack et al. (2018) report that co-digestion of wasted food with animal manure leads to a reduction in total life cycle emissions of AD when compared to digestion of manure only.

Biogas Production

Biogas produced by AD has numerous uses and destinations in the United States. These include producing heat, electricity, and mechanical power for the AD facility, producing electricity for the grid, compression to vehicle fuels to be sold or used by the facility, and producing renewable natural gas. In the EPA survey of AD facilities processing wasted food, 95% of stand-alone, 67% of on-farm, and 86% of WRRF digesters reported using some or all the produced biogas onsite. When given a list of biogas uses with the option to choose multiple uses, the most commonly selected biogas use was combined heat and power (CHP) (56% of responding facilities), followed by fuel for boilers to heat digesters (38%) or to heat other parts of the facility (38%) (U.S. EPA 2023). EPA (2023b) reports that 32% of stand-alone digesters, 50% of farms, and 67% of WRRFs use some of or all the biogas produced for CHP.

Avoided energy benefits associated with the use of AD biogas to displace electricity, heat, and biofuels were consistently found to reduce environmental burdens and, in many instances, led to net environmental benefits. Across a subset of reviewed LCA studies for which electricity recovery numbers were reported, gross electricity production per metric ton of wasted food processed is between 100 and 420 kWh (see Appendix Table B-1). While a considerable quantity of heat can be recovered in addition to the electricity produced, many studies exclude an environmental credit for recovered heat as it is used to operate the digester and no uses are available for excess heat in the studied scenario. Avoided energy benefits are realized on the basis of net energy production (i.e., production minus the energy demand required to operate the digester and associated processes).

Environmental benefits of avoided energy production depend on several factors, including the quantity of energy produced per unit of wasted food, the specific fuel mix that is being displaced, and whether avoided burden credits are or can be reasonably applied for the full quantity of energy recovered.[13] Several authors observed major reductions in GWP impact as a result of biogas production and utilization (Xu et al. 2015; Woon et al. 2016; Khoshnevisan et al. 2018). Individual studies also reported other major reductions in environmental impact, including reductions in acidification potential (Edwards et al. 2017) and ecotoxicity and human toxicity (González et al. 2020). These findings are supported by the analysis of Chiu and Lo (2018), who looked at key parameters for wasted food treatment uncertainty assessment and concluded that biogas production rate, biogas methane content, methane lower heating value, energy recovery efficiency, and energy demand for biogas upgrading all have a statistically significant effect on environmental performance.[14]

Murphy and Power (2006) found that, in instances where a market for CHP heat is available, use of biogas for that purpose is economically preferable to use as transportation fuel. If a market for CHP is not present, then upgrading biogas to transportation fuel is a better option. Other analyses have shown that upgrading biogas to renewable natural gas (RNG) for use as a transportation fuel leads to higher environmental impacts, compared with the same AD system paired with CHP (González et al. 2020) . This finding is due to the energy demand of the additional processing steps required for the RNG upgrade. Utilization of surplus heat (beyond that required for process operation) can offset other forms of heat production such as from kerosene (Murphy and Power 2006), fuel oil (González et al. 2020), natural gas or propane. Novel uses of surplus thermal energy are also possible such as use in greenhouses (Lin et al. 2018), allowing for extension of growing season.

Digestate Post-treatment and Use or Disposal

In the U.S., the liquid fraction of digestate (i.e., liquid separated through dewatering) is commonly applied to agricultural lands as fertilizer (Alexander 2012; U.S. EPA 2023b). This is particularly true for on-farm AD facilities which mainly co-digest wasted food with animal manure. Other uses of the liquid fraction include recirculating digestate through the digester or sending the digestate to wastewater treatment. In the case of AD at WRRFs, the dewatering liquid is recirculated through the wastewater treatment process and eventually released as effluent. The solid fraction of digestate or biosolids can be unprocessed, composted, de-watered, or dried and common uses vary by facility type. In some cases, digestate is not separated into liquid and solid fractions and is land applied as a slurry. In the EPA survey of AD facilities processing wasted food, most often on-farm digesters report creating animal bedding, WRRFs with AD report de-watering and land applying, and stand-alone digestors report composting to create a salable product (Alexander 2012; U.S. EPA 2023b). Of the three AD facility types, WRRFs with AD are most likely to landfill solid digestate or biosolids – 33% report landfilling some or all biosolids (U.S. EPA 2023b). When digestate is applied as fertilizer for crop production or used as animal bedding, a renewable resource is recovered and used. However, depending on contamination and other limiting factors such as access to agricultural land and the seasonality of land application, digestate might be treated as a waste and disposed of (e.g., at landfill or by controlled combustion).

Without post-treatment, digestate or biosolids can be disposed of in a landfill, burned in a controlled combustion facility,[15] or applied to agricultural lands in accordance with state and federal regulations. The means by which compost, digestate, or biosolids is transported from generation to application is important. As shown in Brown and Beecher (2019), transport of Class B biosolids in trucks may have lower transport emissions than retail compost transport in personal vehicles. However, these transport emissions are likely minor relative to other stages of wasted food generation, handling, and processing.

Composting is the most common post-treatment process for digestate considered within the reviewed literature. Benefits of the composting process include waste stabilization, vector control, and volume reduction. Most studies evaluating post-treatment composting consider subsequent land application of generated compost. A study looking at both composting with land application and controlled combustion of digestate found that AD with composting led to better average environmental performance in terms of GWP and particulate matter formation, whereas AD plus controlled combustion led to reduced human toxicity impact as a result of fewer heavy metals being released to the environment (Mayer et al. 2020). Mayer et al. report that the GWP of AD with composting is driven by GHG emissions during post-treatment steps (composting and land application) but is partially offset when the digestate is used as a substitute for synthetic fertilizer.

Land application of (uncomposted) digestate or biosolids produces environmental benefits and impacts in the form of energy use for transportation and application and emissions from application. Depending on post-treatment methods, digestate and biosolids maybe have a high moisture content relative to other materials (e.g., compost; Grigatti et al. 2020) and therefore a higher weight, making them more energy-intensive to transport. The method and timing of digestate land application affect the magnitude of emissions to air and water, as do weather conditions following application (Tiwary et al. 2015). Most nutrient emissions occur during digestate storage and land application (Usack et al. 2018).[16] The literature presents concerns regarding long-term soil contamination from digestate or biosolids land application, particularly in cases where wasted food is co-digested with wastewater solids or other potentially contaminated waste streams. Chemical contaminants include heavy metals, pharmaceuticals and emerging contaminants, such as per- and polyfluoroalkyl substances (PFAS). Note that some metals, such as zinc and copper, are micronutrients. Cycling of these metals can be beneficial, provided they do not accumulate above toxicity thresholds. Physical and chemical contamination of digestate and biosolids can limit their potential for beneficial use. See the discussion on soil health in Section 4.5 for more information.

The potential benefits of digestate land application include avoided fertilizer or peat production, carbon sequestration, and positive contributions to soil health (see Section 4.6). Findings related to the environmental benefits of avoided fertilizer production vary widely across the reviewed studies. Several authors found negligible or minor environmental benefits (Righi et al. 2013; Khoshnevisan et al. 2018). González et al. (2020) identified avoided production of nitrogen fertilizer as the primary source of environmental benefit in acidification, eutrophication, and GWP impact categories. LCA studies typically assume a nitrogen substitution ratio with synthetic fertilizer of between 20% and 50% of nitrogen content. Phosphorus and potassium content is usually assumed to have a similar plant availability to the phosphorus and potassium in mined or synthetic fertilizer products, and therefore a substitution ratio of 1:1 is common.

As an alternative to or in addition to fertilizer replacement, a small number of LCA studies look at using digestate or compost to substitute the use of peat growing media. These studies found that the GWP benefits of avoiding peat use were similar to Morris et al. (2017) or greater than avoided fertilizer benefits (Levis and Barlaz 2011).

Substituting digestate or compost for synthetic fertilizers and/or peat growing media is assessed inconsistently within the LCA literature. Because of the differences in nutrient (N, P, K) plant availability and organic carbon stability and content, digestate and compost may be assigned different substitution ratios when displacing synthetic fertilizer and/or peat growing media.

N substitution:
Similar substitution ratios are assumed for compost and digestate: 20-50% of synthetic N fertilizer. Digestate typically has higher readily available N content (N that can be immediately used by plants), suggesting it may more often fall in the higher end of that range. Compost has lower readily available N content, providing a slower release of N over time, and more often falling in the lower end of that range.

P and K substitution:
For compost and digestate, the same ratio is assumed: 1:1.

(Levis and Barlaz 2011; Yoshida et al. 2012; Stehouwer et al. 2022; Morris et al. 2017)

A portion of the carbon in land-applied digestate is often sequestered beyond 100 years, reducing net GHG emissions according to many GHG accounting frameworks. A review of LCAs of wasted food management identified a range of assumptions that between 10 and 27 kg of CO_2eq is sequestered per metric ton of wasted food digested and land applied (Bernstad Saraiva Schott et al. 2016). For more information regarding the range of carbon sequestration benefits identified see Appendix B-1-1. Adding carbon to soils has benefits beyond sequestration, including improving soil structure, increasing water holding capacity and benefitting soil microbes. But there are no standardized metrics in reviewed LCAs for soil health beyond carbon sequestration; as such, these benefits are not included in the comparisons in Sections 3.4 and 3.5. See Section 3.6 for more information on soil health.

TABLE 2-4. ANAEROBIC DIGESTION: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Energy demand of digester operation.
	Emission of methane associated with biogas leakage during digestion or processing.
	Emission of CAPs from biogas combustion.
	Additional energy and chemical demand for biogas cleaning and upgrading.
	Energy use for post-treatment of digestate or biosolids.
Sources of Environmental Benefit	Reduced production and use of fossil energy due to biogas energy recovery.
	Displacement of conventional fertilizer and peat production if land application of digestate.
	Long-term carbon sequestration and contributions to soil health if land application of digestate (Section 4.6).
Options to Reduce Impact	Optimize biogas production.
	Monitor and minimize fugitive emissions of biogas.
	Use enhanced emission controls on electrical or CHP engines
	Avoid landfilling digestate; when digestate is land applied, follow best management practices regarding digestate application rate, timing, and method to minimize potential damage from agricultural emissions or soil contamination (UNH Extension 2017).
Other Considerations	Digestion technology: Energy demand and methane yield vary depending on digestor type, number of stages, and temperature conditions.[17]
	Net energy production is dependent on energy conversion efficiency and energy demand of AD and post-treatment processes.

2.5 Animal Feed

The animal feed pathway diverts wasted food that was originally intended for human consumption or is a byproduct of human food production (e.g., fry oil, corn husks) to animals, either directly after harvesting or after processing. This pathway does not include animals grazing unharvested fields, which is included under the unharvested/plowed in pathway. Use as animal feed is more common for the U.S. food manufacturing sector than for the retail, food service, and household sector (U.S. EPA 2023a). The nutritive and economic value of animal feed derived from wasted food ranges by feedstock and processing techniques. Wasted food is nutrient rich. When compared with traditional maize used for animal feed, wasted food can have double the crude protein content. Incorporation of wasted food in animal feed can also reduce feed costs and tipping fees for the waste producer (Dou et al. 2018).

Figure 2-6 diagrams the animal feed wasted food management pathway. Wasted food pre-treatment and processing steps include de-packaging, sorting, grinding, and drying; sterilization may be accomplished through heating wasted food or through preservation in a silo, which does not require energy inputs. Finished feed must be free from contamination to ensure animal health (Redlingshöfer et al. 2020). Using wasted food for animal feed can displace the need for primary agricultural production of animal feed from sources such as soy meal, barley, and corn (Tufvesson et al. 2013; Albizzati et al. 2021b).

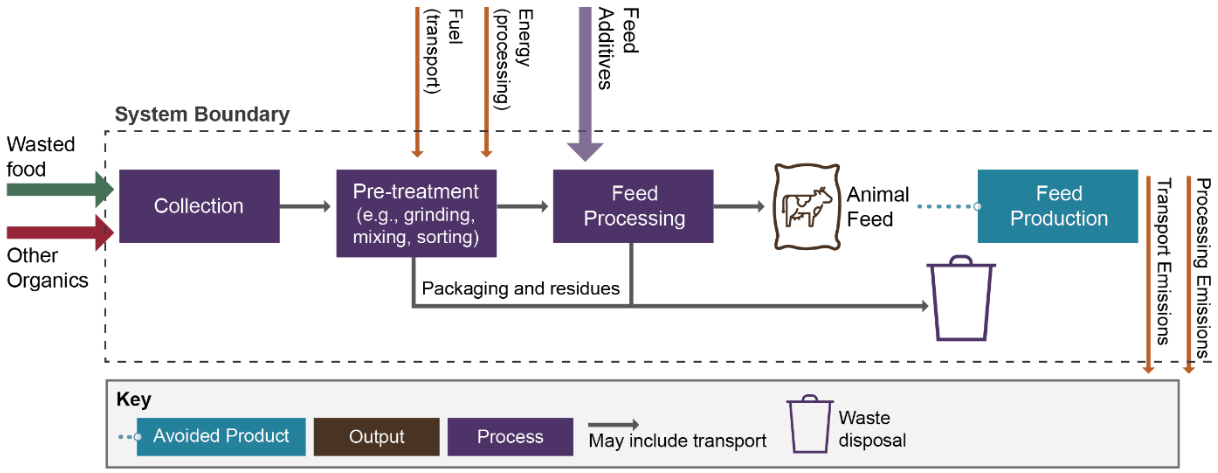

FIGURE 2-6. DIAGRAM OF WASTED FOOD CONVERSION TO ANIMAL FEED

Animal Feed Production Processes

The pathway results in a broad range of potential products from less processed to very highly processed. Wet animal feed is a minimally processed high-moisture feed product, sometimes requiring heat treatment. Dry feed requires further processing to dewater, dry, and pelletize wasted food (Dou et al. 2018).

The environmental impacts of animal feed production depend on the type of wasted food and the type of animal feed being produced. For the same wasted food feedstock, wet animal feed typically undergoes less processing and has reduced energy demand relative to dry animal feed. The high moisture content of wasted food requires increased energy for dry feed dehydration (Salemdeeb et al. 2017). Therefore, dry feed should ideally be produced from wasted food with lower moisture content. Nine of the 14 reviewed animal feed LCA study scenarios pertain to the production of dry animal feed. There were two studies that modeled production of both wet and dry feed from municipal wasted food (Salemdeeb et al. 2017; Albizzati et al. 2021a).

Most of the reviewed studies modeled the direct production of animal feed; animal feed can also be indirectly produced by bioconversion (i.e., feeding wasted food to insects, which are then further processed to produce animal feed). Bioconversion presents opportunities and challenges relative to direct feed production, as the intermediate organisms may be able to accept wasted food with a wider range of moisture, contaminants, and pathogens, but require energy and other inputs.

TABLE 2-5. ANIMAL FEED TECHNOLOGIES

Types	**Dry Feed**	Dry matter content of about 78% (Salemdeeb et al. 2017).
	Wet Feed	Dry matter content of about 30% (Salemdeeb et al. 2017; Albizzati et al. 2021b).
	Protein Concentrated	A dried animal feed with high protein content, similar to soybean meal (46–50% dry matter protein content) (Corona et al. 2018).
	Other	Examples include bakery meal and black soldier fly larvae meal (black soldier flies are fed with wasted food).

Avoided Conventional Feed Production

Environmental credits associated with avoiding conventional animal feed production are a source of considerable environmental benefit for this wasted food pathway and often lead to favorable environmental performance when compared to other pathways.[18,19] A published literature review (Dou et al. 2018) concluded that wasted food is rich in major nutrients for feeding livestock and, when treated properly, is a safe and sustainable alternative to growing dedicated feed crops. Potential benefits include pollution avoidance and reduced use of land and other resources to grow feed crops, according to the review.

Hygiene Considerations

Because raw wasted food intended for animal consumption may contain active disease organisms, several methods of wasted food sterilization have been developed to prevent disease transmission (Dou et al. 2018). Heat treatment of wasted food in animal feed is widely enforced globally (Dou et al. 2018).

Regulations have been developed in the U.S. to assure safety of animal feed. The majority of these regulations pertain only to wasted food that contains animal products. The FDA’s Ruminant Feed Ban Rule (U.S. FDA 2022) prohibits feeding cows, sheep, goats, and other ruminants any feed containing mammalian protein. For pigs, under USDA’s Swine Health Protection Act (SHPA) (USDA 2020), raw wasted food intended for swine consumption must be sterilized via heat treatment (100°C for at least 30 minutes) if it contains or has had contact with meat, poultry, or fish. One study in Spain found that heat treatment at 65°C for 20 minutes adequately reduces E. coli, Salmonella, and S. aureus to levels that are considered safe for animal consumption (García et al. 2005). Another study in China reported that hydrothermal treatment at 110°C for 60 minutes eliminates S. aureus, total coliforms, and total aerobic bacteria (Jin et al. 2012). State governments may choose to enforce more stringent laws for sterilization procedures and go beyond the minimum standards set by U.S. regulation. Additional regulations vary by U.S. state, with 15 states placing more strict guidelines on animal feed than are dictated by federal regulation. Most of this regulation specifically addresses feeding swine. Nine states have a prohibition on feeding animal or vegetable waste to swine (Leib et al. 2016).

TABLE 2-6. ANIMAL FEED: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Energy demand (e.g., wasted food dehydration for dry feed).
	Transportation (wet feed has higher transportation impacts due to its higher mass).
Source of Environmental Benefit	Avoided feed production impacts.
Options to Reduce Impact	Select animal feeds with lower energy demand, such as locally produced wet feed.
	Select pre-treatment methods with the lowest energy and chemical demand.
Other Considerations	Feed type dictates production method and associated impact.
	To justify the avoided feed benefit, the wasted food-based feed must be a suitable substitute for the type of animal feed being displaced.

2.6 Composting

The composting pathway breaks down wasted food, and co-treated wastes, via microorganisms in an oxygen-rich (aerobic) environment to produce a stable, organic material that can be land applied. Successful composting requires a balance of nitrogen-rich and carbon-rich ingredients, adequate but not excessive moisture content, and oxygen flow to achieve desired pile temperatures and aerobic conditions (U.S. EPA 2021h). Compost systems are deployed at a wide variety of scales from small home compost piles to large, highly managed industrial facilities.

Figure 2-7 diagrams the composting management pathway. Wasted food is generally one of multiple feedstocks in a composting operation due to its high nitrogen content. Most composting facilities that accept wasted food require that wasted food be source-separated and separately collected. Some also require pre-treatment (sorting and screening) to remove potential contaminants, along with grinding and blending of feedstocks.[20] The composting process requires some form of turning or aeration to maintain optimal aerobic conditions to reduce the release of airborne and waterborne pollutants (Zhao et al. 2022; Peng et al. 2023). Post-treatment of mature compost may include screening, sorting, grinding, chipping, blending, and/or bagging. The output of the composting process is a nutrient-rich, carbon-rich substrate that can be used as a soil amendment and/or mulch, depending on the carbon-to-nitrogen ratio and the desired end use.

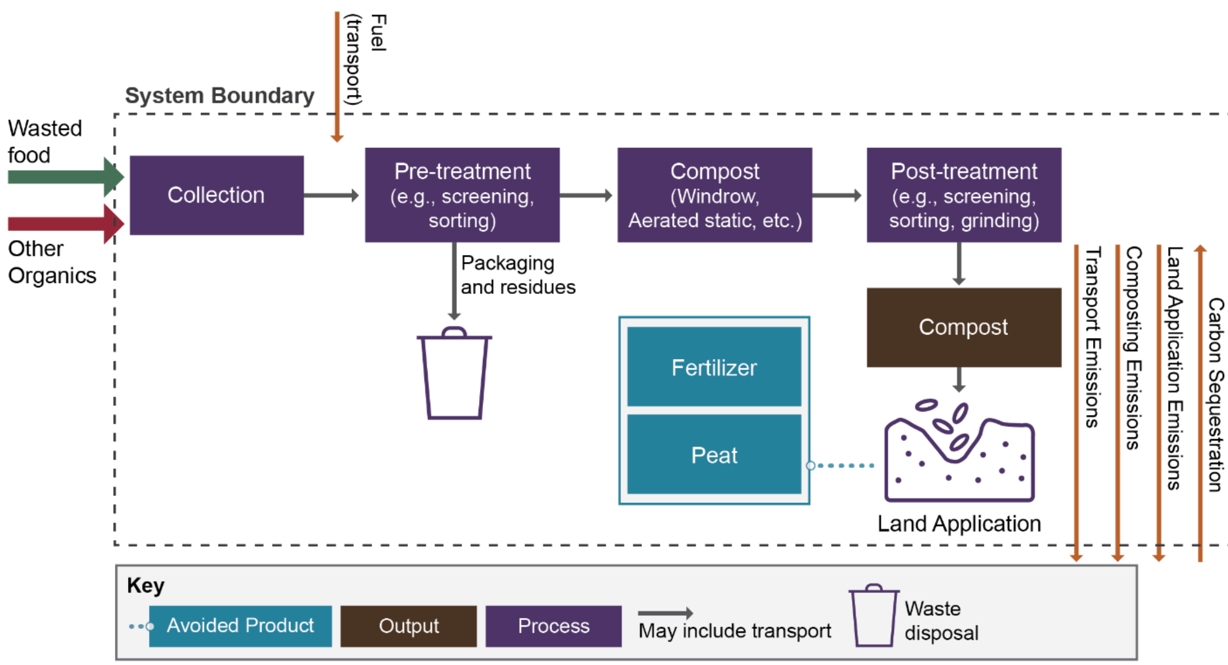

FIGURE 2-7. DIAGRAM OF WASTED FOOD COMPOSTING, INCLUDING A PROMINENT CO-WASTE SUCH AS MUNICIPAL GREEN WASTE

Composting Technologies

There are three common methods of composting in the United States: windrow, aerated static pile, and bioreactor. Many reviewed LCA studies use the term in-vessel to describe bioreactor composting systems.

TABLE 2-7. COMPOSTING TECHNOLOGIES

Types	**Turned Windrow**	Processing of compost in long piles using mechanical turning for aeration.
	Aerated Static Pile (ASP)	Processing of compost in large piles using active and passive aeration. Passive aeration systems are not likely to be used at large scales due to the time and land area requirements (Platt et al. 2014).
	Bioreactor	Processing of compost in an enclosed vessel, often under controlled conditions. Bioreactor systems come in multiple configurations including tunnel, agitated-channel, and rotary drum (Platt et al. 2014). They may be heated to speed decomposition.
Outputs	**Compost**	Nutrient- and carbon-rich soil amendment which can be applied in agriculture, transportation projects, green infrastructure, and other land applications.

Prevalence and Value

There are between roughly 3,000-5,000 composting facilities in the United States, based upon estimates from the EPA Excess Food Opportunities Map (3,887) and BioCycle (4,713) (Goldstein 2017; U.S. EPA 2023d). According to the Excess Food Opportunities Map, at least 16% of U.S. composting facilities accept wasted food (U.S. EPA 2023d). This map includes mostly municipal, commercial, and industrial composting facilities; community composters, institutions and farms with on-site composting operations also accept food scraps.

Of these 3,000-5,000 composting systems, more than 50% are windrow (Platt et al. 2014). The U.S. Composting Council and Environmental Research and Education Foundation (EREF) recently surveyed composting facilities. Of 293 early respondents, over 50% use windrows, ~28% use ASP, ~22% use static piles, and less than 10% use in-vessel (EREF 2023).

The economic value of compost is highly affected by the compost quality, end use, and market (Rynk et al. 2022). On average, compost sells for about $10 U.S. dollars (USD) per cubic yard (Goldstein 2020). A survey of nearly 100 U.S. composts made in part from wasted food demonstrated mean content of 1.5% total nitrogen, 0.37% phosphorus and 0.90% potassium, with mean carbon:nitrogen (C:N) ratio of 15.8 (Stehouwer et al. 2022). In a study of approximately 150 compost manufacturers in the U.S., the U.S. Composting Council estimated the economic value of nutrients recycled through composting to be about $96.6 million USD for nitrogen, phosphorus, and potassium and $20.5 million USD for carbon (U.S. Composting Council 2021). Contamination, either physical or chemical, may also affect compost value. The effort to remove these contaminants can increase the difficulty of processing the wasted food, and not removing them can negatively impact compost quality. Contamination levels in finished compost may also be regulated. California, for example, limits physical contamination to less than 0.5% by weight for physical contaminants larger than 4 millimeters (State of California 2016a). Metal contamination is also regulated with species such as copper and zinc being limited (State of California 2016b).

Aeration Energy

Composting requires the use of energy for aeration and equipment operation. Among methods commonly used for wasted food composting, windrow composting tends to have the lowest energy demand. In windrow compost systems, diesel fuel tends to be the dominant energy source and the amount of energy consumption depends on turning frequency and the type of equipment used. Actively aerated compost methods, such as an ASP or bioreactor, have modest but relatively higher energy demand that tends to come in the form of electricity consumption (Levis and Barlaz 2011). Bioreactor composting systems, especially those operated at above-ambient temperatures, can have considerably greater energy requirements (Lin et al. 2018). Ultimately, energy demands depend on many factors, such as the scale of the operation, the composting method, the equipment used, and feedstock characteristics. Renewable energy may be used by composters and heat recovery may save energy as well.

Looking Forward: Composting Process Emissions

To limit GHGs, California has set target reductions for short-lived climate pollutants (SLCP), like methane and VOC's, that can be generated from organic waste degradation, including composting.[a] SLCP will have to be lowered by 75% by 2025. SLCP can vary greatly between sites, with reported values in California compost facilities ranging from 1-10 lb VOC/wet ton waste (not entirely food waste).[b] Biofilters can help reduce those emissions by 90%+.[c]

[a](California 2023)
[b](CARB 2015)
[c](Brown et al. 2020) article

Process Emissions

Process GHG emissions are associated with bacterial activity in the compost pile (CARB 2017). Large amounts of carbon, nitrogen, phosphorus, and other lower-concentration chemical compounds are cycled through composting processes. In compost piles, most feedstock carbon is either retained in the final product or released to the atmosphere as carbon dioxide. A small fraction of carbon is released as methane.[21, 22] Similarly, feedstock nitrogen is either retained in the finished compost or liberated as ammonia, nitrous oxide, nitrate, or other nitrogen-containing compounds. Emissions released during composting can be a major or minor source of environmental impact depending on impact category and the facilities' operating practices or the modeling assumptions used in LCA studies.[23] For instance, VOC production can be reduced by maintaining composting conditions that are aerobic, with high C:N ratios and moderate moisture (Delgado-Rodríguez et al. 2011). Optimizing emission reductions across all potential pollutants is challenging, and further research is needed to evaluate tradeoffs.[24] Pile structure, chemical composition, moisture content, and other factors can affect composting speed, quality, and environmental performance (Lin et al. 2018).

Compost Land Application

Application of compost to soil can result in numerous benefits to agricultural systems and residential or commercial landscapes. Benefits of compost use include carbon sequestration[25] and avoided production and use of chemical fertilizer or peat.[26] Impacts associated with compost land application typically include transportation, energy use during spreading/incorporation, and emissions to air and water. Organic forms of nitrogen present in compost typically take longer to become plant-available than the nutrients present in synthetic fertilizers. To reflect this fact, LCA studies typically assume a nitrogen substitution ratio with synthetic fertilizer of between 20% and 50%. Phosphorus and potassium content in wasted food compost has similar plant availability to the phosphorus and potassium in mined or synthetic fertilizer products, and therefore a substitution ratio of 1:1 is common. Nitrogen substitution ratios reflect short-term plant availability and may therefore underestimate avoided fertilizer benefit over longer time scales. Rates of mineralization and the timing of nutrient plant availability depend on the matrix in which nutrients are delivered, the method of application, and other site-specific factors. While LCA is therefore not an appropriate tool to model nutrient and soil dynamics, the aggregate LCA results of this review can nevertheless help to identify trends.

While compost typically has less readily available nitrogen than digestate or biosolids, its total N content varies based on feedstock and composting conditions. Literature reviewed shows compost can contain similar (WRAP 2016; Grigatti et al. 2020) or greater (Stehouwer et al. 2022) amounts of total nitrogen compared to digestate, though often contains less total N than biosolids (Brown et al. 2011). Compost releases this nitrogen slowly over time, supporting long-term soil fertility. See Section 4.6 for a more detailed discussion of benefits.

While most LCA studies include environmental credits for avoided fertilizer production, the magnitude of this benefit varies. Morris et al. (2017) identifies a moderate to major reduction in gross GWP impact as a result of avoided fertilizer production and include an additional benefit for displacing peat soil amendment. Only a small number of studies look at using digestate or compost to substitute the use of peat growing media. Compost is assumed to displace peat use 1:1 on a volume basis (Levis and Barlaz 2011). These studies found that the GWP benefits of avoiding peat use were similar to (Morris et al. 2017) or greater than avoided fertilizer benefits (Levis and Barlaz 2011). A study of mixed wasted food and green waste composting found major reductions in gross impact for GWP, particulate matter formation, and carcinogenic effects, with minor reductions in most other impact categories attributable to avoided fertilizer production (Keng et al. 2020). However, a study that examined composting the organic fraction of MSW (OFMSW) found minor to negligible environmental benefit associated with avoided fertilizer production for most impact categories, including GWP (Righi et al. 2013). The median GWP benefit from avoided fertilizer production, across the eight studies for which a disaggregated fertilizer credit could be identified, is -28 kg CO_2eq per metric ton of composted wasted food.

While all the reviewed LCA studies assume that an avoided fertilizer production credit is applied for compost produced from wasted food, in real world circumstances, these benefits will only be realized when the use of compost reduces fertilizer production and use. A survey of commercial composters in the U.S. by the U.S. Composting Council reports the following uses of the total compost produced: landscaping (40%), followed by agriculture, topsoil blenders, parks and roads, and general public (10% each), retail (9%), commercial horticulture (5%), golf courses and nurseries (2% each), and mine reclamation (1%); plus 1% as landfill cover (U.S. Composting Council 2021). In total the survey finds that 100% of compost is land applied to soil (Gilbert 2023), with 1% of that applied as landfill cover which may be considered disposal. However, it is unlikely that compost displaces synthetic fertilizer in all these applications. In some cases, it may be used only as a mulch or clean fill. This survey does not include on-farm composting where compost is used by the farm and thus likely underestimates agricultural use of compost.

The benefits of compost as a soil amendment include the potential for increased carbon sequestration, reduced erosion, increased water retention, and increased soil microbial activity. Beyond use in agriculture and landscaping, compost's water retention and filtration properties are beneficial in stormwater management and green infrastructure. Compost use has proven benefits for urban soils and in remediation. For example, composting can reduce the bioavailability of heavy metals by forming stable chemical bonds and trapping metals in soil aggregates (Goss et al. 2013; Urra et al. 2019; O'Connor et al. 2021; Thakali and MacRae 2021). All these benefits are largely due to compost being rich in carbon and highly effective at building soil organic matter, which is crucial for soil health and function. Compost can also suppress some plant diseases and pests and increase crop yields.

Approximately half of reviewed studies on composting and compost use include a carbon sequestration benefit for the GWP impact category. If the soil is in agricultural use, additional benefits can include reduced fertilizer and herbicide use, reduced water consumption, and increased crop yield (CARB 2017). While the benefits of compost noted here are well accepted, the LCAs reviewed contained no standardized metric for soil health improvement other than carbon sequestration. As such, these benefits are not included in the comparison of LCA studies in Sections 3.4 and 3.5. However, organic matter content is a widely accepted and well-studied metric for overall soil health and is considered elsewhere in this report, including in Section 3.6 and in Chapter 4.

Net Energy Demand

Unlike some other wasted food pathways considered in this review, composting does not typically directly capture energy embedded in wasted food and put it to beneficial use. Composting operations have a net positive energy demand and are negatively influenced when using energy sources with high environmental impacts (Yoshida et al. 2012). A summary of emerging science by Lin et al. (2018) shows the potential to capture low-grade heat energy from compost piles for greenhouse and water heating applications. The authors concluded that if viable systems are developed for capturing a share of composting's waste heat, the energy balance and environmental performance of compost systems could be improved.

TABLE 2-8. COMPOSTING: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Energy demand of aeration.[27]
	Air emissions during composting. The primary pollutants of concern include ammonia, nitrous oxide (N_2O), methane, and other volatile organic compounds (VOCs).[28] Emissions can vary widely depending on composting method, management, and environmental conditions.[29]
Sources of Environmental Benefit	Long-term carbon sequestration and contributions to soil health from land application of compost (Section 4.6).
	Displacement of peat and conventional fertilizer production due to land application of compost.[30]
Options to Reduce Impact	Manage or select systems for energy efficiency. • Windrow systems have low energy demand. • Optimize aeration in actively aerated systems.
	Install bio-filters, which oxidize methane to carbon dioxide and reduce odors. Use of Gore covers also controls odors and reduces emissions of ammonia and volatile organics (Levis and Barlaz 2011).
	Follow best management practices regarding compost application rate, timing, and method to minimize potential damage from agricultural emissions.[31]
Other Considerations	Successful composting requires maintaining aerobic conditions, appropriate moisture content, and a favorable carbon to nitrogen ratio (C:N ratio) (Risse and Faucette 2017).[32]
	Composting's lack of energy recovery limits environmental benefit relative to pathways with energy recovery.

2.7 Controlled Combustion

This pathway encompasses combustion of wasted food in a controlled manner (i.e., incineration) to reduce solid waste volumes and recover energy. This report examines only controlled combustion with energy recovery. By the early 1990s, the majority of controlled combustion facilities in the U.S. had installed emissions control devices and were recovering energy (U.S. EPA 2023c). Emission control devices equip modern MSW incinerators to remove particulates, acid gas, and often dioxin and mercury (National Research Council 2000).

Figure 2-8 diagrams the controlled combustion management pathway and the potential for co-firing with other wastes. Wasted food is typically combusted in a mixture with other solid wastes due to its high moisture content (Trabold and Babbitt 2018). Waste feedstocks that include wasted food, with a high moisture content and low heating value, may require supplemental fuel or pre-drying for complete combustion. Even if a system does not have an explicit pre-drying step, the energy demand of drying would be reflected in lower net energy production.

Controlled combustion byproducts (i.e., fly and bottom ash) may require post-treatment (e.g., drying and heavy metal removal) before beneficial use or disposal. Both kinds of ash are typically landfilled in the United States (U.S. EPA 2023c).

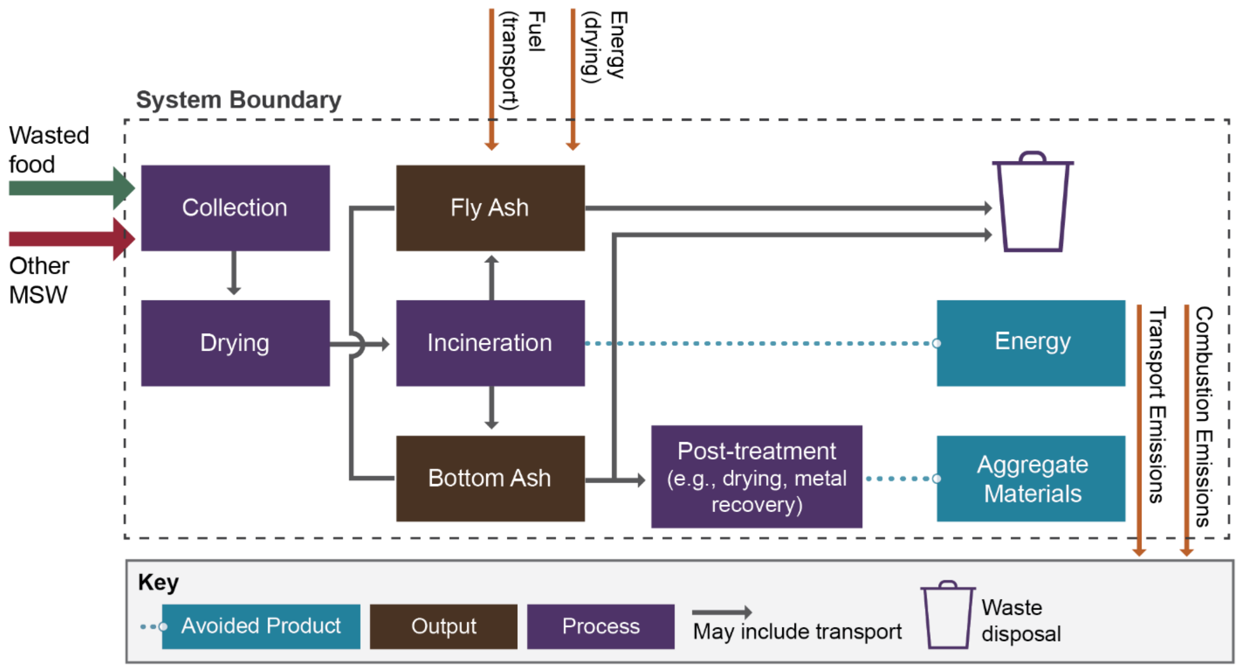

FIGURE 2-8. DIAGRAM OF CONTROLLED COMBUSTION OF WASTED FOOD (AS PART OF MSW) WITH ENERGY RECOVERY

Controlled Combustion Technologies

There are several furnace types used for controlled combustion. Waterfall furnaces that burn commingled waste account for the majority of U.S. municipal waste-to-energy design capacity, 71% (Giraud et al. 2021). A further 19% of U.S. design capacity is fueled using refuse derived fuel (RDF) and combusted in a spreader stoker furnace. Mass burn rotary waterfall and mass burn refractory wall furnaces are the next most common furnace types in the United States, accounting for about 8% of municipal waste-to-energy design capacity (Giraud et al. 2021).

TABLE 2-9. CONTROLLED COMBUSTION TECHNOLOGIES

Types[33]	**Waterfall Furnace**	Combustion of commingled waste (mass burn) on a slanted grate.
	Spreader Stoker	Combustion of RDF on a traveling grate.
	Other	Includes many other furnace types, grate (or alternate) designs, and operating conditions in the three broad categories of mass burn, RDF, and modular.
Outputs	**Fly Ash**	Residues from air pollution control which contain hazardous substances can be landfilled, used as mine backfill, or treated and used in construction.
	Bottom Ash	Can be landfilled or used as structural fill, road base, and asphalt.
	Energy	Electricity, heat, and biofuels which can be used for heating and powering the facility or sold as electricity or fuel.

Prevalence and Value

Controlled combustion of MSW, including wasted food, without energy recovery is not common practice in the U.S. (U.S. EPA 2022b; U.S. EPA 2023c). In the U.S. there are 75 facilities which burn MSW and recover energy (U.S. EPA 2023c). On average they produce 550 kWh and generate $20 to $30 USD per short ton of MSW. Production of energy from wasted food is lower than some other materials in MSW due in part to its high moisture content. Across a subset of reviewed studies for which electricity recovery numbers were reported, gross electricity production per metric ton of wasted food combusted is between 138 and 564 kWh.

Combustion residuals from the incineration of wasted food or MSW containing wasted food include bottom ash and fly ash (and metals that are recovered earlier in the process). Total ash generated ranges from 15% to 25% by weight and from 5% to 15% by volume of the MSW processed (U.S. EPA 2023c). Bottom ash accounts for 80-90% of total residuals by weight. Bottom ash is typically landfilled in the U.S. but can be used as an aggregate in construction materials (Funari et al. 2017; Cho et al. 2020; U.S. EPA 2023c); however, wasted food's contribution to bottom ash is negligible (Slorach et al. 2019a). Fly ash is hazardous due to high levels of heavy metals and dioxins. In the U.S., fly ash is landfilled in facilities designed to prevent leaching into groundwater (U.S. EPA 2023c).

Wasted Food as Feedstock

Efficient incinerator operation depends on the availability of a consistent, suitable waste stream. Two of the most basic suitability criteria include an adequate heating value (energy content) and reasonable moisture content (Trabold and Babbitt 2018). Higher energy content, ideally above 20 MJ/kg total solids, and lower moisture content favor improved incinerator performance (Clavreul et al. 2012). An analysis of fresh fruits and vegetables, which have a high water content, found that of bananas, tomatoes, apples, oranges, and peppers, only controlled combustion of bananas (lowest moisture) led to a net reduction in GWP emissions (Eriksson and Spångberg 2017), demonstrating wasted food's water content make it a poor feedstock for controlled combustion. Research indicates that removal of wasted food from MSW streams before incineration can reduce energy use, increase calorific value, and improve efficiency of controlled combustion per unit of weight (Song et al. 2013; He et al. 2014; Rajagopal et al. 2014; Erses Yay 2015; Lee et al. 2020).

Due to the difficulties that excess moisture poses for wasted food combustion, a pre-drying step is often employed. Benavente et al. (2017) find that rotary drying of olive mill waste prior to controlled combustion is a primary contributor to environmental impact in all categories studied, particularly GWP. Another study finds that pre-drying the OFMSW prior to controlled combustion deteriorates environmental performance and is not preferable (Mayer et al. 2020).

Process Emissions

Incinerators generate large volumes of flue gas process emissions that have the potential to contribute considerably to the environmental impact of wasted food. The human health impacts of direct emissions from incinerator stacks have been of concern for many years. Many emissions are related to other MSW components with which wasted food has been commingled; however, wasted food itself is a source of dioxins at parts-per-billion levels in flue gases, with variation in dioxin levels attributable to wasted food type (Katami et al. 2004).[34] Reviewed LCA studies vary in terms of the hazardous air pollutants that are included in their inventory data. Albizzati et al. (2021b), for example, assesses toxicity related impact categories and includes dioxin, hydrogen fluoride, nickel and polycyclic aromatic hydrocarbons in their air emissions inventory. Mayer et al. (2020) also assesses toxicity impact categories and includes pentachlorophenol, nickel, mercury, lead, dioxins, cyanide, cobalt, chromium, arsenic and antimony in their air emissions inventory. In the latter study, the functional unit is based on the organic fraction of MSW.

Prior to the 1970s, flue gases were often discharged directly. Technological emission controls have been used and improved since that time, beginning with electrostatic precipitation (a technology for removing particulates) and now including flue gas condensation, dioxin/mercury filters, and selective catalytic reduction (Damgaard et al. 2010). These emission controls have often had a positive effect on incinerator emissions. For example, dioxin emissions from combustion of MSW (all MSW, not only wasted food) in the U.S. decreased approximately 90% from 1980 to 2000 (Makarichi et al. 2018).

Several literature reviews have looked for evidence of environmental or human health effects of controlled combustion emissions (National Research Council 2000; World Health Organization 2007; de Titto and Savino 2019). The following conclusions appear widespread: (1) incinerator technology has improved considerably in recent decades, (2) there are known connections between incinerator emissions (particulates, mercury, and dioxin) and potential environmental and human health impacts, (3) limited and generally inadequate studies have identified clear risks to human and environmental health, (4) evidence is unable to definitively support an estimate of the presence or magnitude of risk, and (5) further research should be pursued to expand epidemiological and risk assessment studies using new techniques such as biomarker studies. The development of more definitive conclusions regarding the environmental and human health impacts of waste incinerator operations has been obstructed by difficulties in assessing impacts related to long-term pollutant exposure at low concentrations, poor understanding and variability in the nature and timing of incinerator emissions, widely present confounding factors in the environment (other air emissions, smoking, dietary variation, etc.) and the need for multi-site studies, to name a few. As such, there are ongoing concerns about controlled combustion's impact on communities and the environment (Hooks et al. 2020). Given that 79% of all MSW incinerators in the U.S. are located within low-income communities or communities of color (Baptista and Perovich 2019), emissions from controlled combustion should be considered from technical and environmental justice perspectives.

Energy Recovery

Controlled combustion's primary source of environmental benefit is associated with the recovery of electricity and heat. Recovered energy displaces grid electricity with a biogenic energy source. Avoided energy benefits are calculated based on net energy production (i.e., production minus energy demand required to operate the incinerator and associated processes). Across a subset of reviewed LCA studies for which electricity recovery numbers were reported, gross electricity production per metric ton of wasted food processed is between 138 and 564 kWh (see Appendix Table B-1).

One study analyzed controlled combustion of wasted food both with and without avoided energy benefits. The study found that controlled combustion of mixed waste stream (80% wasted food) resulted in gross environmental impacts when avoided energy benefits were not considered. In the study, packaging materials were not separated from wasted food prior to controlled combustion. When recovered heat and electricity were considered, the analysis showed a greater than 130% reduction in environmental impact (i.e., a net benefit) in 9 out of 10 impact categories (Mondello et al. 2017). GWP was the only impact category to not produce a net benefit, due in part to combustion of packaging materials. GWP realized a 40% reduction in gross impact when avoided energy benefits were included (Mondello et al. 2017). Another study analyzed controlled combustion of mixed municipal organic wastes (wasted food, yard trim, and biosolids) and demonstrated a moderate to major reduction in all considered impact categories when energy recovery was included, with GWP and acidification seeing the greatest reduction in impact (Lee et al. 2020).

TABLE 2-10. CONTROLLED COMBUSTION: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Energy demand of drying and controlled combustion.
	Generation of flue gas containing air pollutants such as particulate matter and dioxins, which are a function of feedstock and emission controls.
	Landfilling of ash results in long-term emissions.
Sources of Environmental Benefit	Reduced production and use of fossil energy due to energy recovery.
	Recovery of fly ash or bottom ash for beneficial use (an avoided product benefit) (Albizzati et al. 2021b).
Options to Reduce Impact	Select or design waste streams with suitable moisture and energy content (e.g., bakery waste).
	Ensure air pollution controls meet standards to limit emissions.

2.8 Land Application

Land application of wasted food, as discussed in this report, refers to the direct application of wasted food to agricultural land without composting, digestion, or treatment via other wasted food pathways discussed in this report. This practice predominantly involves transporting the byproducts of industrial food manufacturing activities, such as sugar beet residues or acid whey from yogurt and cheese production, back to agricultural land for direct integration into soil. It does not include application of wasted food generated further downstream by retail, food service or households.

Figure 2-9 diagrams the land application management pathway. A review of the literature found limited information pertaining to the modeling of environmental impacts and benefits of this wasted food pathway, and scope distinctions between originally edible (e.g., sugar beet pulp) versus inedible (e.g., olive pits) wasted food are not commonly drawn or discussed.

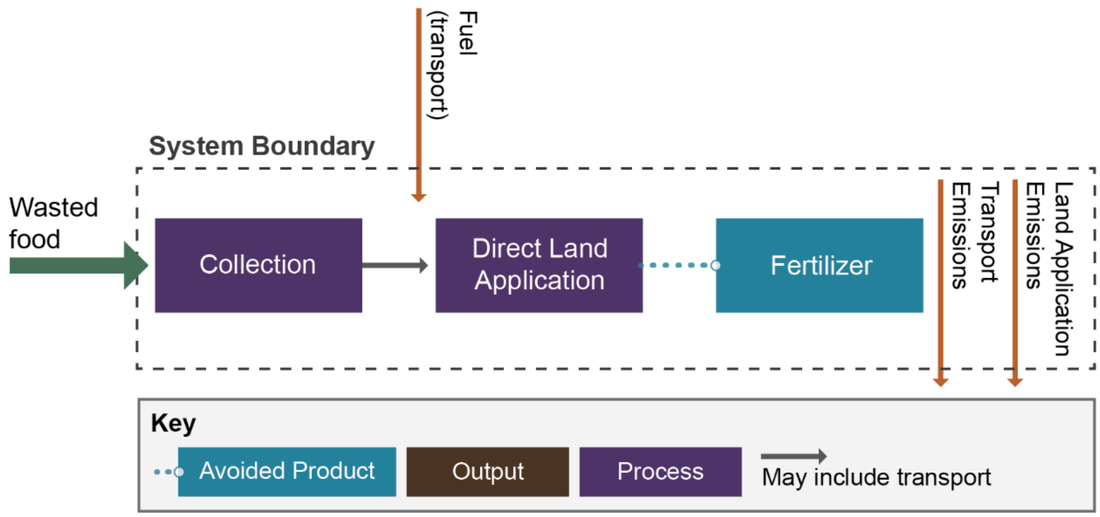

FIGURE 2-9. DIAGRAM OF DIRECT LAND APPLICATION OF WASTED FOOD

Application of raw wasted food to soil can result in numerous benefits to agricultural systems. Benefits include carbon sequestration and avoided production of chemical fertilizer (Klinglmair and Thomsen 2020). Additional soil health benefits may be provided by land application of wasted food and are discussed in Sections 4.4 and 4.6. But these benefits are not commonly assessed in the literature and are not included in comparisons in Sections 3.4 and 3.5. Benefits related to reduced water demand from the application of high moisture content food wastes are also not commonly considered in literature and are not included in any comparisons. Impacts associated with wasted food that is applied to the land typically include transportation, energy use during spreading/incorporation, and emissions to air and water.

An LCA study comparing AD and land application pathways for olive mill wastes found that transportation of waste materials to the field was a major contributor to energy demand and a minor contributor to GWP for land application (Batuecas et al. 2019).[35] The study did not consider any environmental impacts/benefits associated with land application of produced digestate or wasted food. Alternatively, the ReFED model applies the same assumptions for composting and land application, except infrastructure requirements, including assessment of fugitive emissions of methane and nitrous oxide during decomposition and a carbon sequestration benefit (Corona et al. 2020).

The benefits of land application of wasted food will depend on the wasted food feedstock. In the LCA literature, there were a limited number of wasted food types assessed. A study looking at land application of oily wasted food found that multi-year applications led to 9% to 19% increases in soil carbon and that corn yields were maintained for fall applications where sufficient nitrogen was available (Rashid and Voroney 2004).[36] Subsequent research by the same authors examined experimental data and found justification for an avoided fertilizer benefit due to land application of oily wasted food (Rashid et al. 2010).[37] Land application of fats, oils, and grease (FOG) wastes has also been shown to improve microbial activity, thereby increasing aggregate stability and mitigating many forms of soil erosion (Plante and Voroney 1998). The results of a laboratory study evaluating decomposition of potato waste in soil found rapid decomposition leading to fertility benefits (Smith 1986).

TABLE 2-11. LAND APPLICATION: ENVIRONMENTAL DRIVERS

Sources of Environmental Benefit	Long-term carbon sequestration, increased fertility, and contributions to soil health resulting from land application of raw wasted food (Section 4.6).[38]
	Displacement of conventional fertilizer production due to land application of raw wasted food.
Options to Reduce Impact	Follow best management practices regarding wasted food application rate, timing, and method to minimize potential damage from agricultural emissions. [39], [40]
	Minimize transportation distance.

2.9 Landfill

The landfill pathway sends wasted food to an area of land or an excavated site specifically designed and built to receive and contain household and other nonhazardous waste. Wasted food most often enters a landfill commingled with other MSW. This report examines only landfills that capture landfill gas (LFG) and recover energy.

According to EPA's Landfill Methane Outreach Program database, which tracks key data on MSW landfills, LFG collection systems, and energy recovery projects, there were at least 1,279 active landfills accepting waste in the United States in 2021. These numbers do not comprehensively include small facilities that do not meet reporting thresholds. Of the active facilities, 734 have landfill gas collection systems and accepted nearly 90% of landfilled waste in 2020. Nearly 65% of all collected LFG was utilized for energy recovery, which accounts for over 65% of landfilled waste in the U.S. being sent to a facility with an active energy recovery project (U.S. EPA 2022a).

Capturing biogas from waste decomposition creates a renewable energy resource. However, wasted food decay rates vary by composition but are generally faster than other MSW constituents, such as wood or textiles (U.S. EPA 2020b). As such, gas from wasted food decomposition might not be captured and can contribute significantly to fugitive methane emissions (Ishangulyyev et al. 2019). Fugitive emissions from wasted food decomposition occur based on wasted food decomposition rate and the timing of installation of gas capture technology and capping.

Figure 2-10 diagrams the landfill management pathway. Following collection, MSW is deposited in the landfill and covered regularly with cover material then capped permanently once the design capacity of a landfill section is reached. After a brief period of aerobic activity, the interior environment of the landfill becomes anaerobic, and organic materials will partially degrade to LFG, a mixture of methane, carbon dioxide, and other gases.[41] The volume and timing of LFG production can vary widely, depending on the type(s) of waste disposed of, the local climate, and management practices. As methane is produced, it diffuses out of the landfill, both before and after the cover layer is in place. Once the cover layer is installed, some fraction of methane is oxidized to carbon dioxide in the cover layer. The loss of methane is undesirable environmentally, as methane is a potent GHG with a GWP approximately 28 times that of carbon dioxide over a 100-year period (IPCC 2013). Therefore, many landfills capture a portion of LFG and either flare it, reducing fugitive methane emissions, or use it to recover energy, reducing methane emissions and producing a beneficial product. When the landfill is capped, a portion of the LFG is collected. Methane that is not captured as LFG either is oxidized in the cover layer or escapes (U. Lee et al. 2017). Collection and treatment of landfill leachate (liquid effluent from landfills) is necessary.

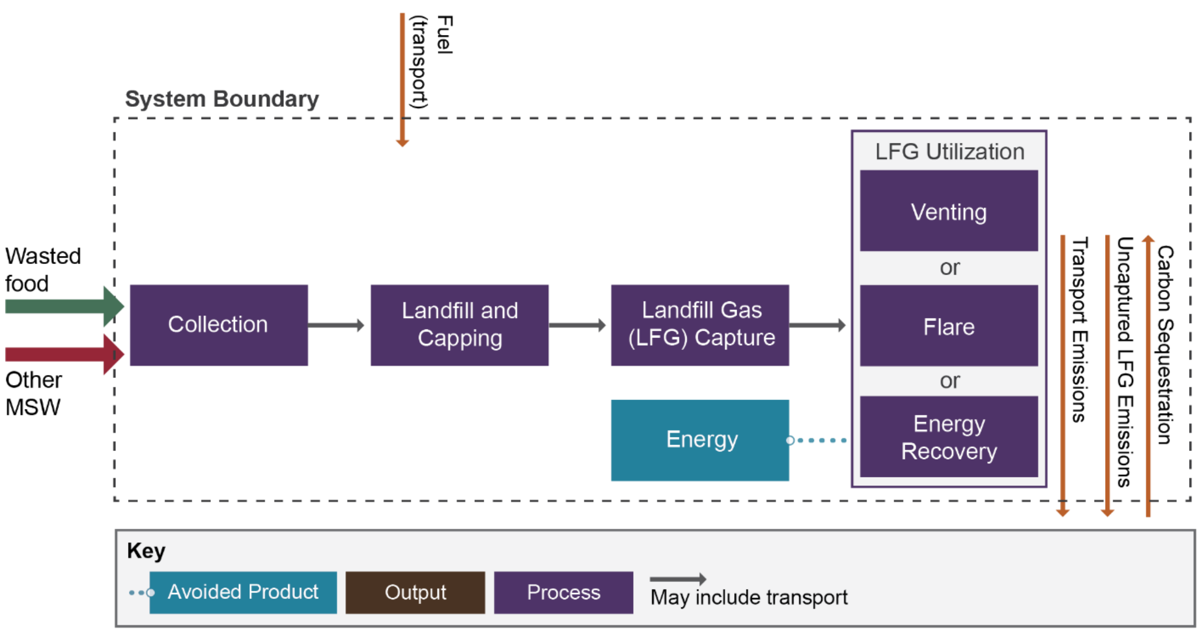

FIGURE 2-10. DIAGRAM OF WASTED FOOD LANDFILLING WITH ENERGY RECOVERY

Prevalence and Value

In the U.S. in 2022, 51% of all landfills that could potentially recover biogas, based on landfill type and age), were doing so. Of these facilities, 68% recovered gas for electricity generation, 15% produced renewable natural gas, and 17% used gas directly, meaning biogas was used onsite or piped to nearby businesses for fueling boilers or other uses (U.S. EPA 2021d).

Average estimates suggest that one million tons of MSW can produce either 0.78 MW of electricity or 216 MMBtu of heat per day (Environmental and Energy Study Institute 2013). The value of electricity and heat will vary by market rates but range between 0.025 and 0.07 USD/kWh and 4 and 8 USD per MMBtu (Environmental and Energy Study Institute 2013). As noted above, much of the energy derives from organic wastes other than wasted food.

Fugitive Emissions

Studies consistently find that fugitive landfill methane emissions account for the majority of GWP impact for the landfill pathway (Yoshida et al. 2012; Edwards et al. 2018). The quantity of fugitive emissions depends on waste type, landfill type, climate, and the type, installation timing, and efficiency of LFG capture and recovery systems. The rate of material degradation in a typical landfill model is defined by a decay factor, which is based on material type and assumed landfill climate factors such as temperature and precipitation.[42] Wasted food, grass clippings, and leaves degrade quickly relative to other organics such as paper, wood and mixed MSW (U.S. EPA 2020b). Faster rates of waste degradation can lead to fugitive methane emissions that occur prior to a landfill being covered for gas recovery, increasing GWP impact (Ishangulyyev et al. 2019). Because of these relative decay rates, the proportional contribution of wasted food to fugitive emissions is higher than to captured gas, relative to many other organic wastes.

Material decay rates affect the timing but not the magnitude of LFG production. The magnitude of methane and carbon dioxide produced in a landfill is determined by the carbon content of landfilled material and the share of landfilled carbon that decomposes over time. Degradable carbon content is typically defined as the quantity of degradable organic carbon (DOC) in the material. An additional factor, DOCf, is used to specify the fraction of DOC that will decompose under landfill conditions. DOCf values vary across waste types and likely within the broad category of wasted food (U. Lee et al. 2017).[43] Wasted food degradability and DOCf are variables that can have a moderate to major effect on net GWP impact results. Both are subject to large uncertainties (U. Lee et al. 2017).

The portion of uncollected LFG methane that is not oxidized to carbon dioxide in landfill cover will be lost as fugitive emissions. Methane oxidation rates vary depending on the stage of LFG capping, ranging from a 10% oxidation rate for landfills without gas collection or final cover, to 20% for landfills with gas collection before final cover, and up to 35% for landfills after final cover installation. Yoshida et al. (2012) found that a 5% increase in LFG methane oxidation, as it filters through the landfill cover, can lead to a 25% decrease in total GHG emissions. A sensitivity analysis conducted by Lee et al. (2017) showed an 18% reduction in GHG emissions when the default oxidation factor of 10% was updated to 36%, which was the mean value identified in a review of methane oxidation factors.[44] The details of LFG modeling have evolved considerably in recent decades, especially for highly degradable wastes such as wasted food, with evidence that past modeling approaches may underestimate landfill methane production and fugitive emissions (Wang et al. 2013; Duren et al. 2019).[45]

Recent work has shown discrepancies between self-reported landfill methane emissions and remotely sensed emissions (Duren et al. 2019), with some landfills severely underestimating emissions. Further work is needed to determine if these underestimated values are based on an incomplete understanding of landfill processes, or whether parameter estimates have been inaccurate, whether there is systematic underreporting bias, or if some other issue is at play.

Landfill Gas Collection

In modern landfills, the majority of LFG is collected and combusted to recover energy or flared. U.S. EPA's WARM model estimates that the average LFG collection efficiency at U.S. landfills is 64.8% and can be closer to 80% at sites that follow stricter California regulations (U.S. EPA 2020). Most analyses assume that recovered energy displaces traditional sources of energy, producing an environmental benefit. One analysis looking at four impact categories (GWP, eutrophication, acidification, and ecotoxicity) concluded that acidification realized the greatest benefit when LFG was collected and used to produce heat and electricity. Gross acidification impact was reduced by approximately 25% due to avoided energy credits (Lee et al. 2020). Minor to moderate reductions in impact were realized for the other three impact categories, with eutrophication seeing the smallest relative reduction in impact.

Carbon Sequestration

Non-degradable carbon in wasted food and the fraction of DOC that does not decompose are sequestered in the landfill, reducing the net GWP of wasted food landfilling. Wasted food has a lower carbon storage potential relative to other degradable waste types due to its lack of lignin content (Ishangulyyev et al. 2019). U.S. EPA's WARM model estimates that approximately 16% of the initial carbon content in wasted food will be stored in the landfill, which is considerably lower than the carbon storage potentials for other organic materials commonly landfilled (e.g., grass: 53%; branches: 77%; leaves: 85%) (U.S. EPA 2020).[46] The LCA models reviewed for this report applied a variety of modeling assumptions regarding carbon sequestration. The differences largely depend on assumptions about material degradability and environmental conditions in the receiving landfill.[47] A separate review of LCA studies examining GHG emissions during wasted food management found nine LCA studies (of 19) that considered carbon sequestration. These studies assumed that between 67 and 168 kg CO_2eq are sequestered per ton of wasted food landfilled (Bernstad Saraiva Schott et al. 2016).

Landfill Leachate

If no liner is in place, or if a liner fails, then escaped leachate can lead to groundwater contamination. Leachate due specifically to wasted food might be of concern due to nutrient content (contributing to eutrophication), but it is difficult to attribute future landfill leachate composition to specific inputs to the landfill. Most studies assume that landfill leachate is collected and either treated onsite or conveyed to a local WRRF. The quantity of leachate produced is a function of incoming waste moisture, rainfall, evapotranspiration, and permeability of the landfill, which will change over time (Edwards et al. 2017). For impact categories where wastewater treatment efficiently removes the contributing pollutants, the share of wastewater that goes untreated can dominate the impact (Edwards et al. 2017).[48]

TABLE 2-12. LANDFILL: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Fugitive methane emissions.
	Pollution of ground water or surface water from untreated landfill leachate.
Source of Environmental Benefit	Reduced production and use of fossil energy due to LFG energy recovery.
Options to Reduce Impact	Maximize gas capture efficiency to reduce gas leakage.[49]
	Use optimized cover materials to increase methane oxidation, which may reduce fugitive emissions (Lee et al. 2017).
	When possible, maximize energy capture from LFG; otherwise maximize LFG flaring. The loss of produced energy credits associated with LFG flaring results in higher GWP impacts for the landfill pathway.
Other Considerations (U.S. EPA 2020a)	LFG leakage from a landfill depends on gas capture efficiency, environmental conditions, and waste characteristics.
	The decomposable carbon content in wasted food impacts the magnitude of methane and carbon dioxide produced in landfills.
	Methane oxidation rates vary depending on the stage of LFG capping.

2.10 Sewer/Wastewater Treatment

The sewer/WWT pathway (also commonly referred to as "down the drain") represents wasted food that is sent down sink drains with or without prior treatment such as grinding in a garbage disposer. Wasted food sent down the drain moves through the sewer and is treated at a WRRF along with other sewage.

Some WRRFs have anaerobic digesters to process wastewater solids/sewage sludge, while others do not. Both WWT with and without AD are included in this pathway. All wasted food travelling down the drain and through the sewer to the WRRF is in the sewer/wastewater treatment pathway, whereas wasted food delivered directly to AD (typically by vehicle) at the WRRF is in the AD pathway.

Figure 2-11 diagrams the WWT management pathway and includes optional AD treatment of the resulting wastewater solids. Other possible wastewater solids treatments include aerobic digestion, alkaline stabilization, thermal drying, composting, pasteurization, and emerging technologies.

When entering this management pathway, wasted food is pre-processed by grinding at the residence or commercial institution and introduced to the sewer system for transport to the WRRF. Once at the WRRF, wasted food is subjected to the same treatment processes as wastewater. Wasted food will typically partition with other wastewater solids (i.e., through settling) and be separated from the resulting liquid effluent in primary, secondary, or tertiary clarification steps. Depending on quality, the resulting wastewater solids can be beneficially applied to land or disposed of in landfills or controlled combustion facilities. Wastewater solids that have been treated to meet federal and state standards for beneficial use are called biosolids.

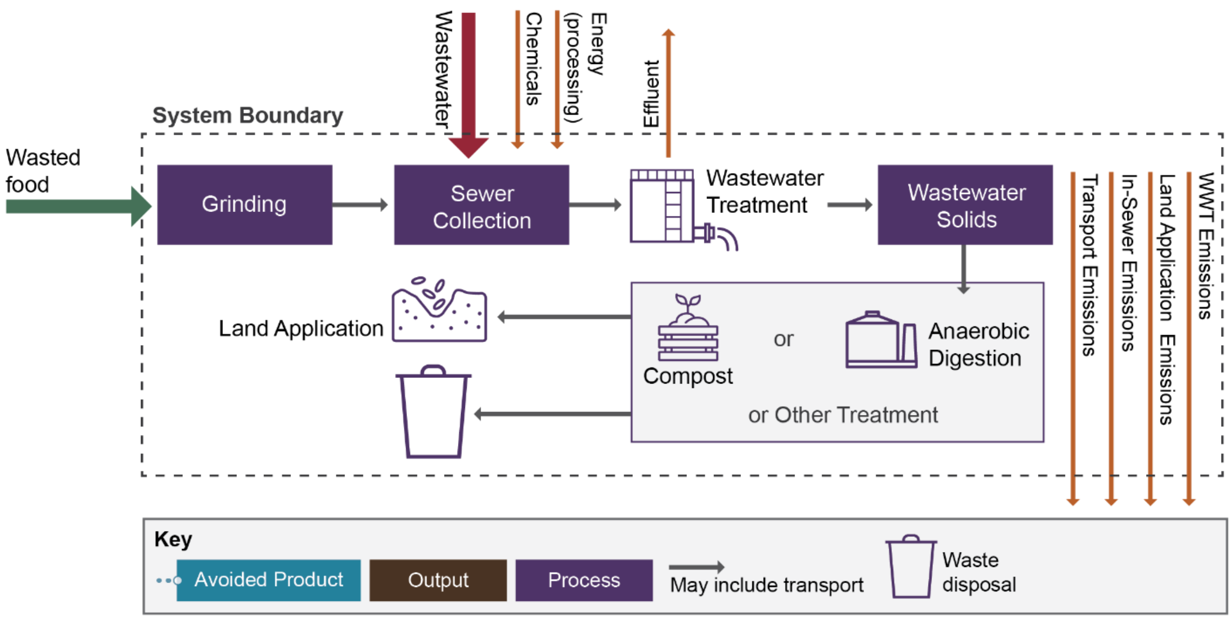

FIGURE 2-11. DIAGRAM OF WASTED FOOD PROCESSING VIA SEWER/WASTEWATER TREATMENT

The benefits and impacts of this pathway depend on the technology and performance of the WRRF and biosolids management. According to the most recent Clean Watersheds Needs Survey, around 29% and 41% of the U.S. population sent wastewater to secondary and advanced treatment systems, respectively, in 2012 (U.S. EPA 2016). While advanced treatment systems (tertiary or greater) are more likely to be capable of enhanced nutrient removal, secondary treatment systems can be designed or retrofitted to remove nutrients (prevalent in wasted food) from wastewater effluent. Generally, though, more WRRF treatment steps lead to higher energy requirements, potentially negating energy recovery from wasted food.

Recent estimates of methane emissions from centralized WRRFs are 1.9 times higher than previous estimates (Moore et al. 2023), suggesting that existing studies may systematically underestimate CH_4 from WRRFs. However, these comprehensive studies (Moore et al. 2023; Song et al. 2023) do not address the contribution of wasted food to sewer and WRRF emissions, so further study is needed. The findings of Song et al. (2023) that leaks, incomplete flaring, and other management issues are the main source of CH_4 at WRRFs with AD suggests that operation and management may be more important than process selection in minimizing GHG, and likely other, impacts.

Prevalence and Value

About 80% of U.S. households are serviced by one of the country's 16,000 operational WRRFs (ASCE 2021). The remaining 20% of households dispose of wastewater in onsite treatment systems (e.g., septic systems) that typically lack the capacity to process wasted food. On average, 150-250 dry metric tons of biosolids are produced per million gallons of wastewater treated (Beecher et al. 2022). This varies depending on wastewater and biosolids treatment technologies. Biosolids represent an economic cost to the wastewater treatment industry, with the wet ton paid for management in 2018 reported as $49 USD composting, $58 USD landfilling, $62 USD land application, and $95 USD controlled combustion (Beecher et al. 2022).

Sewer Collection

Wasted food collection via the sanitary sewer system can lead to anaerobic conditions that favor methane formation for the portion of wasted food that degrades during transportation to the WRRF. Solids type (e.g. wasted food) and loading, temperature, hydraulic residence time, and other factors contribute to methanogenesis in sewers, but establishing relationships between sewer parameters and methanogenesis is complex (Song et al. 2023). Of the studies addressing this, a 2012 study assumed that 15% of wastewater biodegradable chemical oxygen demand (COD) is degraded during transport, 10% anaerobically and 5% aerobically, with the carbon share released as methane and carbon dioxide (Parry and WERF 2012).[50] Employing this assumption, the analysis found that methane emissions during sewer transport are the largest source of GWP in the WWT pathway, with gross emissions exceeding the benefit associated with AD energy recovery. More recently, a comparative LCA of multiple wasted food pathways assumed that 6% of wasted food volatile solids are degraded during sewer transport (Edwards et al. 2018).[51] Other studies have neglected any methane formation potential in sewers altogether. The true magnitude of sewer methane emissions is a function of wasted food degradability and conditions and residence time in the sewer system (Parry and WERF 2012).

Expanding wasted food management via the sewer/WWT pathway incurs the risk of introducing additional FOG waste to sewers, which contributes to bottlenecks and blockages, increasing sewer overflow events and maintenance needs (Wallace et al. 2017). However, studies of long-term impacts for non-FOG wastes have shown that marginal additional sewer deposits from wasted food—when high-specific-gravity wasted food (e.g., bones and egg shells) and/or FOG-rich wasted food are excluded—are small and have a minor impact on sewer operations (Mattsson et al. 2015).

Wasted food in sewer systems may also lead to pipe corrosion in municipalities with small or aging pipes (due to wasted food's high biological oxygen demand (BOD) interacting with sulfates to transform to hydrogen sulfide and sulfuric acid), the need for additional energy in systems which require pumping stations, and/or increased organic pollutants in sewer effluent direct discharges in combined or low-flow systems (U.S. EPA 2021b).

Secondary Treatment

Once at the wastewater treatment facility, wasted food undergoes the same processing steps as the remaining bulk wastewater. The effect that wasted food has on primary and secondary treatment processes is different for every WRRF and must be considered on a case-by-case basis. Parry and WERF (2012) report that 20% to 45% of BOD and 45% to 70% of total suspended solids (TSS) are typically removed during primary clarification. The analysis of Edwards et al. (2018) assumes that 34% of wasted food settles out in the primary clarifiers, 26% is metabolized to carbon dioxide in secondary treatment, while 33% settles out with the waste-activated sludge in secondary clarification. Increased BOD loading to secondary treatment increases aeration electricity demand in secondary treatment (Parry and WERF 2012).

Biogas Production

As discussed in Section 2.4, wasted food holds potential for energy recovery. While only about one out of ten U.S. WRRFs is equipped with anaerobic digesters,[52] more than half of all biosolids generated in the U.S. are treated by AD, since AD is nearly ubiquitous at the nation's largest WRRFs (Beecher et al. 2022). These facilities have the potential to recover wasted food energy, avoiding fossil energy production and crediting the sewer/WWT pathway with associated environmental benefits. However, biogas production at WRRFs that can be attributed to wasted food sent down the drain does not outweigh fugitive methane emissions, resulting in a net environmental impact even if energy is recovered via AD. Elevated CH_4 emissions from WRRFs equipped with AD are mainly caused by biogas leakage from anaerobic digesters and incomplete flaring of excess CH_4 (Song et al. 2023).

Recent estimates of methane emissions from centralized WRRFs are 1.9 times higher than previous estimates (Moore et al. 2023), suggesting that existing studies may systematically underestimate CH_4 from WRRFs. However, these comprehensive studies (Moore et al. 2023; Song et al. 2023) do not specifically address the contribution of wasted food to sewer and WRRF emissions.

Biosolids

Biosolids are wastewater solids/sewage sludge that have been treated to reduce pathogens and contaminants and stabilize organic compounds. Often biosolids are dewatered or thickened in addition to undergoing stabilization via AD, aerobic digestion (such as in aerated lagoons), thermal or air drying, composting, or another treatment method. Biosolids must meet federal, state, and local regulatory standards for pathogens and contaminants in order to be beneficially reused. The federal biosolids rule is contained in Title 40 of the Code of Federal Regulations (CFR) Part 503 and defines two types of biosolids with respect to pathogen reduction. Class B biosolids contain some pathogens and have restrictions on their use to limit public exposure, such as buffer requirements and crop harvesting restrictions. Class A biosolids have virtually no pathogens and fewer restrictions on their use. Most Class A biosolids classify as “Exceptional Quality” (EQ) and are safe for use anywhere, including home gardens and public parks due undetectable levels of pathogens and pollutants. Composted biosolids frequently qualify as EQ. For biosolids to be land applied, heavy metals must not exceed the ceiling concentrations set out in federal and state regulations.

About 53% of biosolids in the U.S. are land applied as fertilizers or soil amendments for agriculture, landscaping, and land reclamation (Beecher et al. 2022). Long-term studies of application of biosolids show clear benefits to soil carbon (Ippolito et al. 2010; Brown et al. 2011; Ippolito et al. 2021; Sullivan et al. 2021). Potential environmental impacts from biosolids land application include transportation and application emissions, including nutrient runoff. When land applied, biosolids provide essential nutrients and organic matter to crops and soils and offer a cheap fertilizer option for farmers. Application rates for biosolids are often determined by a crop’s nitrogen needs (agronomic loading rate). Wasted food can increase nitrogen levels in biosolids which may require nutrient removal during treatment or a decreased application rate. In the reviewed LCAs of sewer/WWT, carbon sequestration from biosolids land application was not assessed. But biosolids application does contribute to soil health and carbon sequestration as discussed in Chapter 4.

If biosolids do not meet quality standards that allow for land application, or appropriate land for application is not available, biosolids may be disposed of by landfill or controlled combustion. Wasted food treated as a component of biosolids is exposed to contamination resulting from other wastewater components, such as PFAS, as described in Section 4.5.

It is important to note wasted food sent down the drain makes up a tiny fraction of biosolids that result from wastewater treatment process.

Wastewater Effluent

Depending on the layout and performance of the specific WRRF, some fraction of all nutrients, BOD/COD, and pollutants in wasted food will be released to surface water with wastewater effluent. For example, in the analysis of Edwards et al. (2018), approximately 1.5% of the original COD load is assumed to be released with treatment plant effluent; significant fractions of N may be released in effluent or to the atmosphere (Morelli et al. 2019). Wasted food has a high nutrient content and can increase the influent nutrient load to be treated at WRRFs, many of which face increasingly stringent nutrient limits due to concerns about eutrophication of local waterbodies (Scheehle and Doorn 2002; Carey and Migliaccio 2009). On the other hand, if applied at appropriate rates, wastewater effluent can be a source of nutrients if used as irrigation water (Sullivan et al. 2021; Vivaldi et al. 2022). If additional wasted food is processed via the sewer/WWT pathway, WRRFs can either increase treatment demands (increasing energy required) to remove additional nutrients or effluent nutrient levels will increase.

TABLE 2-13. SEWER / WASTEWATER TREATMENT: ENVIRONMENTAL DRIVERS

Sources of Environmental Impact	Methane emissions from sewer transport and WWT processes.
	Energy demand of aeration during secondary treatment, which is dependent on the level of biochemical and chemical oxygen demand (BOD and COD, respectively).
	Increased energy demand for nutrient removal at facilities with nutrient permits (wasted food adds to the nutrient load in wastewater).
Source of Environmental Benefit	Reduced production and use of fossil energy due to biogas energy recovery if AD is present.
	Displacement of conventional fertilizer production if land application of biosolids.
	Long-term carbon sequestration and contributions to soil health if land application of biosolids (Section 4.6).
Options to Reduce Impact	Reduce BOD/COD load to secondary treatment (e.g., via enhanced primary clarification) to reduce plant energy demand.
	Utilize the WWT pathway in areas that practice AD with biogas utilization.
	Additional wasted food inputs for certain biological nutrient-removal processes may improve nitrogen removal, reduce net operational costs, and lower aeration energy demand (Kim et al. 2019).
	Avoid landfilling biosolids. Manage biosolids for safe, beneficial reuse by treating to high quality through less energy-intensive methods, such as composting.
Other Considerations	Additional wasted food in sewers may lead to solids deposition and clogging and affect sewer performance.[53]
	Nutrient limits for wastewater effluent vary between WRRFs and will determine the need for and additional impact of nutrient removal.
	Sewer conveyance avoids several logistical and operational challenges associated with wasted food collection that would otherwise be necessary for the AD pathway (e.g., separate collection and transport infrastructure).[54]

2.11 Unharvested/Plowed In

The unharvested/plowed in pathway signifies leaving crops ready for harvest in the field or tilling them into the soil. Most unharvested food is reincorporated into soil, although sometimes livestock graze fields before tilling (Gillman et al. 2019). For grazing, unharvested crops must be paired with specific animals to meet nutritional requirements (Winans et al. 2020). Unharvested crops that are reincorporated into the soil help to build soil organic matter, improving soil health and supporting future crop growth. Further discussion of the soil health benefits provided by the unharvested/plowed in pathway is in Chapter 4.

The quantity of edible food left unharvested is poorly understood. Both FAO and ReFED provide high-level estimates indicating that 20% of fruits and vegetables are left unharvested on agricultural fields. Recent measurements of food loss for 20 crops across 123 fields in California found a loss rate above 30%, which equates to over 11,000 kg of produce per hectare. The authors note that "decisions to leave produce in fields result from a complex mix of consumer preferences, buyer specifications, marketplace economics, labor costs, weather and other factors" (Baker et al. 2019).

Only one LCA study that quantified the environmental impacts and benefits of unharvested field crops was identified, reporting a GWP of 0.31 kg CO_2e / kg tomatoes plowed in (Gillman et al. 2019). Given the lack of available studies, this section provides perspectives based on themes in the literature.

Avoiding Supply-Chain Impacts

Once food is harvested, additional transportation and resources are required to package it, distribute it, and deliver it to humans (Kummu et al. 2012; Bernstad et al. 2017). Therefore, it is environmentally preferable to prevent food that will not be eaten from entering the supply chain in order to avoid these impacts. The unharvested/plowed in pathway provides an alternative to allowing wasted food to enter the supply chain, while still using the organic material for beneficial use.

Energy Consumption

Reincorporating unharvested plant material back into the soil requires energy consumption for equipment operation. The EU (European Union) FUSIONS project performed a basic LCA of wasted tomatoes, comparing plowing-in of unharvested crops with landfilling a packaged, unsold product. This analysis showed that the marginal energy required for tractor use was an order of magnitude smaller than the energy required for transport, packaging, and retail (Scherhaufer et al. 2015).

Soil Reincorporation

Incorporating crops back into the field provides necessary nutrients for subsequent crops (Alexander et al. 2017). If farmers use this input of nutrients to justify reducing subsequent fertilizer applications, then an avoided fertilizer benefit can be attributed to unharvested crop recycling. Leaving crops on the field can also help build soil carbon, especially when combined with less intensive tillage practices (Al-Kaisi and Yin 2005; Omonode et al. 2007). See Section 4.6 for further discussion of qualitative relationships to soil health.

TABLE 2-14. UNHARVESTED / PLOWED IN: ENVIRONMENTAL DRIVERS

Source of Environmental Impact	Energy demands of tillage.
Sources of Environmental Benefit	Avoided fertilizer production: only warranted if the producer reduces fertilizer application due to this practice.
	Long-term carbon sequestration and contributions to soil health resulting from recycling of unharvested crops (Section 4.6).
Option to Reduce Impact	Follow recommended tillage practices to minimize erosion and improve soil health (Claassen et al. 2018).

2.12 Emerging Pathways

While the report focuses on the eleven pathways described above, there are a number of alternative wasted food pathways that are either appropriate only for a very specific wasted food type, or are "emerging", indicating that they rely on processes that might be still at the bench or pilot scale, or that have not yet been widely adopted. These pathways include, for example, hydrothermal carbonization, rendering, and the group of thermochemical processes that includes pyrolysis, torrefaction, and gasification. Because of their dynamic status or lack of available data, these pathways are excluded from the quantitative analysis and some examples are briefly discussed in Appendix E.

2.13 Key Issues Across Pathways

This section describes some of the cross-cutting issues relevant to the environmental impacts and benefits of multiple pathways: technology and operations, collection and transportation, infrastructure, and feedstock characteristics and requirements.

Technology and Operations

While it is beyond the scope of this analysis to comprehensively discuss how different combinations of technologies and operational practices affect the environmental performance of wasted food pathways, the discussion highlights the potential importance of these factors.

Technology

The technological basis and operational and management practices at specific facilities can affect environmental impacts within a particular pathway. Given the range of environmental performance seen in studies, the technologies selected, and the operation of management pathway may be just as important to environmental impact as the selection of the wasted food pathway itself. Within a given pathway, there are technology choices that may strongly influence environmental performance. For example, for composting, energy demand and emissions to air will vary depending on the type of composting system (e.g., aerated static pile or windrow). Within particular technology categories, capture or recovery efficiency may vary. Energy and heat recovery efficiency are critically important in determining avoided energy benefits for AD and controlled combustion pathways. Short- and long-term gas capture efficiency drives the degree to which landfills can lower their GWP impacts.

Operations

Within a given technological approach, the operation of a specific facility will dictate environmental performance. The environmental performance of wasted food pathways is highly variable. Among the assessed studies, there are significant ranges in impacts per mass of wasted food managed. Examples of operational factors that can impact performance include:

- For AD, is biogas flared or combusted in a boiler, electrical engine, or CHP engine?
- For compost, how is the pile constructed (C:N ratio) and are aerobic conditions maintained?
- For AD and sewer/WWT, how are digestate and biosolids treated, and what is the energy demand of that process?
- For any pathway with application to land (compost, AD, land application, sewer/WWT, unharvested), is the wasted food-derived material being applied in keeping with appropriate agronomic practices and reducing fertilizer or peat consumption?
- For controlled combustion, what are the emission controls, and are they operated and maintained adequately?
- For donation, what fraction of incoming food is ultimately wasted?

Collection and Transportation

All pathways except source reduction must collect and transport wasted food for processing.

Sewer Collection

Wastewater treatment is the one wasted food pathway that utilizes sewer, rather than vehicles, for transportation. Potential benefits of the sewer/WWT pathway are avoiding inefficient trucking of heavy wasted food or capturing waste heat from sewers. However, research indicates that fugitive methane from anaerobic decomposition in sewers may negate the energy advantage of pumping wasted food through sewer lines to WRRFs rather than trucking (Parry and WERF 2012; Zan et al. 2020).

Vehicle Collection

Most other wasted food pathways require collection by vehicle. The magnitude of environmental impact associated with waste collection and transportation varies widely across the reviewed studies, indicating that potential transportation impact is influenced by factors beyond wasted food pathway selection, such as the location of collection. While findings vary, the majority of the studies reviewed include transportation/collection within the scope of their analysis. Several authors (Parry and WERF 2012; Ahamed et al. 2016; Edwards et al. 2017; Khoshnevisan et al. 2018; Bartocci et al. 2020) conclude that waste collection is a negligible or minor contributor to a number of impact categories, including GWP, acidification, eutrophication, energy demand, and ecosystem quality. EPA's WARM Model showed that transportation accounted for 7% of the net GWP impact for composting, 4% of the net GWP impact for combustion, and 3% of the net GWP impact for landfilling (U.S. EPA 2020c).

An analysis of wasted food management in Italy analyzed the environmental contributions of wasted food collection across several management pathways and impact categories. The analysis found that in 10 of 12 impact categories the waste collection phase contributed less than 10% of impact for all pathways. Land occupation (i.e., the amount of land required for a system or a process) was strongly influenced by wasted food collection, but impact contributions were still less than 20%. Average transportation distances in this study were approximately 30 kilometers (km). The study's authors promote efficient, localized collection and pre-processing systems as a means to limit environmental impact (Mondello et al. 2017).

Other authors (Righi et al. 2013; Al-Rumaihi et al. 2020; Lee et al. 2020) conclude that waste collection and transportation are a major contributor to environmental impact for several impact categories, including GWP and acidification. For example, Lee et al. (2020) showed moderate to major contributions to gross impact from collection and transportation for GWP, acidification, eutrophication, and ecotoxicity across four pathways (AD, landfill, composting, and controlled combustion). Collecting wasted food and transporting it to the transfer station contributed more to gross impact than did subsequent transportation to the management facility. Their analysis considered a worst-case scenario for collection impact in which collection vehicles are operating at maximum capacity five days per week. Transportation distances after collection ranged from 45 to 93 km. The magnitude of impact associated with collection and transportation was similar across pathways.

Khoshnevisan et al. (2018) performed a sensitivity analysis on transportation distance and showed that when hauling distance was increased from 25 to 150 km, net GWP benefit decreased by approximately 3% and the net benefit for ecosystem quality decreased by more than 10%.

The relative contribution of transportation to gross environmental impact is dependent on the total magnitude of that impact. For example, the relative contribution of wasted food collection and transportation will likely be larger for a well-managed compost system with limited process emissions and energy consumption than for a poorly managed compost system with higher associated impacts. This would be true even in instances where the absolute magnitude of transportation impacts was consistent across both hypothetical scenarios. The analysis of Ahamed et al. (2016) provides an example of this phenomenon, where similar transportation burdens for controlled combustion and AD pathways yield divergent contributions to gross impact due to the greater impact of controlled combustion process operation. This example highlights one of the challenges of accurately gauging the contribution of transportation and other ubiquitous processes in the context of LCA. While transportation may or may not contribute a large relative share of gross impact per unit mass of wasted food managed, the cumulative contribution of transportation to wasted food management impact is likely to be substantial and persistent.

Infrastructure

Infrastructure constitutes the non-consumable physical structures and equipment required to operate a management pathway, such as tanks, piping, mechanical equipment, and accessory buildings. Depending on the study, infrastructure impacts may also include burdens associated with construction (e.g., fuel consumption). Infrastructure environmental impacts accrue in upstream supply-chains and are allocated to the study's functional unit by amortizing over the expected lifespan of the equipment and the total quantity of material processed (Frischknecht et al. 2007). Typically, but not always, infrastructure contributions to overall impacts are small relative to the other issues discussed above.

Contributions of infrastructure to environmental impact can vary considerably across impact categories and between pathways. The analysis of Lee et al. (2020) found that infrastructure made a negligible contribution to gross impact for GWP and acidification across AD, landfill, compost, and controlled combustion pathways. Mayer et al. (2020) reported that the "provision of ancillary or capital goods only affect the GWP very little." Salemdeeb et al. (2017) showed that infrastructure for animal feed production contributes negligibly to fossil fuel depletion for both wet and dry animal feed production. In the latter study, infrastructure contributions to eutrophication were minor and moderate for landfill and controlled combustion pathways, respectively. Ecotoxicity was more strongly influenced by infrastructure, with non-negligible contributions to gross impact for all pathways, including moderate and major contributions for landfill and controlled combustion pathways, respectively.

Feedstock Composition, Contamination and Pre-Treatment

The composition, contamination, or need for pre-treatment of wasted food can all affect the selection and performance of pathways. Characteristics of the wasted food stream inform which pathways are possible. For example, a pathway may require food to be safe for human consumption, mostly free of contaminants such as packaging, or consistent in volume and composition. Those pathways that require low levels of contamination may also necessitate source separation and separate collection.

Table 2-15 provides high-level examples of the qualitative requirements for the different wasted food pathways with respect to the level of homogeneity expected in the waste material and the level of non-food contamination allowed. The table also provides information on the collection requirements for each pathway.

Composition of Wasted Food

The composition of wasted food can affect selection of management pathways as well as performance of those pathways.

- *Edibility:* The donation and upcycling pathways are only applicable for the portion of wasted food safe to consume.
- *Homogeneity:* Certain pathways may require or perform better with homogenous wasted food. For example, the AD process can benefit from a consistent waste stream that provides a stable diet to the biological community.
- *Moisture content:* Environmental performance of controlled combustion and dry animal feed pathways is particularly sensitive to the moisture content of wasted food feedstock. High moisture content leads to increased impacts for these pathways.
- *Energy content:* The controlled combustion and AD pathways will have improved environmental performance for wasted food feedstocks with a high energy content. Other pathways are less influenced by the energy content of wasted food.
- *Oil content:* A comparison of AD and controlled combustion pathways examined the effect of wasted food oil content on pathway preference. Oil content in this study did not refer to FOG; instead, the focus was on wasted food with oil contents between 2.5% and 10%. The authors indicated that oil content is not expected to have a dramatic effect on controlled combustion or AD pathway performance. For AD, the authors noted that without special reactor designs, oil is difficult to digest due to low solubility and biodegradability and floc shielding, which limits oil's availability for biochemical reactions (Ahamed et al. 2016).

- *Nutrient, protein, and carbon content:* For several pathways, the quantity of avoided product and carbon sequestration benefits depend on these aspects of wasted food composition. For example, wasted food with a higher protein content could reasonably be assumed to displace a larger quantity of animal feed, leading to greater avoided product benefits.

A comprehensive discussion of how wasted food composition affects pathway environmental performance is beyond the scope of this analysis.

Contamination of Wasted Food

The presence of contamination (including presence of pathogens and physical and chemical contaminants) in wasted food can influence selection of applicable wasted food pathways and the quality of derived end products. For example, upcycling and donation both require wasted food that is still fit for human consumption, requiring a certain level of hygiene in the management pathway. When this level of food quality is not met, wasted food in the upcycling and donation pathways will be diverted to another pathway, such as landfilling. In general, issues with food quality or contamination often mean that portions of wasted food or wasted food intended for specific pathways are diverted to a pathway with less stringent quality requirements (Vandermeersch et al. 2014; Eriksson and Spångberg 2017). These interconnections are shown in the management pathway flow diagrams in preceding sections, where pre-treatment often produces a residual stream that is sent to disposal.

Contaminants may be pathogens or physical or chemical contaminants. Examples of physical contaminants include noncompostable plastics or metal or glass; per- and polyfluoroalkyl substances (PFAS) is an example of a chemical contaminant that can be found in wasted food streams.

The presence of contamination in a wasted food stream is often linked to whether the source of food is pre- or post-consumer waste. Post-consumer wasted food is typically thought of as having a higher level of contamination and may therefore not be suitable for all pathways, especially in the absence of source separation and other, potentially extensive, pre-treatment steps. While generally true, grocery store waste is an example of pre-consumer wasted food with potentially high levels of contamination (e.g., packaging).

Limiting contamination in wasted food is essential for the operation of certain pathways. The animal feed, AD and land application pathways have historically sourced the majority of processed waste from the pre-consumer manufacturing sector (Figure 1-1), which is likely related to consistency and low contamination levels in pre-consumer waste. The composting pathway as well benefits from low levels of contamination to increase product value. Microplastics are prevalent in wasted food-derived composts and digestates, though amounts, sizes, and types of plastic particles vary (Porterfield et al. 2023). Effective and efficient pre-treatment processes, discussed below, will be required to allow large volumes of post-consumer wasted food to be utilized by pathways with strict contamination tolerances (Table 2-15). For the animal feed pathway, contamination can include the presence of animal products in the wasted food stream as state and federal regulations prohibit feeding animal waste to animals or require pre-treatment in some contexts (Leib et al. 2016).Several of the pathways—such as controlled combustion and landfill—are better able to handle physical contamination in waste streams than other pathways. Physical contamination of wasted food streams can limit the range of applicable management pathways. Chemical contaminants in wasted food and co-treated wastes also drive some environmental impacts (human and ecotoxicity) and affect the use of resulting materials as fertilizers and soil amendments in agricultural systems. See Table 2-15 for more information.

Pre-Treatment of Wasted Food

Pre-treatment of wasted food is often employed to produce a consistent feedstock, free of physical contaminants that could hinder management pathway operation. However, by using additional energy, pre-treatment may increase the environmental impacts of the pathway. Most studies that include pre-treatment focus on three basic steps: sorting, grinding, and mixing. Several studies include additional thermal pre-treatment steps such as pasteurization and sterilization or de-packaging processes. Additional separation equipment such as crushers, screens and magnets may be required ahead of the de-packager, increasing overall pre-treatment energy use.

Source separation (i.e., separation by the generator) and separate collection of the food waste stream is one of the most fundamental pre-treatment sorting steps and can have a considerable effect on the quantity and quality of wasted food available. Pathways requiring high-quality feedstock with low levels of contamination (see Table 2-15) are likely to require this, particularly for commercial and residential wasted food.

Additional pre-treatment sorting may be required for pathways with low or very low tolerance for physical contamination, such as composting and animal feed. For this sorting step, Edwards et al. (2018) found that rejection of additional non-organic materials ranged from 5% to 45%, with an average rejection rate of 21%. Khoshnevisan et al. (2018) examined several methods for combined sorting, mixing, and size reduction, such as biopulping, screw press, and disc screening prior to AD. The authors concluded that pre-treatment steps make a minor contribution to GWP impact. An LCA of co-digested dairy products found that pre-treatment had a negligible effect on a broad range of environmental impacts, including GWP, eutrophication, acidification, and human toxicity (Kopsahelis et al. 2019). A review of advanced AD pre-treatment options found that pre-treatment processes that are mechanical or chemical in nature are environmentally preferable to thermal pre-treatment processes due to the higher energy demand of the latter (Tiwary et al. 2015). Use of residual waste heat is proposed to reduce the environmental impact of thermal pre-treatment processes.

De-packaging as a pre-treatment step for AD and composting pathways was studied in an analysis of Italian supermarket wasted food. Electricity demand of the de-packaging step was a major contributor to gross environmental impact for all pathways and impact categories considered, exceeding impacts associated with collection and transportation for all pathways and also exceeding impacts from treatment process operation for AD (Mondello et al. 2017).

Elginoz et al. (2020) considered the environmental benefits of using wasted food to produce a volatile fatty acid stream, which could then be used as the carbon source for AD (thus displacing the need for methanol production for the digester). The study found that one of the main drivers of impacts for their system was the energy required to thermally pre-treat the wasted food. This finding points to the importance of temperature and moisture: it is energetically expensive to heat wasted food through energy inputs, rather than through beneficial, exothermic biological reactions.

TABLE 2-15. HIGH-LEVEL SUMMARY OF ALLOWABLE CONTAMINATION, REQUIRED HOMOGENEITY, AND COLLECTION REQUIREMENTS FOR WASTED FOOD PATHWAYS

Pathway	Collection	Collection Notes	Contamination Tolerance	Contamination Notes	Homogeneity Required
Donation	Separate	Often packaged	Very low	No contamination or co-wastes; must meet food safety standards	Low
Upcycling	Separate	Generally collected from production or processing	Very low	No contamination or co-wastes; marketable as food once processed	High[a]
AD	Separate or mixed with other organics	Output quality benefits from source separation	Medium[55]	Pre-processing can remove some physical contaminants (e.g., wrappers)	Medium
Animal Feed	Separate		Very low[56]	Wasted food and any co-wastes must meet (or be processed to meet) animal safety requirements; must remove packaging that is unsafe to consume	Medium[57]
Composting	Separate or mixed with other organics	Output quality benefits from source separation	Low[58]	Pre-processing can remove some physical contaminants; quality requirements may depend on ultimate use (e.g., consumer gardens, agricultural fields, general land application)	None
Controlled Combustion	Mixed		High	Any waste permissible	None
Land Application	Separate	Often collected from industrial, pre-consumer waste streams.	Medium	Pre-processing can remove some physical contaminants	None
Landfill	Mixed		High	Any waste permissible	None

Pathway	Collection	Collection Notes	Contamination Tolerance	Contamination Notes	Homogeneity Required
Sewer/WWT	Mixed with other organics (at sink and in sewer)	Some wasted food (e.g., FOG) discouraged for sewer maintenance	Medium	Co-wastes (e.g., sewage) are acceptable and are generally also organic; processes are typically in place for removing foreign materials (e.g., bar screens)	None
Unharvested /Plowed In	Not applicable	Not applicable	Not applicable	No potential for contamination exists	None

[a] High homogeneity implies that for a given product from the pathway (e.g., an energy bar made of spent brewer's grains or a chemical precursor), the input wasted food must be homogeneous. However, across products, the homogeneity is low (i.e., a variety of feedstocks can be used to make different products).

CHAPTER 3
LIFE CYCLE ASSESSMENT

The eleven common wasted food pathways discussed in this report encompass a wide range of technological approaches and generate a spectrum of beneficial products, as described in Chapter 2. While all management pathways fulfill the essential service of managing wasted food, they vary in terms of environmental benefits and impacts. This section reviews relevant LCA literature to compare the environmental performance of wasted food pathways across a range of environmental indicators, focusing particularly on LCA study scenarios where wasted food is the feedstock whenever possible. This analysis focuses exclusively on environmental performance and does not address social and economic factors related to the pathways.

The first section (Section 3.1) describes the environmental indicators included in the LCA literature search, and Section 3.2 provides detail on the LCA review methodology. Section 3.3 provides a qualitative summary of pathways' environmental impacts and benefits. In Section 3.4, we look across studies (inter-study) to understand the range of environmental impact estimates for each wasted food pathway in the literature, while in Section 3.5, we look within studies (intra-study) to compare wasted food pathways. Section 3.6 discusses soil health, the only selected environmental indicator not included in the analysis within the previous two sections. Section 3.7 discusses the limitations of LCA and this LCA review. In Section 3.8 and 0 we synthesize the LCA results and the discussion of environmental drivers in Chapter 2 into a ranking of the eleven wasted food pathways.

3.1 Environmental Indicators

The LCA literature review focused on the eleven environmental indicators described in Table 3-1. These indicators were chosen due to their relevance to the wasted food pathways. Indicators were not chosen based on availability of inventory data or impact factors; inventory data for GWP (e.g., CO_2 emissions) are commonly reported and impact factors have international consensus; inventory for water use is less commonly reported, and impact factors are robust; inventory for soil health can be difficult to capture (e.g., total N vs. bioavailable N are not always distinguished), and impact factors are nascent. Impacts and benefits related to these indicators may result from wasted food management processes and/or from energy use during wasted food management processes, as well as from avoided energy or product (i.e., food, animal feed, fertilizer or peat) use due to outputs made from resources recovered from wasted food during the management pathway.

TABLE 3-1. ENVIRONMENTAL INDICATORS

	**Global Warming Potential (GWP)**	GWP impacts are reported in carbon dioxide equivalents (CO_2eq), which represent the heat-trapping capacity of the GHGs relative to carbon dioxide (CO_2) over a certain time horizon, typically 100 years (IPCC 2013). Biogenic carbon dioxide (e.g., from carbon incorporated by crops during growth stage) that is released back to the environment within the assessed time horizon is often treated as carbon neutral in LCA studies (ISO 2018). Biogenic carbon that is stored beyond the time horizon is typically treated as a credit (negative value) (ISO 2018). GWP impacts of wasted food pathways are primarily related to combustion processes that release fossil CO_2, anaerobic decomposition of organic matter that releases methane (CH_4), and agricultural emissions of nitrous oxide (N_2O).
	Soil Carbon Sequestration	Soil carbon sequestration is the processes by which carbon originating from the atmosphere is stored in the soil in a stable form. This benefit is generally quantified as part of the GWP indicator.
	**Energy Demand**	Energy demand is a cumulative inventory category, which may include non-renewable energy extracted and renewable energy used. It typically includes processing and transportation energy and energy content of non-waste feedstocks. In this report, energy demand includes both the energy requirements of a wasted food management pathway and credits for any energy recovered through that pathway. Energy demand contributes to GWP indicator (above) but provides an additional perspective by including only energy and excluding GHG emissions from anaerobic decay of food, for example.
	**Acidification**	Acidification quantifies the acidifying effect of substances on the terrestrial environment. Sulfur oxides (SO_x), nitrogen oxides (NO_x), and other pollutants such as ammonia (NH_3) may lead to acidification (Bare et al. 2003). Acidification emissions are typically from decomposition (NH_3) or combustion (SO_2, NO_2) processes.
	**Particulate Matter (PM) Formation**	PM formation is driven by primary pollutants (e.g., $PM_{2.5}$) and secondary pollutants such as sulfur oxides (SO_x), nitrogen oxides (NO_x) and ammonia (NH_3) that react in the atmosphere to form PM (Huijbregts et al. 2017). Primary PM emissions are associated with combustion processes, as are secondary emissions of SO_x and NO_x; decomposition processes release NH_3.
	**Human Toxicity**	Human toxicity describes effects of substances on humans. The modeled distribution of substances in environmental media (air, water, soil, vegetation) is coupled with exposure and effect data. Human health impacts are modeled based on carcinogenic and non-carcinogenic effects. Exposure via inhalation and ingestion are typically considered; dose-response effects are largely extrapolated from acute models (Fantke et al. 2018). Human toxicity emissions, in the wasted food context, are typically associated with contaminants released during combustion processes.
	**Ecotoxicity**	Ecotoxicity describes effects of substances on ecological receptors. The modeled distribution of substances in environmental media (air, water, soil, vegetation) is coupled with exposure and effect data. Ecological impacts in the aquatic environment are typically expressed in terms of the potential fraction of species affected (i.e., disappearing). These impacts are typically based on species sensitivity distributions across multiple phyla, accounting for bioavailability of compounds in the aquatic

		environment (Owsianiak et al. 2022). Ecotoxicity emissions, in the wasted food context, are typically associated with contaminants released during combustion processes.
	**Eutrophication**	Eutrophication is the process in which freshwater, marine, and terrestrial systems receive surplus limiting nutrients, resulting in excessive growth of plants and algae and therefore alterations in species composition or productivity (Henderson 2015). Eutrophication impact is modeled by translating nutrient emissions into phosphorus or nitrogen equivalents (e.g., Neq), which relates nutrients to potential primary productivity, and from which species impacts can be calculated (Payen et al. 2019). Eutrophication emissions may result from energy demand and combustion (NO_x), decomposition processes (NH_3), and fertilizer application or wastewater treatment (nitrogen and phosphorus compounds).
	**Water Consumption**	Water consumption typically refers to freshwater that is withdrawn (from surface or groundwater; see Pfister et al. (2017)) and evapotranspirated, incorporated into products or wastes, transferred to different watersheds, or discharged into the sea after use (Bayart et al. 2010). Water withdrawal (e.g., for cooling in power plants) is different from water consumption, as withdrawn water can be returned to the same watershed it was originally extracted from. Many studies use the phrase “water use” to refer to consumption, but this report uses the more precise term “water consumption.”
	**Land Occupation**	Total land occupation (e.g., in square meters during a year) is the amount of land required for a system or a process. Many studies go further and report the effects of that use on biodiversity. Given the variety of modeling approaches to connect land occupation to biodiversity, comparison of biodiversity impacts is challenging and is excluded from this analysis.
	Soil Health	Soil health is a broad, evolving concept that captures the ability of soils to support agricultural activities while continuing to provide other ecosystem services (Kibblewhite et al. 2008). USDA defines it as “the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans” (USDA 2021). Healthy soils provide habitat, cycle water and nutrients, support primary production, and sequester carbon (Lehmann et al. 2020). There is not yet consensus about a definitive set of indicators to measure soil health (Rinot et al. 2019) in LCA. Therefore, no quantitative metrics for soil health were available for this report, and improved soil health is not included in the quantitative LCA rankings—a limitation of this evaluation. Instead, this report qualitatively considers the effect that wasted food pathways have on the following metrics of soil health, as discussed by Lehmann et al. (2020): • Provide recycled organic fertilizers and soil amendments. • Avoid harmful contaminants, pathogens, and pests. • Enhance water storage. • Promote good water quality. • Promote soil biodiversity. • Sequester carbon (included within GWP indicator in this analysis).

3.2 Methodology

Description of Literature

The literature review that serves as the basis for this analysis, returned approximately 3,200 peer-reviewed journal articles and government and non-governmental organization reports. We reviewed approximately 250 in depth and extracted LCA-based data from 74. Of those, 47 provided useable data; data from 23 studies was used in both the inter- and intra-study analyses (in Sections 3.4 and 3.5, respectively); 10 were used only in the inter-study, and 14 were used only in the intra-study.

This review was able to qualitatively assess all major wasted food pathways; however, data were not available for each pathway across all the environmental impact categories of interest. Of the considered wasted food pathways, AD, compost, landfill, and controlled combustion are the best represented in the reviewed literature (see Figure 3-1). Global warming, eutrophication, ecotoxicity, and acidification are the impact categories most assessed across the reviewed studies (see Figure 3-2). Although ecotoxicity was frequently reported, this impact was reported variously as terrestrial, aquatic, marine, and general, using a variety of impact metrics. This variability prevented standardization of impact results. The same variability was true for other categories, such as human toxicity. Carbon sequestration and soil health were noted in some studies that dealt with application of wasted food to land or landfill, but inconsistently (11 studies included sequestration, whereas 25 did not). Those studies that included soil health had infrequent and variable qualitative analysis.

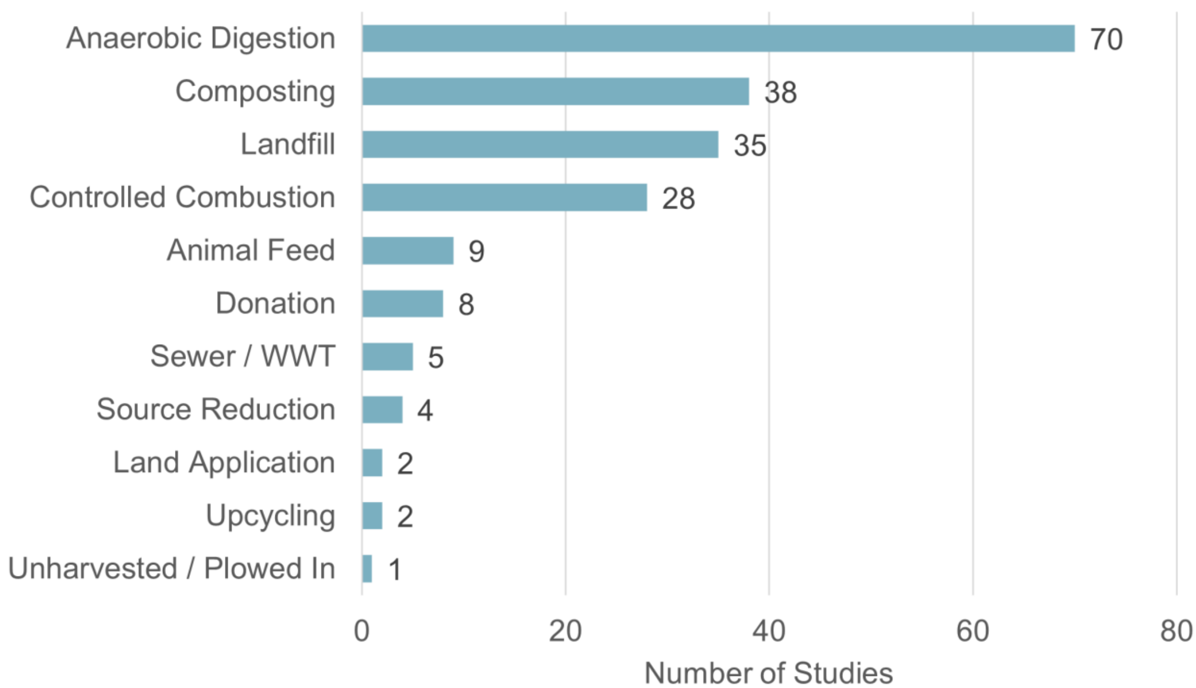

FIGURE 3-1. DISTRIBUTION OF WASTED FOOD PATHWAYS ASSESSED IN REVIEWED STUDIES

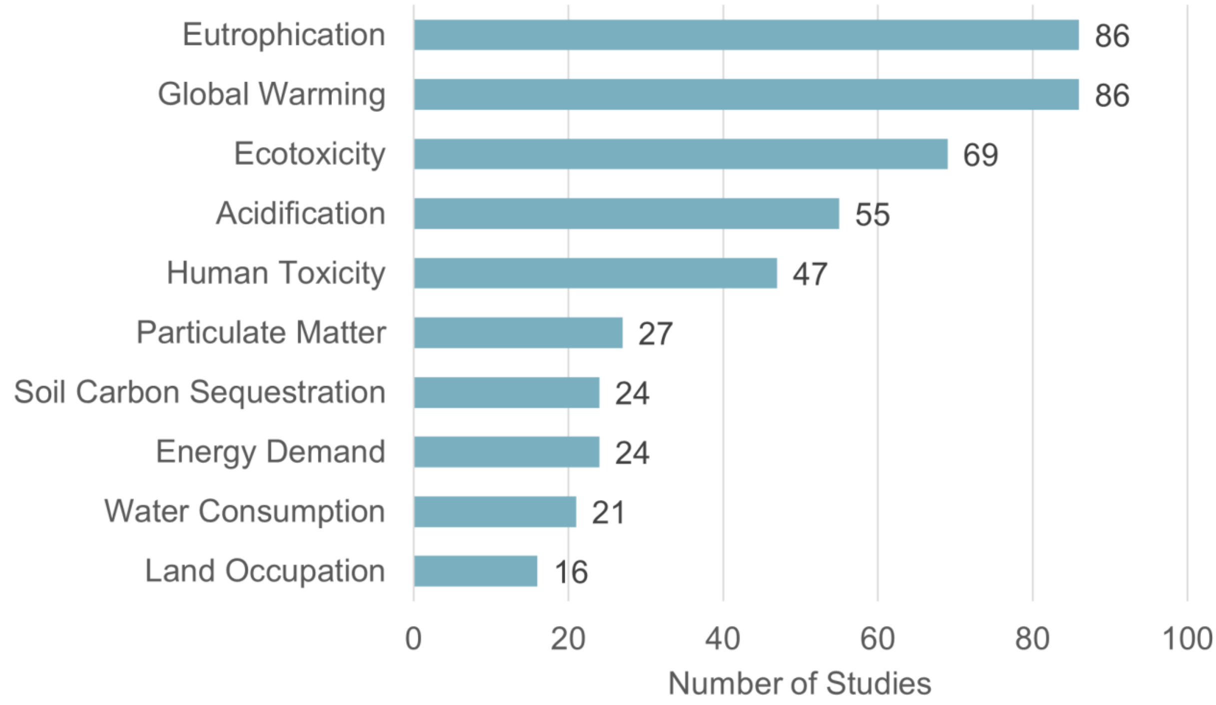

FIGURE 3-2. DISTRIBUTION OF ENVIRONMENTAL INDICATORS ASSESSED IN REVIEWED STUDIES

Each count indicates a single study that addresses the indicator or pathway indicated; single studies may be counted multiple times, as a study could evaluate multiple pathways, scenarios, and/or impacts. Soil health is not included in Figure 3-2, as no studies had quantitative data that could be used in inter-study (Section 3.4) or intra-study (Section 3.5) analysis.

System Boundaries

Within the framework of LCA, system boundaries refer to the definition of a set of processes that are part of the system under consideration and the set of modeling choices that are applied to that system. The diagrams in the previous chapter identify the set of processes that are relevant for each pathway. However, different studies may choose different system boundaries; therefore, studies differ concerning relevant aspects of each management pathway. For example, some studies include waste transportation or modeling of physical infrastructure, while other studies consider these features to be outside the scope of their analysis. Further, the presence of a process or avoided product in a pathway system diagram does not imply that any particular process is necessary to adequately treat wasted food or accurately estimate pathway environmental impact.

Each study also makes specific modeling decisions regarding the performance of the wasted food pathway for which environmental impacts are being estimated. While the system boundaries of a study will specify that wasted food is landfilled, modeling assumptions will specify the details of landfill operation: assumed LFG production, the quantity of energy recovered, methane leakage, and other parameters.

Variation within Pathways

It is beyond the scope of results in this chapter to distinguish between the types and management of technologies within a pathway; however, below we provide details on LCA studies reviewed for transparency and comparison to real-world distribution of types.

“Type,” as introduced in Chapter 2, refers to the specific pathway technology used. For example, composting might use windrow or ASP systems; controlled combustion might involve grate furnaces, spreader stoker furnaces, or other technology. “Scenario” refers to a set of operational and modeling assumptions used to describe a specific, modeled implementation of a pathway type in a study.

- *AD:* Across the LCA studies reviewed in greater detail, at least 55 include one or more AD scenarios. The majority of reviewed scenarios (33) model mesophilic digesters, seven model thermophilic conditions and 22 do not specify the digester temperature regime. Twenty-five scenarios clearly pertain to co-digestion practices. Among the remaining scenarios, it is often difficult to determine if wasted food is the sole feedstock. Within the co-digestion scenarios, wastewater solids are the most common co-digestion feedstock, followed by manure, yard trim and agro-industrial waste(s).
- *Composting:* Within the reviewed studies, at least 37 composting scenarios were identified. Thirteen of those scenarios model windrow composting, eight model an in-vessel composting system, six model aerated static piles, and the remainder model hybrid systems, home systems or do not specify a composting method. Twenty composting scenarios accounted for carbon sequestration in soil, typically assuming that 10-15% of applied carbon is sequestered. Thirty-one scenarios accounted for compost emissions, with two of those not accounting for methane.
- *Controlled Combustion:* Of the 25 reviewed LCA studies with a controlled combustion scenario, 15 scenarios model mass burn combustion facilities, three scenarios model a combustion process that involves some form of pre-drying and ten scenarios do not specify the details of the modeled combustion process. Moisture content for wasted food for controlled combustion is typically between 67-80%, with the majority of studies assuming or sampling wasted food at 70-75% moisture content. Energy content of wasted food going to controlled combustion was typically between 2.5 and 8 MJ/kg LHV. Scenarios that generated electricity with waste heat used heat efficiencies between 15 and 23%. Scenarios that examined wasted food streams with physical impurities (packaging, yard trim, other parts of OFMSW) typically assumed that the stream was at least 80% wasted food (Clavreul et al. 2012; Lee et al. 2020).
- *Landfill:* Within the reviewed LCA studies at least 34 landfill scenarios were identified. Twenty-eight scenarios accounted for gas capture, with varying assumptions about gas capture efficiency. Typical gas capture efficiencies increased over the lifetime of the landfill, starting at 0-50% and ending at 85-95%. Scenarios that used a single gas capture efficiency typically assumed 60-65%.

“Operations” (i.e., an operational practice in use—for example, using a specific set of emission controls or using/not using bulk waste or refused-derived fuels in controlled combustion) are not considered within this analysis, other than in the scope criteria for inter-study analysis (Section 3.4) where studies/facilities that do not include certain key avoided product benefits are excluded.

Modeling Assumptions about Recovered Resources

Avoided Energy

Those management pathways that can recover energy as electricity or heat often receive a significant credit for that recovery. The benefit of energy production depends on the grid mix displaced – that is, recovered energy must offset energy production that has a greater environmental impact for a pathway to receive credit. Depending on modeling assumptions, pathways such as AD and controlled combustion may be net negative in terms of GWP. However, this does not mean that they are net negative relative to the original production of food. Secondly, the benefit received is dependent upon the energy source displaced, and thus is specific to location and time. For example, Slorach et al. (2019a) conducted an analysis of wasted food pathways that recover energy (AD, controlled combustion and landfilling). They found that the avoided energy credits assigned to each pathway had a direct influence on most impact categories including energy demand, GWP, toxicity, eutrophication, and water consumption. That analysis also considered the effect of substituting other energy sources, rather than the current grid mix (in the UK), finding that if the displaced energy is wind or solar, then AD and controlled combustion switch from being net sinks to net sources (i.e., net negative to net positive) of GWP. The study noted that benefits of wasted food pathways are “highly dependent on system credits” (Slorach et al. 2019a).

While a considerable quantity of heat can be recovered in addition to the electricity produced, many studies exclude an environmental credit or include only a partial credit as heat is used to operate treatment processes and no uses are typically available for excess heat. Avoided energy benefits are realized based on net energy production, which constitutes total energy production less energy demand required to operate wasted food treatment systems and associated processes. Due to the mix of input data, modeled data, and assumptions in the reviewed LCAs, absolute energy production varies from study to study. High-level trends from the reviewed studies are summarized below.

Anaerobic Digestion: Most reviewed LCA study scenarios (59 of 64) of AD assess a benefit for avoided electricity production. Twenty-six scenarios include a credit for avoided heat production. Eight scenarios model the production of biofuel from biogas. Across a subset of reviewed studies for which electricity recovery numbers were reported, gross electricity production per metric ton of wasted food processed is between 100 and 420 kWh (Parry and WERF 2012; Yoshida et al. 2012).

Landfill: 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 (Parry and WERF 2012; Huang et al. 2022).

Controlled Combustion: Across 25 reviewed controlled combustion scenarios, 22 specify that they include avoided electricity production and 14 include avoided heat production. The quantity of electricity generated from combusting one metric ton of wasted food varies between 138 and 564 kWh (Mondello et al. 2017; Hoehn et al. 2021).

Avoided Products

The quantity of avoided product (i.e., food, animal feed, fertilizer or peat) assumed to generate environmental credits for a given pathway can vary depending on a variety of characteristics. For example, the protein content of specific wasted food feedstocks determines the quantity of feed avoided by the animal feed pathway.

How avoided products are used, and whether modeling accurately reflects their use, is critical to understanding the environmental performance of each pathway. While soil amendments can offset fertilizer, if compost, digestate, or biosolids are disposed of or used as a landscaping mulch, then claiming an avoided fertilizer benefit may not be appropriate (if the landscape plants would not have otherwise received fertilizer). The context of avoided product use is a factor in determining the magnitude and validity of avoided product credits.

Within the reviewed LCA literature, the majority of studies assume that AD digestate or biosolids are land applied. Among the seventy reviewed study scenarios that model AD, six assume that digestate is incinerated and one assumes that it is landfilled. Nine scenarios did not include digestate management, declaring it out-of-scope. All reviewed studies assume that compost is land applied and all but one study applies an avoided fertilizer production benefit to the assessment of net environmental impacts. Avoiding the use of peat soil amendments is assessed infrequently within the literature.

Directly land-applied wasted food, unharvested/plowed in crops, digestate, biosolids, and compost are different with respect to nutrient content, organic matter content, nutrient release, and other factors. However, the reviewed LCA studies reflect a simplified approach to assigning benefits related to application of these outputs to soil. For instance, throughout the literature, similar ratios are often used for both compost and digestate offsetting synthetic fertilizer production and use and typically fall in the range of 20-50% for nitrogen and 95-100% for potassium and phosphorus (Yoshida et al. 2012). The range for N fertilizer reflects how substitution ratios are intended to reflect plant availability of nutrients in compost or digestate (Levis and Barlaz 2011). With higher readily available nitrogen content, digestate is more commonly viewed as a fertilizer, especially in liquid or slurry form, and therefore may be better suited as a substitute for synthetic N fertilizer, which is immediately available to plants. While the total nitrogen content in compost is typically higher than that of digestate, its readily available nitrogen is lower (Stehouwer et al. 2022). Compost will provide less nitrogen in the first year but more nitrogen over a longer time frame than digestate. Soil amendments produced by other pathways (e.g., biosolids, unharvested/plowed-in crops) can also offset fertilizer use, but these pathways are assessed infrequently in the literature. For all wasted food-derived soil amendments, assumptions about quantity, bioavailability, and loss of nutrients determine the quantity of synthetic fertilizer that is avoided, and credits from fertilizer production and use are calculated inconsistently. Different fertilizer substitution ratios may be applied across or within studies. Given the high-level analysis of LCAs, one should not expect LCA to accurately capture the specificities of any particular wasted food processing operation, especially with respect to soil health and nutrients.

As an alternative to or in addition to fertilizer replacement, a small number of studies look at using digestate or compost to substitute the use of peat growing media. Compost is assumed to displace peat use 1:1 on a volume basis (Levis and Barlaz 2011). Studies assumed that the GWP benefits of avoiding peat use were similar to (Morris et al. 2017) or greater than avoided fertilizer benefits (Levis and Barlaz 2011).

Additionally, wasted food content varies in each material, depending on co-feedstocks. Studies report a range of effects on soils and crops and a variety of environmental impacts. A consistent framework is not applied when assessing the benefit/impact of wasted food-derived nutrients as compared to synthetic fertilizers in LCA research. The reviewed studies vary widely in their approaches when it comes to estimating carbon sequestration and emissions to air and water resulting from land application of compost, digestate, biosolids, or wasted food. These factors make the relative benefits and impacts of land application of these materials difficult to quantify according to any standardized, comparable metrics. Therefore, qualitative analysis of the agronomic and environmental benefits and impacts of land-applied wasted food-derived materials is provided in Section 3.6 and Chapter 4. The research gaps section highlights the need for further studies to better understand the behavior of raw wasted food, unharvested/plowed in crops, digestate, biosolids, and compost when land applied.

Modeling Assumptions about Soil

A meta-analysis of N_2O emission from land application of amendments suggests a generic value 40% lower than IPCC's emission factor, and highlights the importance the C:N ratio of the amendment, the soil characteristics (texture, drainage, organic C and N), and precipitation (Charles et al. 2017). N_2O and other emissions from land-applied material (digestate, compost, biosolids) is an important component of the relevant pathways' GWP and other impacts. Section 3.6 addresses soil amendments and soil health, and we note that there is rich literature describing these systems (e.g., Charles et al. 2017),

Carbon storage potential is influenced by site history, soil type, management practice and land use. Carbon sequestration rates are also impacted by existing levels of soil organic carbon (SOC), with targeted application on previously disturbed or degraded soil resulting in higher levels of net carbon storage (Peltre et al. 2012; Brown and Beecher 2019). The type and composition of organic matter applied to agricultural fields also has a demonstrable impact on SOC storage, and existing SOC affects rates of change (higher initial SOC results in slower SOC gains) (Poulton et al. 2018). Finally, we note that SOC accumulation can be augmented, decreased, or even reversed if management practices are changed(e.g. no-till practices are adopted; soil is left bare/unplanted) (Poulton et al. 2018). Studies assume long-term, if not permanent, changes to soil management practices. The impacts of specific soil management practices, soil types, and land uses on carbon storage potential and GHG emissions is beyond the scope of this report.

3.3 Qualitative Summary of Pathway Environmental Impacts and Benefits

Table 3-2 summarizes the key drivers of environmental impact and benefit (including credits for avoided energy and products) from each wasted food pathway.

In reviewed LCAs, not all impacts and benefits are covered in each study (Figure 3-1 and Figure 3-2). As noted in the previous section, most LCAs reviewed quantified benefits for avoided electricity production where appropriate; some also included avoided heat or fuel production. LCAs also credited pathways for avoided production of food, animal feed, fertilizer, or peat and the sequestration of carbon; however, these credits were not consistently applied.

TABLE 3-2. SUMMARY OF ENVIRONMENTAL IMPACTS AND BENEFITS, BY PATHWAY

Pathway	Environmental Impacts	Environmental Benefits
Energy use is a source of environmental impact for all pathways, except source reduction, and is not listed separately by pathway.		
Source Reduction	-	Avoided Food Production
Donation	GWP: Distribution and storage	Avoided Food Production
Upcycling	Water consumption: Food processing	Avoided Food Production
Anaerobic Digestion	GWP: CH_4 from biogas leakage Eutrophication: WRRF effluent release (from recirculating dewatering liquid)**; land application of digestate or biosolids PM formation, Acidification: Biogas combustion	Avoided Energy Production Avoided Fertilizer Production/Use* Avoided Peat Harvest* Carbon Sequestration* Soil Health*
Animal Feed	GWP and Eutrophication: Feed processing	Avoided Feed Production
Composting	GWP: CH_4 and N_2O emissions Acidification, Eutrophication, PM formation: NH_3 emissions Eutrophication: land application of compost	Avoided Fertilizer Production/Use Avoided Peat Harvest Carbon Sequestration Soil Health
Controlled Combustion	Toxicity: flue gas (Hg, dioxin) PM formation: Flue gas	Avoided Energy Production

Pathway	Environmental Impacts	Environmental Benefits
Land Application	Eutrophication: land application of wasted food	Avoided Fertilizer Production/Use Carbon Sequestration Soil Health
Landfill	GWP: CH_4 from anaerobic decay of food Toxicity and Eutrophication: Untreated landfill leachate PM formation, Acidification: Landfill gas combustion	Avoided Energy Production Carbon Sequestration
Sewer/WWT	GWP: CH_4 from anaerobic decay of food in sewer and from WWT processes Eutrophication: WRRF effluent release; land application of biosolids	Avoided Energy Production** Avoided Fertilizer Production/Use* Avoided Peat Harvest* Carbon Sequestration* Soil Health*
Unharvested/ Plowed in	-	Avoided Fertilizer Production/Use Carbon Sequestration Soil Health

*Appropriate only if digestate or biosolids beneficially used

**If AD at WRRF

Emissions from application of wasted food-derived soil amendments not included above because presumed to be lower than those related to synthetic fertilizers.

"- "minimal environmental impacts expected compared to magnitude of pathway environment benefits.

3.4 Quantitative Comparison of Pathways Across Studies

The following two sections (3.4 and 3.5) present results and discussion related to the environmental indicators described in Table 3-1. Section 3.4 compares wasted food pathways across studies (inter-study) and Section 3.5 compares environmental performance of wasted food pathways within individual studies (intra-study). The two approaches complement one another. The benefit of the inter-study analysis is that it facilitates a wide range of comparisons and provides a high-level comparison of all pathways regardless of study considerations. The benefit of the intra-study analysis is that comparisons are made using consistent scope and modeling.

This section compares wasted food pathways across studies (inter-study) for six impact categories – GWP, energy demand, acidification, eutrophication, water consumption, and land occupation – based on their median environmental impact relative to that of other pathways.

While the methods for assessing certain impacts are well established (GHG emissions), others are nascent (soil health). The latter is a limitation of LCA, discussed further throughout the text. The six environmental indicators presented quantitatively (both in graphics and in tables) in this section were chosen from the 11 described in Section 3.1 based upon two factors. First, they are commonly assessed in LCA studies, and second, when assessed, they are reported in consistent impact units, allowing for comparison (possibly after conversion). The remaining impact categories that are not quantitatively analyzed in this section did not meet these criteria, preventing meaningful comparison of results across studies. For example, particulate matter formation was not consistently reported across the reviewed studies, and human and ecotoxicity were reported in a variety of metrics. However, particulate matter and toxicity (human and eco) are examined in the intra-study rankings. Only soil health could not be examined quantitatively in either section. It is not yet possible to quantitatively address soil health in LCA but given the importance of soil health for food production and the fact that much wasted food can make its way back to the soil, we have included this as an environmental indicator, nonetheless. Further discussion of soil health can be found in Section 3.6.

Methodology for Inter-Study Analysis

Scope Criteria

A limited set of scope criteria, defined in Table 3-3, were applied to enhance comparability of impact data used in Section 3.4. Thirty-four of the LCA studies evaluated met the scope criteria and were included in the quantitative analysis in this section. Of those thirty-four, twenty-three were also used in the intra-study analysis, described below. As a rule, data from studies that met scope criteria were included. The only exceptions to this inclusion were cases in which impact results exceeded the range of realistic values, based on professional judgment, and for which a sufficient justification/explanation was not identified in the original reference.

TABLE 3-3. INTER-STUDY ANALYSIS (SECTION 3.4) SCOPE CRITERIA

All	Studies must not include an environmental credit for avoided disposal (e.g., AD is not credited for avoiding landfill).
	Studies must calculate impacts beginning at the point wasted food arrives at pathway through final disposal or beneficial use. (Studies may also include impacts of transportation to pathway.)
AD	Studies must include environmental credits for avoided energy and fertilizer production.
Animal Feed	Studies must include environmental credits for avoided animal feed production.
Compost	Studies must include environmental credits for avoided fertilizer production.
Controlled Combustion	Studies must include environmental credits for avoided energy production.
Landfill	Studies must include environmental credits for avoided energy production.
Other Pathways	No additional scope criteria applied due to presence of limited study data.

Studies may model electricity or heat (or both) in order to meet the criteria for including avoided energy.

Carbon sequestration was not applied as a scope criterion, but further analysis of results with and without carbon sequestration credits is presented in Table 3-4.

For commonly assessed management pathways, only study results that modeled systems with certain operational practices and included certain avoided products were chosen for comparison in this section, per the criteria shown in Table 3-3. For example, data on controlled combustion were not used if they came from studies that did not include credit for avoided energy production (because it was either not modeled or not implemented). Also, data on AD where digestate or biosolids were disposed were not included. Study data were excluded for all pathways if their calculation of net impact included an environmental credit for avoided disposal, since this would lead to double counting in a direct comparison of impacts of alternate pathways. In general, the majority of the identified studies met these criteria for avoided products – identifying avoided products is common practice in LCA.

As only a few studies included estimates of carbon sequestration associated with each management pathway, this characteristic was not applied as a scope criterion, as it would have further limited the number of studies providing data. An analysis comparing results of studies with and without carbon sequestration is included in the analysis in Table 3-4.

Impact data were screened for outlier data and the studies providing these data were given additional review to ensure that modeling choices align with criteria in and are representative of the wasted food pathways as described in Chapter 2. Beyond the criteria described here, results have not been standardized in terms of scope or other boundary assumptions (e.g., one study may include transportation and another not). Because they involve studies with different scopes and different impact methods, these results provide a broad perspective on the performance of wasted food pathways.

Data Presentation

In the following sections, Figure 3-3 to Figure 3-8 present detailed impact results for the wasted food pathways across a variety of impact categories. These figures show all median values on a single axis. Median impact results are presented per metric ton of wasted food processed in the noted units for each impact category. Often, medians for source reduction and donation are significantly different than the other pathways. Therefore, the figures also have insets to show distributions of the data (as box plots) for the other pathways. The bottom and top of each box represent the first and third quartiles (covering 50%

of the data); median values are shown with a dark line between the quartiles, and the whiskers represent 1.5 times the interquartile range (25th-75th percentile). The discussion of the data underlying the boxplots refers to individual study results. While individual study results are not shown in the boxplots, the data underlying these figures can be found in Appendix D.

In the graphics in this section, the number of data points is indicated with the name of the wasted food management pathway. Some studies report multiple results for scenarios, resulting in multiple data points per study. Sometimes these scenarios yield identical results; in these cases, only one data point has been used for that management pathway in that study. Therefore, the number of data points is always greater than or equal to the number of studies for that pathway.

Graphics and tables present results with a standardized functional unit of 1 metric ton of wasted food (wet basis). The moisture content of considered wasted food is not standardized across studies. The graphics and tables include studies with variations in the type of wasted food considered (e.g., food processing waste versus plate waste). Where a functional unit included a co-waste (e.g., wasted food and yard trim), the study scenario was only included if wasted food was the dominant (>50%) waste material being processed and if the co-waste was also organic. The approach to comparing disparate impact units is described in detail in Appendix C.

These data allow for high-level comparisons among pathways, as designed. However, the data were not intended to allow for rigorous statistical analysis – there are not enough data points to arrive at meaningful confidence intervals. Therefore, in the results that follow, median values and ranges are the main descriptors used.

For resource recovery or avoided products, modeling assumptions can influence a variety of environmental impacts (e.g., if a wasted food pathway recovers energy that leads to avoided electricity production, that avoided electricity will reduce GWP, energy demand, and acidification impacts). Given the influence of resource recovery and avoided products on impacts beyond GWP, observations in the Global Warming Potential section often apply to the other sections but are not repeated in those sections. For example, a high-efficiency biogas capture from AD will lead to a reduced impact for energy demand (directly) and for acidification and eutrophication (indirectly, as energy from the AD may offset NO_x emissions from grid energy sources).

Table 3-4 and Table 3-5 present summary environmental impact results for the wasted food pathways.

Global Warming Potential

Multiple factors impact the GWP of wasted food pathways, including:

- Anaerobic decay of wasted food (fugitive CH_4 from sewer/WWT and landfill).
- Avoided food and animal feed production (CO_2, CH_4, N_2O savings from source reduction, donation, upcycling and animal feed).
- Avoided energy production (CO_2 savings from AD, controlled combustion, and landfill).
- Wasted food management process emissions (CH_4 leakage in AD and CH_4 and N_2O from composting).
- Energy use (CO_2 from all pathways except source reduction).
- Carbon sequestration (composting, landfill, land application and unharvested/plowed in, plus AD and sewer/WWT if soil amendments are beneficially used).
- Nitrous oxide emissions from land application of wasted food or wasted food-derived soil amendments (composting, land application and unharvested/plowed in, plus AD and sewer/WWT if soil amendments are beneficially used).
- Avoided synthetic fertilizer production and use (CO_2 and N_2O from composting, land application and unharvested/plowed in, plus AD and sewer/WWT if soil amendments are beneficially used in ways that reduce application of synthetic fertilizer).
- Avoided peat harvest.
- Emissions associated with landfill or controlled combustion of waste outputs (in AD and sewer/WWT pathways if soil amendments are disposed of rather than beneficially used. Note this was not modelled in reviewed studies of AD since land application of digestate/biosolids was assumed.

Not all these drivers are included in all LCA studies from which data were extracted. See Table 3-3 for scope criteria for inclusion within this analysis.

Figure 3-3 shows GWP impact values for the wasted food pathways. Pathways may achieve net negative values through avoided production of a resource, whether it is food, animal feed, energy, or fertilizer, possibly coupled with the sequestration of carbon in soil. Release of carbon dioxide from wasted food itself is treated as net neutral due to the biogenic (non-fossil) origin of the carbon.

In Figure 3-3 (and all boxplots in subsequent sections) the bottom and top of the boxes represent the first and third quartiles (25th – 75th percentile, covering 50% of the data), the line within the shaded box represents the median value, and the whiskers represent 1.5 times the interquartile range (25th-75th percentile). All data points meet basic scope criteria for each pathway (see Table 3-3), such as application of GHG credits for avoided products and energy, as defined in Table 3-3. Carbon sequestration is not consistently included for some pathways. Numbers in parentheses next to pathway names indicate the number of study scenarios included.

Global Warming Potential (kg CO_2eq/metric ton FLW)

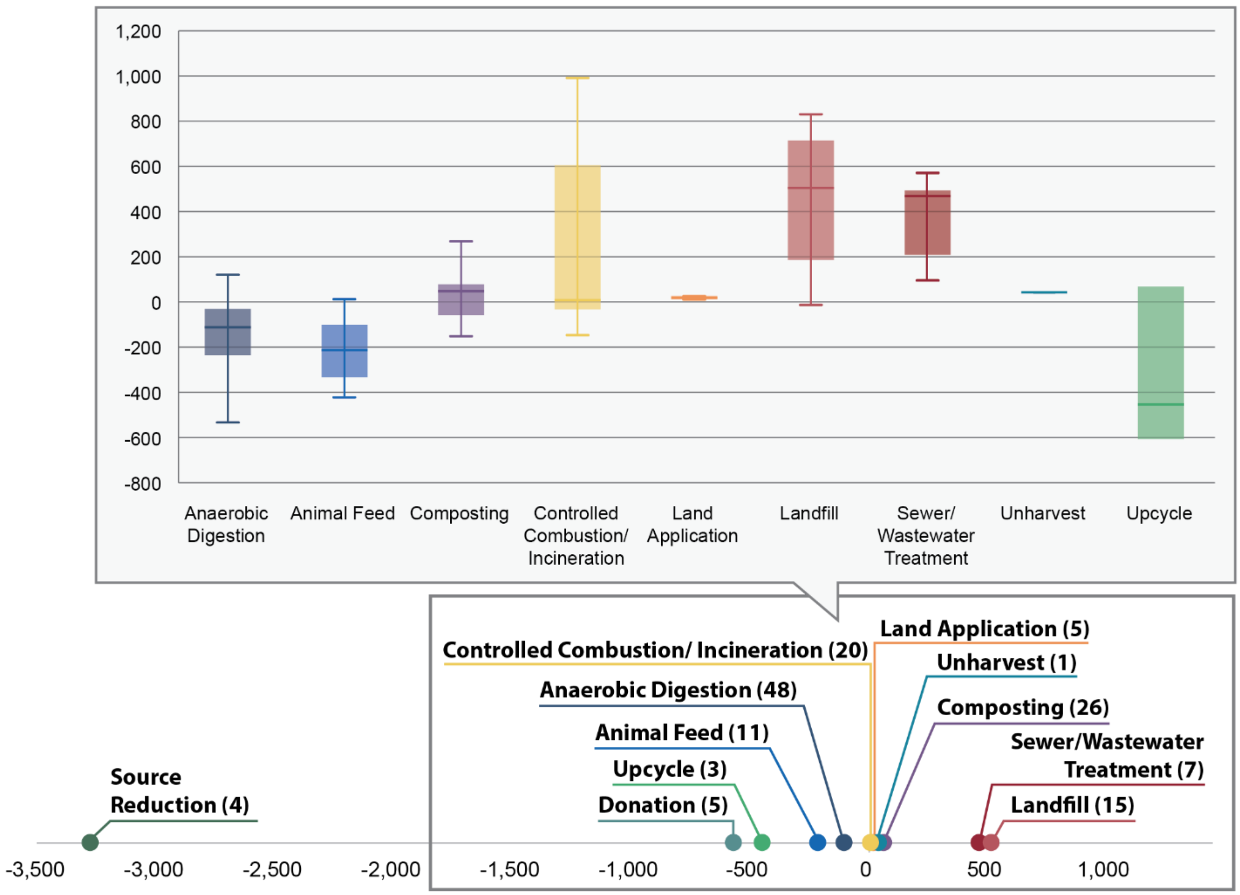

FIGURE 3-3. COMPARISON OF GLOBAL WARMING POTENTIAL OF WASTED FOOD PATHWAYS ACROSS LCA STUDIES

AD data assumes land application of digestate or biosolids. Sewer/WWT studies varied in assumptions about biosolids' destination. Carbon sequestration not consistently assumed for some pathways. See Table 3-4 for details. Donation excluded from boxplot because the range extends an order of magnitude below the next lowest value in the boxplot. See Appendix Table D-1 for data and sources.

The GWP analysis shows that source reduction had order-of-magnitude greater GHG benefits than any other pathway. Sewer/WWT and landfills tend to have the highest GWP values. Among other pathways, donation, upcycling, animal feed and AD have the largest potential for beneficial performance. But the interquartile ranges suggest the potential for similar impacts among the AD and animal feed pathways and the compost, controlled combustion, and land application pathways.

Among the pathways in the inset boxplot, values for GWP can be split into two groups describing broad pathway performance.

- Negative (beneficial) medians and medians close to zero:
  - Source reduction has order-of-magnitude greater GWP than other pathways.
  - Donation and upcycling avoid production of food, avoiding substantial GHG emissions, but also use energy for transportation and, in the case of upcycling, food processing, offsetting some GHG savings.
  - Animal feed values are almost entirely negative, as the energy needed to collect, clean, and process wasted food tends to be less than that needed to produce and process feed crops.
  - For AD, negative values are achieved in systems with high biogas capture and displacement of carbon-intensive energy. All included AD studies assume land application of digestate or

biosolids, thus assigning GHG credits for avoided fertilizer manufacture and use, and not assigning GHG emissions related to landfill or controlled combustion of digestate or biosolids.

- Composting values vary as a function of different operational characteristics and assumptions about fugitive emissions, the degree to which studies assume fertilizer and peat are offset, and whether carbon sequestration credits were applied. Table 3-4 shows inclusion of carbon sequestration can shift values from positive to negative.
- Controlled combustion varies widely, with energy recovery efficiency as a key parameter.
- Land application values, all from ReFED (2023), are all positive, reflecting assumptions about fugitive methane and nitrous oxide (N_2O) emissions.
- Unharvested/plowed in has a single positive value, with limited documentation (ReFED 2023).

- Positive values, with median value above zero:
  - For sewer/WWT, WRRFs do not capture methane produced in sewers and have process emissions of nitrous oxide; these releases offset credits from (potentially) recovered resources, such as energy (if AD present) and nutrients (in biosolids). Table 3-4 shows that carbon sequestration from land-applied biosolids is assumed in two studies; these two have a lower median than the five data points without sequestration but remain positive.
  - Landfills capture a portion of the methane produced, which reduces their net GWP impact, but capture rates are not adequate to offset overall impacts.

For most pathways, there is a range of values, sometimes spanning positive and negative, that reflects modeling choices or system operation.

For example, for animal feed, the lowest value (-420 kg CO_2eq) is for a wasted food process that represents bioconversion of wasted food into animal feed using black soldier fly larvae. The process receives a large credit for avoided soy and fishmeal production, as well as a credit for compost utilization (Mondello et al. 2017). The next lowest values (-385 and -329 kg CO_2eq) are associated with a specialized protein concentrated animal feed (Albizzati et al. 2021b). The largest credit for this system is associated with avoided land use change attributed to avoided feed production. (Land use change impacts are a key consideration associated with agricultural products, though they are often excluded from the system boundaries of LCA studies.)

For AD, the lowest values assume avoided electricity and heat production of Danish coal-based power, so the GWP benefit of this biogas-produced energy is higher than it would be with a lower-carbon power mix (Khoshnevisan et al. 2018). In contrast, the highest AD value reflects an assumption of relatively high emissions from land application of digestate (Albizzati et al. 2021b). Other positive AD values are generally due to scope or operational assumptions. For example, an AD impact of 27 kg CO_2eq is calculated from a study that includes composting of digestate (Al-Rumaihi et al. 2020) and values of 8 to 26 kg CO_2eq from Pace et al. (2018) are calculated with assumptions of relatively low electrical generation efficiency.

For controlled combustion, values above the median are associated with wasted food that has a very low heating value and a system with relatively low energy recovery efficiency; for example, 235 kg CO_2eq (Ahamed et al. 2016).

The sewer/WWT and landfill pathways have positive median values. These two pathways include appreciable methane production, with either uncontrolled or limited capture. Consistent with the scope criteria in Table 3-3, data points pertaining to landfills without gas capture or energy recovery have been excluded from this analysis, given that over 90% of U.S. landfilled waste was disposed of in a facility with an active LFG collection system in 2021, and 65% of LFG is sent for energy recovery (U.S. EPA 2022a).

For landfills, the highest data point is from a study that includes gas capture but assumes low capture rates, demonstrating the impact of landfill operation: with an assumption of 35% gas capture in the first five years of operation and 52% over landfill life, the GWP of landfilling is 2,400 kg CO_2eq/metric ton wasted food (Albizzati et al. 2021b). Other LCA study values for landfilling are <1,000 kg CO_2eq/metric ton wasted food, which is lower than an estimate from the EPA LandGEM model[59]. LandGEM estimates

54 kg methane emitted per metric ton wasted food for landfills with gas collection. Applying GWPs from the Intergovernmental Panel on Climate Change's Fifth Assessment Report (Myhre et al. 2013), this emission corresponds to 1,500 kg CO_2eq/metric ton wasted food for a 100 year time horizon and 4,500 kg CO_2eq over a 20-year time horizon. (Note that these LandGEM estimates do not account for carbon dioxide emissions from landfilling or for energy offset by captured methane, unlike LCA studies considered in this report.)

Carbon Sequestration

Increasing storage of SOC is an important approach to GHG mitigation, as the amount of carbon stored in soils exceeds that in the atmosphere and in living biomass (Ciais et al. 2013). In the context of this report, carbon sequestration refers to the storage of wasted food carbon in the subsurface (e.g., over a period of 100 years). Wasted food carbon reaches the subsurface through the application of wasted food, digestate, biosolids, and compost to land or the burying of wasted food in a landfill. Addition of carbon-rich amendments and resulting increased SOC can improve soil health by increasing water infiltration rates and retention, reducing bulk density, and reducing risk of losses from erosion by improving soil aggregate stability (Brown et al. 2020). Any increase in SOC results in outsized improvements in soil functioning and quality (Poulton et al. 2018). Many studies quantified carbon sequestration within the GWP indicator, but including sequestration was not consistent across all studies. Those studies that do include carbon sequestration have included a benefit (negative GWP) for this sequestration. Across those pathways that deal with carbon sequestration, the presence or absence of sequestration is the one categorical variable that was used (i.e., it was not used as a scope criteria).

Table 3-4 shows the median and ranges of GWP values, with and without carbon sequestration, for wasted food pathways where sequestration is possible. There is significant overlap in these values, indicating that the inclusion or exclusion of sequestration has a moderate effect on overall GWP estimates that is in line with other aspects of management and operational performance.

Across pathways with data for both conditions, impacts with sequestration are lower than results without carbon sequestration (see also Appendix B-1-1). For AD and compost, the shift is on the order of tens of kg CO_2eq / metric ton wasted food. For sewer/WWT, the shift is on the order of hundreds of kg CO_2eq / metric ton wasted food. The median offset for composting is larger than for AD. Given the higher median impact of the composting pathway, sequestration is potentially more consequential to the net GWP of this pathway relative to AD: the inclusion of carbon sequestration could switch the net impact of composting from positive (net source) to negative (net sink).

Within a given study, including carbon sequestration should lead to a net reduction in GWP. In the aggregate, it appears that models that include carbon sequestration show net reductions in GWP relative to those without. However, this is not the case for the landfill pathway, indicating the limited dataset size (only four data points without sequestration) and the complexity of modeling landfills in particular. The scale of the effect can range from minor to major in relation to net impact (for AD and compost, on the order of tens of kg CO_2eq / metric ton wasted food). The shifts in median impact represented in Table 3-4 constitute major shifts (>30% change) in net impact (relative to values including sequestration) for the composting and sewer/WWT pathways and a moderate shift for AD. Minor effects on impact (<10%) are also possible depending on specific assumptions. However, the inclusion or exclusion of carbon sequestration is unlikely to affect the relative ranking of pathways when applied consistently across pathways.

More information on carbon sequestration can be found in Section 4.6.

TABLE 3-4. GWP IMPACTS, WITH AND WITHOUT CARBON SEQUESTRATION, PER METRIC TON WASTED FOOD

	With Sequestration			Without Sequestration		
Pathways	**Median**	**Range (25 to 75 pctile)**	**Data points**	**Median**	**Range (25 to 75 pctile)**	**Data points**
AD	-120	-160 to -41	12	-97	-260 to -30	36
Composting	26	-66 to 55	20	100	67 to 150	6
Land App.	-	n.a.	0	23	22 to 25	5
Landfill	510	190 to 700	11	460	280 to 1000	4
Sewer / WWT	190	140 to 230	2	490	470 to 490	5
Unharvest	-	n.a.	0	47	n.a.	1

AD data assumes land application of digestate or biosolids. Not all sewer/WWT scenarios assume land application of biosolids.

Energy Demand

The energy demand indicator reflects net energy use associated with a wasted food pathway, and should account for:

- Energy use by the pathway, including processing of the wasted food.
- Avoided energy use due to energy generation from wasted food by the pathway; and
- Avoided energy use through avoided production of food, feed or synthetic fertilizer.

In some respects, energy demand parallels GWP: where systems require or produce energy, both energy demand and GWP reflect this (the latter being a function of energy grid). However, the two have an important difference: fugitive losses of methane and nitrous oxide do not factor into energy demand, so pathways with higher losses of these gases may perform more favorably with respect to energy demand. Therefore, energy demand provides an additional perspective on each wasted food pathway's performance. Figure 3-4 displays the energy demand (also referred to as energy use) reported for the wasted food pathways across the reviewed studies.

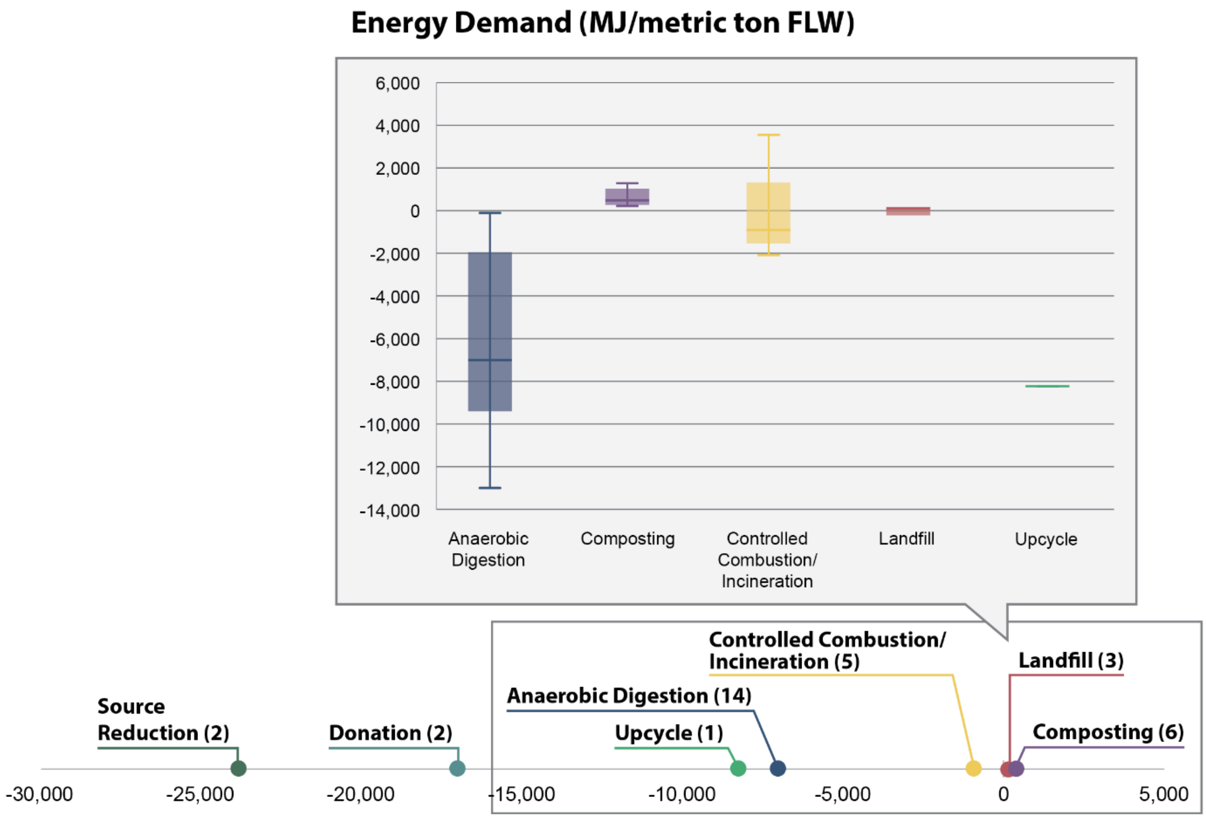

FIGURE 3-4. COMPARISON OF ENERGY DEMAND OF WASTED FOOD PATHWAYS ACROSS LCA STUDIES

AD data assumes land application of digestate or biosolids. See Appendix Table D-1 for data and sources.

Data on energy demand were limited. Overall, the energy demand analysis shows that source reduction, donation and upcycling have the greatest potential for beneficial performance due to energy use savings from avoided food production. No data for animal feed was available from reviewed studies that met scope criteria, but this pathway similarly saves energy through avoided feed production.

Energy Consumption in Composting

The composting pathway can have a wide range of energy consumption depending on the composting method as well as size and complexity of operation. Energy consumption can vary between about 200-600 MJ/metric ton FW for windrow systems compared to 150-550 for aerated static piles.[a] Energy intensities vary with the amount of contaminants in the wasted food feedstock as well. See Section 2.13 for more information on how wasted food contaminants affect the composting pathway.

[a] (Komilis and Ham 2004; Pergola et al. 2018; Morelli et al. 2019)

Beyond these pathways, the data show AD has the greatest potential for beneficial performance due to its energy generation and potential to displace the manufacture and use of synthetic fertilizer. Controlled combustion also demonstrates negative (beneficial) median for energy demand, although data range from negative to positive. Composting and landfilling are the poorest-performing pathways for which data were available, in terms of medians, but controlled combustion's range does extend above that of either pathway, possibly due to high moisture and poor energy value qualities of food as feedstock for controlled combustion. Composting (and other pathways, such as animal feed, for which data were not available) do not typically attempt to capture energy; although landfills do recover some energy, their conversion and capture rates are not as efficient as AD or controlled combustion at harvesting energy from wasted food.

The range of values shown for AD, controlled combustion, and landfill in Figure 3-4 is driven by some of the modeling variations discussed in the previous section for GWP. While all energy demand estimates are negative for the AD pathway, the spread of energy demand results is attributable to variability in assumptions related to energy recovery and energy demand of the AD unit and associated processes (e.g., wasted food pre-processing and biogas upgrading). Studies assuming both avoided electricity and heat production have the lowest net energy demand. These data underscore the importance of operation and management of wasted food pathways that recover resources; these can lead to large variations in environmental performance.

Composting has net positive energy demand, as the energy needed to aerate compost is not offset by avoided energy for fertilizer production (and carbon sequestration benefits are not relevant for energy demand). Several combustion studies show net negative energy demand, but the positive data point for combustion is due to a large estimate of electricity consumption for controlled combustion and a relatively small estimate of energy recovery, which also resulted in high GWP values (Ahamed et al. 2016).

Key Energy Recovery Distinctions between Controlled Combustion and Anaerobic Digestion of Wasted Food

While some energy recovery cases and characteristics of AD, controlled combustion, and landfilling of wasted food are similar, the processes are quite different. Below are some additional features of these technologies:

Incineration generally produces more gaseous pollutants, when compared to the other pathways, and can even volatilize metals (Psomopoulos 2009). Incineration of wasted food streams is flexible though, with the process being able to accept physical contaminants, particularly combustible items such as paper plates and plastic packaging. Both heat and electricity can be generated, though electricity is more commonly produced in waste-to-energy facilities. Wasted food incineration can also be combined with OFMSW treatment, which generally improves energy recovery as wasted food has higher-than-average moisture content for OFMSW (Pham et al. 2015). Energy production from incineration is mostly affected by LHV of the feed (typically 2.5-8 MJ/kg) and recovery efficiency (typically 15-23% heat produced is recovered as electricity).

Anaerobic digestion is a wet biological process. Unlike incineration, its performance is not negatively affected by the high moisture content of foods but can be affected by contaminants found in wasted food (both physical and chemical). While heat is not a main product of AD systems, it can produce biogas to be used for heat or electricity generation, in a similar way to landfills that have gas capture systems. The loading rates for AD can vary, but 1-10 kg solids/m^3 is typical for wet AD (Pham et al. 2015). Dry AD can have higher loading rates, up to a maximum of 30% solids, but has higher operational complexity (Wang et al. 2023). Energy production in AD is affected by the solid loading rate, biogas yield, and methane concentration in the biogas. If a combined heat and power (CHP) system is being used onsite, then the overall efficiency of that system will influence energy recovery as well. Reviewed studies examined CHP systems with a range of overall efficiencies from 70-90%. Landfills that capture biogas can use the same CHP technology as AD.

Acidification

A variety of processes can contribute to acidification in the wasted food pathways, though typical drivers are emissions of NO_x and SO_x from combustion or the emission of NH_3 from biological processes. While avoided production of feed and fertilizer can lead to acidification credits, the application of compost or digestate to land may offset those credits if the study includes air emissions from the applied material. However, emissions from wasted food-derived amendments tend to be lower than those from synthetic fertilizers if appropriate agronomic practices are followed. Section 4.6 provides further discussion. Figure 3-5 shows terrestrial acidification impact.

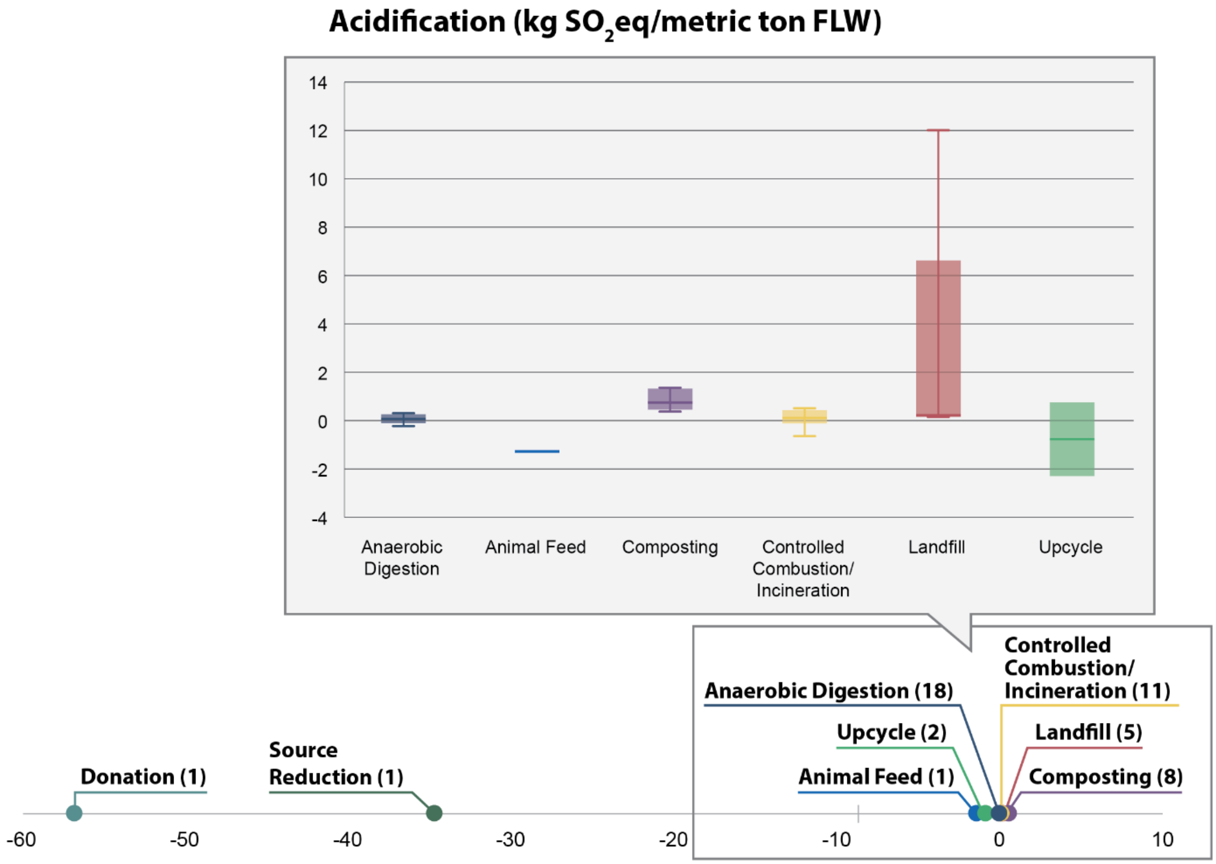

FIGURE 3-5. COMPARISON OF TERRESTRIAL ACIDIFICATION POTENTIAL OF WASTED FOOD PATHWAYS ACROSS LCA STUDIES

AD data assumes land application of digestate or biosolids. See Appendix Table D-1 for data and sources.

For acidification, source reduction and donation out-perform the other pathways by an order of magnitude due to avoided energy use during food production. Among the pathways in the inset boxplot animal feed and upcycling have the most favorable performance. Landfill, compost, AD and combustion perform worse—but the latter two have the potential for beneficial performance. Pathways with energy recovery (especially AD and controlled combustion) can result in offsets of other energy production, resulting in acidification credits through avoided criteria air pollutant emissions. There are three broad groups in the collected data:

- Negative (beneficial) medians:
  - Animal feed modeling that takes credit for avoiding the production of energy-intensive (and thus NO_x- and SO_x-intensive) production of feed crops.
  - Upcycling, which receives credit for avoided production of food crops.
- A mix of positive and negative values, with median value close to zero:

- AD and controlled combustion recover energy; the net impact of each pathway depends on the avoided grid mix and how it compares to direct air emissions associated with biogas and wasted food combustion, respectively.

- Positive values, with median above zero:
  - Although composting can lead to avoided fertilizer production, ammonia may off-gas during operation.
  - Landfill values reflect the balance between avoided energy and electricity and diesel emissions required for processing wasted food.

As with the other impact categories, inspection of data points indicates that modeling assumptions tend to drive high and low values. For example, avoided production of urea and soy meal in animal feed accounts for estimated impact of -1.28 kg SO_2eq (Mondello et al. 2017). Likewise, the lowest AD data point (-0.6 kg SO_2eq) is based on favorable assumptions about avoided heat, electricity, and fertilizer offsets (Mondello et al. 2017). At the other end of the spectrum, the high composting values, 1.2 kg SO_2eq (Morelli et al. 2019) and 1.4 kg SO_2eq (Oldfield et al. 2016), are based on assumption of high emissions of ammonia during the composting process.

Eutrophication

In the wasted food management context, eutrophication is generally due to direct release of nutrients to water bodies and indirect release via energy use. Eutrophication impacts are measured in nitrogen equivalents, Neq. The main contributing factors (both positive and negative) are:

- Energy use and avoided energy use;
- WRRF effluent discharge;
- Untreated landfill leachate;
- Avoided production and application of synthetic fertilizer; and
- Land application of wasted food soil amendments (e.g., wasted food, compost, digestate, biosolids).

Combustion-based energy production may release NO_x to air, which can lead to eutrophication if the NO_x is directly deposited or make its way to an aquatic system. Production of fertilizer is energy intensive and thus also matters for this impact category. When wasted food or effluent from processing (e.g., of animal feed) is sent to sewers, there is an increased nutrient load to WRRFs. Although some WRRFs remove significant fractions of nutrients, at least in the United States, some of the additional nutrient burden is released to waterways, which contributes to eutrophication.

Finally, depending on management practices and related modeling assumptions, soil amendments may provide more steady release of nutrients than synthetic fertilizer. This latter source of benefit/impact is not well or consistently captured in environmental impact modeling. Indeed, a significant portion of the eutrophication impacts of those pathways (AD, composting, and land application) in which wasted food is applied to land is driven by modeling of the fate of wasted food on soil. The off-site movement of nutrients (potentially causing eutrophication), the change in soil organic carbon, the change in soil water retention, and other effects will depend on local soil types, climatic conditions, and method and rate of application. For example, municipal biosolids have requirements for setbacks from waterways to reduce eutrophication impacts. At the level of this analysis, and as discussed in Section 4.6, there is certainly potential for benefits from land application of wasted food and wasted food-derived soil amendments; however, modeling of these processes is inconsistent in the literature.

Figure 3-6 shows eutrophication impacts for the wasted food pathways.

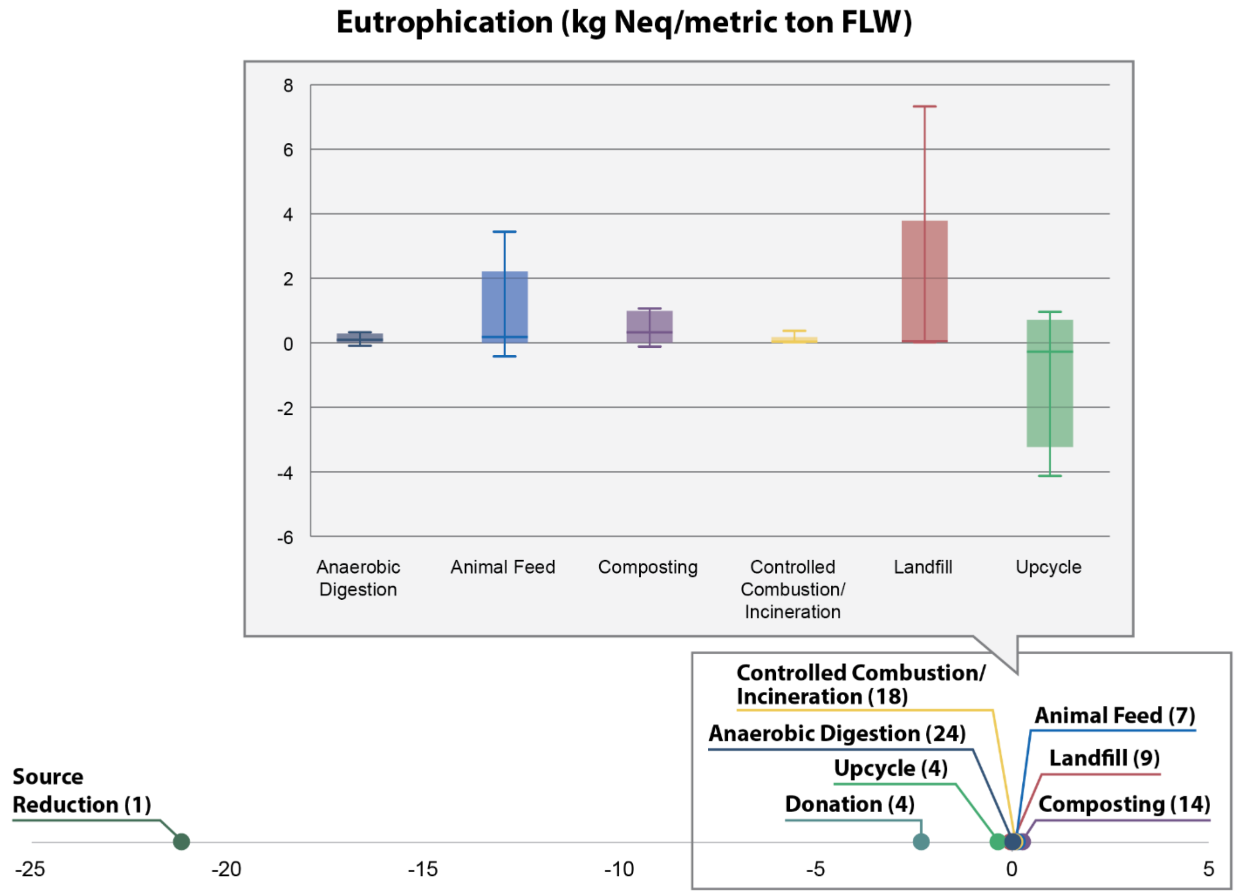

FIGURE 3-6. COMPARISON OF EUTROPHICATION IMPACT OF WASTED FOOD PATHWAYS ACROSS LCA STUDIES

AD data assumes land application of digestate or biosolids. Donation excluded from boxplot because the range extends an order of magnitude below the next lowest value in the boxplot. See Appendix Table D-1 for data and sources.

With inconsistent modeling of the fate of wasted food on soil, and various assumptions about landfill leachate, the collected LCA eutrophication data suggests there is relatively little differentiation among pathways, except for source reduction and donation.

While the AD median is close to zero, the highest value for AD eutrophication (3.06 kg Neq) is from Morelli et al. (2019), which represents co-digestion of wasted food at a WRRF without nutrient removal. Although wasted food is introduced directly into the digesters, a share of wasted food nutrients is released with treatment plant effluent via dewatering return flows. The study also includes estimates of nutrient emissions from land following digestate land application. The next highest AD study value (0.7 kg Neq) is caused by ammonia emissions after agricultural land application of digestate (Slorach et al. 2019a). However, Slorach et al. (2019a) noted the large uncertainty in this process; in their uncertainty analysis, the 10th percentile emission is -0.03 kg Neq and the 90th is 0.69 kg Neq. Thus, the authors report an average that is outside of the 90th percentile of their uncertainty analysis.

The high (7.29 and 7.31 kg Neq) and low (0.016 and 0.02) landfill values are from two related studies that represent marine and freshwater eutrophication, respectively. Landfill eutrophication impacts are in both cases driven by assumptions related to leachate releases (Slorach et al. 2019a; Slorach et al. 2019b).

Water Consumption

Water consumption is not well-documented in most of the reviewed studies. In general, water consumption by wasted food pathways is limited compared to that needed for production of food and feed crops. Thus, water consumption would be driven by avoiding the production of food or feed in pathways that do so.

While a significant amount of water is withdrawn for energy production, a large fraction of this water is returned to the basin from which it comes, and so should not be counted as water consumption (Pfister et al. 2017; Boulay et al. 2018). However, some of the studies that include avoided electricity production appear to have conflated consumption and withdrawal, as noted below. Process cooling water may be thermally polluted, with different impacts from those associated with consumption (Raptis et al. 2017). Thus, water consumption is expected to be low for wasted food pathways; even conveyance to WWT via sewer would have relatively low water consumption allocated to wasted food.

Figure 3-7 shows water consumption for selected wasted food pathways.

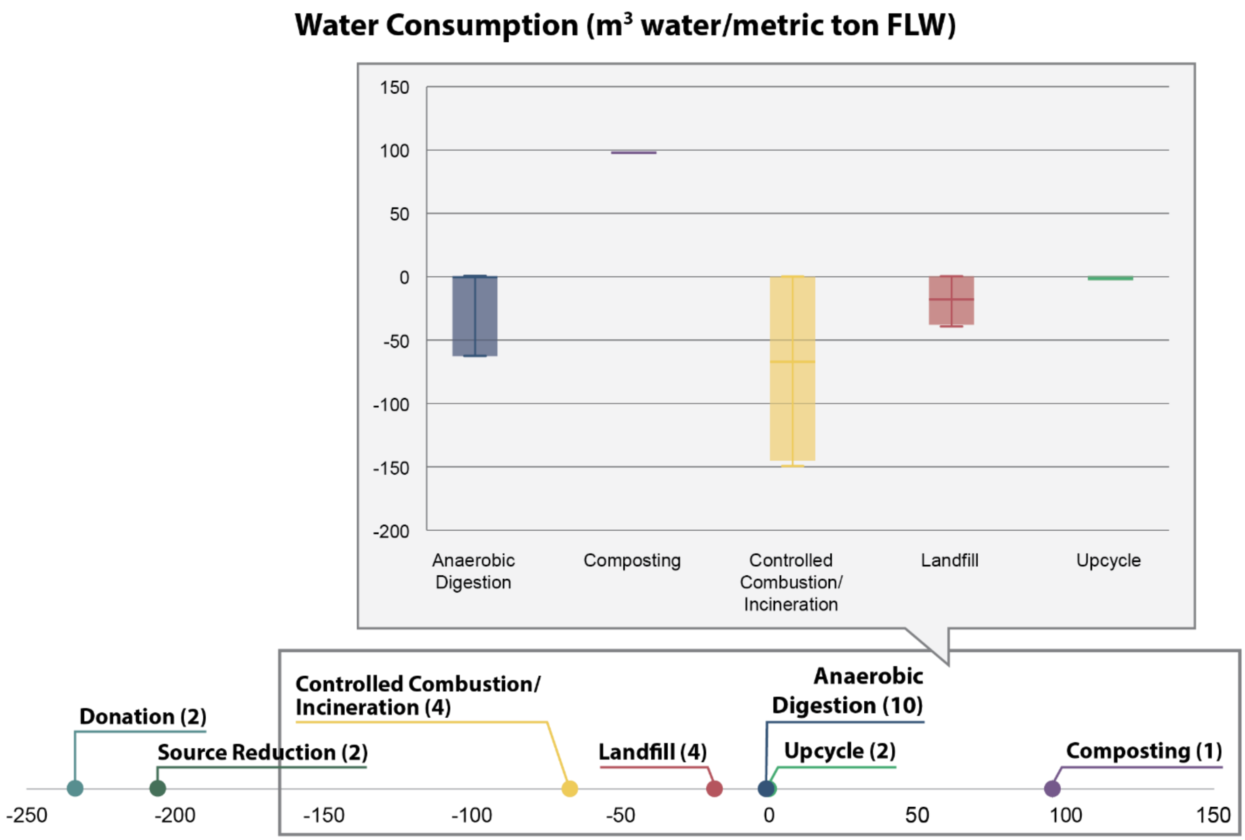

FIGURE 3-7. COMPARISON OF WATER CONSUMPTION OF WASTED FOOD PATHWAYS ACROSS LCA STUDIES

AD data assumes land application of digestate or biosolids. See Appendix Table D-1 for data and sources.

Values for water consumption are limited. Given the lack of standardization in accounting for water (discussed above), the water consumption analysis shows the need for further study; there is no clear distinction among pathways. EPA's From Farm to Kitchen: The Environmental Impacts of U.S. Food Waste report (U.S. EPA 2021c) highlights the importance of water use within agricultural systems. The benefit of reduced water consumption from avoided feed production within the animal feed pathway is poorly captured within the reviewed LCA literature. Relative to the water that could have been used to irrigate food, the water needed for wasted food processing is relatively restricted. As expected, source reduction, donation and upcycling demonstrate water savings due to avoided food production; however, upcycling can also require water for food processing, diminishing savings.

The lowest water consumption values for AD (-250 and -275 m^3, both outside 1.5 x the inter-quartile range) are primarily driven by water consumption associated with avoided electricity production (Slorach et al. 2019a). The authors of that study note that the AD process itself consumes minimal water. All other data points have an absolute value ≤ 1 m^3 water consumed. The positive estimates of AD water consumption (~0.05 to 0.3 m^3) are related to digestate dilution, which is estimated to be about five times as large as is needed by the AD process itself (Tian et al. 2021). The positive value for composting is due to electricity use and is based on a single study (Slorach et al. 2019a), and the negative values for controlled combustion and landfill are also associated with benefits due to avoided electricity. These differences highlight the general inconsistencies encountered in water consumption inventories.

Data for sewer/WWT was not available from reviewed LCAs, but WRRF effluent is sometimes used to irrigate crops, thus saving water consumption from irrigation.

Land Occupation

In general, relative to food production, the management of wasted food has limited land requirements. As noted below, there are likely to be some minor differences between the wasted food pathways, though there are insufficient data to discern meaningful differences.

Figure 3-8 shows the land occupation results for selected. wasted food pathways.

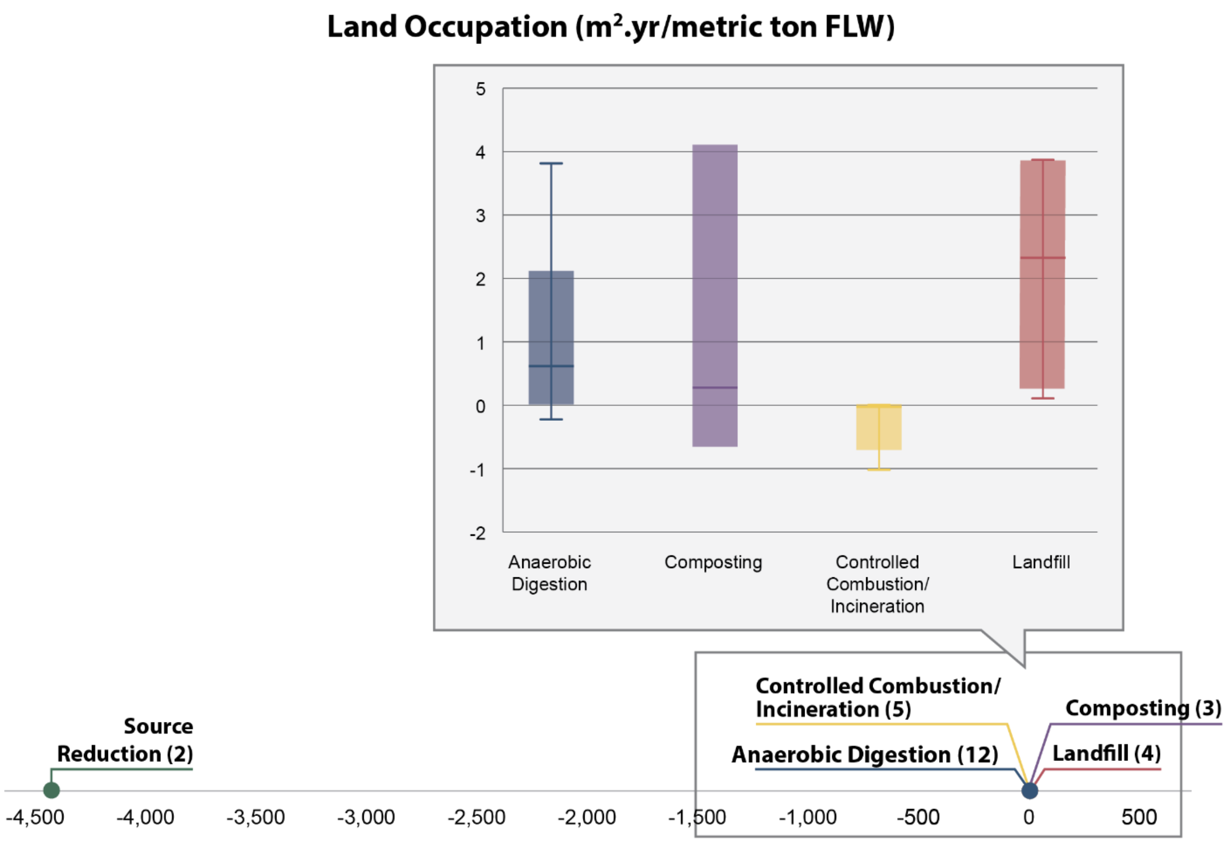

FIGURE 3-8. COMPARISON OF LAND OCCUPATION OF WASTED FOOD PATHWAYS ACROSS LCA STUDIES

AD data assumes land application of digestate or biosolids. See Appendix D-1 for data and sources.

Overall, the land occupation data are too sparse to reveal any trends in impact. The variation among land occupation values reflects the importance of scope considerations, as discussed previously: e.g., inclusion or exclusion of infrastructure; destination of ash produced from combustion (sent to landfill or used to displace construction aggregate). Nonetheless, some of the values for land occupation suggest trends, even though the range of values is restricted. The larger positive values are associated with AD, composting, and landfill. Most composting operations modeled in the literature are windrow or static piles; like landfilling, this can require appreciable land. Large-scale, in-vessel composting systems would require less land. In contrast, controlled combustion takes place in engineered systems with relatively low retention times and thus relatively small land area requirements. It is unclear why AD shows values as high as composting and landfill, speaking to the variability in modeling assumptions across studies.

Land occupation for wasted food management is not typically a limiting factor or an impact of concern in the United States. These land occupation estimates are two orders of magnitude smaller than land requirements of food production. However, while land itself is generally not a limiting factor, we note that siting of wasted food management systems can be challenging, due to real and perceived concerns about local impacts on communities.

Summary of Inter-Study Findings

The numeric values for the medians and ranges displayed in each preceding boxplot figure are provided in Table 3-5 and Appendix D. The relative median life cycle impacts by pathway from the analysis across LCA studies is summarized in Table 3-6.

This section provided high-level comparison of many pathways across LCA studies for six environmental indicators. For many environmental indicators (besides GWP) data were available for only half the pathways. Key findings include:

- Source reduction, donation and upcycling tend to yield environmental benefits rather than impacts across all impact categories, while other pathways indicate a mix of environmental benefits and damages depending on the impact category.
- The environmental benefits of source reduction and donation exceed those of any other wasted food pathway by at least an order of magnitude – except for GWP, where only source reduction exceeds other pathways by an order of magnitude. Some results show donation as having a lower impact than source reduction. This reflects different modeling assumptions across studies and does not indicate that donation is preferable to source reduction.

Among the remaining wasted food pathways, the following observations can be made:

- Upcycling also demonstrates very good environmental performance, with negative medians across all impacts.
- Data on unharvested pathway is very limited. While the pathway avoids the impacts associated with processing, packaging and distribution, the sole data point available showed a small positive GWP value.
- Land application shows positive results across impact categories, but the lack of available studies for land application means this finding is interim and should be revisited as more data are available.

By impact, findings indicate:

- GWP is the most consistently evaluated impact category. For GWP, source reduction, donation and upcycling demonstrate strongest performance, followed by the animal feed and AD pathways which have consistently negative (beneficial) values. The median values for composting, land application, unharvested/plowed in and controlled combustion are all close to zero (≤ 53 kg CO_2eq/metric ton wasted food), suggesting that these pathways' recovery of resources or energy from wasted food generally offsets GWP impacts of the pathway itself. The boxplot in Figure 3-3 demonstrates overlap in impact results for the AD, animal feed, compost, controlled combustion and land application pathways suggesting that modeling assumptions and management practice have a substantive effect on the GWP impact associated with each pathway. Conversely, landfill and sewer/WWT GWP impacts are almost entirely above zero, with median values that suggest they are the poorest-performing wasted food pathways in this impact category.
- Not all studies reviewed quantified carbon sequestration within the GWP indicator; where it was included, it had a moderate effect on overall GWP estimates. This effect could be most consequential for the composting pathway, as the inclusion of carbon sequestration could switch the net impact of composting from positive (net source) to negative (net sink) – and for sewer/WWT, where studies including carbon sequestration from land application of biosolids showed 50% lower GWPs than those without – further examination of all factors is needed to see what may be driving this difference in results.
- For energy demand, beyond source reduction, donation and upcycling, AD has only negative values and a significantly negative median value, reflecting that pathway's recovery of energy. Controlled combustion also shows a consistent, though smaller, recovery of energy. Other pathways have positive impacts, indicating a net demand for energy.
- For acidification and eutrophication, most pathways have a net positive impact, although a single data point suggests animal feed may have a negative acidification impact. For the other pathways, modeling assumptions about a) the NO_x and SO_x intensity of avoided energy sources,

b) the quantity of fugitive ammonia emissions, and c) nutrient losses from land-applied material drive variability across pathways.

- Water consumption and land occupation are infrequently reported and inconsistently modeled, with no clear trends in the collected data.
- For all impact categories, the impacts related to wasted food management are small relative to the impacts associated with food production. Thus, even wasted food pathways with beneficial performance do little to offset the original impact of producing food.

TABLE 3-5. MEDIAN, RANGE (25TH-75TH PERCENTILE), AND SCENARIO COUNT FOR SIX ENVIRONMENTAL IMPACTS PER METRIC TON OF WASTED FOOD, BY PATHWAY

Impact Category	Statistic	Source reduction	Donation	Upcycling	AD	Animal feed	Composting	Controlled Combustion	Land Application	Landfill	Sewer / WWT	Unharvested / Plowed In
Acidification	Median	-35	-57	-0.78	0.055	-1.3	0.76	0.087	-	0.24	-	-
	Range (25 to 75 pctile)	-	-	-	-0.092 to 0.23	-	0.48 to 1.3	-0.086 to 0.39	-	0.17 to 6.6	-	-
	Count	1	1	2	18	1	8	11	-	5	-	-
Energy Demand	Median	-24000	-17000	-8200	-7000	-	470	-940	-	120	-	-
	Range (25 to 75 pctile)	-	-	-	-9400 to -2000	-	270 to 1000	-1500 to 1300	-	-200 to 140	-	-
	Count	2	2	1	14	-	6	5	-	3	-	-
Eutrophication	Median	-21	-2.3	-0.3	0.087	0.15	0.32	0.029	-	0.022	-	-
	Range (25 to 75 pctile)	-	-28 to -1	-3.2 to 0.68	n.d. to 0.27	-0.038 to 2.2	0.00031 to 0.97	0.00075 to 0.16	-	0.013 to 3.8	-	-
	Count	1	4	4	24	7	14	18	-	9	-	-
GWP	Median	-3300	-570	-450	-110	-210	53	14	23	510	470	47
	Range (25 to 75 pctile)	-3900 to -1700	-2000 to -360	-600 to 68	-230 to -30	-330 to -100	-54 to 79	-27 to 600	17 to 27	190 to 720	210 to 490	-
	Count	4	5	3	48	11	26	20	5	15	7	1
Land Occupation	Median	-4400	-	-	0.6	-	0.27	-0.04	-	2.3	-	-
	Range (25 to 75 pctile)	-	-	-	-0.005 to 2.1	-	-0.67 to 4.1	-0.71 to 0.0025	-	0.27 to 3.8	-	-

Impact Category	Statistic	Source reduction	Donation	Upcycling	AD	Animal feed	Composting	Controlled Combustion	Land Application	Landfill	Sewer / WWT	Unharvested / Plowed In
	Count	2	-	-	12	-	3	5	-	4	-	-
Water Consumption	Median	-210	-230	-1.6	-0.4	-	97	-67	-	-18	-	-
	Range (25 to 75 pctile)	-	-	-	-63 to 0.12	-	-	-150 to -0.36	-	-38 to -0.5	-	-
	Count	2	2	2	10	-	1	4	-	4	-	-

A dash ("-") is used for ranges where the study count is less than 2, or medians or counts with study count less than 1.

Some results below show donation as having a lower impact than source reduction; this reflects different modeling assumptions across studies and does not indicate that donation is preferable to source reduction.

Compost, AD, and sewer/WWT data assume land application of compost, digestate, or biosolids.

TABLE 3-6. RELATIVE ENVIRONMENTAL IMPACTS BY WASTED FOOD PATHWAY (MEDIAN IMPACT / METRIC TON WASTED FOOD)

Impact Category	Median Impact/ metric ton FLW	Source Reduction	Donation	AD	Animal Feed	Compost	Controlled Combustion/ Incineration	Landfill	Upcycling
GWP	*kg CO_2eq*	-3,300	-570	-110	-210	53	14	510	-450
Energy Demand	*MJ*	-24,000	-17,000	-7,000	n.d.	470	-940	120	-8,200
Acidification	*kg SO_2eq*	-35	-57	5.50E-02	-1.3	0.76	0.087	0.24	-0.78
Eutrophication	*kg Neq*	-21	-2.3	0.087	0.15	0.32	0.029	0.022	-0.3
Water Consumption	*m^3 water*	-210	-230	-0.4	n.d.	97	-67	-18	-1.6
Land Occupation	*m^2.yr*	-4,400	n.d.	0.6	n.d.	0.27	-0.04	2.3	n.d.

Color scale based on trend in median literature data, not statistical differences. Green = lower impact; yellow to orange = moderate impact; red = higher impact

n.d. = no data

Only pathways which have data for multiple impacts are shown; see Appendix D for more detailed data.

AD data assumes land application of digestate or biosolids.

3.5 Quantitative Comparison of Pathways Within Studies

This section presents the results of pairwise comparisons between five of the most studied wasted food pathways – AD, animal feed, composting, controlled combustion, and landfill – within individual studies for six impact categories – GWP, acidification, eutrophication, particulate matter formation, human toxicity, and ecotoxicity. The intra-study analysis is meant to complement the analysis in the preceding section, which details environmental comparisons between management pathways across different studies. The benefit of intra-study comparisons, as presented here, is that individual studies align the scope of their comparisons across pathways, such that differences in scope (e.g., whether or not transportation is included or harmonization of energy mix) should not account for reported differences in environmental performance. In addition, within study analysis allows for the comparison of two subsets of the animal feed pathway: wet and dry animal feed. See Chapter 2 for further discussion of the differences between these two sub-pathways.

Methodology for Intra-Study Analysis

The intra-study analysis is drawn from the same pool of papers for which data are presented in Section 3.4. However, the scope criteria outlined in the preceding section were not applied as the studies are relied upon to harmonize their own scope. In all, thirty-seven studies, which presented data for multiple pathways, were included in the intra-study analysis. As noted above, twenty-two of these were also used in the inter-study analysis.

To generate pathway rankings most studies compared midpoint data for each impact. For several studies, ranked results were utilized directly. If a study reported that two pathways had statistically similar environmental performance, these pathways were documented as having the same rank. Beyond this, the magnitude of difference in environmental performance between pathways is not reflected in the intra-study results. Most studies do not report results of a statistical analysis that allow determination of significance. To learn more about the magnitude of reported impact differences between pathways, please refer to Appendix Table D-2.

General impact categories such as eutrophication or human toxicity include several individual indicators under a single broad heading. For example, eutrophication includes marine and freshwater eutrophication. Given this, individual studies (such as an analysis that presents both marine and freshwater eutrophication results) can contribute more than one instance of a unique pairwise comparison. Since particulate matter and human and ecotoxicity were not discussed in the previous section, they are discussed briefly below in relation to the wasted food pathways.

Particulate Matter Formation

Particulate matter formation associated with wasted food pathways is driven primarily by energy use and avoided energy use. In addition, composting emits NH_3, a precursor of PM, as does the application of organic soil amendments from AD, composting, and sewer/WWT. See Section 4.6 for additional details on nitrogen losses from soil amendments.

Human Toxicity and Ecotoxicity

Human toxicity and ecotoxicity associated with wasted food pathways are strongly linked to energy consumption and avoided energy production, land application of soil amendments, and in some cases process emissions (e.g., controlled combustion).

Intra-Study Results

Figure 3-9 presents a matrix of figures that each compare the environmental performance of two wasted food pathways, summarizing comparisons from individual studies. The figure depicts the percentage of results in which the underlying studies, documented in Appendix Table D-2, show that either pathway A or pathway B has better or similar environmental performance across six impact categories. The count of study scenarios used to calculate these percentages are documented as bar segment labels. Within the figures, one color is associated with each pathway and the greater area of a pathway's color in the bar chart indicates that the associated pathway has better performance than the listed alternative. White bar segments indicate that a percentage of reviewed studies show that the two listed pathways had statistically similar environmental performance.

Readers interested in the degree to which one pathway is better or worse are referred to the appendices for specific data, and to the inter-study results for general trends. These trends can give a sense of the typical (but often variable) differences between pathways. For example, Figure 3-3 shows that the range of typical GWP values for AD has limited overlap with those for combustion.

Among the six pathways reported in this section, wet and dry animal feed are the two wasted food pathways that demonstrate the best environmental performance relative to all alternative pathways across most impact categories. When comparing wet and dry animal feed directly, wet feed is found to have better environmental performance in all impact categories, due primarily to lower processing energy demand. Animal feed's good environmental performance is linked to avoided agricultural production of grains or legumes used in the avoided feed products. Albizzati et al. (2021b) is the only study to directly compare wet animal feed to landfill and controlled combustion, finding that wet animal feed has higher acidification impacts due largely to land application of digestate associated with rejected meat and fish waste. Dry feed shows some mixed performance compared to other pathways in the toxicity impact categories. Wasted food collection and transport, pre-treatment and processing all contribute to wasted food toxicity impacts (Mondello et al. 2017; Albizzati et al. 2021a), presumably associated with energy consumption. Avoided feed production provides a benefit associated with toxicity impacts.

Anaerobic digestion and controlled combustion are both well studied wasted food pathways within the LCA literature (see Appendix Table D-2). Wet and dry animal feed typically have lower environmental impact than these two pathways, despite the exceptions noted in the preceding paragraph which are based on limited data. The AD and controlled combustion pathways have consistently better environmental performance than the landfill and compost pathways in all impact categories, due primarily to the environmental credits associated with energy recovery (i.e., avoided energy use). The compost pathway does not recover energy and the landfill pathway is less efficient recovering energy from wasted food and produces a larger quantity of fugitive methane emissions. The AD pathway has better or similar environmental performance compared to the controlled combustion pathway in GWP and the two toxicity impact categories. Controlled combustion is often shown to have lower particulate matter and acidification potential impacts than AD. Impacts in both of these categories are driven by ammonia emissions (Slorach et al. 2020) released during digestate treatment and land application.

The compost wasted food pathway does not perform well in the intra-study comparisons relative to the animal feed, AD and controlled combustion pathways. Minor exceptions to this statement include instances of similar environmental performance when comparing the eutrophication impacts of compost and AD, the human toxicity impacts of compost and dry feed, and the GWP impact of compost and controlled combustion. Composting generally performs well compared to the landfill pathway in the toxicity, eutrophication and GWP impact categories. Composting has relatively low impacts due to its low process energy requirements but does not recover any of the energy content present in wasted food, leading to lower environmental benefits when compared to many alternative pathways.

The landfill pathway is consistently shown to have the highest GWP impact when compared to alternative pathways, due largely to the release of fugitive methane emissions. The landfill pathway consistently has higher environmental impacts than dry feed, AD and controlled combustion across all impact categories. Landfill's poor comparative performance relative to these pathways is primarily due to the environmental credits that these pathways receive for avoided feed and energy production. The landfill pathway has higher impacts than the wet feed pathway in particulate matter formation, ecotoxicity, and GWP and similar impacts in human toxicity and eutrophication potential. (Albizzati et al. 2021b) is the only study to look at comparative impacts of the wet feed and landfill pathways in the acidification potential impact category. In this study, wet feed is found to have higher impacts than landfilling due to land application emissions associated with AD of meat and fish waste that is not suitable for the wet feed pathway (i.e., reject stream). Composting generally performs well compared to the landfill pathway in the toxicity, eutrophication and GWP impact categories, but has higher particulate matter and acidification potential due largely to ammonia emissions (Slorach et al. 2020).

Summary of Intra-Study Findings

This section provided results of pairwise comparisons for five of the most common wasted food pathways within LCA studies for six environmental indicators. Key findings include:

- Among the five pathways evaluated, there is no alternative wasted food pathway that consistently has higher climate impacts than does landfill disposal.
- Most studies find that using wasted food as animal feed leads to better environmental outcomes compared to alternative pathways. Wet feed is shown to consistently have better performance than dry feed, due to lower processing requirements, but conclusions are preliminary as only a small number of studies have considered these wasted food pathways.
- AD demonstrates good comparative environmental performance in the GWP and ecotoxicity impact categories. AD's strong environmental performance is usually associated with environmental credits from avoided energy production, which depend on the displaced energy mix. AD's weakest impact category is particulate matter formation, which is linked to biogas combustion and ammonia emissions.
- Composting demonstrates poor comparative environmental performance in acidification and particulate matter formation relative to all alternative pathways. These impacts are strongly associated with ammonia emissions during composting and land application. Composting's best environmental performance is in the GWP category, due to its low energy demand and environmental credits associated with avoided fertilizer production and carbon sequestration. Composts best environmental performance is relative to the landfill pathway.
- Controlled combustion exhibits strong environmental performance relative to the landfill and compost pathways across all six impact categories. Positive environmental performance associated with controlled combustion is strongly linked to avoided energy credits, which depend on the displaced energy mix. The controlled combustion pathway demonstrates mixed environmental performance when compared to AD and animal feed.
- Landfilling demonstrates poor comparative environmental performance in the GWP, eutrophication, ecotoxicity, and human toxicity categories. Landfilling's best environmental performance is in the particulate matter formation and acidification categories where it shows mixed results compared to alternative pathways. Instances of good environmental performance for the landfill pathway are typically linked to avoided energy benefits.

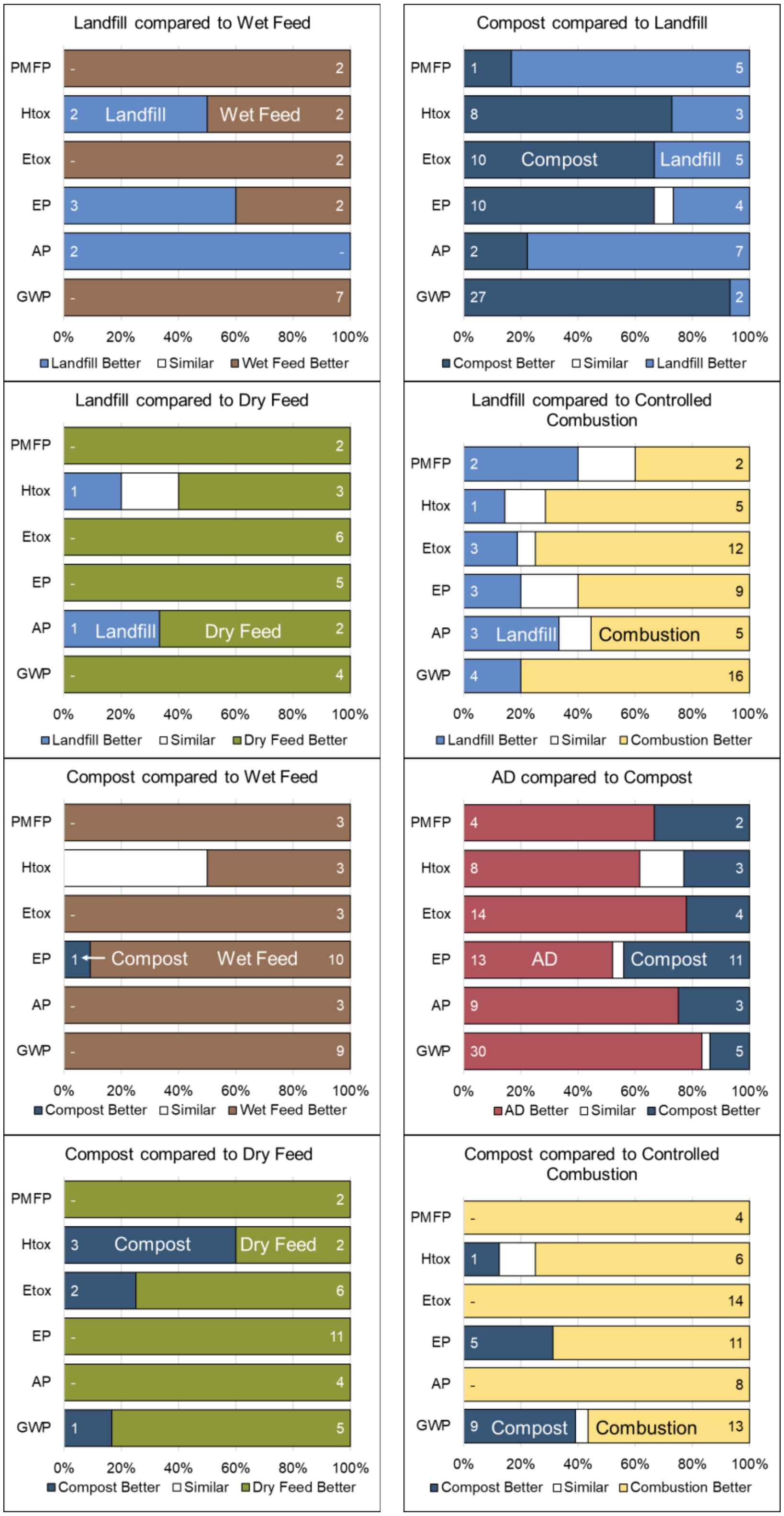

FIGURE 3-9. COMPARISON OF ENVIRONMENTAL IMPACT OF WASTED FOOD PATHWAYS WITHIN STUDIES

AD data assumes land application of digestate or biosolids. See Appendix Table D-2 for data and sources.

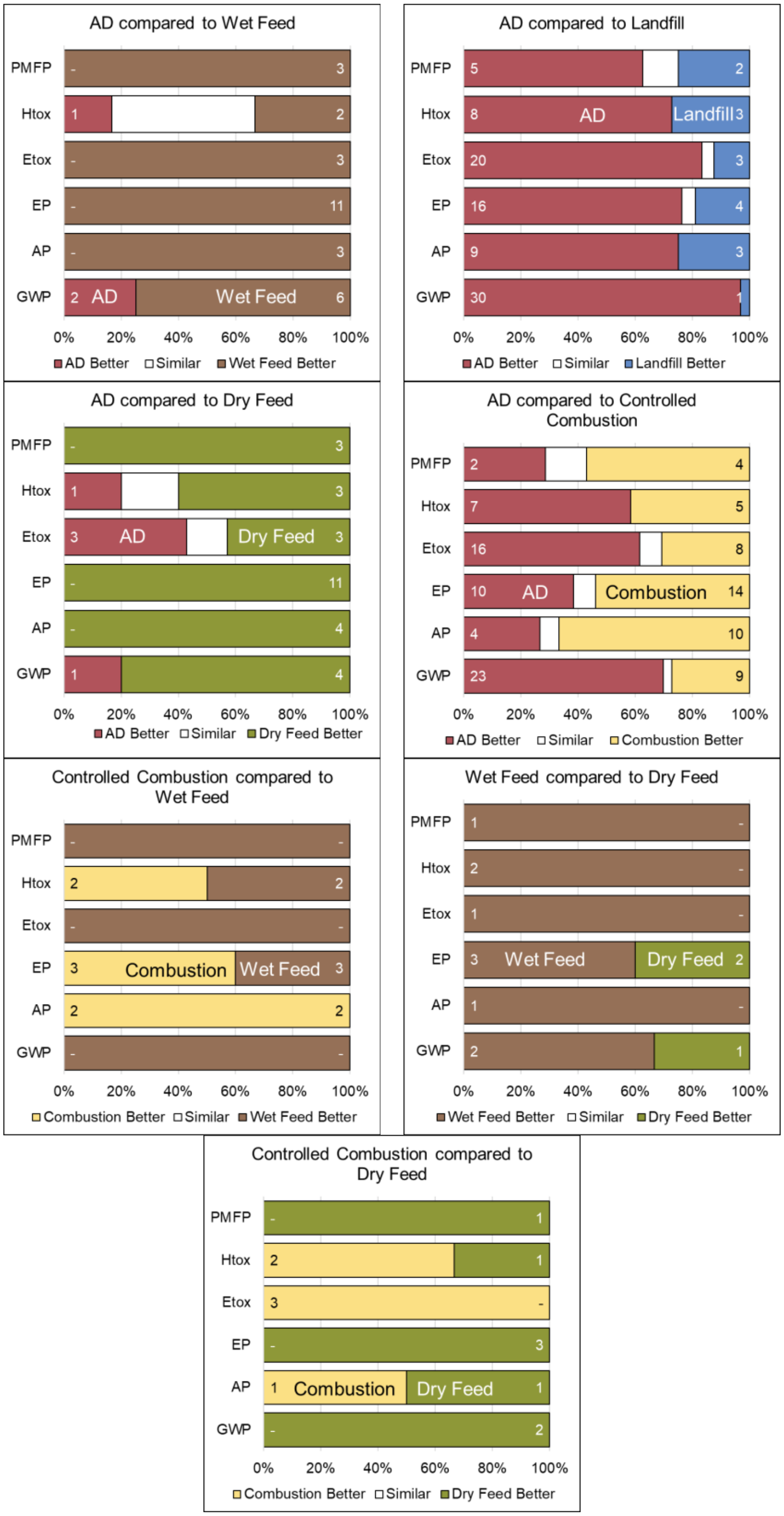

FIGURE 3-9 COMPARISON OF ENVIRONMENTAL IMPACT OF WASTED FOOD PATHWAYS WITHIN STUDIES (CONTINUED)

AD data assumes land application of digestate or biosolids. See Appendix Table D-2 for data and sources.

3.6 Soil Health

Soil health is the only environmental indicator included in our literature search where data available in LCAs did not allow for quantitative inter- or intra-study analyses. Soil health is important on global and local scales for ecosystem services, water conservation, biodiversity protection, and regulation of carbon and nutrient cycles. But soil health is complex, difficult to measure, and dependent on factors such as climate, soil texture and management.

There is not yet consensus about a definitive set of indicators to measure soil health (Rinot et al. 2019) in LCA, but development of soil health indicators is ongoing (Bessou et al. 2020). Therefore, no quantitative metrics for soil health were available for this report, and improved soil health is not included in the quantitative LCA analysis – a limitation of this evaluation. Instead, the LCA review in this chapter qualitatively considers the effect that wasted food pathways have on soil health. The five pathways that apply outputs to the soil (or have the potential to do so) – unharvested/plowed in, land application, AD, composting, and sewer/WWT – can benefit soil health in many ways, including providing organic matter and organic fertilizer, enhancing water storage, and promoting water quality and soil biodiversity. These pathways (as well as landfills) can also result in carbon sequestration in the soil.

Organic Amendments and N_2O

Using wasted food as a substitute for fertilizer can lower impacts in LCA studies by accounting for displaced fertilizer production. However applied wasted food still has some impacts, notably the release of N_2O. Additionally, the application of organic waste, whether untreated wasted food or processed digestate, compost, or biosolids can cause a turnover in native soil organic matter, releasing gases like N_2O (Thangarajan et al., 2013). N_2O volatilization is a concern due to its high GWP at 265 CO_2eq (IPCC 2023). However, N_2O emissions from synthetic fertilizers are typically much greater than from wasted food-derived organic amendments. Inorganic N content, C/N ratio, soil texture, soil drainage, SOM, and precipitation all affect N_2O emissions (Parnaudeau et al. 2009; Charles et al., 2017).

Within reviewed LCAs, environmental credits were possible (but not given consistently) for carbon sequestration and avoided fertilizer production and use. Carbon sequestration is discussed within the GWP impact category of the LCA review. A detailed analysis of sequestration credits can be found in Table 3-4. The numerous other benefits of adding carbon to soils are not captured by carbon sequestration credits. Some of these benefits are detailed below and in Section 4.6.

LCA can provide credit for avoided fertilizer production and use to these pathways, though application of credits is inconsistent. For example, LCA studies may or may not include credits for the pathways above and may apply different credit values for different pathways. Few of the LCAs reviewed on composting and AD included credits for avoided use of peat as a growing media (Levis and Barlaz 2011; Morris et al. 2017), for which compost and appropriately processed digestate can be effective substitutes. Avoided fertilizer production and use does not account for additional soil health benefits of organic amendment use, which are discussed more in Section 4.6.

Nitrogen and phosphorus present in wasted food are important both as resources that can potentially avoid the production and use of synthetic fertilizers and for their connection to several environmental impact categories. Efficient recovery of these essential nutrients can decrease further nutrient mobilization (e.g., reduce chemical nitrogen fixation or phosphorus mining) while also decreasing contributions to eutrophication and acidification. The plant-availability of nutrients (i.e. inorganic fraction) and rate of nutrient leakage differ between synthetic fertilizers and organic amendments (hence LCAs using substitution ratios based on the amount of plant available N, for example, not total N in an organic soil amendment).

For organic amendments, treatment process, type of soil, crop grown, and climatic conditions can all impact nitrogen mineralization rates (the rate at which organic N becomes available for crop uptake) (Brown et al. 2020). However, studies have found that crop response (i.e. crop yield, N uptake, and soil nitrate N) to organic amendments was similar or improved compared to equal application (in kg N/ha) of synthetic fertilizer at rates equivalent to standard agronomic recommendations (Koenig et al. 2011; Cogger et al. 2013). Additionally, slower nutrient mineralization rates positively impact crop response across multiple growing seasons, unlike synthetic fertilizers (Cogger et al. 2013). Application of slower-release fertilizers, such as organic amendments, can increase nitrogen uptake efficiency in soil with previous N-surplus from synthetic fertilizer use. Increased nitrogen uptake efficiency can result in reduction of N-surplus and lower soil-based N_2O emissions (Morais et al. 2021).

The type, composition, and concentration of nutrients and other organic matter in soil amendments influences the amendment's ability to meet local nitrogen and phosphorus demands (Moinard et al. 2021). For instance, compost may not be sufficient to meet a crop's N requirements, or biosolids may contain too much P to be applied in an area with other P inputs, such as animal manure. When nutrient flows are considered within a greater agro-food-waste system, the positive impacts of organic amendments outlined above will depend on the existing nutrient levels in the local pool (van der Wiel et al. 2021). For example, areas with a high density of animal agriculture tend to have high levels of N and P from manures; without exporting some of those nutrients to maintain a balance, the area will have a nutrient surplus and higher N and P emissions. The fertilizing benefits of wasted food-derived soil amendments will only be realized if the amendments are land-applied at appropriate rates and in an area that lacks adequate nutrients for current crop growth and land use.

Soil Organic Matter

Soil organic matter (SOM) is perhaps the most important aspect of soil health (Urra et al. 2019) and a key factor in soil carbon sequestration. SOM content (also known as soil organic carbon content) is a widely used and accepted metric for soil health (Kopittke et al. 2022; Liptzin et al. 2022), as it is indicative of fertility, bulk density, water holding capacity, and cation exchange capacity (which helps to cycle nutrients).

Compost, digestate, biosolids, raw wasted food, and unharvested/plowed-in crops are all sources of organic matter. Increasing soil organic matter promotes nutrient cycling and retention, makes nutrients available for plant uptake and decreases nutrient runoff, and helps to maintain balance in the nitrogen and phosphorus cycles. Carbon can be sequestered in soils when the rate of organic carbon added is greater than the rate of carbon lost through mineralization. Any increase in SOM results in significant improvements to soil quality and function (Poulton et al. 2018).

SOM consists of living organisms in soil, plus root exudates and residues from plants at various stages of decomposition. As soil microbes break down organic matter, the nutrients (e.g., nitrogen, phosphorus) within it are mineralized and become available for plant uptake. Some organic carbon is released as CO_2; some is stored in the soil. Enzymes from microbes and earthworms help build soil aggregates which enhance water infiltration and retention and reduce the risk of soil erosion. The rate at which these processes occur depends on the type of organic matter applied, rate of application (Peltre et al. 2012), soil pH, carbon-to-nitrogen ratio (C:N), soil temperature and moisture, the diversity of microbes, and the physical structure of the soil. Soil science is still understanding how soil organic matter results in stable stores of carbon, such as those required for sequestration. The theory of "humification" of organic matter – the formation due to chemical interactions of large, dark, carbon-rich, soil particles that are slow to decay – is becoming less of a focus (Lehmann and Kleber 2015). Rather than the chemical properties of soils driving carbon stabilization and sequestration, microbial activity and the physical properties of soils may be more influential (Lehmann and Kleber 2015; Witzgall et al. 2021).

Recent research shows that soil structure – the bulk density, porosity, particle size, aggregate size and stability – determines the accessibility of organic matter to microbes. Simply put, where organic matter is easily accessible, microbes will consume it no matter its chemical stability (i.e., labile or recalcitrant). But when organic matter is trapped inside a soil aggregate or otherwise inaccessible to microbes, that organic matter (i.e., carbon) is protected from decomposition and remains in the soil. Decreasing bulk density and increasing porosity and aggregate stability also improves microbial habitat which stimulates microbial activity and diversity and soil function overall (Urra et al. 2019).

Practices to increase SOC are often limited by insufficient local resources (e.g., compost or biosolids), pre-existing levels of SOC high enough to limit the rate of additional SOC sequestration through amendment application, economic accessibility for farmers, and the impact that adoption of those practices might have on crop yield.

Soil health is important for reasons beyond environmental impacts and benefits. Humans rely on soils to grow food, fiber, and energy crops. Healthy soils produce more nutritious food for humans and animals, and degraded soils have been linked to malnutrition and food insecurity (FAO 2022). Increasing the fertility of soils by ensuring adequate amounts and availability of macronutrients – nitrogen, phosphorus, potassium, calcium, magnesium, sulfur – and micronutrients – boron, copper, zinc, iron, etc. – increases both the productivity of plants and their nutritive value.

3.7 Limitations of LCA Review

This section discusses limitations to LCA and this LCA review beyond the lack of quantitative data for soil health. Summary results and high-level conclusions, including the hierarchies of wasted food pathways presented below, are subject to caveats and limitations. While details presented throughout this report aim to identify potential differences in environmental impact that will result from different wasted food pathways, it is beyond the scope of this analysis to systematically present results for specific categories of wasted food or specific sub-technologies or management practices within a given pathway. The range of available technologies, management considerations and the demonstrated overlap or trade-offs in environmental impact that occurs across many pathways indicates that management matters and defies the desire to obtain a strict and simple ranking.

Furthermore, when considering post-consumer wasted food, often the waste stream is a mixture of many food types. When wasted food is mixed with other MSW or is co-processed with other organic waste streams the actual performance of management pathways will reflect not wasted food alone but the bulk waste stream with which it is processed. Many of the studies referenced throughout this document endeavor to model wasted food-specific impacts, however the specific assumptions employed and the degree to which this goal is achieved varies considerably within the literature. Some studies included in the LCA analysis do include another organic co-waste (e.g., yard trim for composting). These studies were only included when wasted food was the primary feedstock. These results, however, reflect the impact of the bulk waste stream. Wasted food composition based on local diets, inclusion of packaging, or the composition of source material can also influence potential treatment pathways, their efficiency, and the LCA results (Mondello et al. 2017).

For many combinations of management pathway and environmental impact category, the number of studies or scenarios modeling these impacts is limited. Given the range of reasonable assumptions, pathway technologies and management practices observed, when only a small number of studies/scenarios were identified it is probable that documented environmental performance may not well represent the average environmental performance of that pathway in real-world conditions.

Despite the preceding limitation, the LCA studies reviewed are broadly intended to reflect “current” pathway performance. While this knowledge is relevant, it may not be sufficient to answer the question of how communities and individuals should seek to manage wasted food in the future. As an example of this topic and exception to this general statement, several studies have looked at the impact that a future, cleaner electricity fuel mix would have on pathway performance. Furthermore, this report has not extensively highlighted opportunities to improve pathway environmental performance. Such opportunities certainly exist for all of the examined wasted food pathways. Examples of opportunities for improved performance are provided below:

- Utilize renewable energy. Applies to all wasted food pathways;
- Use enhanced emission controls on electrical or CHP engines. Applies to multiple pathways including AD, controlled combustion, and landfill;
- Utilize compost biofilters, which oxidize methane to carbon dioxide and reduce odors. Use of Gore covers also controls odors and reduces emissions of ammonia and volatile organics (Levis and Barlaz 2011);
- Expand the use of heat recovery. Applies to multiple pathways including AD, controlled combustion, landfill and composting.

The scope criteria (Table 3-3) used for the inter-study analysis attempt to harmonize environmental performance of the management pathways according to a simple definition of “good” practice from an LCA perspective. This largely means that the management pathway is required to include readily achievable environmental credits. The included scope criteria do not ensure “good” environmental performance, and may depart from current, average pathway performance.

The detailed discussion of environmental drivers in Chapter 2 combined with local knowledge regarding the operation and performance of management pathways in a specific region or community should help individuals interpret the hierarchies of wasted food pathways, presented below, in an appropriate manner.

Additional information on identified research gaps is presented in Chapter 6.

3.8 Summary of LCA Review Findings

This section summarizes the environmental impacts and benefits of wasted food pathways, drawing on the literature search findings described in this chapter and the previous chapter.

LCA Findings, by Indicator

Table 3-7 summarizes the contribution of wasted food pathways to each environmental indicator.

For global warming potential, all pathways are beneficial or near neutral, except for landfill and sewer/WWT. Many indicators, including GWP, are impacted by energy use. All pathways except source reduction use energy, and three pathways – AD, controlled combustion, and landfill – also generate energy from wasted food, offsetting some or all of their energy use. Energy use and avoided energy use affect the following indicators: GWP, energy demand, acidification, eutrophication, PM, human toxicity and ecotoxicity, and water consumption, thus assumptions about energy mix are important to findings. Additional sources of impact beyond energy use include the following: fugitive methane, nitrous oxide, and ammonia emissions. Some pathways produce environmental benefits through avoided production of food, feed, and/or synthetic fertilizer and/or avoided use of synthetic fertilizer.

TABLE 3-7. SUMMARY OF WASTED FOOD PATHWAY IMPACTS AND BENEFITS, BY ENVIRONMENTAL INDICATOR

		GWP (including carbon sequestration)	Energy Demand	Acidification	Eutrophication	Particulate Matter	Human Toxicity and Ecotoxicity	Water Consumption and Land Occupation	Soil Health
Summary		Most pathways beneficial or near neutral, except Landfill and Sewer/ WWT which emit substantial CH_4	All pathways (except source reduction) use energy; AD, controlled combustion, and landfill produce energy	Energy use and avoided energy use largely drive acidification; Composting emits NH_3; Soil amendments from AD > Sewer/WWT > Composting emit NH_3	Nutrient losses directly into waterbodies from Sewer/WWT and Landfill or soil amendments from AD, Composting, Land Application; Indirect losses through energy use (all pathways)	Energy use and avoided energy use largely drive PM formation; Composting emits NH_3	Dependent on technology, co-feedstocks, management practices	Benefits driven by avoiding food/feed production; other impacts relatively small	Benefits driven by land application of wasted food and soil amendments made from wasted food
Source of Environmental Impact or Benefit	Energy Use* by Pathway (all pathways except source reduction)	CO_2 emissions	Energy demand: Composting > Landfill > Controlled Combustion > AD	SO_x and NO_x emissions	NO_x emissions	PM and precursors (SO_x, NO_x, NH_3)	Hazardous air pollutants including mercury and dioxins	Power plants typically use but do not consume water	
	Pathway Process	Fugitive CH_4 emissions from Sewer/WWT and Landfill > Composting and AD; N_2O from Composting and		NH_3 from Composting	Release of nutrients by Sewer/WRRF effluent; NH_3 in Landfill untreated leachate	NH_3 from Composting	Incinerator emissions (dependent on technology and emissions controls) and disposal of ash; Landfill untreated		

		GWP (including carbon sequestration)	Energy Demand	Acidification	Eutrophication	Particulate Matter	Human Toxicity and Ecotoxicity	Water Consumption and Land Occupation	Soil Health
		Sewer/WWT; Carbon sequestration from landfill					leachate; Sewer/WWT effluent		
	Soil Amendments**	Carbon sequestration: Composting, AD and Sewer/WWT			Release of nutrients from application of organic amendments from AD, Composting, Land Application		Dependent on contamination of feedstocks, treatment methods, application rates		Soil health benefits: Compost, AD, Sewer/WWT, Land application, Unharvested/ plowed in
	Avoided Synthetic Fertilizer Manufacturing and Use**	CO_2 savings for Compost, AD and Sewer/WWT Organic amendments from Compost, AD, and Sewer/WWT release less N_2O than synthetic fertilizers	Energy savings for Compost, AD and Sewer/WWT	SO_x, NO_x, and NH_3 savings for Compost, AD and Sewer/WWT	Organic amendments from Compost, AD, Sewer/WWT release less NH_3 emissions and leach and runoff less nitrates than synthetic fertilizers	Organic amendments from Compost, AD, Sewer/WWT release less NH_3 emissions than synthetic fertilizers			

		GWP (including carbon sequestration)	Energy Demand	Acidification	Eutrophication	Particulate Matter	Human Toxicity and Ecotoxicity	Water Consumption and Land Occupation	Soil Health
	Avoided Food or Feed Production	CO_2, CH_4, and N_2O savings for Source reduction > Donation/ Upcycling > Animal feed	Energy savings for Source reduction > Donation/ Upcycling > Animal feed	SO_x, NO_x, and NH_3 savings for Source reduction > Donation/ Upcycling > Animal feed				Savings for Source reduction > Donation/ Upcycling > Animal feed	
	Avoided Energy Use	CO_2 savings for AD > Controlled Combustion > Landfill	AD, Controlled Combustion and Landfill produce net energy	SO_x and NO_x savings for AD > Controlled Combustion > Landfill					

*Including on-site use of energy derived from wasted food by the pathway

**Benefits apply only if digestate or biosolids from AD or Sewer/WWT are beneficially used. If they are disposed of, impacts from landfill or controlled combustion apply.

LCA Findings, by Pathway

The following sub-sections synthesize environmental impacts associated with each wasted food pathway. Note that most pathways covered in this report pertain to wasted food management. The source reduction, donation, and upcycling pathways, though, are functionally different: source reduction avoids the creation of wasted food in the first place, while donation and upcycling capture wasted food and recover it for human consumption. All other wasted food pathways (except animal feed, which indirectly feeds humans by feeding livestock) capture wasted food and take it out of the food supply chain, possibly recovering energy or other beneficial products.

Source Reduction

Source reduction is environmentally preferable to all other pathways and can be implemented at all stages of the supply chain. Avoiding the creation of wasted food saves more resources than can be captured from processing that wasted food via one of the other pathways. There is broad consensus in the literature—and in the findings of this LCA literature review—around this point:

The environmental benefits of source reduction exceed those of any wasted food pathway other than donation and upcycling by at least one order of magnitude.

WRAP also estimates environmental benefits for source reduction and donation that are an order of magnitude greater than the considered pathways (WRAP 2015).[60]

A United Nations report says that "preventing wasted food and … reducing excess food production is a far more effective strategy for minimizing environmental impact than optimizing end-of-life management" (Heller 2019).[61]

Beretta et al. (2017) looked at sources of GWP impact across the food value chain and concluded that the potential benefits of management of wasted food are small (5%–10%) relative to the burdens of initial food production and that preventing wasted food (i.e., through source reduction, donation and upcycling pathways) should be prioritized.

Source reduction, if implemented appropriately, does not reduce the amount of food available for human consumption, but rather reduces waste in the supply chain. However, we note that source reduction is a necessary but incomplete approach for reducing the environmental impacts of wasted food. Even with an optimal production and distribution system, some wasted food will be generated (e.g., inedible food parts such as shells or bones) so other pathways must be used to efficiently manage wasted food.

Donation and Upcycling

The management of wasted food via donation, and to a lesser extent upcycling, demonstrate environmental benefits similar to source reduction. However, both donation and upcycling have inefficiencies, generate waste, and require resources to collect and redistribute (donation) or process (upcycling) the recovered wasted food. Therefore, it is environmentally preferable to avoid wasted food in the first place (via source reduction).

Median impact values show that the magnitude of environmental benefits associated with donation are similar to those of source reduction and exceed the benefits of any other pathway (except upcycling) by at least one order of magnitude.

Damiani et al. (2021) modeled wasted food distribution and found that donation had better environmental performance across 18 impact categories, relative to a mix of controlled combustion, AD, and composting.

An analysis of fresh fruit and vegetable supermarket waste in Sweden found that donation and upcycling had much lower GWP and energy demand than controlled combustion and AD pathways (Eriksson and Spångberg 2017).

The validity of similar conclusions for the upcycling pathway will depend on the energy, resources, and emissions needed to reprocess wasted food for human consumption. Our literature reviewed found only two studies that investigated examples of upcycling; they include reprocessing fruit and vegetable waste into chutney (Eriksson and Spångberg 2017) and converting low grade broccoli crowns and stems into a bread additive or soup base (Eriksson et al. 2021). These studies show that impacts of processing are more likely to affect the net impact of upcycling when they differ substantially from the avoided production process.

Like source reduction, the donation and upcycling pathways receive an environmental credit for avoiding the production of additional food. However, as illustrated by Eriksson and Spångberg (2017), not all the rescued food collected for donation or upcycling helps to avoid food production. There will be some measure of waste present in each system. Eriksson and Spångberg's analysis assumes that 40% of donated food either replaces nothing or is otherwise wasted. Still, the magnitude of benefits associated with avoided food production are such that this loss of food is not enough to diminish the benefit of donation relative to other pathways.

The donation and upcycling pathways also require management, energy, and other resources to recover (donation), refine (upcycling), and distribute the diverted wasted food. While there are environmental impacts associated with collecting, redistributing, or refining food, these impacts tend to be smaller than those from the production of food in the first place. The magnitude of benefits associated with avoided food production is such that this loss of food is not enough to diminish the benefit of source reduction, donation and upcycling relative to other pathways.

Animal Feed

Although the use of wasted food as animal feed is less well-studied than some other management pathways, the reviewed studies demonstrate favorable environmental performance relative to the wasted food pathways beyond source reduction and donation. Broadly, repurposing wasted food for animals can offset production of animal feed, which can be resource intensive.

Inter-study (Section 3.4) results show the potential of the animal feed pathway to yield environmental benefits in the GWP and acidification impact categories. Eutrophication impacts of animal feed range widely in the reviewed studies with net impacts that overlap those of all other reviewed pathways. While no quantitative data were identified for the land occupation category and limited data for water consumption, the contributions of avoided grain production to animal feed's net environmental performance make environmental benefits likely in these categories.

Intra-study (Section 3.5) results show that using wasted food as animal feed generally leads to better environmental outcomes compared to the landfill, AD and compost pathways. However, the landfill pathway has overlapping environmental performance with wet feed in human toxicity and eutrophication and better performance in acidification. Wet feed is shown to consistently have better performance than dry feed, due to fewer processing requirements. Wet and dry feed show trade-offs in environmental impact when compared to controlled combustion. Wet and dry feed both have lower particulate matter and GWP impact than controlled combustion in all reviewed study scenarios. The controlled combustion pathway shows potential for overlapping environmental performance with wet and dry feed in human toxicity, wet feed in eutrophication, and dry feed in acidification. Albizzati et al. (2021b) is the only study to directly compare wet animal feed to landfill and controlled combustion, finding that wet animal feed has higher impacts due largely to land application of digestate associated with meat and fish waste not suitable for incorporation in the feed product.

Given the small number of studies on which these findings are based, more research into different technologies for animal feed production and systems of use (e.g., varying transportation distances) is needed to improve confidence in our understanding of the benefits of using wasted food as animal feed. The importance of avoided feed-grain production to environmental performance also warrants further research to examine the range of potential production systems and determine the appropriate type and quantity of avoided product for different wasted food types.

Anaerobic Digestion

AD is well-represented in the reviewed LCA studies. Median impact results are quite robust for AD, relative to environmental results for other management pathways, given the high number of reviewed studies that include AD as a wasted food management option.

Inter-study results in Section 3.4 demonstrate that among the wasted food pathways, beyond the source reduction, donation and upcycling pathways, AD has the potential to produce some of the largest environmental benefits in the GWP and energy demand impact categories due to its production of energy from wasted food. For acidification, the impact of AD falls in between results identified for other pathways showing a similar range of impacts to the controlled combustion pathway, higher impact than animal feed and lower impacts than the compost and landfill pathways. Eutrophication and land occupation impacts of the AD pathway overlap considerably with identified results for other wasted food pathways.

According to intra-study results in Section 3.5, AD has consistently better environmental performance than the compost and landfill pathways. Wet and dry animal feed generally produce lower environmental impacts (or greater benefits) than the AD pathway. AD demonstrates excellent comparative environmental performance in the GWP and ecotoxicity impact categories when compared to compost and landfill. The AD pathway has better or similar environmental performance compared to the controlled combustion pathway in GWP and the two toxicity impact categories. Controlled combustion is often shown to have lower particulate matter and acidification potential impacts than AD.

The AD pathway produces digestate or biosolids (if made through co-digestion at a WRRF), a nutrient-rich soil amendment, that has the potential to positively influence soil health. This analysis has focused on LCA studies that assume ultimate land application of digestate; however, not all digestate and biosolids are land applied. Where these outputs are disposed of via controlled combustion or landfill, credits for carbon sequestration and avoided fertilizer production and use are not warranted. LCA studies rarely consider this scenario. When digestate is incinerated or landfilled, environmental impacts will vary, nutrient losses from land application will be reduced, there will be no fertilizer offset, and additional energy may be recovered.

Research described in Section 4.6 outlines the potential soil health benefits of digestate use, such as nutrient recycling, improved soil structure, water retention, and water quality improvement. Land application of digestate can also contaminate soil, with subsequent human health impacts, if wasted food or co-treated wastes are contaminated. Research suggests that soil contamination may be more likely when wasted food is co-digested with biosolids or manure. To improve the quality and consistency of environmental impact results, more research on the potential impacts and benefits of digestate land application needs to be performed and incorporated into LCA models.

AD's reported environmental performance also relies on avoided energy credits that will diminish over time as U.S. energy mixes reduce reliance on fossil fuels. This trend is expected to negatively affect the long-term relative performance of this pathway.

Composting

The GWP boxplot in Figure 3-3 demonstrates a large potential for overlap of composting's net impact with that of the AD, animal feed, controlled combustion, and land application pathways in the GWP and eutrophication impact categories. Inter-study results show that composting has a higher, net positive (i.e., impact) energy demand than AD, controlled combustion, and landfill. The same is true for acidification, however the range of compost impact overlaps more considerably with the range of net impacts identified for the landfill pathway.

Intra-study results in Section 3.5 indicate that in the majority of reviewed study scenarios compost has higher environmental impacts than the animal feed, AD, and controlled combustion pathways. Composting demonstrates poor comparative environmental performance in acidification and particulate matter formation relative to all alternative pathways. These impacts are strongly associated with ammonia emissions during composting and land application. The best environmental performance of compost is observed in the GWP category, due to its low energy demand and environmental credits associated with avoided fertilizer production and carbon sequestration. Compost's best environmental performance is relative to the landfill pathway where a majority of study scenarios show that compost has better performance in the toxicity, eutrophication and GWP categories.

Carbon sequestration benefits are an important factor in median estimates of net GWP for the composting pathway and not all studies reviewed included this benefit (Table 3-4). Where carbon sequestration was included, it had a moderate effect on overall GWP estimates. This effect could be most consequential for the composting pathway, as the inclusion of carbon sequestration could switch the net impact of composting from positive (net source) to negative (net sink). Another review of wasted food management LCAs found that researchers assumed composting had more potential to sequester wasted food carbon than AD but less potential than landfilling (Bernstad Saraiva Schott et al. 2016).

Composting has the highest energy demand among the assessed wasted food pathways for which data were available. Although many composting operations require relatively low energy inputs, this pathway does not recover any of the energy present in wasted food, affecting its performance relative to other pathways.

Figure 3-9 shows that across intra-study comparisons, composting has higher or equivalent environmental impacts than controlled combustion in the majority of reviewed study scenarios across all assessed impact categories. These conclusions are subject to several caveats. First, the major disadvantage of the composting pathway is that it does not recover any of the energy present in wasted food. The U.S. depends heavily on fossil fuels for energy; this benefits the net environmental performance of the AD, controlled combustion, and landfill pathways, for which energy recovery is common.[62] In the future, as cleaner energy sources become more prevalent, the relative environmental performance of composting will improve, as wasted food pathways that derive benefit from avoided energy production will show fewer benefits. Second, the composting pathway produces a nutrient- and carbon-rich soil amendment that has the potential to positively influence soil health. Research presented in Section 4.6 outlines the soil health benefits of compost use, such as nutrient recycling, improved soil structure, water retention, and water quality improvement. The interactions of compost in the soil and accompanying agronomic practices are complex and are not treated consistently in the reviewed LCA literature. To improve the quality and consistency of environmental impact results, basic scientific research on the potential impacts and benefits of compost land application needs to be translated to LCA models.

Controlled Combustion

Based on this literature review, the availability of effective emission control devices and the ability to recover energy result in a higher ranking for controlled combustion (i.e., incineration) than for landfill. Controlled combustion of MSW, including wasted food, without energy recovery is not common in the United States (U.S. EPA 2022b; U.S. EPA 2023c).

Inter-study results in Section 3.4 show that controlled combustion has wide variation in GWP performance and overlapping estimates of eutrophication with the AD, animal feed and compost pathways. For acidification the impact of controlled combustion falls in between results identified for other pathways showing a similar range of impacts to the AD pathway, higher impact than animal feed and lower impacts than the compost and landfill pathways. Land occupation and water consumption impacts of the controlled combustion pathway are low compared to other wasted food pathways.

Controlled combustion exhibits strong environmental performance relative to the landfill and compost pathways across all six impact categories, where the majority of study scenarios show that controlled combustion's net impact is lower than that of landfill or compost. In the GWP and particulate matter impact categories, reviewed results indicate similar or overlapping environmental performance for controlled combustion with the compost and landfill pathways, respectively. Good environmental performance associated with controlled combustion is strongly linked to avoided energy credits, which depend on the displaced energy mix. The controlled combustion pathway demonstrates trade-offs in environmental performance and the potential for overlapping impacts when compared to AD and animal feed. Note that this analysis did not quantitatively consider soil health benefits.

Clavreul et al. (2012) looked at wasted food moisture content and heating values, finding that wasted food sources with lower moisture content and greater heating values favor controlled combustion, whereas wasted food sources with higher moisture content and lower heating values favor AD for the GWP impact category. Below 16 MJ/kg total solids, AD is preferred, while above 20.5 MJ/kg total solids, controlled combustion is preferred. Between these values, the technology preference is determined by moisture content. The range of variability in wasted food characteristics across studies likely plays a significant role in the variability identified in environmental impact results for all wasted food pathways.

Controlled combustion's reported environmental performance also relies on avoided energy credits that will diminish over time as U.S. energy mixes reduce reliance on fossil fuels. This trend is expected to negatively affect the long-term relative performance of this pathway.

As noted in Section 2.7, though there have been changes in emission control technologies, siting of controlled combustion facilities has disproportionately affected disadvantaged communities. Therefore, controlled combustion—to a greater degree than other wasted food pathways—has associated concerns about localized human health impacts.

Landfill

The findings of this literature review corroborate the general finding that landfilling of readily degradable wasted food will generate considerable quantities of fugitive methane emissions. Given the difficulty in efficient capture of that methane, these emissions typically drive GWP impact for that system. Results for several other impact categories also provide a strong environmental basis for preferring other wasted food pathways instead of landfill disposal of wasted food.

Looking across study scopes in Figure 3-3, wasted food landfilling consistently leads to high GWP impacts relative to other management pathways, except sewer/WWT. Intra-study results corroborate the high GWP impacts that result from wasted food landfilling, showing that in greater than 90% of reviewed study scenarios, GWP comparisons between landfill and four other common wasted food pathways (AD, animal feed, composting, and controlled combustion), landfilling demonstrated higher GWP impact.

In Section 3.5, landfilling demonstrates poor comparative environmental performance in the GWP, eutrophication, ecotoxicity, and human toxicity categories. Landfilling's best environmental performance is in the particulate matter formation and acidification categories where it shows mixed results compared to alternative pathways. Instances of good environmental performance for the landfill pathway are typically linked to avoided energy benefits, limited energy requirements and comparatively lower ammonia emissions.

Landfilling of MSW, including wasted food, is a heterogeneous, dynamic process. Therefore, landfilling is less well-suited to resource recovery than the other wasted food pathways. With respect to GWP, it is difficult to efficiently capture methane generated in these systems, especially given wasted food's rapid decomposition rate, which may be faster than the rate at which landfills are capped to recover methane. Although landfilling may have relatively low energy requirements, it requires a significant land area, has low relative energy recovery, consistently leaches pollutants to water via leachate, and therefore tends to have poorer environmental performance than other wasted food pathways.

Land Application, Unharvested/Plowed in and Sewer/WWT

The land application, unharvested/plowed in and sewer/WWT pathways are poorly studied relative to the other pathways listed above. The land application and unharvested /plowed in pathways were found to have a small GWP impact in the range of values observed for AD, composting and controlled combustion. The sewer/WWT pathway was found to have a high GWP, comparable to that of the landfill pathway, in part due to estimated methane production within the sewer system, which was estimated assuming 10% of COD (Parry and WERF 2012) or 6% of volatile solids (Edwards et al. 2018) degrade anaerobically during sewer transport. All of these pathways would benefit from further study.

3.9 LCA Conclusions

This section synthesizes the results of the review of LCA literature in this report through two rankings of the pathways. These rankings are based on the environmental performance of the pathways with wasted food as a feedstock.

Figure 3-10 presents a ranking based solely on GWP. The GWP ranking is based largely on the distribution of data shown in Figure 3-3, with additional considerations regarding the biophysical bases for the pathways and the degree to which the pathways are represented in the literature.

Figure 3-11 presents a ranking based upon LCA results from all eleven impact categories considered--including GWP (including soil carbon sequestration), energy demand, acidification, eutrophication, particulate matter, human toxicity, ecotoxicity, land occupation, water consumption, and soil health.

As shown in Figure 3-10 and Figure 3-11, the LCA rankings for GWP and for all eleven impacts are very similar. For GWP, animal feed and AD rank just above composting, controlled combustion, and land application, whereas when the ranking criterion is extended to all LCA impact categories, these five pathways are considered to comprise only one tier. For indicators beyond GWP, fewer data are available and greater uncertainties exist, leading to less differentiation among pathways in the hierarchy. The broader hierarchy also considered soil health qualitatively, since it is not captured in the quantitative LCA analysis. As the intra-study analysis shows, there are meaningful differences between some of the pathways grouped as single tiers above; however, application specifics, such as operating conditions, should be considered if refining beyond the tiers presented.

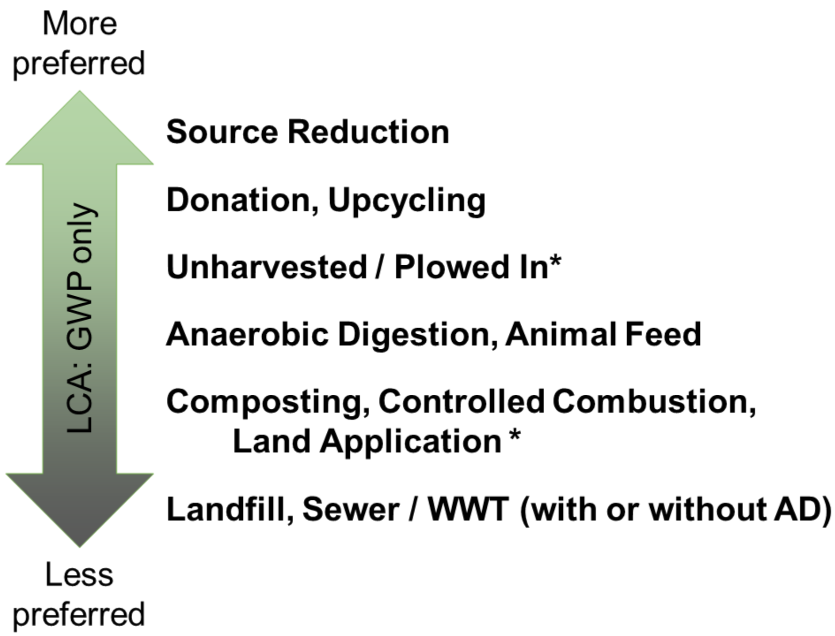

FIGURE 3-10. GLOBAL WARMING POTENTIAL: RANKING OF WASTED FOOD PATHWAYS

*indicates limited data available for a pathway
The ranking reflects each pathway's performance in global warming potential.

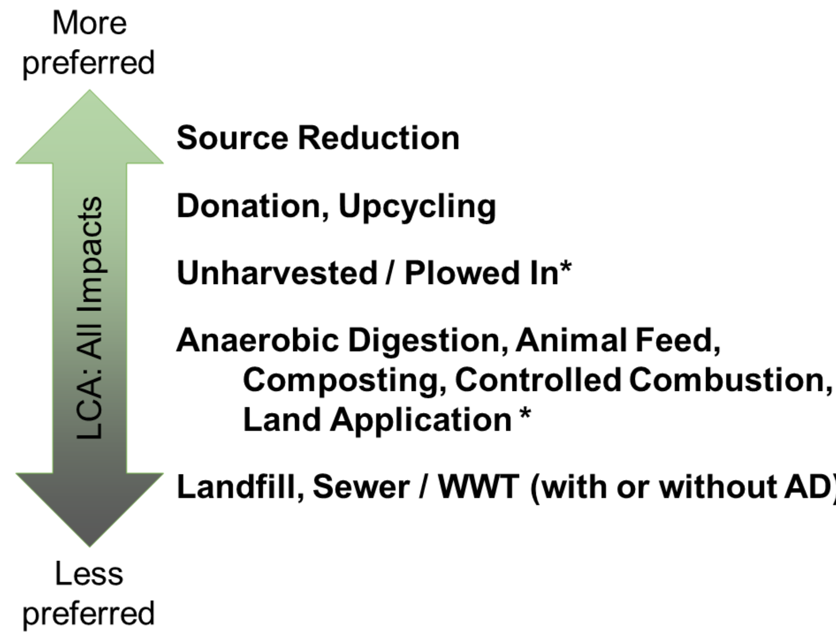

FIGURE 3-11. ALL LCA INDICATORS: RANKING OF WASTED FOOD PATHWAYS

*indicates limited data available for a pathway
The ranking reflects each pathway's performance in 11 impact areas: global warming potential, carbon sequestration, energy demand, acidification, eutrophication, particulate matter, human toxicity, ecotoxicity, land occupation, water consumption, and soil health.

Key LCA Findings:

- The environmental benefits of source reduction and donation exceed those of any other wasted food pathway by at least one order of magnitude (except for GWP, where only source reduction exceeds all other pathways by an order of magnitude).
- Landfill and sewer/wastewater treatment (with or without AD) stand out for their substantial fugitive methane emissions from the decay of wasted food in anaerobic conditions, and, for that reason, rank at the bottom of the LCA-based hierarchy.
- The unharvested/plowed in pathway is ranked high in both hierarchies. However, this pathway is limited to on-farm wasted food and is only recommended when food is unmarketable or otherwise likely to be wasted. In this context, the pathway provides an opportunity to avoid the environmental impacts of downstream processing, packaging, and distribution, while also returning nutrients to the soil. Limited data did not allow for in-depth analysis of this pathway.
- Upcycling is also not well-studied; however, the substantial benefits of avoided food production outweigh the resource use (energy, water) of processing wasted food, resulting in net benefits (i.e., negative medians) for all impacts.
- The differences among the remaining pathways are small compared to the differences between them and the highest-ranked pathways (source reduction and donation) and the lowest-ranked pathways (landfill and sewer/WWT). In particular, the performance of AD, animal feed, composting, controlled combustion, and land application are quite similar.
- The median GWP of all pathways except landfill and sewer/WWT were found to be negative (i.e., beneficial) or near zero.
- For GWP, carbon sequestration credits were inconsistently applied in LCAs reviewed and when applied, provide a moderate benefit to composting and anaerobic digestion, potentially moving compost from net source of GWP to net sink of GWP. When applied to sewer/WWT the benefit is larger, demonstrating the importance of beneficial use of biosolids to the GWP performance of the pathway.
- Per ton of wasted food, AD can produce more energy from wasted food than controlled combustion, which produces more energy than landfill gas capture systems. The moisture of wasted food reduces efficiency of energy recovery by controlled combustion and can require pre-drying of feedstock before processing, increasing the energy use of the pathway. Methane can off-gas from wasted food before installation of landfill gas capture systems, reducing potential for energy recovery from wasted food and resulting in fugitive methane emissions.
- Five pathways can contribute to soil health and carbon sequestration: AD, composting, land application, sewer/WWT, and unharvested/plowed in. However, for two of these pathways (AD and sewer/WWT), the nutrients recovered are sometimes landfilled or incinerated rather than beneficially used. LCA assumptions vary, but AD is typically credited with carbon sequestration and avoided fertilizer manufacturing and use in LCAs. This is legitimate only if digestate or biosolids (if produced through co-digestion) are applied to land. Among the sewer/WWT studies reviewed, two did include carbon sequestration offsets; five did not (Table 3-4). As shown in that table, studies that do include carbon sequestration tend to show a modest decrease in GWP.
- The environmental performance of individual pathways is highly variable and is affected by operating parameters, such as technology type or temperature and pre- and post-treatment of outputs. For example, composting is well-studied but inconsistently modeled by LCA, and process emissions are variable due to composting method, facility management, and environmental conditions.
- GWP and energy demand impacts are reduced when pathways produce energy, due to avoided fossil fuel use. In the future, as the energy mix changes to include fewer fossil fuels and thus to become less carbon intensive, pathways which recover energy (AD, controlled combustion, and landfill) will see reduced GWP benefits while other pathways that only use energy (e.g., animal feed, composting) will see decreased GWP impacts.

CHAPTER 4
CIRCULARITY ASSESSMENT

Circularity is a central component of EPA's vision for materials and wasted food management. In contrast to a traditional linear (i.e., "take-make-use-dispose") economy, a circular economy is one where economic growth can be decoupled from resource extraction by capturing greater value from existing products and materials (Figure 4-1). Many organizations and governments have adopted circularity principles over the last decade, including the United Nations, the World Economic Forum, China, and the European Union (Bocken et al. 2016; Geissdoerfer et al. 2017). In the U.S., the Save Our Seas Act 2.0 directed EPA to promote and support circular economy strategies for post-consumer materials management (U.S. Congress 2020).

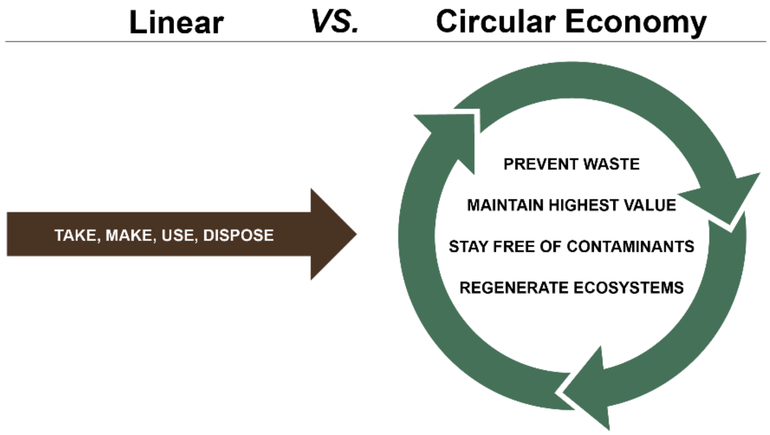

FIGURE 4-1. COMPARING A LINEAR ECONOMY AND A CIRCULAR ECONOMY

The concept of circularity draws from several schools of thought, including industrial ecology, cradle-to-cradle, regenerative design, and others (Geissdoerfer et al. 2017), and a review of the development of a circular economy is provided elsewhere (Geissdoerfer et al. 2017; Murray et al. 2017; Geisendorf and Pietrulla 2018). There are now over 100 definitions of circular economy (Elia et al. 2017; Geissdoerfer et al. 2017; Kirchherr et al. 2017; Do et al. 2021; Tanveer et al. 2021; WEF 2023) and over 60 metrics (Parchomenko et al. 2019; De Pascale et al. 2021) to evaluate circularity in the scientific literature. The definition of a circular economy has evolved over the last twenty years, from simply meaning closed loop manufacturing (where wastes become inputs) to including concepts of restoration and regeneration of ecosystems (Geissdoerfer et al. 2017; Geisendorf and Pietrulla 2018; Tanveer et al. 2021).

The Ellen MacArthur Foundation has developed some of the most comprehensive and cited resources for outlining, implementing, and assessing circularity, including the ReSOLVE framework which focuses on optimizing systems to reduce waste, maintaining value of products and materials, keeping materials within the system by which they were created, designing out negative externalities, and preserving and regenerating natural systems (Ellen MacArthur Foundation 2013). In the Ellen MacArthur Foundation model, a circular economy seeks to address natural capital challenges (e.g., soil degradation, biodiversity loss, and freshwater quality and depletion), resource challenges (e.g., nutrient losses), and system challenges (e.g., emissions and limited carbon carrying capacity).

In the Save Our Seas Act 2.0 (U.S. Congress 2020), which guides this analysis, a circular economy is defined, in line with descriptions supplied by the Ellen Macarthur Foundation, as one that "uses a systems-focused approach and involves industrial processes and economic activities that:

- are restorative or regenerative by design;
- enable resources used in such processes and activities to maintain their highest values for as long as possible; and
- aim for the elimination of waste through the superior design of materials, products, and systems (including business models)".

These principles can apply to both technical (i.e., engineered or industrial) systems, such as electronics manufacturing, and biological systems, such as the food system (Ellen MacArthur Foundation 2013). In the case of the food system, analyses must account for food's essential role in human life—i.e., that its primary purpose is to be consumed—and its perishability. While technical products and their components (e.g., electronics, appliances) in a circular economy should remain in the economy for as long as possible through repair, reuse, re-manufacture, and recycling, biological products (including food) should instead be designed to re-enter the environment safely to restore the ecosystem from which they were withdrawn (Do et al. 2021).

This analysis does not address the entirety of a circular food system. Instead, it focuses on the potential contributions of the wasted food pathways, as defined earlier in this report (Table 1-1), to the circularity of the food system. The generation and management of wasted food have been identified as key stages of the food system where circular economy principles can be implemented (Jurgilevich et al. 2016). The report will exclude other parts of the food system, such as the management and effects of human waste from consuming food (e.g., outputs from respiration or excretion).

4.1 Circularity Themes

The key circularity themes used to evaluate the wasted food pathways in this report were derived from the Save Our Seas Act 2.0 and informed by the Ellen MacArthur Foundation ReSOLVE framework and the scientific literature regarding circularity of biological systems (Ellen MacArthur Foundation 2013; Ellen Macarthur Foundation 2019; U.S. Congress 2020; Do et al. 2021; Tanveer et al. 2021). In this chapter, the selected themes—waste prevention, value, purity, and regeneration— are utilized to determine a ranking of the wasted food pathways, by degree of circularity.

Definitions of the key circularity themes are presented in Table 4-1 and further expanded in the following sections. While the themes are presented here as four distinct concepts, it should be noted that they often overlap and relate. While waste prevention is preferred, some amount of wasted food is inevitable. In a circular economy, when waste cannot be prevented, waste outputs should maintain their highest potential value, stay free of contaminants, and help to regenerate ecosystems.

TABLE 4-1. KEY CIRCULARITY THEMES

Circularity Theme	Definition
Waste Prevention	Waste is minimized through improvements in system efficiencies.
Value	Outputs maintain their highest potential value.
Purity	Outputs stay free of contaminants to enable reuse.
Regeneration	Outputs help to regenerate ecosystems.

Waste Prevention

Preventing waste is paramount to circularity and, within the food system, it can occur in a variety of ways, such as source reduction, donation, or upcycling. All three of these pathways offer the potential to reduce the amount of food production required to feed a population, thus reducing the negative externalities associated with food production (Sherwood 2020), including greenhouse gas emissions and climate change, consumption and degradation of freshwater resources, loss of biodiversity and ecosystem services, and degradation of soil health and air quality. According to EPA (U.S. EPA 2021c), the production, processing, and distribution of food that is ultimately wasted in the U.S. each year embodies:

- 140 million acres of agricultural land – an area the size of California and New York combined.
- 5.9 trillion gallons of blue water – equal to annual water use of 50 million American homes.
- 778 million pounds of pesticides.
- 14 billion pounds of fertilizer – enough to grow all the plant-based foods produced each year in the United States for domestic consumption.
- 664 billion kWh energy – enough to power more than 50 million U.S. homes for a year.
- 170 million $MTCO_2e$ GHG emissions (excluding landfill emissions) – equal to the annual CO_2 emissions of 42 coal-fired power plants.

Value

In a circular economy, outputs should maintain their highest potential value. The highest value of food is as sustenance for humans (directly or indirectly through feeding of livestock). Once food is not consumed, the highest value is dependent on the context for decision making and could be evaluated in numerous ways including economically, culturally, nutritionally, or physically.

Valuable materials such as the nutrients most relevant to agriculture – nitrogen, phosphorus, and potassium —can be recovered from wasted food and used as fertilizers. Synthetic nitrogen fertilizers currently consume one percent of global energy demand (Sherwood 2020), and anthropogenic interference with the nitrogen cycle has already exceeded estimated planetary boundaries (Steffen et al. 2015; Cobo et al. 2018). For instance, in some areas of the globe, over-application of N fertilizer made possible by the N fertilizer synthesis process has led to an imbalance in the N available to plants and the N those plants need and can take up, resulting in runoff of excess N. Also, phosphorus is a limited resource (i.e., it cannot be synthesized like nitrogen fertilizers), and viable reserves are expected to be depleted and limit food production within 100 years (Cobo et al. 2018; Alewell et al. 2020; Sherwood 2020). Biomass from wasted food can be used as an alternative feedstock to produce nitrogen and phosphorus fertilizers, reducing fossil fuel use and supplementing the global supply of phosphorus.

In a circular economy, material recovery is generally valued over energy recovery (Potting et al. 2017; Do et al. 2021) due to the embodied value in materials and the global transition towards more renewable energy sources. As the energy mix becomes less emissions-intensive, the value of providing alternative energy from wasted food will decrease; however, the fossil-fuel inputs required as feedstock for manufacturing synthetic nitrogen fertilizer, the limited global phosphorus supply, and reliance on soils for food production, means there will continue to be high value in recovering nutrients from wasted food. In this analysis, nutrient retention is considered more valuable than energy recovery from wasted food, especially over the long term. Carbon is another component of wasted food that can be recovered and applied to soil to build organic matter and store carbon in soils, and this use for the carbon in wasted food is valued more highly in this analysis than carbon's use to create renewable energy. Nutrient cycling in soils is reliant on carbon as an energy source for microbes and a factor in soil structure, linking carbon recovery with nutrient recovery in wasted food pathways that can contribute to soil health. This was discussed in Section 3.6 on soil health and is further discussed with the regeneration theme in this chapter.

Circularity also credits "cascading uses" (i.e., the sequential and consecutive use of resources). For example, one could remove valuable components from wasted food and upcycle them into new food for human consumption, then process the remaining mass of wasted food through composting to recover nutrients like nitrogen and phosphorus.

Purity

The purity (or "pure circle") theme of circularity stresses the prevention of physical and/or chemical contamination so that materials can be reused more easily. Physical contamination includes things like plastic, metal or glass. Examples of chemical contaminants include pharmaceuticals or persistent organic pollutants like per- and polyfluoroalkyl substances (PFAS). This concept is highly related to value, as contamination may prevent the beneficial use of recovered materials or limit the range of possible uses. For the purposes of this analysis, potential contaminants, such as pesticides or PFAS, introduced into food before it becomes surplus or waste (e.g., during the production stage) are considered out of scope. However, potential contaminants may also be introduced to wasted food streams accidentally or intentionally as they are collected and managed. For example, non-compostable packaging may be accidentally placed into a wasted food stream, or wasted food may intentionally be co-digested with wastewater solids – and either of those scenarios may introduce a contaminant like PFAS or plastics to the wasted food stream (U.S. EPA 2021e; U.S. EPA 2021f). In addition to making reuse more difficult, physical and chemical contaminants in the food waste stream can reduce life cycle environmental performance of a wasted food pathway by requiring additional energy use for pre-treatment, reducing the amount of food waste recycled, generating technical problems during recycling, and contaminating soils where recycled outputs are applied (Le Pera et al. 2023). Some wasted food pathways can screen out, destroy, inactivate, or immobilize contaminants, reducing potential risks from contamination. Likewise, improving sorting out of physical contaminants can improve the life cycle environmental performance of a wasted food pathway (Angouria-Tsorochidou et al. 2023).

Regeneration

The theme of regeneration is critical to circularity; however, recent reviews found inconsistencies in the definition and use of the term (Morseletto 2020; Schreefel et al. 2020). This analysis will rely on the definition of regeneration from Elevitch et al. (2018), which is consistent with many of the common themes identified by the reviews. Regeneration is an “approach that has the capacity for self-renewal and resiliency, contributes to soil health, increases water percolation and retention, enhances and conserves biodiversity, and sequesters carbon.” This analysis will focus on the potential contributions of the wasted food pathways to regeneration; broader agricultural practices are considered out of scope.

Regeneration may occur through the wasted food pathways by the addition of slow-release nutrients or carbon to the soil. Carbon can help to build soil organic matter (SOM), increase water holding capacity, and enhance nutrient cycling. Most agricultural soils have lost significant portions – 30-75% – of soil organic carbon (Lal et al. 2007), which leads to increased risk of erosion and compaction and less resistance to drought. Soil amendments derived from wasted food can supply valuable organic carbon to soils and help restore the health and functionality of agricultural lands.

Wasted food pathways which promote regeneration can improve the resilience of agricultural systems in the face of climate change through improved soil health. Soil moisture conservation, reclamation of degraded soils, and soil carbon management are effective practices for climate change mitigation and adaptation (IPCC 2023).

Other Circularity Themes

The four circularity themes applied in this chapter were chosen based upon their connection to the definitions of circularity from the Save Our Seas Act 2.0 and the Ellen Macarthur Foundation and their applicability to biological systems. Other circularity criteria presented in the literature (Do et al. 2021) that are not easily applicable to the food system or to wasted food management were not applied. For example, the concept of longevity (i.e., keeping a product in use for as long as possible) does not apply to a perishable and consumable product like food.

In addition, some definitions of circularity call out the need to design out negative externalities from systems. Potting (2017) offers the “rule of thumb” that more circularity equals more environmental benefit, since a primary goal of circularity is to reduce the consumption of natural resources and materials and minimize waste; however, circularity does not guarantee positive environmental results. For example, recovering materials can require additional water, energy or transportation (Del Borghi et al. 2020). LCA can help to identify and weigh some of these unintended or indirect consequences. The results of this circularity assessment will be used in conjunction with the LCA results in the previous chapter, which may capture additional externalities, to inform the conclusions of this report.

4.2 Methodology

Although there are quantitative methods for evaluating circularity, these are nascent for biological cycles and food (Elia et al. 2017; Moraga et al. 2019). A systematic search of the literature provided no comparisons of wasted food management pathways using quantitative circularity metrics. Therefore, this report qualitatively evaluates the relationships between the pathways and key themes in circularity discussed above. The assessment relies on the literature search described in detail in Appendix A. Alternative qualitative methods and accompanying results are discussed in Section 4.7.

The following sections aim to identify the connections between the four key themes of circularity and the wasted food pathways. The analysis focuses on the environmental aspects of circularity and does not consider social and economic factors. Where possible, the analysis considers the specific effect of wasted food in each pathway (e.g., incinerating wasted food rather than incinerating MSW or another MSW component such as plastics) rather than the aggregate circularity of the pathway. In Section 4.9 the findings are synthesized to develop a novel circularity hierarchy based on each pathway's agreement with the circularity themes.

4.3 Waste Prevention

Only source reduction, donation, and upcycling minimize the generation of wasted food that will not be directly eaten by humans. The other pathways offer alternative uses for wasted food. Source reduction, by definition, prevents the generation of wasted food and more closely aligns supply and demand. Donation and upcycling aim to ensure food is still consumed by humans, although some amount of wasted food is inevitable in these pathways (e.g., food spoiling at food banks or in recipients' homes). Since source reduction, donation and upcycling minimize the need for waste management strategies, only the remaining pathways will be examined in connection to the other circularity themes.

If crops would ultimately be wasted (i.e., not sold and consumed), then the unharvested/plowed in pathway offers an opportunity to prevent downstream environmental impacts from processing, packaging and distributing food. However, unlike the three pathways noted above, it cannot reduce food production or its more substantial environmental impacts. Therefore, unharvested/plowed in is grouped with the remaining pathways and is less circular than donation or upcycling of crops which feed people.

Figure 4-2 provides a ranking of pathways based on their alignment with the waste prevention theme of circularity.

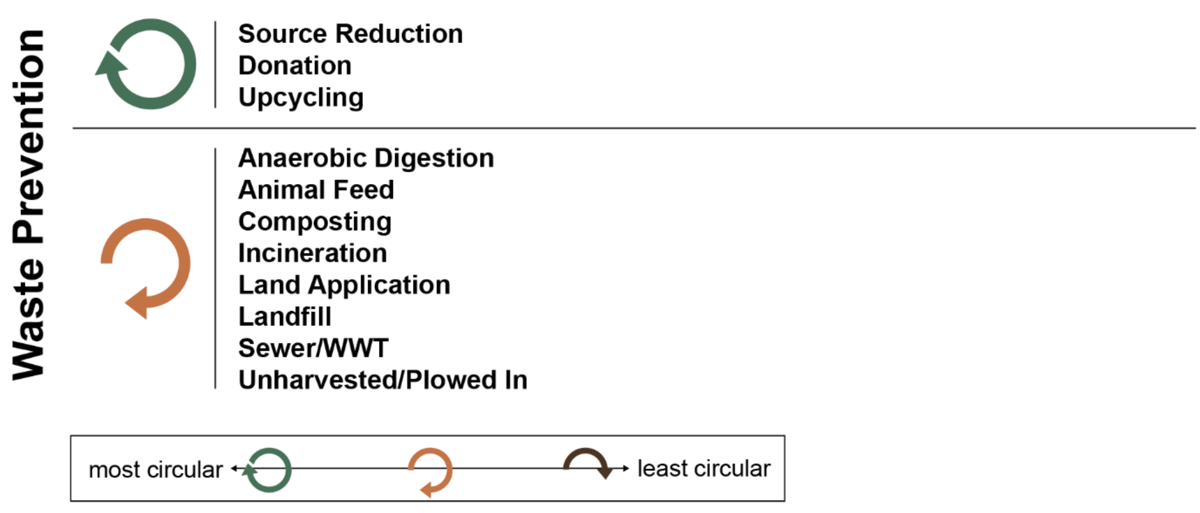

FIGURE 4-2. WASTE PREVENTION: ALIGNMENT OF WASTED FOOD PATHWAYS

Within lower tier the pathways are presented in alphabetical order.

4.4 Value

Nourishing Humans

Food grown for human consumption is most valuable when it can be used to provide human sustenance. Transforming wasted food into animal feed indirectly maintains food's original function (e.g., energy, nutrients) by providing animal feed to livestock production, which produces food for human consumption. The circularity of the animal feed pathway is endorsed by researchers supporting the World Economic Forum, who included the "use of only biomass unsuited for direct human consumption for animal feed" as one of their three core principles for a circular food system (WEF 2023).

Recovering Nutrients

When food can no longer be used to nourish humans or animals, circularity generally values the recovery of material over energy recovery. In this analysis, pathways that recover organic matter and critical nutrients, using them as fertilizer or soil amendments, are considered more circular than those that produce only energy.

Five pathways recover organic matter and valuable nutrients from wasted food that can be applied to land as fertilizers or soil amendments – AD, composting, land application, sewer/WWT, and unharvested/plowed in. Wasted food can be a source of various critical macro- and micronutrients required for plant growth, but it is especially rich in nitrogen. These pathways also recover carbon, which is critical for nutrient cycling and retention as well as other soil functions. Carbon recovery from wasted food is discussed further in Section 4.6.

Table 4-2 provides a summary of the nutrient-rich outputs produced by these five pathways, along with the potential destinations of each output, for reference.

To provide value (and thus qualify as circular), the nutrients recovered in these pathways must be beneficially used, rather than disposed of. Note that disposal by landfill and controlled combustion are potential destinations for recovered nutrients from two of the five pathways (AD and sewer/WWT).

Three pathways provide direct nutrient recovery. The unharvested/plowed in pathway leaves unharvested crops in the field, and the land application pathway applies waste from the food processing sector to agricultural land. Compost (i.e., the end product of the composting pathway) is also predominantly land applied. According to a survey of U.S. composters, 99% compost is used in ways that utilize recovered nutrients (e.g., landscaping, agriculture), with remaining 1% applied as landfill cover (U.S. Composting Council 2021).

The other two pathways that recover nutrients –AD and sewer/WWT—may use or dispose of nutrient outputs. As a reminder, the nutrient-rich outputs from stand-alone AD facilities and AD systems on farms that digest or co-digest wasted food are called digestate, whereas the nutrient-rich outputs from AD facilities that co-digest wasted food at water resource recovery facilities (WRRFs) are called biosolids.

TABLE 4-2. DESTINATIONS FOR NUTRIENT OUTPUTS FROM WASTED FOOD PATHWAYS

Pathway	Nutrient Output	Alternative Destinations, Beyond *Direct Application to Soil**
Anaerobic Digestion (stand-alone and on-farm digesters)	Digestate	**Animal Bedding** **Composting** Controlled Combustion Landfill Sewer/WWT
Anaerobic Digestion (co-digestion at WRRFs)	Biosolids	**Composting** Controlled Combustion Landfill Sewer/WWT** (re-circulated and/or discharged in effluent)
Composting	Compost (including compost made from digestate or biosolids)	-
Land Application	Raw Wasted food from food processing industry	-
Sewer/WWT (with or without AD)	Biosolids	**Composting** Controlled Combustion Landfill Sewer/WWT** (re-circulated and/or discharged in effluent)
Unharvested/Plowed In	Crops	-

Pathways noted in green are beneficial uses of nutrients.

*Digestate or biosolids may receive treatment other than and/or in addition to composting before direct application to soil.

**Sewer/WWT can be beneficial use if discharge used as irrigation water.

The sewer/WWT pathway also produces biosolids from organic waste (including wasted food) sent down the drain and through the sewer to a WRRF. Wastewater solids from the sewer/WWT pathway may undergo a variety of treatments, including AD, aerobic digestion, lime stabilization, composting, and thermal drying, among others. For the sewer/WWT pathway, a portion of nutrients from wasted food sent down the drain is separated from wastewater solids into the liquid effluent at a WRRF, which is treated prior to reuse or discharge.

Digestate includes solid and liquid fractions which are often, but not always, separated after digestion (i.e. through dewatering), with the majority of nitrogen contained in the liquid fraction (Havukainen and Dace 2023). In 2021, the EPA performed a voluntary survey of AD facilities in the U.S. that process wasted food and included questions about the use of solid and liquid digestate fractions from their facilities (U.S. EPA 2023). Respondents shared that the solid fraction may be beneficially used or disposed of, with disposal more likely at WRRFs than at stand-alone or on-farm digesters. The liquid fraction is typically applied to agricultural land at on-farm digesters (100% of facilities reported this as one of their practices), whereas it may be applied to land or discharged to a WRRF by stand-alone digesters (33% and 57%, respectively of stand-alone facilities reported each practice as one method for handling their digestate) (U.S. EPA 2023). WRRFs most often recirculate the liquid fraction of digestate (i.e., dewatering liquid) back through the wastewater treatment process. After discharge to and/or treatment by a WRRF, nutrients, now in biosolids or effluent, may be beneficially used or disposed of, as discussed below. Researchers have noted that digestate utilization is currently a “bottleneck” for development of biogas industry (Lu and Xu 2021).

Biosolids are wastewater solids/sewage sludge – the solid fraction of the final product of wastewater treatment (effluent is the liquid) – that have been treated (with or without AD) to meet federal standards for contaminants and pathogens, allowing for their beneficial use. Biosolids treated to higher quality (i.e., Class A or EQ) have fewer restrictions on their use. In EPA’s AD survey, 16% of reporting AD facilities at WRRFs indicated that they produce Class A biosolids, and 84% reported producing Class B biosolids (U.S. EPA 2023). To the extent that biosolids are not used as fertilizers or soil amendments, the circularity of this pathway is limited. In 2018, 45% of biosolids generated in the U.S. (including those that are produced at facilities that do not co-digest wasted food with wastewater solids) were sent to landfill[63] or controlled combustion rather than being beneficially used (Beecher et al. 2022). Nationwide, treatment of biosolids to Class A quality (rather than Class B) is increasing, which may improve the circularity of this pathway in the future (Beecher et al. 2022).

Wasted food, compost, digestate and biosolids may all be applied to agricultural land to provide nutrients to the soil and improve soil health. These materials also have many uses outside of agriculture as well, such as land restoration or reclamation (e.g., at abandoned mine sites) and green infrastructure. For agricultural land, nutrient application should be aligned with a crop’s nutrients needs, most often nitrogen.

The nutrient content in wasted food can be a challenge for management pathways. For example, wasted food contains more nitrogen than wastewater solids and increases the nutrient load in biosolids and effluent from wastewater treatment processes when added through sewers or co-digested at a WRRF. One limitation to the increased use of sewer/WWT and AD at WRRFs to process wasted food is the ability of wastewater treatment technologies to manage additional nutrient loads, especially in regions with concerns of nutrient loading to marine and freshwater ecosystems. Emerging technologies that “strip” nutrients during the wastewater treatment or AD process can help to control the amount of nitrogen and phosphorus in the final materials and produce concentrated fertilizers. Stripped, concentrated nutrients are then sold and land applied – perhaps mixed with other plant macro- or micro-nutrients – as substitutes for synthetically-derived nutrients.

In the sewer/WWT pathway, two factors reduce the total nutrient recovery/content of biosolids (relative to original feedstocks, including wasted food. First, nitrogen and phosphorus are removed during secondary wastewater treatment to limit nutrients in the effluent and thus potential runoff to surface waters (Morris et al. 2014). Second, some nutrients remain dissolved in effluent (another potential method of nutrient recovery) and do not become a part of the biosolids. The effluent is discharged or used as irrigation water in agriculture or landscaping and can be source of nutrients. One study in Italy showed that significant portions of N and P required by grapes and olives could be supplied by irrigating with reclaimed water from 11 different WRRFs (Vivaldi et al. 2022). In addition to nitrogen and phosphorus, effluent from WRRFs contains potassium and other salts that plants need to grow (Sullivan et al. 2021). While nutrient recovery is not the primary goal of WRRF effluent reuse, the nutrient concentrations are considered when also applying fertilizers or organic soil amendments to avoid exceeding crop needs, thus potentially reducing synthetic fertilizer use.

Recovering Other Materials

The solid fraction of digestate can be processed into animal bedding (in lieu of land application), which, after use, will be managed with manure and urine (and may ultimately be processed through AD or composting).

Controlled combustion results in a material output (ash) that can be beneficially used in certain circumstances; however, this is currently infrequent in the United States. Ash is often contaminated with heavy metals, dioxins, and other pollutants. There are two types of ash – bottom ash and fly ash – and both are generally landfilled (Cho et al. 2020). Bottom ash accounts for the majority of incinerator ash and can be used as an aggregate in construction materials (Funari et al. 2017; U.S. EPA 2023c). Metals may be recovered from ash, but these are typically not derived from wasted food feedstock.

Other pathways for wasted food are emerging to create chemicals, bioplastics and other materials, and these pathways would also be considered circular.

Recovering Energy

Three pathways recover energy from wasted food – AD, controlled combustion, and landfill – with varying efficiency. Generally speaking, AD produces the most energy per ton of wasted food of the three pathways. Wasted food's methane potential, biodegradability, and high-water content make it an ideal substrate for AD (Momayez et al. 2019). When added to AD systems at WRRFs or on-farm AD systems and co-digested with wastewater solids or manure, respectively, wasted food has been shown to synergistically increase biogas production (Kim et al. 2017; Baek et al. 2020). AD can recover energy and nutrients, demonstrating the cascading uses (i.e., the sequential and consecutive use of resources) principle promoted by circularity. However, energy recovery may be the primary intention of the facility (Beghin-Tanneau et al. 2019) (e.g., due to economics), and thus the nutrient content of wasted food (in digestate) may be underutilized. Most AD research has focused on optimizing energy production, leaving digestate use as a secondary concern (Cesaro 2021). To meet objectives of a circular economy there must also be valorization (i.e., beneficial use) of the digestate (Cesaro 2021).

Energy is also recovered by controlled combustion and landfill pathways. However, the energy recovery efficiency tends to be lower than with AD. The moisture content of wasted food makes it less ideal for energy recovery from controlled combustion than from AD (Klinglmair and Thomsen 2020; Lu and Xu 2021). Landfills are typically considered to be the end points of linear systems; they bury waste and fail to beneficially reuse nutrients. In this assessment, we consider only landfills that recover energy from waste decomposition; however, as mentioned earlier in report, wasted food may decompose prior to the placement of landfill gas capture systems, resulting in more fugitive methane emissions than captured energy.

Figure 4-3 provides a ranking of wasted food pathways based on their alignment with the value theme of circularity.

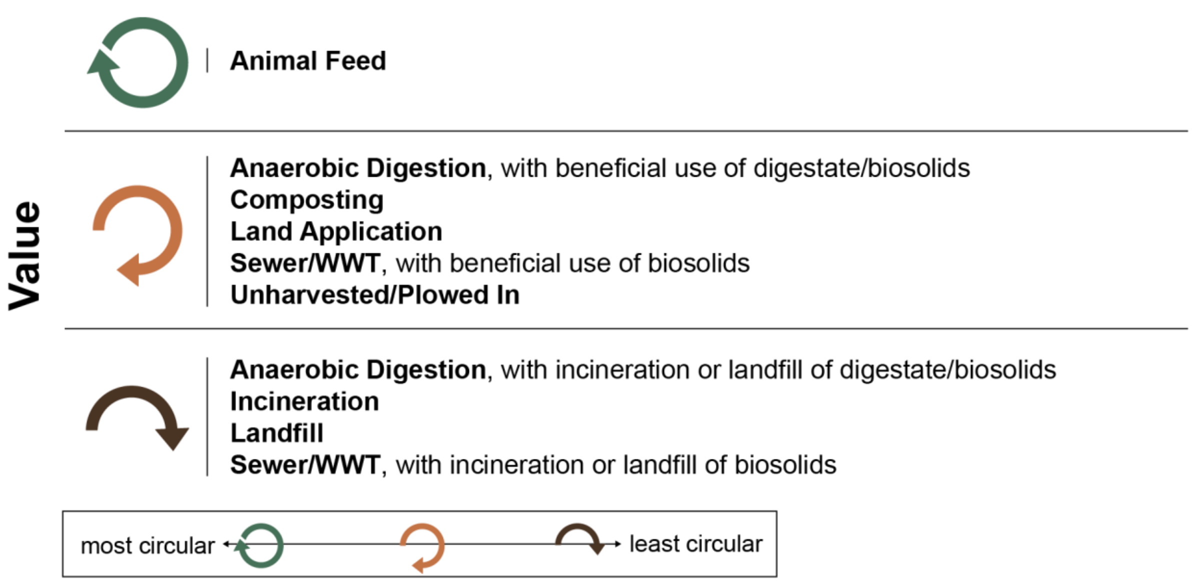

FIGURE 4-3. VALUE: ALIGNMENT OF WASTED FOOD PATHWAYS

Within each tier the pathways are presented in alphabetical order. Sewer/WWT with incineration or landfill of biosolids that also does not include AD would rank below the bottom tier here, as it does not recover energy. Landfills and controlled combustion without energy recovery are not included as they are out of scope for this report. Since source reduction, donation and upcycling minimize the need for waste management strategies, they are not ranked with regard to the other circularity themes (value, purity and regeneration).

4.5 Purity

Pure Circle

Pathways which manage only wasted food (i.e., no co-mingled wastes) can most effectively maintain a "pure circle" limiting physical and chemical contamination. The unharvested/plowed in pathway avoids contamination by leaving wasted food in its original location for tilling or grazing. The animal feed pathway applies strict health and safety requirements to ensure wasted food is safe for animals to eat. In addition, wasted food is often sterilized to ensure its safety for livestock. The land application pathway preserves purity by managing only wasted food from the food processing sector. This waste stream tends to be consistent and homogenous with little contamination since the wasted food is typically captured prior to packaging or distribution.

Stand-alone AD facilities can also maintain a pure circle. Some stand-alone AD facilities (often called industry-designated digesters) are designed and sited to manage wasted food streams from food processing facilities and do not accept other wastes, while other facilities (called multi-source digesters) accept wasted food and other wastes from a variety of sources. Digestate made only from wasted food has fewer pathogens and heavy metals than digestate from manure, for example (O'Connor et al. 2021). Wasted food digestates typically do not require sanitation before land application, due to low contamination levels (Dutta et al. 2021). If digesters accept additional non-wasted food streams, they can face challenges of contamination described below.

Potential for Contamination

Beyond the pathways discussed above, the degree of purity maintained is more complex to judge. Purity is highly connected to value, as more beneficial uses of material outputs are possible with lower contamination. Thus, we will examine the sources of and potential remedies for physical and chemical contamination within the four remaining pathways that recover materials – AD (except industry-dedicated digesters), composting, controlled combustion, and sewer/WWT. Landfills will not be considered in this section since they do not recover material for reuse.

Contaminants may be introduced into wasted food streams when mixed with other waste streams during collection and/or management. Wasted food streams from sources other than the food processing industry are more likely to contain contaminants.

When wasted food is collected, it can be source-separated (i.e., separated by the generator) and separately collected, or it can be collected as part of a mixed waste stream like MSW. Wasted food may also be mechanically separated from MSW after collection. Generally, source-separated wasted food has less chemical and physical contamination than mechanically separated waste, which has less contamination than mixed wastes such as MSW (Thakali and MacRae 2021; Le Pera et al. 2022a) and can produce a higher quality product (Angouria-Tsorochidou et al. 2023). However, source-separated streams can still contain contaminants. Items like non-compostable packaging or utensils may be placed into the wasted food stream accidentally and may introduce contaminants such as PFAS or plastics (U.S. EPA 2021e; U.S. EPA 2021f). In addition, food waste, even when source-separated from other waste streams, may still be packaged and need to be de-packaged before processing.

The composting and AD pathways typically accept only source-separated (i.e., by the generator) and separately collected wasted food streams, in order to avoid contamination with and from other MSW. These pathways also commonly pre-treat wasted food streams through screening, sorting or mechanical de-packaging to remove physical contaminants like (non-compostable) packaging.

In contrast, the other two pathways – sewer/WWT and controlled combustion – accept wasted food only as part of a mixed waste stream. The sewer/WWT pathway receives and processes wastewater (including food sent down the sink drain) and controlled combustion receives and processes MSW (including wasted food thrown out with the trash).

In the composting, AD (except industry-dedicated digesters), sewer/WWT, and controlled combustion pathways, wasted food is intentionally mixed with other waste streams for processing, which can introduce chemical contaminants. For example, composting requires a mix of nitrogen-rich (e.g., wasted food) and carbon-rich (e.g., wood waste or yard trim) feedstocks, and, except in certain specialized systems, compost cannot be made from wasted food alone. To evaluate the risks from contamination, one must consider the types of waste with which the wasted food is mixed, the contaminants commonly found in those wastes, each pathway's ability to mitigate those contaminants (including possibility of post-treatment), and the specific application of the recovered material. See Table 4-3 for a list of wastes typically managed with wasted food and the common contaminants associated with them.

TABLE 4-3. WASTE STREAMS COMMONLY MANAGED WITH WASTED FOOD

Waste	Potential Contaminants	Pathway that may accept it
Wood waste and yard trim (aka green waste)	Persistent herbicides	Composting
Animal manure (including feces, urine, and animal bedding)	Pathogens Persistent herbicides Pharmaceuticals	AD (on-farm and stand-alone) Composting
Wastewater solids (aka sewage sludge)	Heavy metals Microplastics Pathogens Persistent organic pollutants (including PFAS) Pharmaceuticals	AD (at WRRF) Composting Sewer/WWT
Digestate	(Depends on source of digestate; may include contaminants associated with wastewater solids or manure)	Composting
Biosolids	(Same as wastewater solids above, though biosolids will have by definition received some treatment and be labeled by Class depending on level of pathogen treatment)	Composting
MSW	Heavy metals Pathogens Persistent organic pollutants (including PFAS) Pharmaceuticals Plastic	Controlled Combustion

Table includes both physical and chemical contaminants. Table does not include landfill pathway since materials are not recovered.

Data Sources: (Goss et al. 2013; Urra et al. 2019; Thakali and MacRae 2021; U.S. EPA 2021e; U.S. EPA 2021f)

In general, wasted food streams are less contaminated than wastewater solids or manure (Lu and Xu 2021; O'Connor et al. 2021; O'Connor et al. 2022b); thus, keeping wasted food separate from these wastes enhances purity. Wasted food is most often composted with yard trim and wood chips, given their complementary composition (wasted food is nitrogen-rich, while yard trim and wood chips are carbon-rich) (Dutta et al. 2021); whereas manure and wastewater solids are nitrogen-rich like wasted food. In the AD pathway, feedstocks are dependent on the type of digester. On-farm digesters process manure, while digesters at WRRFs process wastewater solids. Stand-alone AD facilities may process only wasted food or accept other feedstocks.

Each of the waste streams used as feedstocks in the wasted food management pathways brings its own chemical contaminant risks. Chemical contaminants in feedstocks for soil amendments include PFAS and other persistent organic pollutants, heavy metals, persistent herbicides, pathogens, and pharmaceuticals (Goss et al. 2013; Urra et al. 2019; Thakali and MacRae 2021).

PFAS and heavy metals may be present in wastewater solids and biosolids because WRRFs receive wastes that contain traces of products used in daily life from homes, businesses, and industries. WRRFs may also receive higher concentrations of PFAS from industrial wastewaters. A literature review by EPA demonstrated that PFAS contamination was generally higher in biosolids and compost made from biosolids than in compost made from wasted food and green waste (U.S. EPA 2021e). When applied to land, PFAS has the potential to be taken up by plants and crops and/or leach into groundwater; however, a full risk assessment of PFAS-contaminated biosolids or compost applied to land is not yet available. There is no evidence in the literature that any wasted food management pathway destroys PFAS (U.S. EPA 2021e).

Compost and digestate that contain persistent herbicides (e.g., clopyralid, aminocyclopyrachlor, and picloram) can curb growth of certain plants, such as those in the nitrogen-fixing legume family and nightshade family. Documented cases of persistent herbicides in compost have identified green waste, manure or hay as the source (U.S. EPA 2021e).

Pathogens and pharmaceuticals may be found in manure and in wastewater solids and biosolids. By regulation, biosolids have been treated to meet federal standards for pathogen levels. Class A biosolids have undergone a Process to Further Reduce Pathogens (PFRP), whereas Class B biosolids generally contain low levels of pathogens. Pharmaceuticals found in manure and wastewater solids include antibiotics and associated antibiotic-resistant genes (ARGs) and bacteria (ARBs) which can persist into crops treated with contaminated soil amendments (Urra et al. 2019).

Composting is recognized for its ability to destroy many pathogens, suppress plant disease and weed seeds, and decompose many chemical contaminants such as pesticides, pharmaceuticals, and persistent organic pollutants through high temperatures, biological activity, and the maturation process (O'Connor et al. 2021; Thakali and MacRae 2021; Le Pera et al. 2022a; Stehouwer et al. 2022). For example, composting promotes degradation and/or mineralization of organic pollutants by providing nutrients and energy to microbial populations (Urra et al. 2019). Composting also decreases concentration of many pesticides, with the exception of persistent herbicides. Composting can degrade antibiotic residues, and research suggests it may also decrease the bioavailability of remaining residues. Composting may decrease presence of ARB and ARGs (Urra et al. 2019), but additional research is needed. Composting can also reduce the bioavailability of heavy metals by forming chemically stable metallo-humic complexes and aggregates (Goss et al. 2013; Urra et al. 2019; O'Connor et al. 2021; Thakali and MacRae 2021).

Thermophilic (but not mesophilic) AD is able to achieve a similar degree of decontamination as composting for many contaminants (Urra et al. 2019; O'Connor et al. 2021; Thakali and MacRae 2021); however, the majority of AD in the United States is run at mesophilic conditions (U.S. EPA 2023) which can reduce contamination to a lesser degree than thermophilic AD (Urra et al. 2019). According to EPA's recent survey, 17% reporting anaerobic digesters operate at thermophilic conditions (U.S. EPA 2023b). Post-treatment of digestate can also help to remove contaminants (e.g., pasteurization can kill pathogens (Thakali and MacRae 2021)).

The decontamination abilities of composting extend to the composting of digestate and biosolids (Le Pera et al. 2022b; Le Pera et al. 2022a). For example, Class A Exceptional Quality (EQ) biosolids (with less contamination than Class A biosolids) are typically created through composting. Class A EQ biosolids do not have restrictions on their use, due to their high level of treatment and the virtual absence of pathogens and pollutants.

In addition to AD and composting, other treatment methods exist for wastewater solids and affect the quality and purity of the final product. Aerobic digestion (digestion in an oxygen-rich environment), alkaline stabilization (adding lime), and drying (by air-drying, rotary drum, or thermal drying) are a few methods by which wastewater solids may be stabilized to reach Class B or Class A standards. For biosolids to be beneficially used (i.e., as fertilizer or soil amendment), they must not only be Class A or B, but also must not exceed regulatory concentration limits for heavy metals.

Controlled combustion can also destroy many organic contaminants, but not heavy metals (e.g., from MSW), which remain in the waste output (ash) and make it unlikely to be reused. Risk assessments have not yet been done on many of these contaminants in soil amendments for land application. Potential pathways for human exposure to contaminants in soil amendments include uptake in crops and transport in air (as bioaerosol or particulate matter) or into surface or groundwater (Goss et al. 2013).

Figure 4-4 provides a ranking of wasted food pathways based on their alignment with the purity theme of circularity.

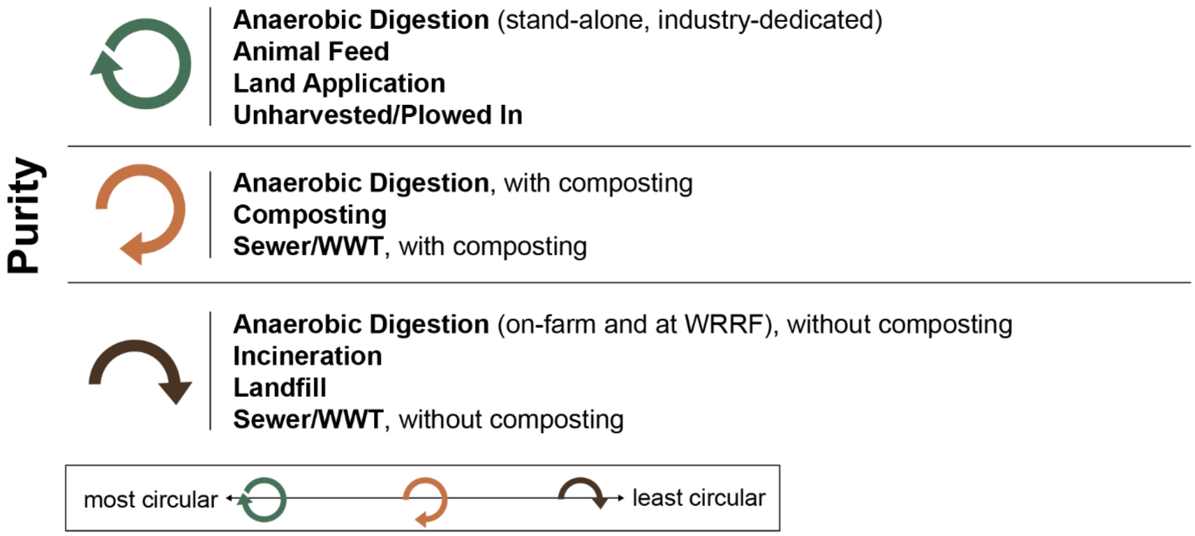

FIGURE 4-4. PURITY: ALIGNMENT OF WASTED FOOD PATHWAYS

Within each tier the pathways are presented in alphabetical order.
Ranking considers typical waste collection practices (e.g., source separation, mechanical separation, or no separation of wasted food streams) and co-waste streams for wasted food in each pathway as well as the decontamination abilities of each pathway. Since source reduction, donation and upcycling minimize the need for waste management strategies, they are not ranked with regard to the other circularity themes (value, purity and regeneration).

4.6 Regeneration

The nutrient-rich materials from the wasted food pathways—compost, biosolids, digestate, unharvested/plowed in crops, and land application—range in composition and structure. While all the materials discussed here provide benefits to soil (O'Connor et al. 2021), some promote regeneration of ecosystems to a greater degree than others. This discussion builds upon that in Section 4.4, where we discussed nutrient (NPK) recovery and the beneficial use (or disposal) of recovered nutrients. It also connects to the discussion of soil health in Chapter 3.6, where the definition of soil health included enhancing water storage, promoting good water quality, maintaining or enhancing soil biodiversity and sequestering carbon, in addition to providing recycled organic fertilizers and soil amendments and avoiding harmful contaminants, pathogens, and pests (Lehmann et al., (2020).

In addition to providing nutrients for immediate use by plants, these nutrient-rich materials can, to varying degrees, contribute to ecosystem regeneration by providing slow-release nutrients and organic matter, improving soil health, increasing water percolation and retention, enhancing and conserving biodiversity, and sequestering carbon.

All the materials derived from wasted food build soil organic matter, which is an important factor in soil fertility (promoting nutrient cycling), soil structure (aggregate stability, bulk density, porosity) and carbon sequestration (Morris et al. 2014). While the benefits of land applying any material will depend on soil type, crop type, timing within the growing season, and climatic conditions (Urra et al. 2019), considering these variances is beyond the scope of this analysis. In addition, soil amendments may be applied to the surface of soils or incorporated into soils (e.g., through tilling or injection) and differentiating effects of these methods is outside the scope of this assessment as well.

Sequestration of carbon results when the rate of carbon addition is higher than the rate of carbon mineralization. Organic amendments add soil carbon directly through application, as well as indirectly through increased plant productivity which increases biomass. Increasing soil carbon is important for climate adaptation (e.g., water holding capacity) and mitigation (prevention of carbon dioxide emissions). Soil organic carbon is also an essential source of fuel for soil microbes to mineralize nutrients necessary for crop growth. Applying carbon-rich amendments can fulfill two goals that are sometimes in tension with each other: building soil carbon stocks for long-term health and sequestration, and increasing microbial activity and carbon mineralization to enhance crop yield (Liptzin et al. 2022). It is important to note that the potential to sequester additional carbon by applying soil amendments is limited by the current health of the soil. Disturbed or degraded soils offer more sequestration potential than healthy soils. For example, greater soil carbon sequestration may be possible on croplands and abandoned mine sites than on untilled rangelands (Gravuer et al. 2019). Similarly, greater soil carbon sequestration is possible on disturbed urban lands like new housing developments and highway right-of-ways than on well-tended yards and landscapes (Brown and Beecher 2019).

While there are differences among the particular wasted food-derived amendments discussed here, all of these amendments demonstrate clear benefits compared to synthetic fertilizers (Brown et al. 2011; Möller 2015; Urra et al. 2019; Ippolito et al. 2021). These amendments also avoid the energy-intensive process and greenhouse gas emissions of producing mineral and synthetic fertilizers (Kraj and Smol 2023).

Compost

The regenerative benefits of compost are well recognized in the literature. Given the amount of research on its regenerative benefits, compost provides a point of comparison for other amendments throughout this analysis.

Amending soil with compost increases soil organic matter, soil microbial biomass, fertility, porosity, aeration, water holding capacity, and earthworm populations, and decreases soil bulk density and risk of erosion. Mature compost is stable (i.e., not undergoing rapid degradation) and significantly contributes to carbon sequestration and long-term carbon storage. For example, a recent study demonstrated that compost amendments to rangelands improved aboveground production by more than 40 percent and belowground carbon content by 50 percent (Kutos et al. 2023).Compost increases organic carbon in soils by directly adding organic matter and also by promoting plant productivity, which adds more organic matter to soils. Compost's positive impact on soil carbon has been proven through studies in a range of agricultural systems and ecosystems and over different time periods (Ippolito et al. 2010; Brown et al.

2011; Ryals et al. 2014; Kutos et al. 2023). A study on one-time compost application to valley and coastal grasslands showed soil organic carbon had increased at the end of the first growing season and after three years (Ryals et al. 2014). Another study showed an increase in soil organic carbon persisted 14 years after a one-time application of biosolids compost to a semi-arid grassland, with higher rates of compost application resulting in higher levels of carbon (Ippolito et al. 2010). In urban soils, compost application has also resulted in carbon storage (Brown et al. 2011). The organic matter in compost is stable and less prone to volatilization or mineralization than uncomposted organic matter, which makes compost highly effective at increasing soil organic carbon (Gilbert et al. 2020).

Compost also provides slow release of nutrients to promote long-term soil health and renewal (Thyagarajan et al. 2014; WRAP 2016; Urra et al. 2019; Grigatti et al. 2020; Lu and Xu 2021; Le Pera et al. 2022a; Stehouwer et al. 2022). Compost's water holding ability can increase an ecosystem's climate resilience, storing water to protect against drought and absorbing water to mitigate flood damage.

On a mass basis, nitrogen is the most prevalent of the macro and micro-nutrients provided by wasted food-derived soil amendments. Compost contains nitrogen primarily in its organic form, which must be mineralized into inorganic nitrogen (i.e., nitrate NO_3- or ammonium NH_4+) before most plants can use it. Roughly 10% of nitrogen will be available to plants upon application of compost, with up to an additional 10% typically mineralizing in the first year (Ozores-Hampton et al. 2022; Stehouwer et al. 2022).

Mature compost is biologically stable, making it suitable for use on lands with high human contact, such as gardens and public green spaces. The relative stability of other amendments depends on post-treatment methods (which may include composting). In addition, compost can be applied at a high rate because of its high C:N ratio – more carbon can therefore be added to the soil before the crop's nitrogen needs are met.

Compost made from food waste (in part) demonstrates mean 15.1 C:N ratio (Stehouwer et al. 2022).

Digestate: Anaerobic Digestion (on-farm and stand-alone)

On-farm and stand-alone digesters produce digestate which is commonly applied to land (AD at WRRFs produce biosolids, discussed in subsequent section). Studies specifically comparing digestate and compost illustrate the differences in the composition and regenerative effects of the two types of soil amendments (WRAP 2016; Nicholson et al. 2017; Case and Jensen 2019; Yan et al. 2023). Since feedstock can affect the composition and effects of amendments, all studies referenced in this section specifically examined digestate and compost made in part from wasted food, unless otherwise noted. Available research suggests that digestate may be less effective than compost in several key areas of regeneration, such as nutrient retention (i.e., keeping nutrients for use by plants and soil microbes), carbon sequestration, and long-term soil health.

Digestate contains primarily inorganic nitrogen (compost is primarily organic N), which is readily available and can be used by plants immediately (Stehouwer et al. 2022). Notably, digestate made in part from wasted food typically has higher ammonium (i.e., inorganic nitrogen) content than digestate made from other common feedstocks like crop residues and animal manure (Dutta et al. 2021; O'Connor et al. 2022a).

Two field studies illustrate the key differences in composition of compost and digestate made from wasted food, demonstrating that compost had higher total nitrogen levels, but lower readily available nitrogen levels, than digestate (WRAP 2016). See Table 4-4 for details.

TABLE 4-4. MEAN NITROGEN COMPOSITION OF WASTED FOOD-DERIVED SOIL AMENDMENTS

Data Source	WRAP 2016		Nicholson et al. 2017	
	Digestate	**Compost**	**Digestate**	**Compost**
Feedstock	Wasted food and manure	Wasted food and green waste	Wasted food	Wasted food and green waste
Total Nitrogen (kg/ton wasted food)	4.67	11.8	7.4	10
Readily available Nitrogen* (kg/ton wasted food)	3.78	0.81	6.3	0.85
Readily available Nitrogen* (% of total N)	81%	7%	86%	8.0%

*Readily Available Nitrogen data from WRAP includes NH_4 and NO_3. RAN data from Nicholson et al. includes only NH_4; however, Nicholson measured NO_3 and found it to be minimal (<0.1) in all cases.

Note: Both studies were led by the same researcher.

A study by Tambone (2010) reinforced these general findings about the composition of compost and digestate, finding samples of digestate made primarily from wasted food (80% organic fraction of MSW and 20% pig slurry) contained 26% to 48% inorganic nitrogen (as a fraction of total nitrogen), while samples of compost made from the organic fraction of MSW and green waste contained only 10% inorganic nitrogen.

These differences in nitrogen content give distinct strengths and weaknesses to each amendment. While both compost and digestate can increase soil nutrients, crop yields, and crop quality over a short time period (WRAP 2016), digestate's greater readily available nitrogen content makes it well-suited to be a substitute for synthetic fertilizers (Tambone et al. 2010; Morris et al. 2014). In addition, the nitrogen-phosphorus-potassium (NPK) ratio of digestate made from wasted food can be similar to synthetic fertilizers (Dutta et al. 2021). In contrast, compost provides long-term reserves of organic nitrogen, improving nutrient retention and soil health over multiple growing seasons (Stehouwer et al. 2022), but, at typical application rates, does not fulfill crops' complete short-term nitrogen needs and is often used in conjunction with fertilizers (Le Pera et al. 2022b).

Yet, digestate's high proportion of inorganic nitrogen also increases the potential for nitrogen losses to the environment (Cooke et al. 2001; Le Pera et al. 2022a; Yan et al. 2023). Nitrogen losses can take three forms – ammonia emissions, nitrous oxide emissions (a GHG with 298x more global warming potential than carbon dioxide over 100-yr period), and nitrate leaching and runoff – all of which are harmful to the environment. These nitrogen losses contribute to acid deposition, formation of particulate matter, climate change, and eutrophication of surface and ground water. Losses can occur during storage and transport or after application of digestate (Le Pera et al. 2022a).

Evidence indicates nitrogen losses are greater from digestate than from compost. Researchers suggest 60% to 70% total nitrogen may be lost in direct land application of digestate made from wasted food given its higher ammonium content (Dutta et al. 2021). Also, nitrogen loss from digestate is more dependent upon intrinsic characteristics of the digestate such as inorganic N content than on any external conditions like soil type (Rigby and Smith 2013).

Two studies measured nitrogen losses to the environment from digestate and compost application. A field study comparing nitrogen losses from digestate and compost under similar conditions at multiple sites demonstrated lower nitrogen losses as ammonia emissions from compost (3.3% mean emissions factor, as % total nitrogen applied) than digestate (38% or 42% mean emissions factor, depending on application method). Cumulative nitrate leaching was also significantly lower for compost than digestate in the study (Nicholson et al. 2017). A laboratory study examined leachate from application of digestate and compost finding that compost leached less nitrogen than digestate over 50 days (Yan et al. 2023). Researchers note that leaching and volatilization of nitrogen during or after application “remains a challenge for…widespread use” of digestate (Yan et al. 2023).

The same two studies were inconclusive on the differences in nitrous oxide emissions from compost and digestate, indicating nitrous oxide emissions are low (less than 1%) for both amendments and, in the case of compost, not significantly different than background values (Nicholson et al. 2017). Since nitrous oxide forms from ammonium or nitrate, greater amounts of inorganic nitrogen can mean greater potential for emissions. In general, nitrous oxide emissions are heavily dependent on climatic conditions, such as temperature, and application rate and timing within the growing season, and emissions can be predicted and controlled based on those factors (Crolla et al. 2013; Lazcano et al. 2016). For example, applying digestate in the fall when soils are cooler and wetter will limit nitrous oxide emissions compared to applying in spring when soils are warmer and better aerated (Crolla et al. 2013).

Digestate typically has lower carbon content (as percentage of dry matter) and a lower C:N ratio than compost, likely limiting its potential (relative to compost) to sequester carbon and improve soil structure and health (O’Connor et al. 2021). Differences in amounts of carbon sequestered through application of compost or digestate may be small and, unless applied to similar sites, would also be dependent on other factors including soil type, climatic conditions, and level of soil degradation. More research is needed in this area.

Anaerobic digestion utilizes 20% to 95% of carbon in feedstocks to make energy (Möller 2015), thus diminishing its carbon stock; more research is needed to determine losses under different operating parameters. The composting process also results in carbon losses, primarily as biogenic carbon dioxide.

One lab study of municipal wasted food sent to a “wet” mesophilic anaerobic digester shows how the carbon in resulting digestate is less than in the wasted food feedstock due to conversion to biogas (Table 4-5).

TABLE 4-5. CARBON CONTENT OF WASTED FOOD AND WASTED FOOD-DERIVED DIGESTATE

	Municipal wasted food	Wasted food digestate
Carbon (as % dry matter)	46.18 ± 0.1	26.98 ± 0.4
Nitrogen (as % dry matter)	3.02 ± 0.11	3.26 ± 0.13
C:N ratio	15.29	8.28

Data source: Paul et al. (2018)

Research suggests that compost can increase soil organic matter more quickly than digestate through its higher carbon content (WRAP 2016). There is evidence that digestate can decrease bulk density and increase water retention capacity and aggregate stability of soil (compared to untreated soil); however, these effects have only been demonstrated over short time frames (Möller 2015; O’Connor et al. 2021). Overall, evidence suggests digestate has greater potential as a fertilizer substitute but lesser potential to provide long-term benefits to soil health relative to compost (O’Connor et al. 2021).

Application rates may also inform the soil health benefits of amendments. Often digestate is applied at a rate determined by nitrogen needs of a crop (Morris et al. 2014), which may not provide as much carbon as a compost application per unit of land, since nitrogen in digestate is more readily available and therefore its application rate is more restricted.

Biosolids: Anaerobic Digestion (at WRRFs) and Sewer/WWT

Biosolids are created from wastewater solids at WRRFs. For the sewer/WWT pathway, wasted food is disposed of down a sink drain, co-mingling with sewage and other wastewater, and travelling through the sewer and WRRF operations (i.e., wastewater treatment stages). For the AD at WRRFs pathway, wasted food is brought directly to the anaerobic digester at a WRRF (typically by truck) to be digested along with wastewater solids. In this case, the wasted food that is co-digested with the wastewater solids does not go through the wastewater treatment stages. In both pathways, wasted food is never the dominant feedstock. It is always a small fraction of the total feedstock, and thus does not drive the characteristics or effects of the biosolids. Wasted food can represent a greater proportion of feedstock in AD at WRRFs than in sewer/WWT.

Biosolids contain primarily organic nitrogen with mineralization rates (conversion from organic nitrogen to a more bioavailable form) dependent on treatment process, type of soil, crop grown, and climatic conditions. Application rates can also impact crop response compared to synthetic N. Slower mineralization rates extend crop response to biosolids N across multiple growing seasons (Brown et al. 2020). Biosolids from both pathways have been proven to increase soil organic matter, aeration, water infiltration and holding capacity, microbial activity, and reduce erosion, compared to a control (i.e., no treatment), but also to cause nitrogen and phosphorus runoff and leaching and nitrous oxide emissions (Thangarajan et al. 2013; Morris et al. 2014; Oladeji et al. 2020; Ippolito et al. 2021).

In general, the nitrogen composition of biosolids (and nitrogen losses from application of biosolids) falls between that of digestate and compost. A laboratory study comparing biosolids, compost, and digestate from varying feedstocks found biosolids (treated through AD, composting, drying/pelletization, and dewatering) generally had lower percentage of nitrogen in inorganic form and lower nitrogen releases than digestate (Case and Jensen 2019). Another study confirmed biosolids have less nitrogen in the form of ammonium than digestate made from wasted food (Dutta et al. 2021).

The regenerative potential of biosolids may vary considerably based upon treatment method (including post-treatment). Treatment method has been demonstrated to significantly affect the nitrogen availability in biosolids, with aerobically digested biosolids > thermally dried biosolids > lime-treated biosolids > mesophilic anaerobically digested biosolids > composted biosolids for readily-available nitrogen as a percentage of total nitrogen (Oladeji et al. 2020). Greater nitrogen availability can lead to greater runoff, leaching, and air emissions, at certain application rates. As with digestate, the rate of biosolids application is usually determined by a crop’s nitrogen needs.

The carbon content of biosolids may also be affected by treatment type. Carbon is lost during AD (converted to biogas) and aerobic processes at WRRFs, reducing the total solids and the total carbon in the final biosolids (Moller 2015; Morris et al. 2014).

Composting Digestate and Biosolids

The regenerative benefits of composting can also be realized by composting digestate and biosolids from the AD and sewer/WWT pathways. There is broad agreement in the literature that composting these outputs can stabilize organic matter, decrease volume (making storage and application less expensive), increase soil health benefits of application, and conserve nitrogen and phosphorus, thus reducing nutrient pollution to air and water (Urra et al. 2019; Grigatti et al. 2020; Lu and Xu 2021; Le Pera et al. 2022a). Due to the stabilization and dilution of nitrogen in the composting process, composts can be applied at a higher rate than other amendments without exceeding a crop’s nitrogen requirements or increasing the risk of nitrogen pollution.

An LCA comparing composting of wasted food to AD of wasted food followed by composting of the wasted food-derived digestate at a plant in Italy showed benefits across most categories, including GHG emissions, for AD followed by composting. The study also confirmed similarity (including nutrient content) of the two compost products (Le Pera et al. 2022a).

A two-year greenhouse study comparing the composition and effect of biosolids and composted biosolids (both from sewer/WWT pathway) applied to clay and sandy loam soils showed composting decreased cumulative nitrate leaching without significant effect on plant growth. The composted biosolids had lower ammonium content (as percentage of total nitrogen content) and higher carbon-to-nitrogen ratio (increasing from 7.5 to 8.9) than uncomposted biosolids. Both amendments were applied at nitrogen-based application rates to ensure adequate nitrogen. Importantly, the researchers suggest that, even in high-rate applications, composted biosolids should not cause “leaching concern” (Oladeji et al. 2020).

While an exhaustive study and comparison of biosolids treatment methods is beyond the scope of this analysis, many examples exist in the literature of composting providing benefits relative to other treatment methods, with regard to energy use (e.g., from thermal drying and ammonia stripping (Case and Jensen 2019; Lu and Xu 2021)) and nitrogen losses (Askri et al. 2016; Case and Jensen 2019).

In EPA's survey of AD facilities processing wasted food, 55% stand-alone digesters, 25% on-farm digesters, and 14% co-digestion systems at WRRFs reported composting as one of the practices used for managing solid digestate or biosolids when asked how they manage their solid digestate (20 stand-alone facilities, 12 farms, and 58 WRRFs responded; multiple answers allowed) (U.S. EPA 2023).

Unharvested/Plowed-In Crops

There is limited research on the regenerative potential of the unharvested/plowed in pathway. However, two other avenues of research can provide valuable insights. First, since parts of unharvested/plowed-in crops are food (i.e., the parts typically sold to consumers), research comparing wasted food feedstocks for composting and AD with finished compost and digestate made from wasted food is relevant. Researchers have found that composted feedstocks have longer-term positive effects on soil than uncomposted feedstocks (Urra et al. 2019). Uncomposted food is less stable than compost, decomposing more quickly and leaving only one-third of organic matter on soil after one year (Ozores-Hampton et al. 2022). Digestate made from wasted food has also been found to be more stable than raw wasted food (Möller 2015); however, raw wasted food would add more carbon to the soil than digestate, since digestion transforms some of the carbon into energy (Möller 2015). Raw wasted food and digestate made from wasted food contain similar amounts of nitrogen but different carbon-to-nitrogen ratios.

Second, similar to the unharvested/plowed in pathway, the agricultural practice of maintaining crop residues (i.e., the part of a plant that is not harvested) and planting cover crops[64] leave plant material in the field unharvested to be tilled or grazed to improve soil health and reduce inputs. Cover crops and crop residues improve the physical, biological, and chemical properties of soil by providing carbon and nutrients to the soil (Blanco-Canqui and Lal 2009; Turmel et al. 2015; Adetunji et al. 2020). These benefits lead to carbon sequestration, moisture retention, reduction in erosion, weed control, and greater soil fertility. However, compared to wheat and legume crop residues, biosolids have been shown to more effectively build soil organic carbon (Wuest and Gollany 2013). It should be noted, because cover crops are grown to improve soil health and not for human consumption, they are predominantly legumes and grasses. These will provide different and potentially greater benefits to soil than food crops which are unharvested/plowed in.

Depending on the unharvested crop type, its effects on soil health may be more similar to raw wasted food or to cover crops. In general, unharvested crops should be expected to provide feed for livestock and carbon and nutrients to soil (though in less stable form than compost, biosolids or digestate), thus providing, at a minimum, short-term benefits to soil health and plant productivity.

Raw Wasted Food: Land Application

The land application pathway is also not well-studied. The types of wasted food from food processing facilities that may be applied to land vary widely, and studies typically focus on one specific waste stream. Regeneration benefits appear to be mixed, depending on the type of waste applied and other ecosystem and agronomic characteristics. There is potential for increased soil carbon, microbial activity, and aggregate stability (Plante and Voroney 1998; Rashid and Voroney 2004) as well as nitrogen pollution (Batuecas et al. 2019), and wasted food application may or may not meet needs for crop growth without additional fertilizer or organic amendments (Rashid and Voroney 2004; San Miguel et al. 2012).

Other Pathways

The animal feed pathway contributes to manure production, and so indirectly provides nutrients and other benefits to soil. As noted, manure may be co-digested with wasted food in the on-farm AD and stand-alone AD pathways, but the majority of manure is managed separately from wasted food. The relative soil health benefits of manure compared to compost, digestate, and biosolids made in part from wasted food are beyond the scope of this report.

The remaining two wasted food management pathways –controlled combustion and landfill –are not regenerative and thus are not discussed in this section. While landfills do sequester carbon, they also emit significant amounts of methane and do not return nutrients or organic carbon to soils.

Regeneration Summary

In summary, the weight of evidence for compost's regenerative abilities is strong. Composting wasted food and composting digestate or biosolids made in part from wasted food provide numerous, well-documented regenerative benefits to soil. The regenerative abilities of biosolids and digestate may be more limited and are heavily dependent on post-treatment for stabilization and nutrient retention. Research suggests that digestate is best used as a short-term nutrient boost akin to a fertilizer, whereas compost is expected to provide longer-term benefits. It is possible that digestate will, over a longer time period, provide broader soil health benefits, but this has not been proven. Biosolids likely fall between the compost and digestate on the spectrum of regeneration. While there is more research and evidence of the regenerative benefits of biosolids than that of digestate, it is also clear that composted biosolids provide greater regenerative benefits than uncomposted biosolids, keeping compost as the "gold standard" for regenerative soil amendments.

In addition, compost, digestate, biosolids, land-applied wasted food, and unharvested/plowed-in crops all add organic matter to soils, which promotes nutrient cycling and enhances soil structure. The physical structure of soils – porosity, aggregate stability, bulk density, etc. – is important for water retention and percolation and for carbon sequestration. Compost contains more organic matter in a less volatile form (partly due to its higher dry matter content), making it better suited to build soil organic matter more quickly.

Because synthetic fertilizers contain nutrients in inorganic forms that are readily available to plants, they provide short-term benefits for crop growth. In comparison, organic nitrogen and phosphorus require mineralization by microbes in order to be available for plant uptake, therefore remaining in the soil for a longer period of time. Synthetic nitrogen and phosphorus fertilizers are more prone to runoff which can impact water quality. The effect of synthetic nitrogen fertilizer on soil organic matter (SOM) varies greatly in the literature, with some studies showing nitrogen fertilizer increasing SOM through increased plant productivity and other studies showing nitrogen fertilizer decreasing SOM by enhancing mineralization (Mahal et al. 2019).

While composting improves the regenerative potential of digestate and biosolids, all three amendments are more regenerative than synthetic fertilizer. For example, a 22-year comparison of uncomposted biosolids versus synthetic fertilizers in semi-arid dryland wheat fallow rotations showed biosolids improved soil health, through indicators such as soil carbon and microbial biomass, while fertilizer had little effect (Ippolito et al. 2021). These amendments also retain more nutrients, allowing less nitrogen to volatilize, leach or runoff into the environment (Sogn et al. 2018; Oladeji et al. 2020). In addition, unlike the most commonly used synthetic fertilizers, soil amendments made from wasted food and/or other plant material, manure, or biosolids can provide additional nutrients beyond nitrogen, phosphorus and potassium, which plants need to thrive (Morris et al. 2014).

Figure 4-5 provides a ranking of wasted food pathways based on their alignment with the regeneration theme of circularity (or by the extent to which they are regenerative).

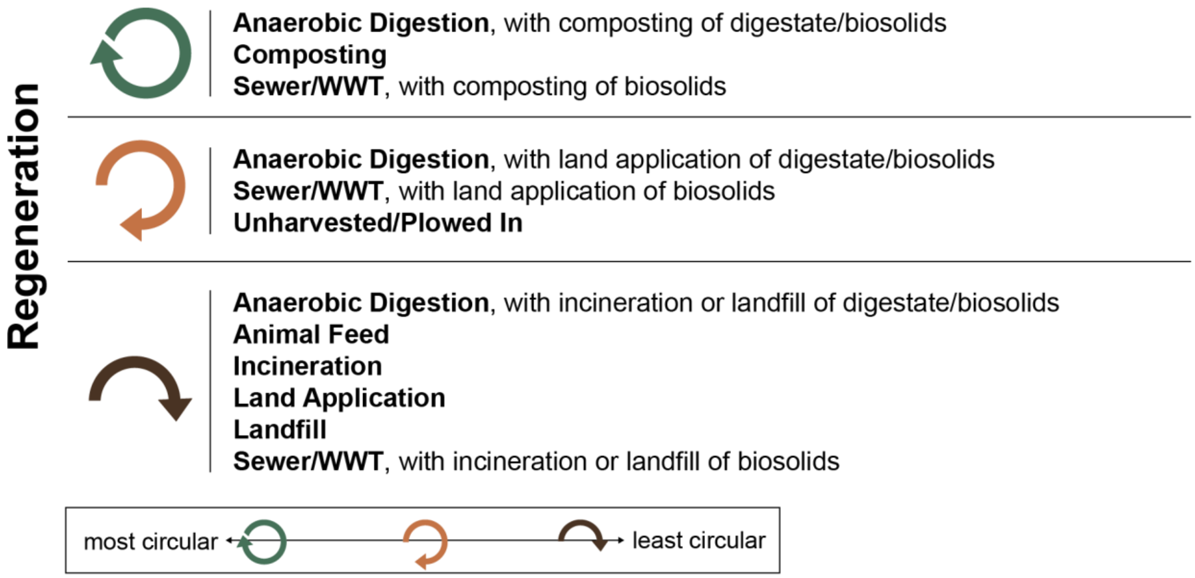

FIGURE 4-5. REGENERATION: ALIGNMENT OF WASTED FOOD PATHWAYS

Within each tier the pathways are presented in alphabetical order.
.Regeneration is defined here as an “approach that has the capacity for self-renewal and resiliency, contributes to soil health, increases water percolation and retention, enhances and conserves biodiversity, and sequesters carbon” (Elevitch et al. 2018).
Since source reduction, donation and upcycling minimize the need for waste management strategies, they are not ranked with regard to the other circularity themes (value, purity and regeneration).

4.7 Alternative Methods for Circularity Assessments

While no studies that quantitatively evaluated the wasted food pathways for circularity were available in the literature, two studies qualitatively examined and ranked wasted food pathways in a circular economy framework. Wang et al. (2021) declared six pathways either “linear” or “circular” based upon a definition of circular economy from Jurgilevitch (2016) which focused on reducing wasted food, utilizing byproducts, and nutrient recycling (Jurgilevich et al. 2016; Wang et al. 2021). Landfill and controlled combustion were categorized as linear, given their waste of nutrients; and source reduction, donation, AD, animal feed, and composting were categorized as circular. No rankings were provided within the two categories.

Tanveer et al. (2021) mapped eight of the wasted food pathways discussed in this report to the R framework that serves as the basis for the United Nations Environmental Programme’s Circularity Framework (Potting et al. 2017; UNEP 2019). The R framework is an expansion of the traditional reduce-reuse-recycle framework that incorporates circular economy strategies. In order of priority, the framework suggests the following circular strategies: refuse, rethink, reduce, reuse, repair, refurbish, remanufacture, repurpose, recycle, and recover. See Table 4-6 for findings.

Several pathways aligned with more than one R, making clear ranking difficult. In general, the authors found the source reduction and donation pathways to be the most circular, followed by animal feed and upcycling, then AD and composting, then controlled combustion, then landfill as least circular. Tanveer et al. (2021) did not consider the land application, sewer/WWT or unharvested/plowed in pathways. For this ranking, the analysis assumes beneficial use of nutrients from AD; without this, AD is ranked as energy recovery (R9), a lower ranking.

TABLE 4-6. MAPPING OF WASTED FOOD PATHWAYS TO THE CIRCULAR STRATEGIES OF THE "R" FRAMEWORK

	Circular "R" strategies, in order of priority, from left to right										
	R0: Refuse	R1: Rethink	R2: Reduce	R3: Reuse	R4: Repair	R5: Refurbish	R6: Remanufacture	R7: Repurpose	R8: Recycle	R9: Recover	Dispose
Source Reduction	✓		✓								
Donation		✓		✓							
Animal Feed				✓				✓			
Upcycling					✓	✓	✓	✓			
Anaerobic Digestion*									✓	✓	
Composting									✓		
Controlled Combustion/ Incineration										✓	
Landfill											✓

Data source: Tanveer et al. (2021)

*If digestate is applied to land as fertilizer, AD is matched with R8: Recycle. If the purpose of AD is only energy recovery, AD is matched with R9: Recover.

R strategies are listed in order of priority from R0 to R9, followed by disposal. Pathway names were converted to those used in this report where definitions were similar. For example, "converted into human food" is noted as upcycling here. The land application, sewer/WWT, and unharvested pathways were not considered in this analysis.

More specific case studies exist in the literature as well. For example, a qualitative assessment comparing use of cereal bran waste for upcycling, biogas production, and animal feed production found that upcycling outperformed the other waste management pathways in terms of system circularity (Grippo et al. 2019). However, these specific findings cannot be translated into more general findings applicable to this analysis.

4.8 Limitations of Circularity Assessment

The circularity assessment both benefits from and is limited by its qualitative nature. A qualitative analysis allows for investigation in areas without standardized data like soil health but limits the quantitative rigor of the analysis. Data gaps in some areas important to circularity exist. While extensive information about many of the pathways is available in the scientific literature, some of the environmental areas of greatest importance to circularity—like soil health and nutrient retention/losses—have not received as much attention as global warming potential or energy production of the pathways. In addition, the science of carbon sequestration in soil is an active and quickly evolving field. Additional work on both the climate mitigation (via carbon sequestration) and climate adaptation (via improved soil health) potential of wasted food pathways will strengthen circularity analyses in the future.

Another evolving area of research is on “nutrient circularity” (Harder et al. 2021). Studies assess flows of nutrients (e.g. nitrogen and phosphorus) in a given region’s food system, considering the imports, exports, and local production and consumption of food and feed crops, live animals, and “organic residuals” (wasted food, crop residuals, manures, human excreta/biosolids) (Harder et al. 2021; Moinard et al. 2021; van der Wiel et al. 2021). Some of the materials considered in nutrient circularity analyses are outside the scope of this report (e.g., manures). However, continued tracking and quantification of nutrients throughout a system and region can add additional nuance to future circularity analyses of wasted food management pathways.

Much recent work has been done on contamination of feedstocks, including wasted food, to the pathways, but still little is known about the risk posed to human health and the environment from application of contaminated soil amendments. This analysis relies on data on the relative levels of key contaminants in various feedstocks to assess purity but could not consider the risks posed by those contaminant levels due to lack of data.

4.9 Circularity Assessment Conclusions

This chapter analyzed each wasted food pathway's potential contribution toward a more circular food system, along four themes:

- *Waste Prevention*: The most circular strategy is to design waste out of processes, minimizing the need to manage waste.
- *Value:* The intended purpose and highest value of food is as human sustenance, including its use to feed livestock. Once food can no longer nourish humans, the next highest value is the recovery and beneficial use of the nutrients most important to agriculture – nitrogen, phosphorus and potassium – and carbon to build soil organic matter. Energy and other materials may also be recovered from wasted food.
- *Purity:* Keeping wasted food free of physical and chemical contaminants allows for greatest beneficial use of recovered materials. Purity is best maintained by collecting and processing wasted food streams separately from other wastes, since many waste steams (e.g., manure and wastewater solids) are more contaminated than wasted food. Also, some pathways can mitigate contamination by destroying or inactivating some contaminants. Food must also be removed from all non-compostable packaging, which can introduce physical and chemical contaminants to the waste stream.
- *Regeneration:* The carbon- and slow-release nutrient-rich outputs of some pathways can help to regenerate ecosystems when applied to land. Improved soil health can help mitigate climate change (e.g., through carbon sequestration) and improve climate resilience of ecosystems (e.g., through soil moisture conservation).

The overlaps and interactions between these four themes can generate additional complexities in comparing wasted food pathways. For example, adding organic carbon to soil fuels two processes that compete for that carbon: building long-term soil carbon stocks and driving nutrient mineralization to increase crop yield (Liptzin et al. 2022). This points to a tension between the value of these organic amendments (in terms of readily available nutrients and plant productivity) and their regenerative effects (in terms of carbon sequestration and nutrient retention).

Figure 4-6 provides a summary of findings from the qualitative assessment. The rankings for the AD and sewer/WWT pathways are dependent upon the use of the digestate/biosolids, and so three scenarios for each pathway (where nutrient outputs are composted, land applied, or disposed of) are displayed.

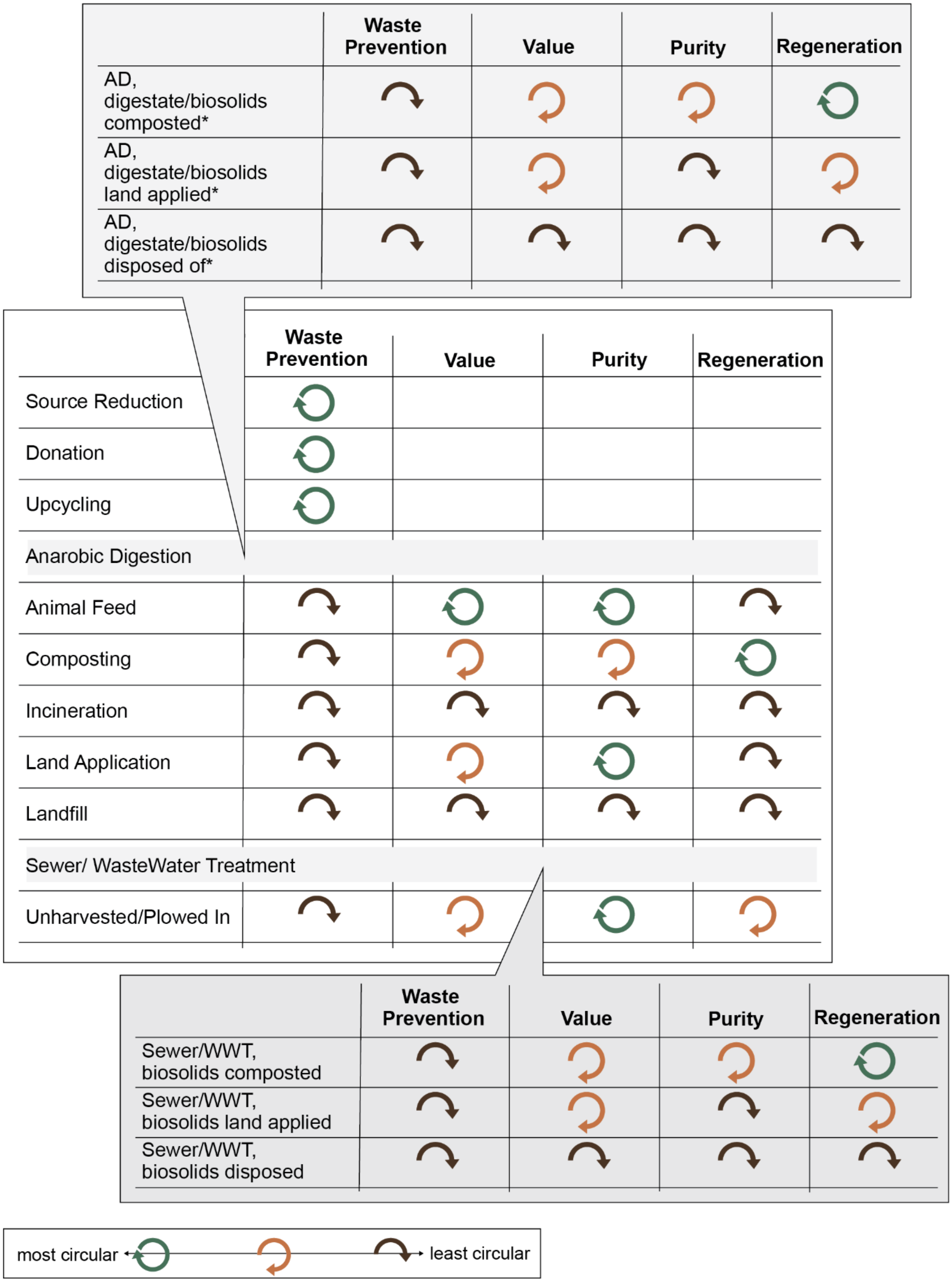

FIGURE 4-6. SUMMARY OF CIRCULARITY THEME RANKINGS, BY PATHWAY

*Stand-alone digesters that only accept wasted food are ranked higher for purity.

Based upon the findings of the qualitative assessment, a ranking of wasted food pathways is presented here, with focus on environmental aspects of circularity. The hierarchy relies on analysis of specific properties of food and how they affect a pathway's potential for circularity.

Figure 4-7 provide rankings of 11 wasted food pathways, by agreement with circularity themes, from most circular to least circular.

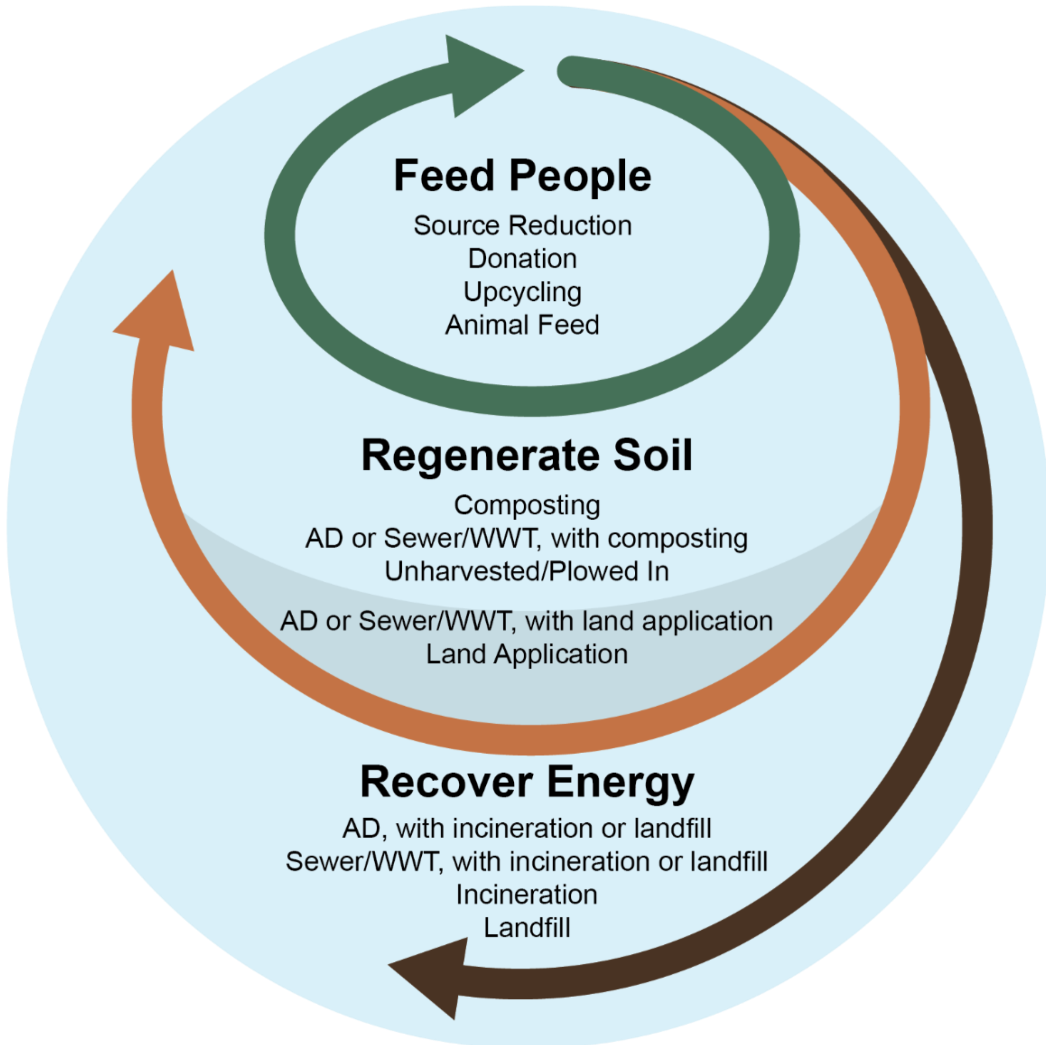

FIGURE 4-7. RANKING OF WASTED FOOD PATHWAYS BASED ON CIRCULARITY

Figure depicts ranking of wasted food pathways, by contribution toward a more circular economy. Analysis considered four circularity themes: (1) preventing waste, (2) maintaining highest value, (3) maintaining purity, and (4) helping to regenerate ecosystems. Limited data were available on land application and unharvested/plowed in pathways.

Key findings of the circularity assessment include:

- **Source Reduction**, **Donation**, and **Upcycling** top the hierarchy since they minimize the need for waste management.
- **Animal Feed** shares the top tier since it maintains the original intended purpose and highest value for food– to nourish humans, albeit indirectly by providing feed to livestock. All four of these pathways share the potential to reduce the amount of food and feed production required to feed people, thus reducing the substantial environmental externalities associated with agriculture.
- **Composting** is the most circular wasted food management option available for wasted food generated anywhere along the supply chain from production to consumers. This includes composting of digestate and biosolids (see next bullet). Composting stabilizes wasted food, providing many regenerative benefits to ecosystems, such as the slow release of nutrients, carbon sequestration, and improved soil health and structure through addition of organic matter. The composting process can also degrade or immobilize many contaminants.
- **Unharvested/Plowed In** ranks as equally circular to composting, though is only available to farmers. If the crops were to be ultimately wasted, then leaving crops in the field for grazing or tilling into the soil returns the nutrients and carbon to the soil and prevents the unnecessary downstream impacts of food processing, packaging, and distribution.
- The circularity of the **Anaerobic Digestion** and **Sewer/WWT** pathways is dependent upon the destination for the nutrient-rich outputs (digestate and biosolids) produced by these pathways.
    - If digestate or biosolids are composted, the pathways rank higher, equal to composting. Compost made from digestate or biosolids shares the beneficial qualities of compost made from wasted food, though more contamination may be introduced in the process.
    - If digestate or biosolids are applied to land (without first being composted), the pathways rank just below composting but above landfill and controlled combustion. Digestate provides many benefits to soil; however, its ability to improve soil health over the long-term to the degree that compost can has not been clearly demonstrated. It can, however, reduce significant environmental externalities by replacing synthetic fertilizer.
    - If digestate or biosolids are disposed of at a landfill or incinerator, the pathway ranks lower, alongside landfill and controlled combustion. Since AD recovers energy, the AD pathway is more circular than sewer/WWT without AD.
- **Land Application** of raw wasted food from the food processing sector ranks alongside AD and sewer/WWT with land application of digestate or biosolids. Land application uses specific streams of food processing waste which may vary widely in composition and regenerative potential; yet the streams are typically free of contaminants since they are collected and managed alone. Additional research is needed on this pathway.
- **Controlled Combustion** and **Landfill** are the least circular wasted food pathways. Both pathways scored consistently in the bottom tier across all four circularity themes. Both can recover energy from wasted food, though less efficiently than AD, and neither result in beneficial use of nutrition nor nutrients from wasted food. Only sewer/WWT can rank beneath these pathways, and only if there is no energy recovery (via anaerobic digestion) and no beneficial use of nutrients (including no effluent used as irrigation).

The results of this analysis align generally with the mapping to the R strategies performed by Tanveer et al. (2021) and presented in the previous section; however, this analysis adds the concept of regeneration which is important to biological systems. This analysis also provides a detailed comparison of the recovered materials, noting differences in the evidence of regenerative abilities of compost and digestate.

CHAPTER 5
DISCUSSION AND CONCLUSIONS

In its *National Recycling Strategy*, EPA emphasized the need to reduce the life cycle impacts of materials (including food) and decouple natural resource use from economic growth by implementing circular economy strategies. In this report, two complementary analyses are presented to evaluate 11 common wasted food pathways with respect to these goals. Both the review of life cycle assessments from the literature (Chapter 3) and the circularity assessment (Chapter 4) were utilized to develop the new EPA ranking of wasted food pathways (called the Wasted Food Scale) presented later in this chapter.

Section 5.1 presents a comparison of the LCA and circularity assessment methodologies and results, and Section 5.2 highlights the effect of wasted food as a feedstock in the pathways. Section 5.3 synthesizes the report's findings into a new wasted food pathways hierarchy based upon environmental impacts and contributions towards a circular economy.

5.1 Comparing Methodologies and Results

This section compares the LCA and circularity assessment methodologies and results, demonstrating the value of combining these complementary approaches.

Life Cycle Assessment

LCA is a well-established methodology that relies upon quantitative data to analyze the cradle-to-grave environmental performance of a process or product. For this report, results from 47 LCAs of the wasted food pathways were reviewed and 11 environmental indicators were considered. At least one LCA was available for each pathway, although for several pathways only one environmental indicator (global warming potential) was assessed.

LCA works particularly well for impacts like GHG emissions, where data are readily available, and results are easily comparable through GWP calculations. A ranking of the pathways based solely on net GHG emissions is presented in Section 3.9. Confidence can be higher for GWP results than for many other indicators, as more practitioners have examined this impact category.

LCA's strengths include identifying trade-offs between environmental considerations for decision-makers and including the environmental benefits of avoided products (i.e., substituting alternative products made through the pathways for conventional products with the same function) (Lindgreen et al. 2021). In the LCAs reviewed here, wasted food pathways received credits for avoiding the production of food, animal feed, energy, and fertilizer through substitution with products made through the pathways, where appropriate. Credits for energy generation by a pathway often included a broad range of environmental benefits, such as reduced GHG emissions, acidification, eutrophication, particulate matter, and toxicity from avoided fossil fuel combustion. Credits for fertilizer production can include reduced energy demand from avoided synthetic nitrogen fertilizer manufacturing and reduced acidification and eutrophication from avoided synthetic nitrogen fertilizer application.

LCA is less valuable in areas where quantitative data does not exist in a standardized format that is easily comparable across pathways. Of the 11 environmental indicators highlighted in this report, only four allowed for meaningful quantitative comparisons across LCA studies – global warming potential, energy demand, acidification, and eutrophication. Data for water consumption and land occupation were available for quantitative comparison across LCA studies but found to be inconsistently modeled. Data for additional categories (human toxicity, ecotoxicity, and particulate matter formation) were suitable for intra-study comparisons.

For one environmental indicator – soil health – specifically relevant to wasted food pathways, standardized quantitative data were not available in the reviewed LCA studies. A key product of many wasted food pathways is a soil amendment to provide organic matter and nutrients to improve soil health and plant productivity (i.e., fertilize) and to provide ecosystem services. There is much published research on the soil health benefits of these amendments, but quantification is not yet standardized, therefore meaningful quantitative comparison across pathways was not possible in the life cycle assessment. However, soil carbon sequestration, which is related to soil health, was considered as part of GWP calculations where sufficient studies were available for applicable pathways (i.e., those that produce a soil amendment).

Circularity Assessment

Circularity prioritizes preventing waste and utilizing outputs for their highest value rather than managing them as wastes. Outputs should be free of contaminants and, in a biological system such as the food system, help to regenerate ecosystems. Wasted food is a valuable output for which many opportunities exist to reuse and recycle the contents. Circularity assessments allow for comparison of competing uses (Sevigne-Itoiz et al. 2021), considering the unique value of food, to retain the highest value possible. For example, circularity values recovery and beneficial use of soil nutrients and aiding ecosystem regeneration over energy recovery from wasted food.

Circularity assessment is a newer field than LCA. Currently no appropriate quantitative metrics exist for assessing the circularity of biological systems. Thus, a qualitative approach was used in this report. The qualitative nature of the circularity assessment in this report allowed for the use of scientific information from primary studies and reviews, without requiring standardization of data. In this way, soil health and ecosystem services benefits from application of soil amendments created from wasted food could be included in the assessment. Insights could also be synthesized from the primary scientific literature about pathways not well-covered by the reviewed LCAs, such as unharvested/plowed in and land application (only one and two LCAs available, respectively), and incorporated.

The circularity assessment can account for the relative performance of pathways, even when external conditions affect that performance. For example, studies show soil type and climatic conditions greatly affect leaching and runoff of nitrogen from soil amendments. However, pathways' relative performance stays constant (e.g., how much leaching might change, but which pathway leaches the most does not). In addition, each LCA typically reflected one set of operating parameters for a pathway, whereas the qualitative circularity assessment could differentiate among choices of operating parameters within a single pathway. For example, the majority of reviewed LCAs assumed beneficial use of nutrient-rich digestate created by AD; however, disposal (e.g., landfill or controlled combustion) of these outputs is also common in the AD pathway.

In contrast to LCA, circularity assessments do not intend to call out every negative externality, and thus the literature suggests decision-makers use them in concert with LCA to ensure environmental performance (Rigamonti and Mancini 2021). While circular strategies save virgin resources, they may not perform better environmentally across all categories, due to energy or water requirements (Lindgreen et al. 2021; Gallo et al. 2023). Thus, existing research supports the need to combine these approaches to inform decisions even when circularity is the primary goal.

It should be noted that neither methodology used in this report considered social and economic factors, which were beyond the scope of this report. Additional tools, such as life cycle sustainability assessment or life cycle costing, could be used in the future to integrate these criteria.

Figure 5-1 displays a comparison of the LCA and circularity methodologies.

Life Cycle Assessment (LCA)

- ✓ Well-established
- ✓ Quantitative
- ✓ Measures overall environmental performance
- ✓ Identifies trade-offs (i.e., burden shifting)
- ✓ Can include avoided impacts
- × Cannot compare studies in areas without standardized quantitative data, like soil health
- × Typically does not consider whether resources are limited or there are competing uses for a resource

Circularity Assessment

- ✓ Newer discipline
- ✓ Qualitative (for biological systems, such as the food system)
- ✓ Measures continued use of resources (i.e., resource cycling) to reduce virgin resource consumption
- ✓ Considers highest value for material use among competing uses, factoring in resource scarcity
- ✓ Can include scientific findings in areas without standardized quantitative data, like soil health
- × Does not guarantee higher environmental performance

FIGURE 5-1. COMPARING LCA AND CIRCULARITY ASSESSMENT METHODOLOGIES

Combining LCA and Circularity Methodologies

LCA and circularity assessment methodologies have been used synergistically in a variety of ways in the literature – from simply considering the results of the two independent analyses together, to using a more formal approach like multi-criteria decision analysis to integrate results, to formally integrating quantitative circularity indicators into LCA (Rigamonti and Mancini 2021; Gallo et al. 2023). LCA can be performed first to limit options under consideration in the circularity analysis by environmental performance; or LCA can be used to assess and compare strategies identified as circular by the circularity assessment. LCA can also be used to identify improvements needed in the environmental performance of desired circularity strategies (Peña et al. 2021).

Regardless of how they are combined or integrated, using both methodologies can reveal trade-offs between resource circularity and environmental impacts and among environmental impacts, helping decision-makers avoid shifting problems from one environmental area to another (e.g., decreasing global warming potential but increasing acidification). Using both quantitative and qualitative methodologies can also provide benefits, allowing for consideration of important environmental criteria where quantitative data are not standardized.

In this report, the LCA review and circularity assessment were independently performed, then the results were considered together to inform the creation of a hierarchy of wasted food pathways. The LCA review enabled comparison of the environmental footprint of circular and linear pathways, and the circularity assessment provided insight into environmental areas not covered by the LCAs, including resource conservation and soil health.

Comparing Results

A summary of the rankings informed by the LCA and circularity assessments and presented in Chapter 3 and Chapter 4, respectively, is provided in Figure 5-2.

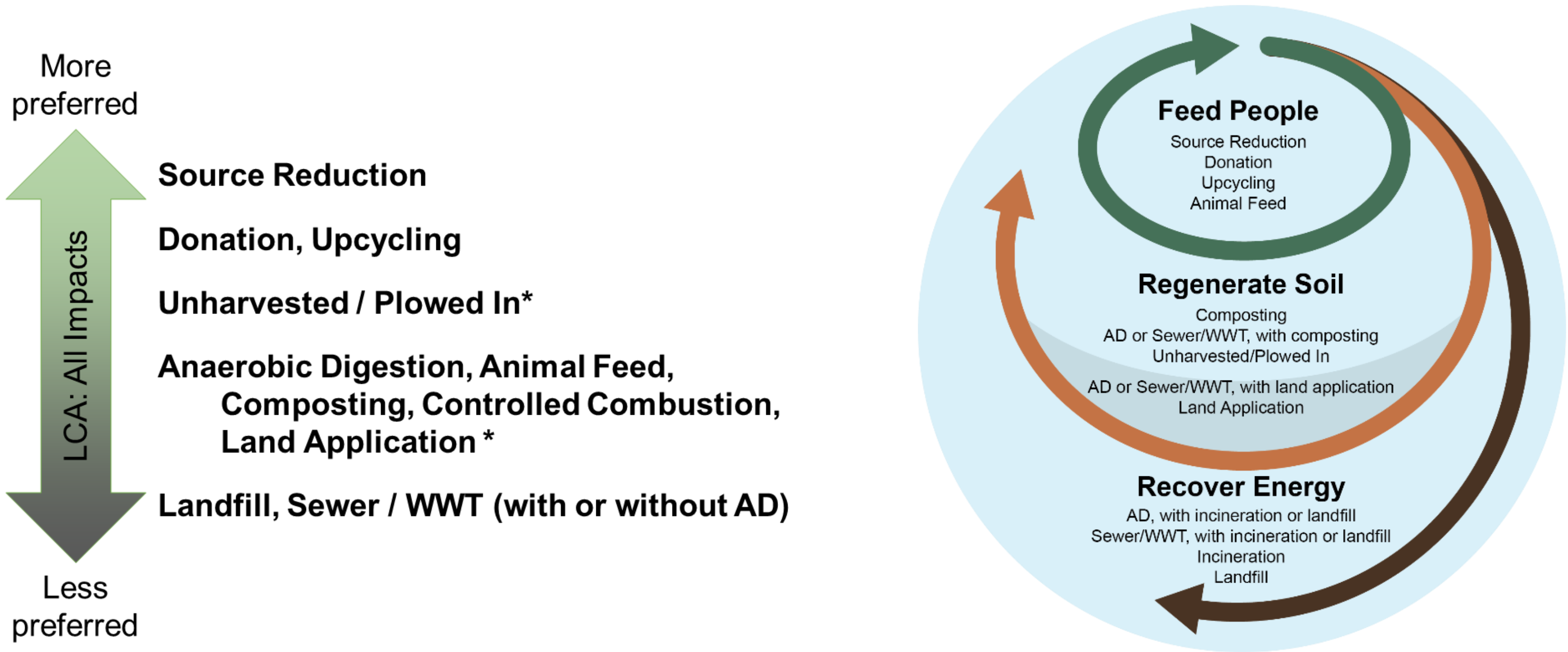

FIGURE 5-2. SUMMARY OF LCA (LEFT) AND CIRCULARITY (RIGHT) ASSESSMENT RANKINGS OF WASTED FOOD PATHWAYS

Pathways listed in tiers from most preferable to least preferable, from top to bottom, in each diagram.
In LCA diagram, pathways within tiers are listed in alphabetical order.
In circularity diagram, pathways within tiers are listed in preferential order.
*denotes limited data

In both methodologies, source reduction, donation, and upcycling achieve the highest rankings. These pathways nourish humans, avoid (or minimize) the waste of food, and have the potential to displace some of the additional food production required to feed a growing population (and consequently avoid the substantial environmental impacts of food production).

At the other end of the spectrum, both methodologies ranked landfills as one of the least preferable options (i.e., in the bottom tier) for wasted food management. Fugitive methane emissions from rapid anaerobic decay of wasted food at landfills, along with other negative environmental effects such as untreated leachate leading to water pollution, demonstrate that landfills are one of the worst environmental performers among the pathways. In addition, landfilling is a textbook example of a linear (i.e., not circular) pathway, as no materials are recovered and recycled. While some energy is recovered from landfills, much of the produced methane from wasted food escapes as fugitive emissions before it can be captured and converted to energy.

For the other seven pathways, results did not align exactly between the circularity analysis and LCA. Key differences between the results of the two methodologies for other pathways arise from (1) how they value energy use and avoided energy use; (2) how they consider emissions and releases during the waste management process; and (3) how they include soil health and ecosystem benefits in the analysis.

Energy Use and Avoided Energy Use

The overall LCA results are driven largely by GHG and energy demand, due to the availability of standardized data for most pathways and clear differences in GHG performance among many of the pathways. Thus, a GHG-only hierarchy is presented in the report as well for transparency (see Figure 3-10). The GHG impacts in the LCAs typically integrated data on (1) emissions during the pathway; (2) avoided emissions due to the use of products (food, feed, energy, and/or soil amendments) created by the pathway; and (3) carbon sequestration potential (including through application of soil amendments, if produced by the pathway).

Beyond source reduction, donation, and upcycling, the GHG rankings were most dependent on two factors: the use and production of energy (i.e., energy demand) and fugitive methane emissions from rapid anaerobic decay of wasted food. While all wasted food pathways use energy to varying degrees, three pathways generate energy from wasted food – AD, controlled combustion and landfill. The sewer/WWT pathway may also produce energy if AD is included. Per unit of wasted food, AD produces more net energy (i.e., subtracting the energy used on-site to run the pathway) than controlled combustion, which produces more net energy than landfill.

Only fugitive methane emissions from the anaerobic decay of wasted food outweigh the global warming potential impacts of energy use and avoided energy use in the wasted food pathways. Sewer/WWT and landfill ranked most damaging of all pathways for global warming potential due to fugitive methane emissions from sewer transport to WRRFs and fugitive methane emissions released before a landfill gas collection system is installed, respectively. Where anaerobic decay was not a factor and energy was produced from wasted food, beneficial GWP performance is often observed (e.g., AD and controlled combustion). Data on landfills was robust, whereas data on sewer/WWT is still limited.

In the LCA analysis, more than half of the other environmental criteria (including all four criteria where meaningful quantitative comparisons were possible) were impacted by the use of energy and the production of net energy and its assumed displacement of traditional fossil fuel energy sources. In some cases, energy use was the primary driver of emissions in wasted food pathways. Energy demand is a direct measure of energy use and/or net energy generated by pathways, and energy use (or avoided energy use) contributes to (or reduces) acidification, eutrophication, particulate matter, human and eco-toxicity, and water consumption. This makes assumptions about displaced energy supply very influential on final LCA results. Parsing out the effects not dependent upon energy will be important to understanding the impacts of these pathways as the sources of energy change in the United States.

In contrast, the circularity assessment valued material recovery over energy recovery, and results were less dependent on assumptions about the current energy mix. Circularity prioritizes pathways that recover and beneficially use materials to regenerate ecosystems through carbon sequestration, soil health improvements, and addition of slow-release nutrients. The circularity assessment did not, however, consider the associated environmental impacts and benefits of energy use and net production, respectively.

Process Emissions and Waste Outputs

The LCA also identified key process emissions from the wasted food pathways beyond those related to energy use. As noted earlier, sewer/WWT and landfill emit methane from the anaerobic decay of wasted food. In addition, composting emits ammonia and volatile organic compounds (VOCs); controlled combustion generates flue gas containing particulate matter and dioxins; landfill may release untreated leachate causing toxicity; and sewer/WWT (and AD at WRRFs) results in the release of effluent which may cause eutrophication. The circularity assessment did not comprehensively examine emissions during the wasted food pathway, from energy use or other processes.

Several pathways can also result in waste outputs that must be sent to landfill or controlled combustion. For example, ash from controlled combustion is typically landfilled and may contain hazardous components, and digestate and biosolids from AD and sewer/WWT may be disposed of in a landfill. Some of the reviewed LCAs consider the effect of waste from controlled combustion in terms of toxicity. Stack emissions, impurities in the wasted food stream, and energy consumption or avoided energy production have all been cited as drivers of toxicity impact (Mondello et al. 2017; Mayer et al. 2020; Albizzati et al. 2021b). Studies of AD assumed land application of digestate or biosolids (i.e., no creation of waste). In the circularity assessment, pathways were credited for beneficial use of outputs over their disposal, but the environmental impacts of disposal were not explicitly considered. Pathways with waste outputs (i.e., AD or sewer/WWT where digestate or biosolids are sent to landfill or controlled combustion; or sending wasted food directly to landfill or controlled combustion) rank lower than other pathways in the circularity assessment since resources do not maintain value.

Soil Health

Five wasted food pathways offer the potential to provide carbon and nutrients to soil and improve soil structure and health – AD, composting, land application, sewer/WWT, and unharvested/plowed in. Unlike the reviewed LCA studies, circularity assigned value to improvements in soil health and ecosystems from the application of wasted food or soil amendments made from wasted food. The LCA review’s dependence on standardized quantitative data limited its abilities in this area, and none of the LCAs reviewed considered soil health or ecosystem regeneration quantitatively.

In LCAs, AD and composting were sometimes credited with replacing manufacture and use of synthetic fertilizers or peat and/or sequestering carbon by returning wasted food to the soil. When applied, these credits have a moderate effect on GWP, with their application potentially raising composting to net negative (beneficial) effect. The sewer/WWT pathway received these credits in two of seven of the reviewed LCA scenarios.

The circularity assessment was able to incorporate more detailed findings from the scientific literature about the potential benefits to soil health and ecosystems from these pathways. In addition, the circularity assessment considered scenarios where nutrient-rich outputs are disposed of, where fertilizer and carbon sequestration credits should not be applied in LCA.

All the types of soil amendments created from wasted food are better for soil health than synthetic fertilizers, and all result in less nitrogen loss through leaching, runoff, and volatilization, than synthetic fertilizers (Brown et al. 2011; Morris et al. 2014; Möller 2015; Urra et al. 2019; Ippolito et al. 2021). This is primarily due to the organic matter content in these amendments – something synthetic fertilizers lack. Soil organic matter is a widely accepted indicator of soil health as it affects the biological, chemical, and physical characteristics and ability of soils to provide ecosystem services and support the growth of food, fiber, and fuel (Urra et al. 2019; Witzgall et al. 2021; Kopittke et al. 2022; Liptzin et al. 2022). While synthetic fertilizers can help to increase soil organic matter by promoting crop growth, studies have also shown that synthetic fertilizers have negative effects on soil organic matter (Mahal et al. 2019). Application of amendments derived from wasted food builds soil organic matter, which decreases bulk density, increases porosity, water retention and infiltration, promotes nutrient cycling and retention, and stabilizes soil aggregates, reducing the risk of erosion. LCAs demonstrate that the use of organic amendments can reduce global warming potential, acidification, and eutrophication relative to synthetic fertilizer use. Among the soil amendment types, the composition and degree of benefits vary, with digestate best able to substitute for synthetic fertilizers and compost best able to build long-term soil health, regenerate ecosystems, and release nutrients slowly over time. Leaching and runoff of nitrogen appears to be greater from digestate than compost. Compost can also be less contaminated than digestate or biosolids (depending on feedstock) due to composting's greater ability to degrade or decrease the bioavailability of many contaminants (Thakali and MacRae 2021).

5.2 Effects of Wasted Food on Pathway Performance

The findings in this report relate specifically to the management of wasted food in the 11 pathways. Where possible, the analysis considers the specific properties of food and how they affect a pathway's impacts and benefits. In many pathways food is not the only feedstock and, in some pathways, like sewer/WWT, wasted food is never a major feedstock. Wasted food is an ideal feedstock in some pathways, reinforcing benefits offered by the pathway; while in other cases it might contribute directly to (or increase) the environmental impacts of a pathway.

For example, energy may be recovered from wasted food through three pathways, at varying efficiencies. Wasted food is well-suited for AD (due to its moisture content and energy value) and its use with other feedstocks (e.g., manure or biosolids) provides synergies allowing for more energy generation overall. In contrast, the moisture of wasted food makes it a poor feedstock for energy generation by controlled combustion; it may require drying (which uses additional energy), and it produces less energy through controlled combustion than some other components of MSW. At a landfill, much of the methane produced by wasted food occurs prior to installation of landfill gas capture systems, leading to GHG emissions rather than recovered energy.

Wasted food is unique among MSW components in that it is nitrogen-rich, making it valuable for plant productivity, and carbon-rich, making it advantageous for soil health and providing an opportunity for soil carbon sequestration when process outputs are used as a soil amendment. However, the nitrogen content of wasted food can also lead to environmental impacts. For instance, adding wasted food to sewer/WWT pathway increases energy use during secondary treatment (due to increased biological and chemical oxygen demand from addition of nutrients) and can increase nutrients released in effluent. Also, digestate from wasted food is higher in ammonium than other digestates, which can lead to greater leaching and runoff of nitrogen when digestate is applied to land.

In many cases, more value can be generated by source-separating wasted food before management. For example, the AD and composting pathways typically require source separation to remove physical contaminants. Wasted food streams tend to contain fewer pathogens and chemical contaminants like PFAS or heavy metals than other common feedstocks (e.g., manure or wastewater solids) for the wasted food pathways. This reduces opportunities for toxicity in pathway outputs, expanding potential uses for outputs such as soil amendments. In other cases, like controlled combustion, where wasted food is not an ideal feedstock, wasted food will likely only be managed through this pathway inside a broader waste stream like MSW. For that reason, decision-makers should consider the broader impacts of controlled combustion when determining whether to choose it as a wasted food pathway. While wasted food contains fewer contaminants than some other MSW feedstocks, the combustion of wasted food can lead to dioxins in flue gas.

5.3 Synthesizing Results into New Ranking of Wasted Food Pathways

The two approaches presented in this report – LCA and circularity assessment – are complementary and together inform the conclusions of this report and the Wasted Food Scale presented in Figure 5-3. EPA's Wasted Food Scale ranks wasted food pathways from most to least environmentally preferable. The Scale is based upon each pathway's environmental performance and contribution to a circular economy and does not consider economic and social factors. Pathways were evaluated based upon the use of food as a key feedstock wherever possible. The Wasted Food Scale replaces EPA's previous ranking of wasted food pathways (the Food Recovery Hierarchy).

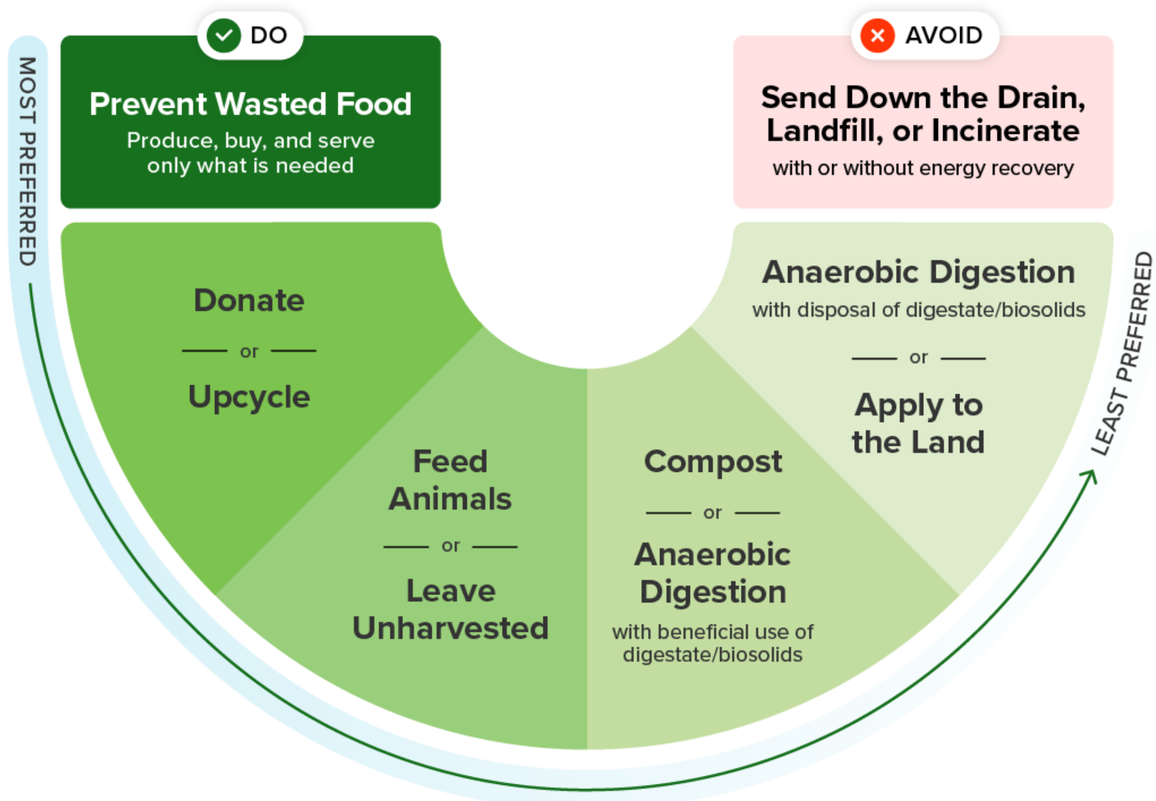

FIGURE 5-3. THE WASTED FOOD SCALE: EPA'S NEW RANKING OF WASTED FOOD PATHWAYS BASED UPON LCA AND CIRCULARITY ASSESSMENT

This scale of the wasted food pathways is divided into tiers, from most preferable (#1 below) to least preferable (#6 below), based upon the life cycle environmental impacts and contributions toward a circular economy of each pathway. Pathways are grouped within a tier where this report found levels of environmental impact and circularity to be similar.

1. *Source Reduction:* Source reduction demonstrates environmental benefits an order of magnitude greater than all other pathways (except donation in some cases) by reducing the amount of additional food that must be produced. The benefits of most other pathways are small relative to the environmental impacts of food production; thus, they can do little to offset the environmental impacts of food production. Source reduction can be achieved at any stage of the supply chain, and source reduction options should be exhausted before other pathways are considered.

2. *Donation and Upcycling:* Among the pathways considered, donation and upcycling exhibit benefits closest to those of source reduction. Like source reduction, donation and upcycling can reduce the amount of additional food that must be produced and, because of this, demonstrate beneficial performance across all impacts studied. However, unlike source reduction, these two pathways require additional energy use (e.g., for transportation, cold storage, or processing), reducing net environmental benefits (i.e., by increasing energy demand, and thus particulate matter, acidification, eutrophication, water consumption, and human and eco-toxicity if fossil fuels are used). In addition, researchers estimate that up to 40 percent of donated food may be wasted (Alexander and Smaje 2008), further reducing benefits of donation relative to source reduction. An estimate of food waste in upcycling pathway is not available in the literature.

3. *Animal Feed and Unharvested/Plowed In:* Using wasted food as animal feed can displace the production of traditional animal feed (e.g., growing soy, corn or barley) which brings substantial environmental savings. The impacts of recycling wasted food into animal feed are largely due to energy use (e.g., heat treatment to meet safety standards). If crops would ultimately be wasted (i.e., not consumed), the unharvested/plowed in pathway provides an opportunity to return nutrients and carbon directly to the soil and avoid environmental impacts of processing, packaging and distributing food.

4. *Composting and Anaerobic Digestion with beneficial use of digestate/biosolids:* Both composting and AD produce soil amendments with potential to improve soil health, sequester carbon, and displace some use of synthetic nitrogen fertilizers. Compost, digestate and biosolids (made through co-digestion) require less energy to make than synthetic fertilizers and, upon application, release less nitrogen to the environment through air emissions, runoff and leaching. While compost is typically not applied at rates which would allow complete substitution of synthetic fertilizers, it can reduce the amount of synthetic fertilizer needed and provide long-term slow release of nitrogen.

   While the level of evidence for long-term soil health benefits, including improved biodiversity, water holding capacity, and protection against erosion, is strongest for compost, AD can create both energy and soil amendments, demonstrating the “cascading uses” principle of circularity, hence we have ranked them equally in the scale. (Note: Wasted food sent down the drain and through the sewer to a WRRF that has AD is defined as the sewer/WWT pathway, not the AD pathway, and is ranked separately.)

5. *Land Application and Anaerobic Digestion with disposal of digestate/biosolids:* When the nutrient-rich output of AD (i.e., digestate or biosolids) is disposed (typically by landfill) rather than beneficially used, the AD pathway ranks lower. All the LCAs of AD reviewed in this paper assumed beneficial use of the digestate or biosolids, thus their findings reflect beneficial use rather than disposal of the digestate or biosolids. When digestate and biosolids are disposed we must subtract those benefits and add the environmental impacts of landfilling.

   AD can create more net energy from wasted food than the other energy-producing pathways, thus it is ranked higher than controlled combustion and landfill; however, unlike some other renewable energy resources (e.g., solar or wind), combustion during AD generates criteria air pollutant emissions similar to natural gas combustion.

The benefits and impacts of land application of food processing waste streams (i.e., raw wasted food) may vary widely based upon the composition of the wasted food and more research is needed. In many cases, more benefits could be achieved by managing the waste stream through other pathways such as AD with beneficial use of digestate.

6. *Controlled Combustion, Landfill, and Sewer/WWT:* Sewer/WWT and landfill stand out for their sizeable methane emissions. Wasted food decays rapidly in anaerobic conditions. Sewer transport allows for uncontrolled methane releases and wasted food at landfills begins to decompose prior to capping and placement of landfill gas capture systems. These emissions far exceed the benefits of energy recovery technologies (i.e., AD at WRRFs or gas capture systems at landfills) or the carbon sequestration potential of landfills. WRRFs with AD have lower global warming potential than those without, but the difference is not big enough to affect the final ranking of the sewer/WWT pathway. Sewer/WWT and landfill have other environmental impacts as well, including eutrophication from effluent releases and water pollution from untreated leachate, respectively. While sewer/WWT offers the possibility of nutrient recovery, biosolids are currently disposed of roughly half the time. Given that wasted food sent down the drain results in methane emissions from sewer transport and increased energy use during secondary treatment at the WRRF, it is probably more environmentally beneficial to source-separate wasted food and collect and transport it directly to anaerobic digesters rather than to send it down the drain to a WRRF with AD.

   While controlled combustion performs better than landfill and sewer/WWT across most categories in the LCA review, it fails to contribute to circularity beyond modest energy production. The moisture content of wasted food makes it a poor feedstock for energy generation by controlled combustion; it may require drying (which uses additional energy), and it produces less energy through controlled combustion than some other components of MSW. Research in multiple studies indicates that removal of wasted food from MSW streams before incineration can reduce energy use, increase calorific value, and improve efficiency of controlled combustion (Song et al. 2013; He et al. 2014; Rajagopal et al. 2014; Erses Yay 2015; Lee et al. 2020). Controlled combustion also generates hazardous waste (fly ash) and can create localized air pollution impacts.

Key Conclusions

While the performance gap between the most and least preferable pathways – source reduction, donation, and upcycling compared to incineration, landfill, and sewer/WWT – is considerable, the differences in environmental performance among the middle-performing pathways are much smaller. For example, in the LCA review the median performance of all six of the remaining pathways (AD, animal feed, composting, controlled combustion, land application, and unharvested/plowed in) are beneficial or near neutral for global warming potential. While all pathways use energy and release CO_2, and some pathways release other GHGs (CH_4 and N_2O in composting and CH_4 in AD), this is generally offset by the benefits of avoided production of energy, food, animal feed and/or fertilizer, avoided fertilizer use, and/or carbon sequestration. However, these five pathways each exhibit one or more important tradeoffs across environmental impact categories and/or between circularity and environmental performance. In the circularity assessment, a similar pattern emerged, with many of the same pathways rising to rank highest (source reduction, donation, upcycling and animal feed) for maintaining food's highest value and lowest due to it recovering little value from wasted food (landfill and incineration), with smaller differences in the pathways in between.

As the U.S. becomes less dependent on fossil fuels for energy, the environmental value of producing energy from wasted food will decrease. The LCA results are driven largely by global warming potential and energy demand, due to widely available data and, in the case of energy demand, the effect of energy use effect on many of the selected indicators. Energy use by pathways contributes to more than half of the indicators assessed in the LCA chapter of this report, including energy demand, acidification, eutrophication, particulate matter, and human and eco-toxicity performance. Thus, the energy mix assumed (when calculating effect of energy use and/or avoided energy production) has a substantial effect on the outcome of LCA studies. As the U.S. increases the use of lower-impact energy sources in its portfolio, the environmental "cost" of energy use and the environmental "benefit" of avoiding energy production will both decrease. This would likely, for instance, lower the environmental performance of controlled combustion and AD but improve the environmental performance of composting. For example, if energy produced by AD was displacing wind power, rather than fossil fuel-derived power, the pathway would be net positive (increase GHG emissions), rather than net negative (reduce GHG emissions), for global warming potential (Slorach et al. 2019a).

The benefits of most pathways are small relative to the impacts of food production; thus, they can do little to offset the original impact. For example, anaerobic digestion, the most efficient energy recovery pathway for wasted food, recovers only around 20 percent of the energy required to produce each unit of food.

The environmental performance of individual pathways is highly variable and is affected by operating parameters, such as technology type or temperature and pre- and post-treatment of outputs. For example, composting is well-studied but inconsistently modeled by LCA, and process emissions are variable due to composting method, facility management, and environmental conditions. Also, for many pathways, pre- and post-treatment can add substantial impacts which may or may not be included in modeling. For example, a study of de-packaging showed its energy use was greater than that of collection and transportation for AD and composting pathways (Mondello et al. 2017). Additionally, digestate and biosolids from AD and sewer/WWT pathways often require post-treatment, which can be energy-intensive, before they can be beneficially used.

Recycling wasted food into soil amendments offers opportunities to make long-term improvements in soil health and help regenerate ecosystems by recovering nitrogen and carbon and returning them to the soil. When food cannot be used to nourish humans, the composting pathway, including composting of digestate and biosolids, offers the greatest benefits in this area. The carbon in wasted food can be used to make long-term improvements in soil structure and health or to generate energy. Application of soil amendments derived from wasted food builds soil organic matter, which decreases bulk density, increases porosity, water retention and infiltration, promotes nutrient cycling and retention, and stabilizes soil aggregates, reducing the risk of erosion. These improvements are regenerative and can help to improve the resilience of agricultural systems in the face of climate change. According to the IPCC, soil moisture conservation, reclamation of degraded soils, and soil carbon management are effective practices for climate change mitigation and adaptation (IPCC 2023). Synthetic fertilizers lack the organic matter content of these amendments and cannot provide similar benefits, other than indirectly through promoting crop growth. In addition, the nitrogen in wasted food can be used to provide nutrients to crops over the short or long-term in lieu of synthetic fertilizers. The application of organic amendments made from wasted food (compost, digestate, or biosolids) lose less nitrogen through volatilization, leaching, or runoff than synthetic fertilizers, reducing acidification, eutrophication, particulate matter emissions, and global warming potential.

The circularity and environmental performance of wasted food pathways are generally positively correlated. While circularity does not guarantee environmental performance, for the wasted food pathways, circularity and environmental performance are not in conflict, and progress can be made on both fronts with similar pathways. When the LCA review considered all 11 environmental indicators presented in this report (including soil health), the results are not wildly different than those of the circularity assessment. The one exception is the sewer/WWT pathway. While this pathway can contribute to a circular economy by recovering and beneficially using nutrients in biosolids, that potential must be balanced by consideration of the magnitude of sewer methane emissions.

Emerging pathways including hydrothermal carbonization (HTC); pyrolysis, torrefaction and gasification; and rendering were excluded from the main LCA and circularity analyses due to a lack of available environmental data. The pathway technologies and key features of each are introduced in Appendix E. All the considered emerging pathways have the potential to produce high value end products including electricity, solid and liquid biofuels, chemical feedstock, and feed and fertilizer products. Each end product has the potential to displace traditional sources of these materials and the net environmental performance of these pathways depends strongly on how they perform relative to the avoided product options. For example, HTC produces a coal-like solid fuel that has been shown to generate net environmental benefits in the acidification and GWP impact categories when displacing coal (Berge et al. 2015). The authors note that if other energy mixes are being displaced, a consideration of environmental priorities is necessary. As noted elsewhere in this report, such considerations are likely to be of increasing importance as electricity grid mixes move towards reliance on cleaner fuels. Still, any pathway with the potential to convert a low value waste stream into a high value product that can reduce reliance on fossil resources holds environmental promise. Given the complexity of these processes, their range of potential operating conditions and pre- and post-treatment options, more research is needed to understand the key factors driving environmental impact, product yield, and process economics such that suitable use cases can be identified.

CHAPTER 6
IMPLICATIONS AND RESEARCH GAPS

This chapter provides context for the conclusions presented in the previous chapter – comparing the new Wasted Food Scale to other wasted food hierarchies, suggesting ways to use the new scale and implications of the scale, and noting priority research needs to better inform decision-makers and policymakers about the 11 wasted food pathways.

6.1 Comparing Conclusions to Other Hierarchies

The Wasted Food Scale presented in this report is based upon the latest science regarding the wasted food pathways' life cycle environmental impacts and contributions toward a circular economy. A comparison of the scale to the previous EPA ranking (i.e., the Food Recovery Hierarchy) and to two international hierarchies is presented in Figure 6-1. These hierarchies may be based on different criteria than this hierarchy.

Previous U.S. EPA Wasted Food Pathway Ranking

The new Wasted Food Scale agrees with the previous EPA ranking (i.e., the Food Recovery Hierarchy) developed in the 1990s in several ways. The scale confirms the position of source reduction, donation, and animal feed as the most desirable pathways. It also confirms the position of landfills in the bottom tier of the hierarchy. Differences between the Wasted Food Scale and the previous EPA ranking include:

- *Pathways included:* The Wasted Food Scale considers four additional common wasted food pathways— land application, sewer/WWT, unharvested/plowed in, and upcycling—but removes the broad pathway of "industrial uses" since there is wide variation in the environmental performance of wasted food pathways that could fall under this umbrella. Anaerobic digestion, now the most common industrial use, is considered on its own in the scale. Since the time of the previous EPA ranking's development, many new pathways which could be considered "industrial uses" have emerged. Information on some of these emerging pathways can be found in Appendix E.
- *Value of end products:* The Wasted Food Scale moves composting to the same tier as anaerobic digestion due to the numerous well-researched benefits of applying compost, including providing slow-release nutrients, improving soil health, sequestering carbon, and contributing to regeneration of disturbed ecosystems. The scale also distinguishes within pathways based upon the end use (land application or disposal) of the products created, such as digestate or biosolids.

International Wasted Food Hierarchies

The two international hierarchies most cited in the literature are from the United Kingdom (UK) and the European Commission's (EC) Bioeconomy Knowledge Center (EC 2018; DEFRA 2021). Like the scale presented in this report, both prioritize source reduction, donation, and animal feed above other pathways. They also rank sewer/WWT and landfill in the bottom tier. However, they both rank controlled combustion with energy recovery higher than sewer or landfill, rather than including it in the same tier as the hierarchy in this report does.

The international hierarchies also both rank AD and composting in the same tier, like the scale presented in this report but unlike the previous EPA ranking. The EC framed the two pathways as "nutrient recovery" (which ranked higher than "energy recovery" in the hierarchy) so it appears the EC assumes that digestate or biosolids would be beneficially used and not disposed of. This is another approach to the distinguishing of AD with and without beneficial use of digestate/biosolids, akin to the presentation in this report's hierarchy.

Only the UK hierarchy includes land application, and it is ranked below AD and composting and above controlled combustion, landfill, or sewer, consistent with the findings of this report. While this report ranked upcycling, both the UK and EC included what we define in this report as upcycling (i.e., making food products for humans from wasted food) as part of a broader category, making comparison with its ranking in this report's hierarchy difficult. The EC ranks a "valorization" pathway, and the UK ranks a "biomaterials" pathway, both of which include the creation of food and non-food products like chemicals.

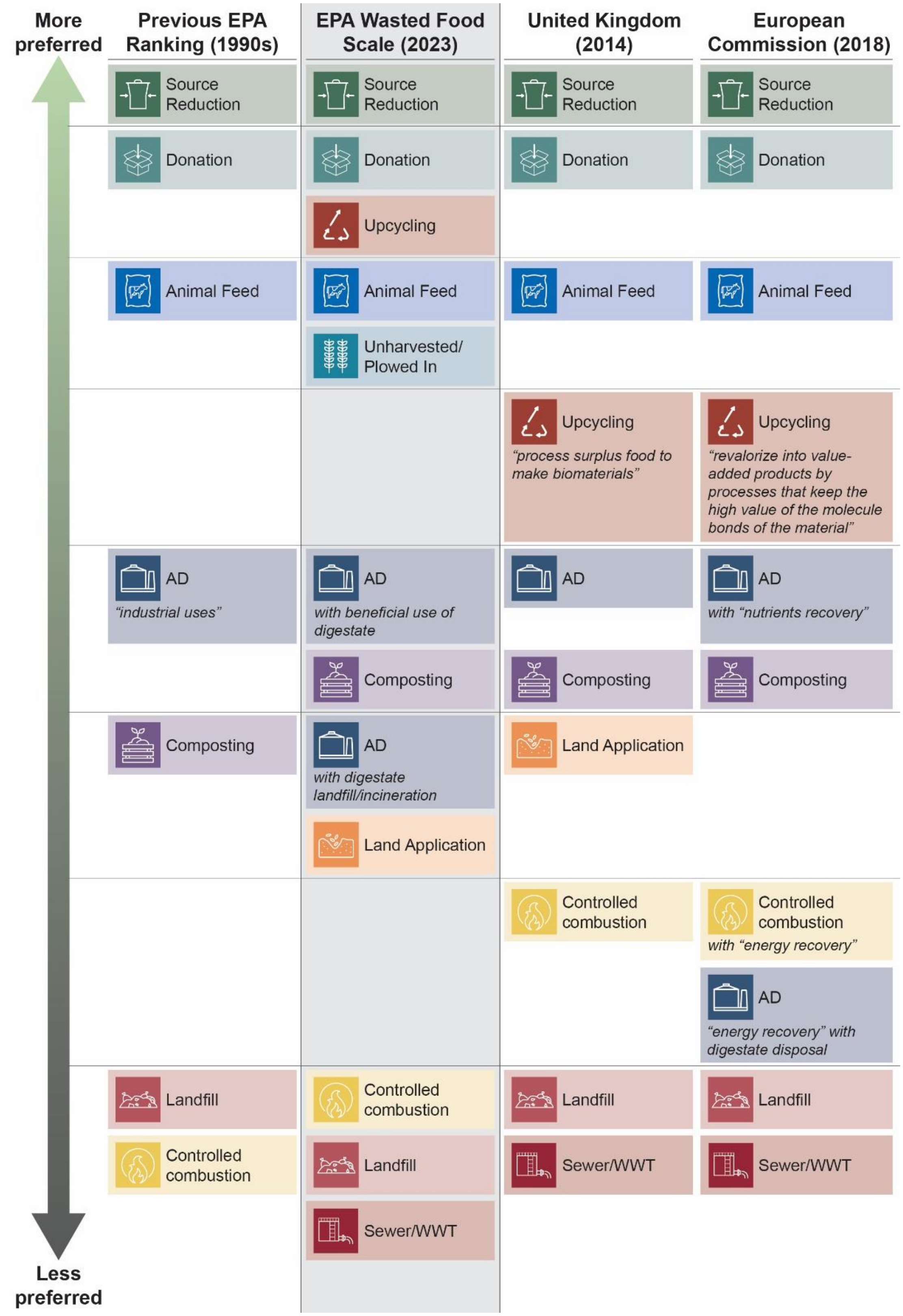

FIGURE 6-1. COMPARISON OF WASTED FOOD PATHWAY HIERARCHIES

Pathways shown in the same box in a column were ranked equally.
Data sources: UK DEFRA (2021); European Commission Bioeconomy Knowledge Center (2018 EC 2018)

6.2 Using the Hierarchy

The literature provides two key insights into the most effective use of waste hierarchies to achieve environmental improvements.

- Users of the Wasted Food Scale should be encouraged to begin by considering the top tier, exhausting source reduction options before moving on to the next tier, rather than aiming to move up a tier from their current location (Van Ewijk and Stegemann 2016).
- The audience for the Wasted Food Scale should include decision-makers from outside as well as within the waste management community. Once waste is discarded and collected (and under control of waste managers), opportunities for source reduction, donation and upcycling have passed, and opportunities for recycling may be limited (e.g., due to mixed collection).

In addition, a variety of factors beyond environmental performance may limit and/or influence decisions about which pathways to choose for management of wasted food in a specific context, including:

- Wasted food characteristics such as edibility, homogeneity, contamination, packaging, and content of moisture, energy, oil, nutrients, protein, and carbon;
- Stage of food supply chain at which food is wasted;
- Availability of local infrastructure and end-markets for recycled materials made from wasted food (e.g., soil amendments or energy);
- Cost;
- Local policies, such as bans on landfilling wasted food, requirements to recycle food, climate or waste goals;
- Local priorities, such as concern about hazardous air pollutants or eutrophication of local waterbodies;
- Localized environmental impacts and environmental justice considerations; and
- Stakeholder preferences.

Given these considerations, along with the potential differences in performance due to operating parameters, it is not possible to definitively rank the pathways across all potential conditions. The scale in this report is meant only to inform the user of the environmental performance of the most common pathways.

6.3 Implications

The evidence presented in this report suggests that, to achieve greater environmental performance regarding wasted food, the focus should be on (1) increasing source reduction, donation and upcycling and (2) diverting wasted food from landfills and wastewater resource recovery facilities (with or without anaerobic digesters). Source reduction (and often donation) have environmental benefits an order of magnitude greater than other pathways and can be implemented at many stages of the supply chain. However, as discussed in EPA's Farm to Kitchen report, donation cannot be relied on as sole solution, since not all wasted food is fit for donation and the amount of wasted food generated in the U.S. far outweighs the needs of food-insecure Americans (U.S. EPA 2021c).

The alignment of current public policies with the hierarchy should be examined. Often policies focus more upon landfill diversion and donation than source reduction (Van Ewijk and Stegemann 2016; Redlingshöfer et al. 2020), missing opportunities to encourage greater environmental savings. Non-food policies, such as renewable energy goals and incentives, can also discourage source reduction of wasted food (Redlingshöfer et al. 2020), and these potential trade-offs should be studied. Also, the effect of landfill diversion policies should be assessed to ensure wasted food is not shifting from landfill to sewer/WWT, which this report found to have similar magnitude of global warming potential performance.

6.4 Research Gaps

Although numerous studies have been published examining the environmental impacts and value of wasted food pathways, data gaps persist. Filling these data gaps is essential to achieving the *2030 Food Loss and Waste Reduction Goal* and the *National Recycling Goal* with maximum environmental benefits.

We see two areas as top priorities for further laboratory and field research:

- *Methane generation from sewer conveyance and wastewater treatment of wasted food.* Further research is needed on the amount of methane generated from wasted food in the sewer/WWT pathway under a variety of conditions (e.g., residence time, oxygenation, temperature). Current research indicates that between 6% and 10% of volatile solids or COD, respectively, may be released as methane during sewer transport (Parry and WERF 2012; Edwards et al. 2018). More robust measurements of sewer methane emissions are needed and should be incorporated into LCAs of this pathway to increase confidence in pathway impacts.
- *Environmental benefits and impacts of applying wasted food or wasted food-derived amendments to soil.* Five wasted food pathways have the potential to provide nutrients and other organic matter to soil and improve soil health directly: the unharvested/plowed in pathway leaves crops in the field; land application applies raw wasted food to soil; composting creates compost to be applied to soil; and AD and sewer/WWT create digestate or biosolids which can be used as soil amendments. These pathways can also have negative environmental impacts through nitrogen losses or contaminants. Further research is needed to understand the environmental benefits and impacts of these outputs, relative to one another and relative to synthetic fertilizer, in situations where these outputs are used as partial or full replacement for fertilizer. Soil health benefits requiring further study include increasing carbon sequestration, biodiversity, water percolation and storage, porosity and nutrient retention. Environmental impacts requiring further study include nitrogen losses to the environment (through ammonia or nitrous oxide emissions and nitrate leaching and/or runoff), as well as risks to human health or the environment from contaminants that may be present in soil amendments. Research should extend to uses beyond application to agricultural soil, such as site reclamation or remediation and green infrastructure. Additional work is then needed to incorporate these findings into the LCA framework.

Additional areas that warrant further research include:

- *Effect of technology type and specific management and operational practices on pathway performance, especially for composting and AD pathways.* Current limitations make it difficult to compare pathways at a higher level of resolution that considers technology type or specific management and operational practices. Emerging technologies such as nutrient recovery from WRRF effluent and heat recovery from composting provide more options for facilities to reduce their environmental impact. For pathways with wide variability in operational choices, better understanding the environmental effects of choices at the facility level could improve environmental performance by guiding sound management.
- *Environmental impacts of two understudied pathways – upcycling and land application.* More research is needed to understand the typical range of operating conditions, inputs and outputs, and environmental impacts and benefits that characterize these two understudied pathways. For land application, understanding the effects of different wasted food characteristics (e.g., nutrient, carbon, or oil content) on soil health (e.g., biodiversity, bulk density, and carbon sequestration) and the environment (e.g., potential for nitrogen losses) could help optimize for which wasted food streams this pathway is selected. Upcycling is an emerging pathway that shows great potential to avoid the impacts of additional food production. More research is needed on environmental performance to increase confidence in current results.
- *Environmental impacts of emerging pathways.* Emerging pathways such as HTC, pyrolysis, torrefaction and gasification as well as the established pathway of rendering were excluded from the main LCA and circularity analyses due to a lack of available environmental data. The pathway technologies and key features of each are introduced in Appendix E. These processes are complex and can be operated under a range of conditions (e.g., temperature and pressure) and in combination with various pre- and post-treatment steps. Research is needed to help commercialize these systems for use processing wasted food and more broadly. Additional

environmental research could help to identify applications and operational contexts (e.g., feedstock mix, energy source, product mix) that yield broad environmental benefits or reductions in impact.

- *Effect of future energy mix on pathway performance. Investments in wasted food pathways will necessarily be considerable and determine wasted food management methods for decades to come.* By focusing primarily on current energy mix, LCAs and other environmental assessments of wasted food pathways might make recommendations that are not supported in future conditions. For example, citing other research, Slorach et al. (2020) noted that many of the benefits of AD depend on environmental credits for replacing fertilizer and energy production, which can be expected to change in coming years. Incorporating future energy mix scenarios in more wasted food management LCAs will help increase confidence in the expected change in pathway performance due to this important driver of impact. While directionality of results is easily predicted (impacts of energy use decrease, while benefits of producing energy decrease), how pathways compare to one another is not.
- *Effect of PFAS and microplastic contaminants.* As discussed elsewhere in the report, PFAS and microplastic contamination in wasted food streams prior to entering pathways can affect the suitability of wasted food for management pathways. The potential for wasted food management to contribute to broader environmental and human health impacts is of concern, too, if the pathway can reintroduce contaminants to the environment. The risk associated with emerging contaminants, such as microplastics and PFAS, in wasted food pathways has not been well-studied and may be difficult to quantify. As techniques to measure these contaminants become more effective and more available, they should be considered with respect to the environmental performance of wasted food pathways. In addition, research is needed on the potential introduction of microplastics to wasted food streams during pre-processing activities such as use of depackaging equipment.
- *Local implications of wasted food management.* This report has focused on high-level comparison of wasted food pathways. We have considered impacts at a variety of spatial scales. For example, GWP impacts occur globally, some acidifying compounds may be transported at continental scales, and other compounds may only travel tens or hundreds of kilometers. Human and ecotoxicity impacts, for example, tend to fall in the latter category. Toxicity was not consistently reported in the literature, preventing comparisons across studies. Regardless, LCA (and thus this study) is best suited to evaluate broad impacts; for questions about impacts to specific communities, risk assessment is often used. Any wasted food pathway can have impacts to the local community. While negative impacts are most often associated with landfilling and controlled combustion of solid waste (e.g., Hooks et al. (2020)), emissions, odors, noise, etc. are part of wasted food management. Existing or proposed wasted food management should also be considered from an environmental justice perspective.
- *Benefits of replacing peat with compost, digestate or biosolids.* Avoiding the use of peat soil amendments is assessed infrequently within the literature but demonstrates potential to generate considerable avoided product benefits for the compost and AD pathways. More LCA research is needed to establish a consensus around the magnitude of benefit that results from replacing peat as a growing media. At this time the extent to which digestate and compost are being used in this application is unclear and modeling of both pathways would benefit from market research on this use of soil amendments.
- *Alignment of current public policies with the hierarchy and trade-offs between environmental goals and other policy priorities.*

CHAPTER 7 REFERENCES

Adetunji, A. T., B. Ncube, R. Mulidzi, and F. B. Lewu. 2020. Management impact and benefit of cover crops on soil quality: A review. *Soil and Tillage Research* 204: 104717. doi:https://doi.org/10.1016/j.still.2020.104717.

Ahamed, A., K. Yin, B. J. H. Ng, F. Ren, V. W.-C. Chang, and J.-Y. Wang. 2016. Life cycle assessment of the present and proposed food waste management technologies from environmental and economic impact perspectives. *Journal of Cleaner Production* 131: 607–614. doi:10.1016/j.jclepro.2016.04.127.

Albizzati, P. F., D. Tonini, and T. F. Astrup. 2021a. A quantitative sustainability assessment of food waste management in the European Union. *Environmental Science & Technology* 55. American Chemical Society: 16099–16109. doi:10.1021/acs.est.1c03940.

Albizzati, P. F., D. Tonini, and T. F. Astrup. 2021b. High-value products from food waste: An environmental and socio-economic assessment. *The Science of the Total Environment* 755: 142466. doi:10.1016/j.scitotenv.2020.142466.

Alewell, C., B. Ringeval, C. Ballabio, D. A. Robinson, P. Panagos, and P. Borrelli. 2020. Global phosphorus shortage will be aggravated by soil erosion. *Nature Communications* 11: 4546. doi:10.1038/s41467-020-18326-7.

Alexander, C., and C. Smaje. 2008. Surplus retail food redistribution: An analysis of a third sector model. *Resources, Conservation and Recycling* 52: 1290–1298. doi:10.1016/j.resconrec.2008.07.009.

Alexander, P., C. Brown, A. Arneth, J. Finnigan, D. Moran, and M. D. A. Rounsevell. 2017. Losses, inefficiencies and waste in the global food system. *Agricultural Systems* 153: 190–200. doi:10.1016/j.agsy.2017.01.014.

Alexander, R. 2012. Digestate utilization in the U.S. *BioCycle*: *the Organics Recycling Authority*.

Al-Kaisi, M. M., and X. Yin. 2005. Tillage and crop residue effects on soil carbon and carbon dioxide emission in corn-soybean rotations. *Journal of Environmental Quality* 34: 437–445. doi:10.2134/jeq2005.0437.

Al-Rumaihi, A., G. McKay, H. R. Mackey, and T. Al-Ansari. 2020. Environmental impact assessment of food waste management using two composting techniques. *Sustainability* 12 (4): 1595. doi:10.3390/su12041595.

American Biogas Council. 2018. Biogas industry market snapshot.

Angouria-Tsorochidou, E., S. Walk, I. Körner, and M. Thomsen. 2023. Environmental and economic assessment of household food waste source-separation efficiency in a German case study. *Cleaner Waste Systems* 5: 100092. doi:10.1016/j.clwas.2023.100092.

ASCE. 2021. 2021 Infrastructure report card: Wastewater. *ASCE's 2021 infrastructure report card*.

Askri, A., P. Laville, A. Trémier, and S. Houot. 2016. Influence of origin and post-treatment on greenhouse gas emissions after anaerobic digestate application to soil. *Waste and Biomass Valorization* 7: 293–306. doi:10.1007/s12649-015-9452-6.

Baek, G., D. Kim, J. Kim, H. Kim, and C. Lee. 2020. Treatment of cattle manure by anaerobic co-digestion with food waste and pig manure: Methane yield and synergistic effect. *International Journal of Environmental Research and Public Health* 17. doi:10.3390/ijerph17134737.

Baker, G. A., L. C. Gray, M. J. Harwood, T. J. Osland, and J. B. C. Tooley. 2019. On-farm food loss in northern and central California: Results of field survey measurements. *Resources, Conservation and Recycling* 149: 541–549. doi:10.1016/j.resconrec.2019.03.022.

Baptista, A. I., and A. Perovich. 2019. *U.S. municipal solid waste incinerators: An industry in decline*. New York, NY: Tishman Environment and Design Center at The New School.

Bare, J. 2011. TRACI 2.0: The tool for the reduction and assessment of chemical and other environmental impacts 2.0. *Clean Technologies and Environmental Policy* 13: 687–696. doi:10.1007/s10098-010-0338-9.

Bare, J. C., G. A. Norris, D. W. Pennington, and T. E. McKone. 2003. TRACI: The tool for the reduction and assessment of chemical and other environmental impacts. *Journal of Industrial Ecology* 6: 49–78. doi: 10.1162/108819802766269539.

Bartocci, P., M. Zampilli, F. Liberti, V. Pistolesi, S. Massoli, G. Bidini, and F. Fantozzi. 2020. LCA analysis of food waste co-digestion. *Science of the Total Environment* 709: 136187.

Batuecas, E., T. Tommasi, F. Battista, V. Negro, G. Sonetti, P. Viotti, D. Fino, and G. Mancini. 2019. Life cycle assessment of waste disposal from olive oil production: Anaerobic digestion and conventional disposal on soil. *Journal of Environmental Management* 237: 94–102. doi:10.1016/j.jenvman.2019.02.021.

Bayart, J.-B., C. Bulle, L. Deschênes, M. Margni, S. Pfister, F. Vince, and A. Koehler. 2010. A framework for assessing off-stream freshwater use in LCA. *The International Journal of Life Cycle Assessment* 15: 439–453. doi:10.1007/s11367-010-0172-7.

Beecher, N., J. Beecher, J. Burke-Wells, M. Lono-Batura, N. Goldstein, G. Kester, and B. Toffey. 2022. *National Biosolids Data Project: Biosolids management in the U. S.*

Beghin-Tanneau, R., F. Guerin, M. Guiresse, D. Kleiber, and J. D. Scheiner. 2019. Carbon sequestration in soil amended with anaerobic digested matter. *Soil & Tillage Research* 192: 87–94. doi:10.1016/j.still.2019.04.024.

Belcher, D., and B. Aldrich. 2006. Regulations governing land application of food waste in New York State. Ithaca, N.Y.: Cornell University Manure Management. http://northeast.manuremanagement.cornell.edu/Pages/General_Docs/Fact_Sheets/Land_Application_Food_Waste_Regs_2006_factsheet.pdf. Accessed August 6 2021.

Benavente, V., A. Fullana, and N. D. Berge. 2017. Life cycle analysis of hydrothermal carbonization of olive mill waste: Comparison with current management approaches. *Journal of Cleaner Production* 142: 2637–2648. doi:10.1016/j.jclepro.2016.11.013.

Beretta, C., M. Stucki, and S. Hellweg. 2017. Environmental impacts and hotspots of food losses: Value chain analysis of Swiss food consumption. *Environmental Science & Technology* 51: 11165–11173. doi:10.1021/acs.est.6b06179.

Berge, N. D., L. Li, J. R. V. Flora, and K. S. Ro. 2015. Assessing the environmental impact of energy production from hydrochar generated via hydrothermal carbonization of food wastes. *Waste Management* 43: 203–217. doi:10.1016/j.wasman.2015.04.029.

Bernstad, A. K., A. Cánovas, and R. Valle. 2017. Consideration of food wastage along the supply chain in lifecycle assessments: A mini-review based on the case of tomatoes. Waste Management & Research: The Journal of the International Solid Wastes and Public Cleansing Association, ISWA 35 (1): 29–39. doi:10.1177/0734242X16666945.

Bernstad Saraiva Schott, A. K., and T. Andersson. 2015. Food waste minimization from a life-cycle perspective. *Journal of Environmental Management* 147: 219–226. doi:10.1016/j.jenvman.2014.07.048.

Bernstad Saraiva Schott, A., H. Wenzel, and J. la Cour Jansen. 2016. Identification of decisive factors for greenhouse gas emissions in comparative life cycle assessments of food waste management – An analytical review. *Journal of Cleaner Production* 119: 13–24. doi:10.1016/j.jclepro.2016.01.079.

Bessou, C., A. Tailleur, C. Godard, A. Gac, J. L. de la Cour, J. Boissy, P. Mischler, A. Caldeira-Pires, et al. 2020. Accounting for soil organic carbon role in land use contribution to climate change in agricultural LCA: which methods? Which impacts? *The International Journal of Life Cycle Assessment* 25: 1217–1230. doi:10.1007/s11367-019-01713-8.

Blanco-Canqui, H., and R. Lal. 2009. Crop residue removal impacts on soil productivity and environmental quality. *Critical Reviews in Plant Sciences* 28: 139–163. doi:10.1080/07352680902776507.

Bocken, N. M. P., I. de Pauw, C. Bakker, and B. van der Grinten. 2016. Product design and business model strategies for a circular economy. *Journal of Industrial and Production Engineering* 33. Taylor & Francis: 308–320. doi:10.1080/21681015.2016.1172124.

Bolzonella, D., P. Pavan, P. Battistoni, and F. Cecchi. 2003. The under sink garbage grinder: A friendly technology for the environment. *Environmental Technology* 24: 349–359. doi:10.1080/09593330309385567.

Boulay, A.-M., J. Bare, L. Benini, M. Berger, M. J. Lathuillière, A. Manzardo, M. Margni, et al. 2018. The WULCA consensus characterization model for water scarcity footprints: Assessing impacts of water consumption based on available water remaining (AWARE). *The International Journal of Life Cycle Assessment* 23: 368–378. doi:10.1007/s11367-017-1333-8.

Brown, S., and N. Beecher. 2019. Carbon accounting for compost use in urban areas. *Compost Science and Utilization* 27: 227–239. doi:10.1080/1065657X.2019.1674224.

Brown, S., K. Kurtz, A. Bary, and C. Cogger. 2011. Quantifying benefits associated with land application of organic residuals in Washington State. *Environmental Science & Technology* 45: 7451–7458. doi:10.1021/es2010418.

Brown, S., A. Carpenter, and L. Souther. 2020. What are composting VOCs? And how are they measured? *BioCycle: the Organics Recycling Authority*.

California, S. of. 2023. Short-Lived Climate Pollutants (SLCP): Organic waste methane emissions reductions. *CalRecycle Home Page*. https://calrecycle.ca.gov/climate/slcp/. Accessed August 15.

CalRecycle. 2019. *SB 1383 Infrastructure and market analysis*. DRRRR-2019-1652. California Department of Resources Recycling and Recovery.

CARB. 2015. *ARB Emissions Inventory Methodology for Composting Facilities*. California Air Resources Board.

CARB. 2017. *Method for estimating greenhouse gas emission reductions from diversion of organic waste from landfills to compost facilities*. California Air Resources Board.

Carey, R. O., and K. W. Migliaccio. 2009. Contribution of wastewater treatment plant effluents to nutrient dynamics in aquatic systems: A review. *Environmental Management* 44: 205–217. doi:10.1007/s00267-009-9309-5.

Case, S. D. C., and L. S. Jensen. 2019. Nitrogen and phosphorus release from organic wastes and suitability as bio-based fertilizers in a circular economy. *Environmental Technology* 40: 701–715. doi:10.1080/09593330.2017.1404136.

Cerda, A., A. Artola, X. Font, R. Barrena, T. Gea, and A. Sánchez. 2018. Composting of food wastes: Status and challenges. *Bioresource Technology* 248: 57–67. doi:10.1016/j.biortech.2017.06.133.

Cesaro, A. 2021. The valorization of the anaerobic digestate from the organic fractions of municipal solid waste: Challenges and perspectives. *Journal of Environmental Management* 280: 111742. doi:https://doi.org/10.1016/j.jenvman.2020.111742.

Charles, A., P. Rochette, J. K. Whalen, D. A. Angers, M. H. Chantigny, and N. Bertrand. 2017. Global nitrous oxide emission factors from agricultural soils after addition of organic amendments: A meta-analysis. *Agriculture, Ecosystems & Environment* 236: 88–98. doi:10.1016/j.agee.2016.11.021.

Chiu, S. L. H., and I. M. C. Lo. 2018. Identifying key process parameters for uncertainty propagation in environmental life cycle assessment for sewage sludge and food waste treatment. *Journal of Cleaner Production* 174: 966–976. doi:10.1016/j.jclepro.2017.10.164.

Cho, B. H., B. H. Nam, J. An, and H. Youn. 2020. Municipal solid waste incineration (MSWI) ashes as construction materials-A review. *Materials* (Switzerland) 13. doi:10.3390/ma13143143.

Ciais, P., C. Sabine, G. Bala, L. Bopp, V. Brovkin, J. Canadell, A. Chhabra, et al. 2013. Carbon and other biogeochemical cycles. In *Climate change 2013: The physical science basis. Contribution of Working Group I to the fifth assessment report of the Intergovernmental Panel on Climate Change*, ed. T. F. Stocker, D. Qin, G.-K. Platner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, et al., 465–570. Cambridge, U.K., and New York, NY: Cambridge University Press.

Claassen, R., M. Bowman, J. McFadden, D. Smith, and S. Wallander. 2018. *Tillage intensity and conservation cropping in the United States*. Economic Information Bulletin Number 197. U.S. Department of Agriculture.

Clavreul, J., D. Guyonnet, and T. H. Christensen. 2012. Quantifying uncertainty in LCA-modelling of waste management systems. *Waste Management* 32: 2482–2495. doi:10.1016/j.wasman.2012.07.008.

Cobo, S., A. Dominguez-Ramos, and A. Irabien. 2018. Trade-offs between nutrient circularity and environmental impacts in the management of organic waste. *Environmental Science & Technology* 52: 10923–10933. doi:10.1021/acs.est.8b01590.

Cogger, C. G., A. I. Bary, E. A. Myhre, and A.-M. Fortuna. 2013. Biosolids applications to tall fescue have long-term influence on soil nitrogen, carbon, and phosphorus. *Journal of Environmental Quality* 42: 516–522. doi:10.2134/jeq2012.0269.

Cooke, C. M., L. Gove, F. A. Nicholson, H. F. Cook, and A. J. Beck. 2001. Effect of drying and composting biosolids on the movement of nitrate and phosphate through repacked soil columns under steady-state hydrological conditions. *Chemosphere* 44: 797–804. doi:https://doi.org/10.1016/S0045-6535(00)00293-9.

Corona, A., M. Ambye-Jensen, G. C. Vega, M. Z. Hauschild, and M. Birkved. 2018. Techno-environmental assessment of the green biorefinery concept: Combining process simulation and life cycle assessment at an early design stage. *Science of the Total Environment* 635: 100–111. doi:10.1016/j.scitotenv.2018.03.357.

Corona, A., A. Ernstoff, C. Segato, and M. Zgola. 2020. *Greenhouse gas emissions of food waste: Methodology*. ReFED. URL: https://refed.com/downloads/quantis-ghg-methodology-vfinal-2020-11-03.pdf

Crolla, A., C. Kinsley, and E. Pattey. 2013. 13 - Land application of digestate. In *The Biogas Handbook*, ed. A. Wellinger, J. Murphy, and D. Baxter, 302–325. Woodhead Publishing.

Damgaard, A., C. Riber, T. Fruergaard, T. Hulgaard, and T. H. Christensen. 2010. Life-cycle-assessment of the historical development of air pollution control and energy recovery in waste incineration. In *Thermal treatments of solid wastes*, special issue, *Waste Management* 30: 1244–1250. doi:10.1016/j.wasman.2010.03.025.

Damiani, M., T. Pastorello, A. Carlesso, S. Tesser, and E. Semenzin. 2021. Quantifying environmental implications of surplus food redistribution to reduce food waste. *Journal of Cleaner Production* 289: 125813. doi:10.1016/j.jclepro.2021.125813.

De Pascale, A., R. Arbolino, K. Szopik-Depczyńska, M. Limosani, and G. Ioppolo. 2021. A systematic review for measuring circular economy: The 61 indicators. *Journal of Cleaner Production* 281: 124942. doi:10.1016/j.jclepro.2020.124942.

DEFRA. 2021. *Statutory guidance: Food and drink waste hierarchy: Deal with surplus and waste*. London, U.K.: Department for Environmental Food & Rural Affairs (DEFRA).

Del Borghi, A., L. Moreschi, and M. Gallo. 2020. Circular economy approach to reduce water–energy–food nexus. *Current Opinion in Environmental Science & Health* 13: 23–28. doi:https://doi.org/10.1016/j.coesh.2019.10.002.

Delgado-Rodríguez, M., M. Ruiz-Montoya, I. Giraldez, R. López, E. Madejón, and M. J. Díaz. 2011. Influence of control parameters in VOCs evolution during MSW trimming residues composting. *Journal of Agricultural and Food Chemistry* 59: 13035–13042. doi:10.1021/jf2036017.

Do, Q., A. Ramudhin, C. Colicchia, A. Creazza, and D. Li. 2021. A systematic review of research on food loss and waste prevention and management for the circular economy. *International Journal of Production Economics* 239: 108209. doi:10.1016/j.ijpe.2021.108209.

Dou, Z., J. D. Toth, and M. L. Westendorf. 2018. Food waste for livestock feeding: Feasibility, safety, and sustainability implications. *Global Food Security* 17: 154–161. doi:10.1016/j.gfs.2017.12.003.

Duren, R. M., A. K. Thorpe, K. T. Foster, T. Rafiq, F. M. Hopkins, V. Yadav, B. D. Bue, et al. 2019. California's methane super-emitters. *Nature* 575: 180–184. doi:10.1038/s41586-019-1720-3.

Dutta, S., M. He, X. Xiong, and D. C. W. Tsang. 2021. Sustainable management and recycling of food waste anaerobic digestate: A review. *Bioresource Technology*: 125915. doi:https://doi.org/10.1016/j.biortech.2021.125915.

Ebner, J. H., R. A. Labatut, M. J. Rankin, J. L. Pronto, C. A. Gooch, A. A. Williamson, and T. A. Trabold. 2015. Lifecycle greenhouse gas analysis of an anaerobic codigestion facility processing dairy manure and industrial food waste. *Environmental science & technology* 49. United States: 11199–11208. doi:10.1021/acs.est.5b01331.

EC. 2018. *Review of the 2012 European bioeconomy strategy*. European Commission Directorate-General for Research and Innovation.

EC, 2018. 2018. *Directive 2018/851 of the European Parliament and of the Council of 30 May 2018 Amending Directive 2008/98/EC on Waste*. Vol. 2018/851.

Edwards, J., M. Othman, E. Crossin, and S. Burn. 2017. Anaerobic co-digestion of municipal food waste and sewage sludge: A comparative life cycle assessment in the context of a waste service provision. *Bioresource Technology* 223: 237–249. doi:10.1016/j.biortech.2016.10.044.

Edwards, J., M. Othman, E. Crossin, and S. Burn. 2018. Life cycle assessment to compare the environmental impact of seven contemporary food waste management systems. *Bioresource Technology* (England) 248: 156–173. doi:10.1016/j.biortech.2017.06.070.

Elevitch, C. R., D. N. Mazaroli, and D. Ragone. 2018. Agroforestry standards for regenerative agriculture. *Sustainability* 10. doi:10.3390/su10093337.

Elginoz, N., K. Khatami, I. Owusu-Agyeman, and Z. Cetecioglu. 2020. Life cycle assessment of an innovative food waste management system. *Frontiers in Sustainable Food Systems* 4. doi:10.3389/fsufs.2020.00023.

Elia, V., M. G. Gnoni, and F. Tornese. 2017. Measuring circular economy strategies through index methods: A critical analysis. *Journal of Cleaner Production* 142: 2741–2751. doi:https://doi.org/10.1016/j.jclepro.2016.10.196.

Ellen MacArthur Foundation. 2013. *Towards the circular economy*.

Ellen Macarthur Foundation. 2019. *Cities and circular economy for food*.

Embrandiri, A., R. P. Singh, H. M. Ibrahim, and A. A. Ramli. 2012. Land application of biomass residue generated from palm oil processing: Its potential benefits and threats. *The Environmentalist* 32: 111–117. doi:10.1007/s10669-011-9367-0.

Environmental and Energy Study Institute. 2013. *Fact sheet: Landfill methane*.

EREF. 2023. Composting Facility Survey. *Environmental Research and Education Foundation*.

Eriksson, M. and J. Spångberg. 2017. Carbon footprint and energy use of food waste management options for fresh fruit and vegetables from supermarkets. In *Urban mining and circular economy*, special issue, *Waste Management* 60: 786–799. doi:10.1016/j.wasman.2017.01.008.

Eriksson, M., L. Bartek, K. Löfkvist, C. Malefors, and M. E. Olsson. 2021. Environmental assessment of upgrading horticultural side streams—The case of unharvested broccoli leaves. *Sustainability* 13: 5327. doi:10.3390/su13105327.

Ermolaev, E., C. Sundberg, M. Pell, and H. Jönsson. 2014. Greenhouse gas emissions from home composting in practice. *Bioresource Technology* (England) 151: 174–182. doi:10.1016/j.biortech.2013.10.049.

Erses Yay, A. S. 2015. Application of life cycle assessment (LCA) for municipal solid waste management: a case study of Sakarya. *Journal of Cleaner Production* 94: 284–293. doi:https://doi.org/10.1016/j.jclepro.2015.01.089.

Fantke, P., L. Aylward, J. Bare, W. A. Chiu, R. Dodson, R. Dwyer, A. Ernstoff, B. Howard, et al. 2018. Advancements in life cycle human exposure and toxicity characterization. *Environmental Health Perspectives* 126: 125001. doi:10.1289/EHP3871.

FAO. 2022. *Soils for nutrition: State of the art*. Food and Agricultural Organization of the United Nations.

Frischknecht, R., H. Althaus, C. Bauer, G. Dóka, T. Heck, N. Jungbluth, D. Kellenberger, and T. Nemecek. 2007. The environmental relevance of capital goods in life cycle assessments of products and services. *The International Journal of Life Cycle Assessment* 12 (1): 1–11. doi:10.1065/LCA2007.02.309.

Funari, V., J. Mäkinen, J. Salminen, R. Braga, E. Dinelli, and H. Revitzer. 2017. Metal removal from Municipal Solid Waste Incineration fly ash: A comparison between chemical leaching and bioleaching. *Waste Management* 60. Special Thematic Issue: Urban Mining and Circular Economy: 397–406. doi:10.1016/j.wasman.2016.07.025.

Gallo, F., A. Manzardo, D. Camana, A. Fedele, and A. Scipioni. 2023. Integration of a circular economy metric with life cycle assessment: methodological proposal of compared agri-food products. *The International Journal of Life Cycle Assessment*. doi:10.1007/s11367-022-02130-0.

García, A. J., M. B. Esteban, M. C. Márquez, and P. Ramos. 2005. Biodegradable municipal solid waste: Characterization and potential use as animal feedstuffs. *Waste Management* 25 (8): 780–787. doi:10.1016/j.wasman.2005.01.006.

Geisendorf, S., and F. Pietrulla. 2018. The circular economy and circular economic concepts—a literature analysis and redefinition. *Thunderbird International Business Review* 60: 771–782. doi:10.1002/tie.21924.

Geissdoerfer, M., P. Savaget, N. M. P. Bocken, and E. J. Hultink. 2017. The circular economy – A new sustainability paradigm? *Journal of Cleaner Production* 143: 757–768. doi:https://doi.org/10.1016/j.jclepro.2016.12.048.

Gilbert, J. 2023. Personal communication. Carbon Clarity.

Gilbert, J., M. Ricci-Jürgensen, and A. Ramola. 2020. *Benefits of compost and anaerobic digestate when applied to soil*. International Solid Waste Association.

Gillman, A., D. C. Campbell, and E. S. Spang. 2019. Does on-farm food loss prevent waste? Insights from California produce growers. *Resources, Conservation and Recycling* 150: 104408. doi:10.1016/j.resconrec.2019.104408.

Giraud, R. J., P. H. Taylor, and C. Huang. 2021. Combustion operating conditions for municipal waste-to-energy facilities in the U.S. *Waste Management* 132: 124–132. doi:10.1016/j.wasman.2021.07.015.

Goedkoop, M., R. Heijungs, M. A. J. Huijbregts, A. D. Schryver, J. Struijs, and R. van Zelm. 2013. *ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level; First edition (version 1.08); Report 1: Characterisation*. The Netherlands: Ministry of Housing, Spatial Planning, and Environment (VROM).

Goldstein, N. 2017. The state of organics recycling. *BioCycle: the Organics Recycling Authority*.

Goldstein, N. 2020. Compost Markets in The U.S. *BioCycle: the Organics Recycling Authority*.

González, R., J. G. Rosas, D. Blanco, R. Smith, E. J. Martínez, R. Pastor-Bueis, and X. Gómez. 2020. Anaerobic digestion of fourth range fruit and vegetable products: Comparison of three different scenarios for its valorisation by life cycle assessment and life cycle costing. *Environmental Monitoring and Assessment* 192: 551. doi:10.1007/s10661-020-08521-w.

Goss, M. J., A. Tubeileh, and D. Goorahoo. 2013. A review of the use of organic amendments and the risk to human health. *Advances in Agronomy* 120: 275–379. doi:10.1016/B978-0-12-407686-0.00005-1.

Gravuer, K., S. Gennet, and H. L. Throop. 2019. Organic amendment additions to rangelands: A meta-analysis of multiple ecosystem outcomes. *Global Change Biology* 25: 1152–1170. doi:10.1111/gcb.14535.

Grigatti, M., L. Barbanti, M. U. Hassan, and C. Ciavatta. 2020. Fertilizing potential and CO2 emissions following the utilization of fresh and composted food-waste anaerobic digestates. *Science of The Total Environment* 698: 134198. doi:10.1016/j.scitotenv.2019.134198.

Grippo, V., S. Romano, and A. Vastola. 2019. Multi-criteria Evaluation of Bran Use to Promote Circularity in the Cereal Production Chain. *Natural Resources Research* 28. doi:10.1007/s11053-019-09457-w.

Guinée, J., M. Gorrée, R. Heijungs, G. Huppes, R. Kleijn, A. de Koning, L. van Oers, et al. 2002. *Handbook on life cycle assessment: Operational guide to the ISO standards (CML 2002 documentation)*. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Gwyther, C. L., A. P. Williams, P. N. Golyshin, G. Edwards-Jones, and D. L. Jones. 2011. The environmental and biosecurity characteristics of livestock carcass disposal. *Waste Management* 31: 767–778. doi:10.1016/j.wasman.2010.12.005.

Hanson, C., B. Lipinski, K. Robertson, D. Dias, I. Gavilan, P. Greverath, S. Ritter, et al. 2016. *Food loss and waste accounting and reporting standard, version 1.0*. Food Loss and Waste Protocol.

Harder, R., M. Giampietro, and S. Smukler. 2021. Towards a circular nutrient economy. A novel way to analyze the circularity of nutrient flows in food systems. *Resources, Conservation and Recycling* 172: 105693. doi:10.1016/j.resconrec.2021.105693.

Havukainen, J., and E. Dace. 2023. Chapter 8 - Waste to energy and circular economy: the case of anaerobic digestion. In *Sustainable and Circular Management of Resources and Waste Towards a Green Deal*, ed. M. N. Vara Prasad and M. Smol, 105–115. Elsevier.

He, C., K. Wang, Y. Yang, and J.-Y. Wang. 2014. Utilization of sewage-sludge-derived hydrochars toward efficient cocombustion with different-rank coals: Effects of subcritical water conversion and blending scenarios. *Energy & Fuels* 28: 6140–6150. doi:10.1021/ef501386g.

Heller, M. 2019. *Waste not, want not: Reducing food loss and waste in North America through life cycle-based approaches*. Washington, DC: United Nations Environment Programme.

Henderson, A. D. 2015. Eutrophication. In *Life cycle impact assessment*, ed. M. Z. Hauschild and M. A. J. Huijbregts, 177–196. LCA Compendium—The Complete World of Life Cycle Assessment. Netherlands: Springer Science & Business Media.

Hodge, K. L., J. W. Levis, J. F. DeCarolis, and M. A. Barlaz. 2016. Systematic evaluation of industrial, commercial, and institutional food waste management strategies in the United States. *Environmental Science & Technology* 50: 8444–8452. doi:10.1021/acs.est.6b00893.

Hoehn, D., M. Margallo, J. Laso, I. Ruiz-Salmon, A. Fernandez-Rios, C. Campos, I. Vazquez-Rowe, R. Aldaco, et al. 2021. Water footprint assessment of food loss and waste management strategies in Spanish regions. *Sustainability* 13: 7538. doi:10.3390/su13147538.

Hooks, S. K., M. Hrysko, C. Strickland, B. Dixon, I. M. Karaye, K. Kirsch, and J. Horney. 2020. Can global policy impact local environmental justice? Using CASPER to assess perceived health effects of an incinerator. *Human Ecology* 48 (1): 119–125. doi:10.1007/s10745-020-00133-9.

Huang, Y., C. Zhao, B. Gao, S. Ma, Q. Zhong, and L. Wang. 2022. Life cycle assessment and society willingness to pay indexes of food waste-to-energy strategies. *Journal of Environmental Management* 305: 114364. doi:10.1016/j.jenvman.2021.114364.

Huijbregts, M. A. J., Z. J. N. Steinmann, P. M. F. Elshout, G. Stam, F. Verones, M. Vieira, M. Zijp, A. Hollander, R. van Zelm. 2017. ReCiPe 2016: A harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment* 22: 138–147. doi:10.1007/s11367-016-1246-y.

IPCC. 2013. *Climate change 2013: The physical science basis. Contribution of Working Group I to the fifth assessment report of the Intergovernmental Panel on Climate Change*. Cambridge, U.K., and New York, NY: Cambridge University Press.

IPCC. 2023. *Climate change 2023: Synthesis Report. A Report of the Intergovernmental Panel on Climate Change. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Edited by H. Lee and J. Romero. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

Ippolito, J. A., K. A. Barbarick, M. W. Paschke, and R. B. Brobst. 2010. Infrequent composted biosolids applications affect semi-arid grassland soils and vegetation. *Journal of Environmental Management* 91. Elsevier: 1123–1130.

Ippolito, J. A., T. F. Ducey, K. Diaz, and K. A. Barbarick. 2021. Long-term biosolids land application influences soil health. *Science of The Total Environment* 791: 148344. doi:https://doi.org/10.1016/j.scitotenv.2021.148344.

Ishangulyyev, R., S. Kim, and S. H. Lee. 2019. Understanding food loss and waste—Why are we losing and wasting food? *Foods* (Switzerland) 8. doi:10.3390/foods8080297.

ISO. 2006a. *ISO 14040:2006 Environmental management—Life cycle assessment—Principles and framework*. The International Organization for Standardization.

ISO. 2006b. *ISO 14044:2006 Environmental management—Life cycle assessment—Requirements and guidelines*. The International Organization for Standardization.

ISO. 2018. *ISO 14067:2018 Greenhouse gases—Carbon footprint of products—Requirements and guidelines for quantification.* The International Organization for Standardization.

Jäckel, U., K. Thummes, and P. Kämpfer. 2005. Thermophilic methane production and oxidation in compost. *FEMS Microbiology Ecology* 52: 175–184. doi:10.1016/j.femsec.2004.11.003.

Jain, P., J. Wally, T. G. Townsend, M. Krause, and T. Tolaymat. 2021. Greenhouse gas reporting data improves understanding of regional climate impact on landfill methane production and collection. *PLOS One* 16 (2): e0246334. doi:10.1371/journal.pone.0246334.

Jin, Y., T. Chen, and H. Li. 2012. Hydrothermal treatment for inactivating some hygienic microbial indicators from food waste–amended animal feed. *Journal of the Air & Waste Management Association* 62 (7): 810–816. doi:10.1080/10962247.2012.676999.

Jolliet, O., M. Margni, R. Charles, S. Humbert, J. Payet, G. Rebitzer, and R. K. Rosenbaum. 2003. IMPACT 2002+: A new life cycle impact assessment methodology. *International Journal of Life Cycle Assessment* 8: 324–330.

Jurgilevich, A., T. Birge, J. Kentala-Lehtonen, K. Korhonen-Kurki, J. Pietikäinen, L. Saikku, and H. Schösler. 2016. Transition towards circular economy in the food system. *Sustainability* 8. doi:10.3390/su8010069.

Kapo, K. E., M. Paschka, R. Vamshi, M. Sebasky, and K. McDonough. 2017. Estimation of U.S. sewer residence time distributions for national-scale risk assessment of down-the-drain chemicals. *Science of the Total Environment* 603–604: 445–452. doi:10.1016/j.scitotenv.2017.06.075.

Katami, T., A. Yasuhara, and T. Shibamoto. 2004. Formation of dioxins from incineration of foods found in domestic garbage. *Environmental Science & Technology* 38: 1062–1065. doi:10.1021/es030606y.

Kaur, G., L. Luo, G. Chen, and J. W. C. Wong. 2019. Integrated food waste and sewage treatment—A better approach than conventional food waste-sludge co-digestion for higher energy recovery via anaerobic digestion. *Bioresource Technology* 289: 121698. doi:10.1016/j.biortech.2019.121698.

Keng, Z. X., S. Chong, C. G. Ng, N. I. Ridzuan, S. Hanson, G.-T. Pan, P. L. Lau, et al. 2020. Community-scale composting for food waste: A life-cycle assessment-supported case study. *Journal of Cleaner Production* 261: 121220. doi:10.1016/j.jclepro.2020.121220.

Khoshnevisan, B., P. Tsapekos, M. Alvarado-Morales, S. Rafiee, M. Tabatabaei, and I. Angelidaki. 2018. Life cycle assessment of different strategies for energy and nutrient recovery from source sorted organic fraction of household waste. *Journal of Cleaner Production* 180: 360–374. doi:10.1016/j.jclepro.2018.01.198.

Kibblewhite, M. G., K. Ritz, and M. J. Swift. 2008. Soil health in agricultural systems. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363: 685–701. doi:10.1098/rstb.2007.2178.

Kim, M., M. M. I. Chowdhury, G. Nakhla, and M. Keleman. 2017. Synergism of co-digestion of food wastes with municipal wastewater treatment biosolids. *Waste management* 61. United States: 473–483. doi:10.1016/j.wasman.2016.10.010.

Kim, M., G. Nakhla, and M. Keleman. 2019. Modeling the impact of food wastes on wastewater treatment plants. *Journal of Environmental Management* 237: 344–358. doi:10.1016/j.jenvman.2019.02.065.

Kirchherr, J., D. Reike, and M. Hekkert. 2017. Conceptualizing the circular economy: An analysis of 114 definitions. *Resources, Conservation and Recycling* 127: 221–232. doi:10.1016/j.resconrec.2017.09.005.

Klinglmair, M., and M. Thomsen. 2020. Using food waste in organic fertilizer: modelling biogenic carbon sequestration with associated nutrient and micropollutant loads. *Sustainability* 12. Multidisciplinary Digital Publishing Institute: 7399. doi:10.3390/su12187399.

Koenig, R. T., C. G. Cogger, and A. I. Bary. 2011. Dryland winter wheat yield, grain protein, and soil nitrogen responses to fertilizer and biosolids applications. *Applied and Environmental Soil Science* 2011. Hindawi: e925462. doi:10.1155/2011/925462.

Komilis, D. P., and R. K. Ham. 2004. Life-cycle inventory of municipal solid waste and yard waste windrow composting in the United States. *Journal of Environmental Engineering* 130. American Society of Civil Engineers: 1390–1400. World. doi:10.1061/(ASCE)0733-9372(2004)130:11(1390).

Kopittke, P. M., A. A. Berhe, Y. Carrillo, T. R. Cavagnaro, D. Chen, Q.-L. Chen, M. Román Dobarco, F. A. Dijkstra, et al. 2022. Ensuring planetary survival: the centrality of organic carbon in balancing the multifunctional nature of soils. *Critical Reviews in Environmental Science and Technology* 52: 4308–4324. doi:10.1080/10643389.2021.2024484.

Kopsahelis, A., C. Kourmentza, C. Zafiri, and M. Kornaros. 2019. Life cycle assessment (LCA) of end-of-life dairy products (EoL-DPs) valorization via anaerobic co-digestion with agro-industrial wastes for biogas production. *Journal of Chemical Technology & Biotechnology* 94: 3687–3697. doi:https://doi.org/10.1002/jctb.6174.

Kraj, K., and M. Smol. 2023. Chapter 25 - How to achieve climate neutrality—the impact of fertilizer usage on climate change. In *Sustainable and Circular Management of Resources and Waste Towards a Green Deal*, ed. M. N. Vara Prasad and M. Smol, 349–356. Elsevier. doi:10.1016/B978-0-323-95278-1.00028-0.

Kumar, K., C. J. Rosen, S. C. Gupta, and M. McNearney. 2009. Land application of sugar beet by-products: Effects on nitrogen mineralization and crop yields. *Journal of Environmental Quality* 38: 319–328. doi:10.2134/jeq2008.0005.

Kummu, M., H. de Moel, M. Porkka, S. Siebert, O. Varis, and P. J. Ward. 2012. Lost food, wasted resources: Global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use. *Science of the Total Environment* 438: 477–489. doi:10.1016/j.scitotenv.2012.08.092.

Kutos, S., E. Stricker, A. Cooper, R. Ryals, J. Creque, M. Machmuller, M. Kroegar, and W. L. Silver. 2023. Compost amendment to enhance carbon sequestration in rangelands. *Journal of Soil and Water Conservation* 78. Soil and Water Conservation Society: 163–177. doi:10.2489/jswc.2023.00072.

Lal, R., R.F. Follett, B.A. Stewart, and J.M. Kimble. 2007. Soil carbon sequestration to mitigate climate change and advance food security. *Soil Science*: 172: 943-956. doi: 10.1097/ss.0b013e31815cc498

Lazcano, C., A. Tsang, T. A. Doane, G. S. Pettygrove, W. R. Horwath, and M. Burger. 2016. Soil nitrous oxide emissions in forage systems fertilized with liquid dairy manure and inorganic fertilizers. *Agriculture, Ecosystems & Environment* 225: 160–172. doi:https://doi.org/10.1016/j.agee.2016.04.009.

Le Pera, A., M. Sellaro, and E. Bencivenni. 2022a. Composting food waste or digestate? Characteristics, statistical and life cycle assessment study based on an Italian composting plant. *Journal of Cleaner Production* 350: 131552. doi:https://doi.org/10.1016/j.jclepro.2022.131552.

Le Pera, A., M. Sellaro, E. Bencivenni, and F. D'Amico. 2022b. Environmental sustainability of an integrate anaerobic digestion-composting treatment of food waste: Analysis of an Italian plant in the circular bioeconomy strategy. *Waste Management* 139: 341–351. doi:https://doi.org/10.1016/j.wasman.2021.12.042.

Le Pera, A., M. Sellaro, F. Sicilia, R. Ciccoli, B. Sceberras, C. Freda, E. Fanelli, and G. Cornacchia. 2023. Environmental and economic impacts of improper materials in the recycling of separated collected food waste through anaerobic digestion and composting. *Science of the Total Environment* 880: 163240. doi:10.1016/j.scitotenv.2023.163240.

Lee, D., E. Sönmez, M. I. Gómez, and X. Fan. 2017. Combining two wrongs to make two rights: Mitigating food insecurity and food waste through gleaning operations. *Food Policy* 68: 40–52. doi:10.1016/j.foodpol.2016.12.004.

Lee, E., D. S. B. L. Oliveira, L. S. B. L. Oliveira, E. Jimenez, Y. Kim, M. Wang, S. J. Ergas, and Q. Zhang. 2020. Comparative environmental and economic life cycle assessment of high solids anaerobic co-digestion for biosolids and organic waste management. *Water Research* (England) 171: 115443. doi:10.1016/j.watres.2019.115443.

Lee, U., J. Han, and M. Wang. 2017. Evaluation of landfill gas emissions from municipal solid waste landfills for the life-cycle analysis of waste-to-energy pathways. *Journal of Cleaner Production* 166: 335–342. doi:10.1016/j.jclepro.2017.08.016.

Lehmann, J., and M. Kleber. 2015. The contentious nature of soil organic matter. *Nature* 528: 60–68. doi:10.1038/nature16069.

Lehmann, J., D. A. Bossio, I. Kögel-Knabner, and M. C. Rillig. 2020. The concept and future prospects of soil health. *Nature Reviews Earth & Environment* 1: 544–553. doi:10.1038/s43017-020-0080-8.

Leib, E. B., O. Balkus, C. Rice, M. Maley, R. Taneja, R. Cheng, N. Civita, and T. Alvoid. 2016. *Leftovers For Livestock: A legal guide for using food scraps as animal feed*. The Harvard Food Law and Policy Clinic and the Food Recovery Project, University of Arkansas School of Law. https://chlpi.org/wp-content/uploads/2013/12/Leftovers-for-Livestock_A-Legal-Guide_August-2016.pdf. Accessed August 29 2022.

Levis, J. W., and M. A. Barlaz. 2011. What is the most environmentally beneficial way to treat commercial food waste? *Environmental Science & Technology* (United States) 45: 7438–7444. doi:10.1021/es103556m.

Lin, L., F. Xu, X. Ge, and Y. Li. 2018. Improving the sustainability of organic waste management practices in the food-energy-water nexus: A comparative review of anaerobic digestion and composting. *Renewable and Sustainable Energy Reviews* 89: 151–167. doi:10.1016/j.rser.2018.03.025.

Lin, Z., J. K. Ooi, and K. S. Woon. 2022. An integrated life cycle multi-objective optimization model for health-environment-economic nexus in food waste management sector. *Science of The Total Environment* 816: 151541. doi:10.1016/j.scitotenv.2021.151541.

Lindgreen, E. R., G. Mondello, R. Salomone, F. Lanuzza, and G. Saija. 2021. Exploring the effectiveness of grey literature indicators and life cycle assessment in assessing circular economy at the micro level: a comparative analysis. *International Journal of Life Cycle Assessment* 26: 2171–2191. doi:10.1007/s11367-021-01972-4.

Liptzin, D., C. E. Norris, S. B. Cappellazzi, G. M. Bean, M. Cope, K. L. H. Greub, E. L. Rieke, P. W. Tracy, et al. 2022. An evaluation of carbon indicators of soil health in long-term agricultural experiments. *Soil Biology and Biochemistry* 172: 108708. doi:https://doi.org/10.1016/j.soilbio.2022.108708.

Lombardi, M., and M. Costantino. 2021. A hierarchical pyramid for food waste based on a social innovation perspective. *Sustainability* 13: 4661. doi:10.3390/su13094661.

Lopez, V. M., F. B. De la Cruz, and M. A. Barlaz. 2016. Chemical composition and methane potential of commercial food wastes. *Waste Management* 56: 477–490. doi:10.1016/j.wasman.2016.07.024.

Lu, J., and S. Xu. 2021. Post-treatment of food waste digestate towards land application: A review. *Journal of Cleaner Production* 303: 127033. doi:10.1016/j.jclepro.2021.127033.

Mahal, N. K., W. R. Osterholz, F. E. Miguez, H. J. Poffenbarger, J. E. Sawyer, D. C. Olk, S. V. Archontoulis, and M. J. Castellano. 2019. Nitrogen fertilizer suppresses mineralization of soil organic matter in maize agroecosystems. *Frontiers in Ecology and Evolution* 7. doi:10.3389/fevo.2019.00059.

Makarichi, L., W. Jutidamrongphan, and K. Techato. 2018. The evolution of waste-to-energy incineration: A review. *Renewable and Sustainable Energy Reviews* 91: 812–821. doi:10.1016/j.rser.2018.04.088.

Mancini, E., and A. Raggi. 2021. A review of circularity and sustainability in anaerobic digestion processes. *Journal of Environmental Management* 291: 112695. doi:10.1016/j.jenvman.2021.112695.

Mattsson, J., A. Hedström, and M. Viklander. 2014. Long-term impacts on sewers following food waste disposer installation in housing areas. *Environmental Technology* 35: 2643–2651. doi:10.1080/09593330.2014.915346.

Mattsson, J., A. Hedström, R. M. Ashley, and M. Viklander. 2015. Impacts and managerial implications for sewer systems due to recent changes to inputs in domestic wastewater—A review. *Journal of Environmental Management* 161: 188–197. doi:10.1016/j.jenvman.2015.06.043.

Mayer, F., R. Bhandari, and S. Gäth. 2019. Critical review on life cycle assessment of conventional and innovative waste-to-energy technologies. *Science of the Total Environment* 672: 708–721. doi:10.1016/j.scitotenv.2019.03.449.

Mayer, F., R. Bhandari, S. A. Gäth, H. Himanshu, and N. Stobernack. 2020. Economic and environmental life cycle assessment of organic waste treatment by means of incineration and biogasification. Is source segregation of biowaste justified in Germany? *Science of the Total Environment* 721: 137731. doi:10.1016/j.scitotenv.2020.137731.

Moinard, V., F. Levavasseur, and S. Houot. 2021. Current and potential recycling of exogenous organic matter as fertilizers and amendments in a French peri-urban territory. *Resources, Conservation and Recycling* 169: 105523. doi:https://doi.org/10.1016/j.resconrec.2021.105523.

Möller, K. 2015. Effects of anaerobic digestion on soil carbon and nitrogen turnover, N emissions, and soil biological activity. A review. *Agronomy for Sustainable Development* 35: 1021–1041. doi:10.1007/s13593-015-0284-3.

Momayez, F., K. Karimi, and M. J. Taherzadeh. 2019. Energy recovery from industrial crop wastes by dry anaerobic digestion: A review. *Industrial Crops and Products* 129: 673–687. doi:https://doi.org/10.1016/j.indcrop.2018.12.051.

Mondello, G., R. Salomone, G. Ioppolo, G. Saija, S. Sparacia, and M. C. Lucchetti. 2017. Comparative LCA of alternative scenarios for waste treatment: The case of food waste production by the mass-retail sector. *Sustainability* 9: 827. doi:10.3390/su9050827.

Moore, D. P., N. P. Li, L. P. Wendt, S. R. Castañeda, M. M. Falinski, J.-J. Zhu, C. Song, Z. J. Ren, et al. 2023. Underestimation of Sector-Wide Methane Emissions from United States Wastewater Treatment. *Environmental Science & Technology* 57: 4082–4090. doi:10.1021/acs.est.2c05373.

Moraga, G., S. Huysveld, F. Mathieux, G. A. Blengini, L. Alaerts, K. Van Acker, S. de Meester, and J. Dewulf. 2019. Circular economy indicators: What do they measure? *Resources, Conservation and Recycling* 146: 452–461. doi:https://doi.org/10.1016/j.resconrec.2019.03.045.

Morais, T. G., R. F. M. Teixeira, C. Lauk, M. C. Theurl, W. Winiwarter, A. Mayer, L. Kaufmann, H. Haberl, et al. 2021. Agroecological measures and circular economy strategies to ensure sufficient nitrogen for sustainable farming. *Global Environmental Change* 69: 102313. doi:https://doi.org/10.1016/j.gloenvcha.2021.102313.

Morelli, B., S. Cashman, S. Arden, X. C. Ma, J. Turgeon, J. Garland, and D. Bless. 2019. *Life cycle assessment and cost analysis of municipal wastewater treatment expansion options for food waste anaerobic co-digestion*. EPA/600/R-19/094. Washington, DC: U.S. Environmental Protection Agency, Office of Research and Development.

Morelli, B., S. Cashman, X. C. Ma, J. Turgeon, S. Arden, and J. Garland. 2020. Environmental and cost benefits of co-digesting food waste at wastewater treatment facilities. *Water Science and Technology: A Journal of the International Association on Water Pollution Research* 82: 227–241. doi:10.2166/wst.2020.104.

Morris, J. 2014. *Evaluation of climate, energy, and soils impacts of selected food discards management systems*. Oregon Department of Environmental Quality.

Morris, J., S. Brown, M. Cotton, and H. Scott Matthews. 2017. Life-cycle assessment harmonization and soil science ranking results on food-waste management methods. *Environmental Science & Technology* 51: 5360–5367. doi:10.1021/acs.est.6b06115.

Morseletto, P. 2020. Restorative and regenerative: Exploring the concepts in the circular economy. *Journal of Industrial Ecology* 24: 763–773. doi:https://doi.org/10.1111/jiec.12987.

Moshtaghian, H., K. Bolton, and K. Rousta. 2021. Challenges for upcycled foods: definition, inclusion in the food waste management hierarchy and public acceptability. *Foods* 10: 2874. doi:10.3390/foods10112874.

Mu, D., N. Horowitz, M. Casey, and K. Jones. 2017. Environmental and economic analysis of an in-vessel food waste composting system at Kean University in the U.S. *Waste Management* 59: 476–486. doi:10.1016/j.wasman.2016.10.026.

Murphy, J. D., and N. M. Power. 2006. A technical, economic and environmental comparison of composting and anaerobic digestion of biodegradable municipal waste. *Journal of Environmental Science and Health* 41: 865–879. doi:10.1080/10934520600614488.

Murray, A., K. Skene, and K. Haynes. 2017. The circular economy: An interdisciplinary exploration of the concept and application in a global context. *Journal of Business Ethics* 140: 369–380. doi:10.1007/s10551-015-2693-2.

Myhre, G., D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, et al. 2013. Anthropogenic and natural radiative forcing. In *Climate change 2013: The physical science basis. Contribution of Working Group I to the fifth assessment report of the Intergovernmental Panel on Climate Change*, ed. T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, et al., 659–740. Cambridge, U.K., and New York, NY: Cambridge University Press.

NARA. 2019. *The story of rendering*. North American Renderers Association.

National Research Council. 2000. *Waste incineration and public health*. Washington, DC: The National Academies Press.

Nayak, A., and B. Bhushan. 2019. An overview of the recent trends on the waste valorization techniques for food wastes. *Journal of Environmental Management* 233: 352–370. doi:10.1016/j.jenvman.2018.12.041.

Nicholson, F., A. Bhogal, L. Cardenas, D. Chadwick, T. Misselbrook, A. Rollett, M. Taylor, R. Thorman, and J. Williams. 2017. Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land. *Environmental Pollution* 228: 504–516. doi:10.1016/j.envpol.2017.05.023.

Nick Hacheney and Sally Brown. 2017. Correctional Facility Adopts Multiple Food Scraps Practices. *BioCycle: the Organics Recycling Authority*. October.

NRCS and Agricultural BMP Task Force. 2017. *Manual of best management practices (BMPs) for agriculture in New Hampshire*. Concord, NH: New Hampshire Department of Agriculture, Markets, and Food.

NYSDOH. 2019. *Important things to know about landfill gas*. New York State Department of Health.

O’Connor, J., S. A. Hoang, L. Bradney, S. Dutta, X. Xiong, D. C. W. Tsang, K. Ramadass, A. Vinu, et al. 2021. A review on the valorisation of food waste as a nutrient source and soil amendment. *Environmental Pollution* 272: 115985. doi:10.1016/j.envpol.2020.115985.

O’Connor, J., B. S. Mickan, J. Rinklebe, H. Song, K. H. M. Siddique, H. Wang, M. B. Kirkham, and N. S. Bolan. 2022a. Environmental implications, potential value, and future of food-waste anaerobic digestate management: A review. *Journal of Environmental Management* 318: 115519. doi:https://doi.org/10.1016/j.jenvman.2022.115519.

O’Connor, J., B. S. Mickan, K. H. M. Siddique, J. Rinklebe, M. B. Kirkham, and N. S. Bolan. 2022b. Physical, chemical, and microbial contaminants in food waste management for soil application: A review. *Environmental Pollution* 300: 118860. doi:https://doi.org/10.1016/j.envpol.2022.118860.

Ogden, S., A. Robb, D. Roberts, G. Palmer, and O. Bell. 2022. *Comparing the costs of alternative waste treatment options*. PSE500-022. WRAP.

Oladeji, O., G. Tian, P. Lindo, K. Kumar, A. Cox, L. Hundal, H. Zhang, and E. Podczerwinski. 2020. Nitrogen release and plant available nitrogen of composted and un-composted biosolids. *Water Environment Research* 92: 631–640. doi:https://doi.org/10.1002/wer.1260.

Oldfield, T. L., E. White, and N. M. Holden. 2016. An environmental analysis of options for utilising wasted food and food residue. *Journal of Environmental Management* 183: 826–835. doi:10.1016/j.jenvman.2016.09.035.

Omonode, R. A., T. J. Vyn, D. R. Smith, P. Hegymegi, and A. Gál. 2007. Soil carbon dioxide and methane fluxes from long-term tillage systems in continuous corn and corn–soybean rotations. *Soil and Tillage Research* 95: 182–195. doi:10.1016/j.still.2006.12.004.

Opatokun, S. A., A. Lopez-Sabiron, G. Ferreira, and V. Strezov. 2017. Life cycle analysis of energy production from food waste through anaerobic digestion, pyrolysis and integrated energy system. *Sustainability* 9. Multidisciplinary Digital Publishing Institute: 1804. doi:10.3390/su9101804.

Oregon DEQ. 2017. *Strategic plan for preventing the wasting of food*. Oregon Department of Environmental Quality (ORDEQ).

Oregon DEQ. 2019. *Life cycle assessment of edible food rescue*. Oregon Department of Environmental Quality (ORDEQ).

Owsianiak, M., M. W. Ryberg, M. Renz, M. Hitzl, and M. Z. Hauschild. 2016. Environmental performance of hydrothermal carbonization of four wet biomass waste streams at industry-relevant scales. *ACS Sustainable Chemistry & Engineering* 4: 6783–6791. doi:10.1021/acssuschemeng.6b01732.

Owsianiak, M., M. Z. Hauschild, L. Posthuma, E. Saouter, M. G. Vijver, T. Backhaus, M. Douziech, T. Schlekat, et al. 2022. Ecotoxicity characterization of chemicals: Global recommendations and implementation in USEtox. *Chemosphere*: 136807. doi:10.1016/j.chemosphere.2022.136807.

Ozores-Hampton, M., J. Biala, G. Evanylo, B. Faucette, L. Cooperband, N. Roe, J. A. Creque, and D. Sullivan. 2022. Chapter 16 - Compost use. In *The Composting Handbook*, ed. R. Rynk, 777–846. Academic Press.

Pace, S. A., R. Yazdani, A. Kendall, C. W. Simmons, and J. S. VanderGheynst. 2018. Impact of organic waste composition on life cycle energy production, global warming and water use for treatment by anaerobic digestion followed by composting. *Resources, Conservation and Recycling* 137: 126–135. doi:10.1016/j.resconrec.2018.05.030.

Pahla, G., F. Ntuli, and E. Muzenda. 2017. Torrefaction of landfill food waste for possible application in biomass co-firing. *Waste Management* 71: 512–520. doi:10.1016/j.wasman.2017.10.035.

Parchomenko, A., D. Nelen, J. Gillabel, and H. Rechberger. 2019. Measuring the circular economy—A multiple correspondence analysis of 63 metrics. *Journal of Cleaner Production* 210: 200–216. doi:10.1016/j.jclepro.2018.10.357.

Pardo, G., R. Moral, E. Aguilera, and A. del Prado. 2015. Gaseous emissions from management of solid waste: A systematic review. *Global Change Biology* 21: 1313–1327. doi:10.1111/gcb.12806.

Paritosh, K., S. K. Kushwaha, M. Yadav, N. Pareek, A. Chawade, and V. Vivekanand. 2017. Food waste to energy: An overview of sustainable approaches for food waste management and nutrient recycling. *BioMed Research International* 2017: 2370927. doi:10.1155/2017/2370927.

Parnaudeau, V., S. Génermont, C. Hénault, A. Farrugia, P. Robert, and B. Nicolardot. 2009. Measured and Simulated nitrogen fluxes after field application of food-processing and municipal organic wastes. *Journal of Environmental Quality* 38: 268–280. doi:10.2134/jeq2007.0486.

Parry, D. L. 2014. *Co-digestion of organic waste products with wastewater solids: Final report with economic model*. OWSO5R07. Alexandria, Virginia: Water Environment Research Foundation.

Parry, D. L., and WERF. 2012. *Sustainable food waste evaluation*. OWSO5R07e. Alexandria, Virginia: Water Environment Research Foundation.

Paul, S., A. Dutta, F. Defersha, and B. Dubey. 2018. Municipal Food Waste to Biomethane and Biofertilizer: A Circular Economy Concept. *Waste and Biomass Valorization* 9: 601–611. doi:10.1007/s12649-017-0014-y.

Payen, S., B. Civit, H. Golden, B. Niblick, A. Uwizeye, L. Winter, and A. D. Henderson. 2019. Acidification and eutrophication. In *Global guidance for life cycle impact assessment indicators: Volume 2*, ed. R. Frischknecht and O. Jolliet, 60–79. Paris, France: United Nations Environment Program/Society for Environmental Toxicology and Chemistry Life Cycle Initiative.

Peltre, C., B. T. Christensen, S. Dragon, C. Icard, T. Kätterer, and S. Houot. 2012. RothC simulation of carbon accumulation in soil after repeated application of widely different organic amendments. *Soil Biology and Biochemistry* 52: 49–60. doi:https://doi.org/10.1016/j.soilbio.2012.03.023.

Peña, C., B. Civit, A. Gallego-Schmid, A. Druckman, A. Caldeira-Pires, B. Weidema, E. Mieras, et al. 2021. Using life cycle assessment to achieve a circular economy. *The International Journal of Life Cycle Assessment*. doi:10.1007/s11367-020-01856-z.

Peng, L., R. Tang, G. Wang, R. Ma, Y. Li, G. Li, and J. Yuan. 2023. Effect of aeration rate, aeration pattern, and turning frequency on maturity and gaseous emissions during kitchen waste composting. *Environmental Technology & Innovation* 29: 102997. doi:10.1016/j.eti.2022.102997.

Pergola, M., A. Persiani, A. M. Palese, V. Di Meo, V. Pastore, C. D'Adamo, and G. Celano. 2018. Composting: The way for a sustainable agriculture. *Applied Soil Ecology* 123. HUMUSICA 3 - Reviews, Applications, Tools: 744–750. doi:10.1016/j.apsoil.2017.10.016.

Pfister, S., A.-M. Boulay, M. Berger, M. Hadjikakou, M. Motoshita, T. Hess, B. Ridoutt, et al. 2017. Understanding the LCA and ISO water footprint: A response to Hoekstra (2016) "A critique on the water-scarcity weighted water footprint in LCA." *Ecological Indicators* 72: 352–359. doi:10.1016/j.ecolind.2016.07.051.

Pham, T. P. T., R. Kaushik, G. K. Parshetti, R. Mahmood, and R. Balasubramanian. 2015. Food waste-to-energy conversion technologies: Current status and future directions. *Waste Management* 38: 399–408. doi:10.1016/j.wasman.2014.12.004.

Plante, A. F., and R. P. Voroney. 1998. Decomposition of land applied oily food waste and associated changes in soil aggregate stability. *Journal of Environmental Quality* 27: 395–402. doi:10.2134/jeq1998.00472425002700020021x.

Platt, B., N. Goldstein, C. Coker, and S. Brown. 2014. *State of composting in the US: What, why where & how*. Institute for Local Self-Reliance.

Poeschl, M., S. Ward, and P. Owende. 2012. *Environmental impacts of biogas deployment – Part II: life cycle assessment of multiple production and utilization pathways*. Journal of Cleaner Production 24: 184–201. doi:10.1016/j.jclepro.2011.10.030.

Porterfield, K. K., S. A. Hobson, D. A. Neher, M. T. Niles, and E. D. Roy. 2023. Microplastics in composts, digestates, and food wastes: A review. *Journal of Environmental Quality* 52: 225–240. doi:10.1002/jeq2.20450.

Posch, M., J. Seppälä, J.-P. Hettelingh, M. Johansson, M. Margni, and O. Jolliet. 2008. The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA. *International Journal of Life Cycle Assessment* 13: 477–486. doi:10.1007/s11367-008-0025-9.

Potting, J., M. P. Hekkert, E. Worrell, and A. Hanemaaijer. 2017. *Circular economy: Measuring innovation in the product chain*. PBL Netherlands Environmental Assessment Agency.

Poulton, P., J. Johnston, A. Macdonald, R. White, and D. Powlson. 2018. Major limitations to achieving “4 per 1000” increases in soil organic carbon stock in temperate regions: Evidence from long-term experiments at Rothamsted Research, United Kingdom. *Global Change Biology* 24: 2563–2584. doi:https://doi.org/10.1111/gcb.14066.

Psomopoulos, C. S. 2009. Waste-to-energy: A review of the status and benefits in USA. *Waste Management* 29. Pergamon: 1718–1724. doi:10.1016/j.wasman.2008.11.020.

Quirós, R., G. Villalba, P. Muñoz, J. Colón, X. Font, and X. Gabarrell. 2014. Environmental assessment of two home composts with high and low gaseous emissions of the composting process. *Resources, Conservation and Recycling* 90: 9–20. doi:10.1016/j.resconrec.2014.05.008.

Rajagopal, R., A. Ahamed, and J.-Y. Wang. 2014. Hydrolytic and acidogenic fermentation potential of food waste with source segregated feces-without-urine as co-substrate. *Bioresource technology* 167. England: 564–568. doi:10.1016/j.biortech.2014.06.024.

Raptis, C. E., J. M. Boucher, and S. Pfister. 2017. Assessing the environmental impacts of freshwater thermal pollution from global power generation in LCA. *Science of the Total Environment* 580: 1014–1026. doi:10.1016/j.scitotenv.2016.12.056.

Rashid, M. T., and R. P. Voroney. 2004. Land application of oily food waste and corn production on amended soils. *Agronomy Journal* 96: 997–1004. doi:10.2134/agronj2004.0997.

Rashid, M. T., R. P. Voroney, and M. Khalid. 2010. Application of food industry waste to agricultural soils mitigates green house gas emissions. *Bioresource Technology* 101: 485–490. doi:10.1016/j.biortech.2009.06.113.

Redfield, A. C. 1934. On the proportions of organic derivatives in sea water and their relation to the composition of plankton. In *James Johnstone memorial volume*, ed. F. J. Cole, 176–192. Liverpool, U.K.: University Press of Liverpool.

Redlingshöfer, B., S. Barles, and H. Weisz. 2020. Are waste hierarchies effective in reducing environmental impacts from food waste? A systematic review for OECD countries. *Resources, Conservation and Recycling* 156: 104723. doi:10.1016/j.resconrec.2020.104723.

ReFED. 2021. *Insights Engine Food Waste Monitor*.

ReFED. 2022. Federal Animal Feed Policy. http://policyfinder.refed.com. Accessed August 25 2022.

ReFED. 2023. *Insights engine solutions database: 2023.* https://insights-engine.refed.org/solution-database. Accessed June 1, 2023.

Rigamonti, L., and E. Mancini. 2021. Life cycle assessment and circularity indicators. *International Journal of Life Cycle Assessment* 26: 1937–1942. doi:10.1007/s11367-021-01966-2.

Rigby, H., and S. R. Smith. 2013. Nitrogen availability and indirect measurements of greenhouse gas emissions from aerobic and anaerobic biowaste digestates applied to agricultural soils. *Waste Management* 33: 2641–2652. doi:10.1016/j.wasman.2013.08.005.

Righi, S., L. Oliviero, M. Pedrini, A. Buscaroli, and C. Della Casa. 2013. Life cycle assessment of management systems for sewage sludge and food waste: Centralized and decentralized approaches. *Journal of Cleaner Production* 44: 8–17. doi:10.1016/j.jclepro.2012.12.004.

Rinot, O., G. J. Levy, Y. Steinberger, T. Svoray, and G. Eshel. 2019. Soil health assessment: A critical review of current methodologies and a proposed new approach. *Science of the Total Environment* 648: 1484–1491. doi:10.1016/j.scitotenv.2018.08.259.

Risse, M., and B. Faucette. 2017. *Food waste composting: Institutional and industrial applications.* University of Georgia Extension. Bulletin 1189.

Ryals, R., M. Kaiser, M. S. Torn, A. A. Berhe, and W. L. Silver. 2014. Impacts of organic matter amendments on carbon and nitrogen dynamics in grassland soils. *Soil Biology and Biochemistry* 68. Elsevier: 52–61.

Rynk, R., L. Cooperband, C. Oshins, H. Wescott, J. Bonhotal, M. Schwarz, R. Sherman, and S. Brown. 2022. Chapter 1 - Why compost? In *The Composting Handbook*, ed. R. Rynk, 1–26. Academic Press. doi:10.1016/B978-0-323-85602-7.00001-7.

Salemdeeb, R., E. K. H. J. Zu Ermgassen, M. H. Kim, A. Balmford, and A. Al-Tabbaa. 2017. Environmental and health impacts of using food waste as animal feed: A comparative analysis of food waste management options. *Journal of Cleaner Production* 140: 871–880. doi:10.1016/j.jclepro.2016.05.049.

San Miguel, C., D. Giménez, U. Krogmann, and S. W. Yoon. 2012. Impact of land application of cranberry processing residuals, leaves and biosolids pellets on a sandy loam soil. *Applied Soil Ecology* 53: 31–38. doi:10.1016/j.apsoil.2011.11.001.

Scheehle, E. A., and M. R. J. Doorn. 2002. Improvements to the U.S. wastewater methane and nitrous oxide emissions estimates. U.S. Environmental Protection Agency, Office of Air and Radiation.

Scherhaufer, S., S. Lebersorger, A. Pertl, G. Obersteiner, F. Schneider, L. Falasconi, F. D. Menna, et al. 2015. *Criteria for and baseline assessment of environmental and socio-economic impacts of food waste: Final report*. BOKU University of Natural Resources and Life Sciences, Institute of Waste Management.

Schneider, F. 2013. The evolution of food donation with respect to waste prevention. In *Urban mining*, special issue, *Waste Management* 33: 755–763. doi:10.1016/j.wasman.2012.10.025.

Schreefel, L., R. P. O. Schulte, I. J. M. de Boer, A. P. Schrijver, and H. H. E. van Zanten. 2020. Regenerative agriculture – the soil is the base. *Global Food Security* 26: 100404. doi:10.1016/j.gfs.2020.100404.

Seo, Y.-C., M. T. Alam, and W.-S. Yang. 2018. Gasification of municipal solid waste. In *Gasification for low-grade feedstock*. 10.5772/intechopen.73685.

Seppälä, J., M. Posch, M. Johansson, and J.-P. Hettelingh. 2006. Country-dependent characterisation factors for acidification and terrestrial eutrophication based on accumulated exceedance as an impact category indicator. *International Journal of Life Cycle Assessment* 11: 403–416. doi:10.1065/lca2005.06.215.

Sevigne-Itoiz, E., O. Mwabonje, C. Panoutsou, and J. Woods. 2021. Life cycle assessment (LCA): informing the development of a sustainable circular bioeconomy? *Philosophical Transactions of The Royal Society A-Mathematical Physical and Engineering Sciences* 379. doi:10.1098/rsta.2020.0352.

Sherwood, J. 2020. The significance of biomass in a circular economy. *Bioresource Technology* 300. England: 122755. doi:10.1016/j.biortech.2020.122755.

Shih, M.-F., C.-Y. Lin, and C.-H. Lay. 2021. Comparison of Potential Environmental Impacts and Waste-to-Energy Efficiency for Kitchen Waste Treatment Scenarios in Central Taiwan. *Processes* 9. Multidisciplinary Digital Publishing Institute: 696. doi:10.3390/pr9040696.

Shurson, G. C. 2020. "What a Waste"—Can We Improve Sustainability of Food Animal Production Systems by Recycling Food Waste Streams into Animal Feed in an Era of Health, Climate, and Economic Crises? *Sustainability* 12. Multidisciplinary Digital Publishing Institute: 7071. doi:10.3390/su12177071.

Silver, W. L., S. E. Vergara, and A. Mayer. 2018. *Carbon sequestration and greenhouse gas mitigation potential of composting and soil amendments on California's rangelands*. CCC4A-CNRA-2018–002. California Natural Resources Agency.

Slorach, P. C., H. K. Jeswani, R. Cuéllar-Franca, and A. Azapagic. 2019a. Environmental and economic implications of recovering resources from food waste in a circular economy. *Science of the Total Environment* 693: 133516. doi:10.1016/j.scitotenv.2019.07.322.

Slorach, P. C., H. K. Jeswani, R. Cuéllar-Franca, and A. Azapagic. 2019b. Environmental sustainability of anaerobic digestion of household food waste. *Journal of Environmental Management* 236: 798–814. doi:10.1016/j.jenvman.2019.02.001.

Slorach, P. C., H. K. Jeswani, R. Cuéllar-Franca, and A. Azapagic. 2020. Assessing the economic and environmental sustainability of household food waste management in the UK: Current situation and future scenarios. *Science of the Total Environment* 710: 135580. doi:10.1016/j.scitotenv.2019.135580.

Smith, J. H. 1986. Decomposition of potato processing wastes in soil. *Journal of Environmental Quality* 15: 13–16. doi:10.2134/jeq1986.00472425001500010004x.

Sogn, T. A., I. Dragicevic, R. Linjordet, T. Krogstad, V. G. H. Eijsink, and S. Eich-Greatorex. 2018. Recycling of biogas digestates in plant production: NPK fertilizer value and risk of leaching. *International Journal of Recycling of Organic Waste in Agriculture* 7: 49–58. doi:10.1007/s40093-017-0188-0.

Song, C., J.-J. Zhu, J. L. Willis, D. P. Moore, M. A. Zondlo, and Z. J. Ren. 2023. Methane emissions from municipal wastewater collection and treatment systems. *Environmental Science & Technology* 57: 2248–2261. doi:10.1021/acs.est.2c04388.

Song, Q., Z. Wang, and J. Li. 2013. Environmental performance of municipal solid waste strategies based on LCA method: a case study of Macau. *Journal of Cleaner Production* 57: 92–100. doi:https://doi.org/10.1016/j.jclepro.2013.04.042.

State of California. 2016a. *Compostable Materials Handling Operations and Facilities Regulatory Requirements*. *California Code of Regulations*. Vol. 14 CCR § 17868.3.1.

State of California. 2016b. *Compostable Materials Handling Operations and Facilities Regulatory Requirements*. *California Code of Regulations*. Vol. 14 CCR § 17868.2.

State of Washington. 2019. *Washington Administrative Code 173-350-220*.

Steffen, W., K. Richardson, and J. Rockstrom. 2015. Planetary boundaries: Guiding human development on a changing planet. *Science* 347: 1259855. doi:10.1126/science.1259855.

Stehouwer, R., L. Cooperband, R. Rynk, J. Biala, J. Bonhotal, S. Antler, T. Lewandowski, and H. Nichols. 2022. Chapter 15 - Compost characteristics and quality. In *The Composting Handbook*, ed. R. Rynk, 737–775. Academic Press.

Sullivan, D. M., A. I. Bary, C. G. Cogger, I. J. Madsen, and L. J. Brewer. 2021. *Biosolids in Dryland Cropping Systems*. PNW 716. Pacific Northwest Extension: Oregon State University, University of Idaho, Washington State University.

Sulis, F., F. Agostinho, C. M. V. B. Almeida, and B. F. Giannetti. 2021. Recognizing the wealth of non-marketable food in distribution centres: The environmental benefits of donation. *Journal of Cleaner Production* 318: 128482. doi:10.1016/j.jclepro.2021.128482.

Sundin, N., C. Persson Osowski, I. Strid, and M. Eriksson. 2022. Surplus food donation: Effectiveness, carbon footprint, and rebound effect. *Resources, Conservation and Recycling* 181: 106271. doi:10.1016/j.resconrec.2022.106271.

Tambone, F., B. Scaglia, G. D’Imporzano, A. Schievano, V. Orzi, S. Salati, and F. Adani. 2010. Assessing amendment and fertilizing properties of digestates from anaerobic digestion through a comparative study with digested sludge and compost. *Chemosphere* 81: 577–583. doi:10.1016/j.chemosphere.2010.08.034.

Tanveer, U., S. Ishaq, and A. Gough. 2021. Circular economy in agri-food sector: food waste management perspective. In *Challenges and Opportunities of Circular Economy in Agri-Food Sector*, ed. D. R. S. D. A. P. Mor. Springer.

Thakali, A., and J. D. MacRae. 2021. A review of chemical and microbial contamination in food: What are the threats to a circular food system? *Environmental Research* 194: 110635. doi:10.1016/j.envres.2020.110635.

Thakali, A., J. D. MacRae, C. Isenhour, and T. Blackmer. 2022. Composition and contamination of source separated food waste from different sources and regulatory environments. *Journal of Environmental Management* 314: 115043. doi:10.1016/j.jenvman.2022.115043.

Thangarajan, R., N. S. Bolan, G. Tian, R. Naidu, and A. Kunhikrishnan. 2013. Role of organic amendment application on greenhouse gas emission from soil. *Science of The Total Environment* 465. Soil as a Source & Sink for Greenhouse Gases: 72–96. doi:10.1016/j.scitotenv.2013.01.031.

Thyagarajan, T., M. Puri, J. Vongsvivut, and C. J. Barrow. 2014. Evaluation of bread crumbs as a potential carbon source for the growth of thraustochytrid species for oil and omega-3 production. *Nutrients* 6: 2104–2114. doi:10.3390/nu6052104.

Tian, H., X. Wang, E. Y. Lim, J. T. E. Lee, A. W. L. Ee, J. Zhang, and Y. W. Tong. 2021. Life cycle assessment of food waste to energy and resources: Centralized and decentralized anaerobic digestion with different downstream biogas utilization. *Renewable and Sustainable Energy Reviews* 150: 111489. doi:10.1016/j.rser.2021.111489.

de Titto, E., and A. Savino. 2019. Environmental and health risks related to waste incineration. *Waste Management & Research* 37 (10): 1–11. doi:10.1177/0734242X19859700 Article information.

Tiwary, A., I. D. Williams, D. C. Pant, and V. V. N. Kishore. 2015. Emerging perspectives on environmental burden minimisation initiatives from anaerobic digestion technologies for community scale biomass valorisation. *Renewable and Sustainable Energy Reviews* 42: 883–901. doi:10.1016/j.rser.2014.10.052.

Tong, H., Y. Shen, J. Zhang, C.-H. Wang, T. S. Ge, and Y. W. Tong. 2018. A comparative life cycle assessment on four waste-to-energy scenarios for food waste generated in eateries. *Applied Energy* 225: 1143–1157. a9h. doi:10.1016/j.apenergy.2018.05.062.

Trabold, T., and C. W. Babbitt, eds. 2018. *Sustainable food waste-to-energy systems*. Elsevier.

Tufvesson, L. M., M. Lantz, and P. Börjesson. 2013. Environmental performance of biogas produced from industrial residues including competition with animal feed—Life-cycle calculations according to different methodologies and standards. *Journal of Cleaner Production* 53: 214–223. doi:10.1016/j.jclepro.2013.04.005.

Turmel, M.-S., A. Speratti, F. Baudron, N. Verhulst, and B. Govaerts. 2015. Crop residue management and soil health: A systems analysis. *Agricultural Systems* 134: 6–16. doi:https://doi.org/10.1016/j.agsy.2014.05.009.

UN General Assembly. 2015. *Transforming Our World: The 2030 Agenda for Sustainable Development*. Vol. A/Res/70/1.

UNEP. 2019. Circularity framework. *United Nations Environment Program*.

UNH Extension. 2017. *Manual of best management practices for land application of biosolids*. University of New Hampshire Cooperative Extension.

Urra, J., I. Alkorta, and C. Carbisu. 2019. Potential benefits and risks for soil health derived from the use of organic amendments in agriculture. *Agronomy* 9: 542. doi:10.3390/agronomy9090542.

U.S. Composting Council. 2021. USCC Compost Manufacturer's Environmental Impact Analysis. *Newsroom*. July 12.

U.S. Congress. 2020. *Save Our Seas 2.0 Act*. Public Law 116-224.

U.S. EPA. 2016. *Clean watersheds needs survey 2012: Report to Congress*. EPA-830-R-15005. United States Environmental Protection Agency.

U.S. EPA. 2020a. *2018 wasted food report*. EPA 530-R-20-004. United States Environmental Protection Agency.. https://www.epa.gov/sites/production/files/2020-11/documents/2018_wasted_food_report.pdf. Accessed July 1 2021.

U.S. EPA. 2020b. *Documentation for greenhouse gas emission and energy factors used in the waste reduction model (WARM): Management practices chapters*. WARM Version 15. Washington, DC, USA: United States Environmental Protection Agency, Office of Resource Conservation and Recovery. https://www.epa.gov/warm/documentation-chapters-greenhouse-gas-emission-energy-and-economic-factors-used-waste-reduction.

U.S. EPA. 2020c. *Documentation for greenhouse gas emission and energy factors used in the waste reduction model (WARM): Organic materials chapters*. WARM Version 15. Washington, DC, USA: United States Environmental Protection Agency, Office of Resource Conservation and Recovery. https://www.epa.gov/warm/documentation-chapters-greenhouse-gas-emission-energy-and-economic-factors-used-waste-reduction.

U.S. EPA. 2020d. Food loss and waste 2030 champions. United States Environmental Protection Agency. https://www.epa.gov/sustainable-management-food/united-states-food-loss-and-waste-2030-champions. Accessed 4/1/2022.

U.S. EPA. 2021a. *Anaerobic digestion facilities processing food waste in the United States (2017 & 2018)*. EPA/903/S-21/001. United States Environmental Protection Agency..

U.S. EPA. 2021b. *Commercial pre-processing technologies*. Emerging Issues in Food Waste Management. United States Environmental Protection Agency.

U.S. EPA. 2021c. *From farm to kitchen: The environmental impacts of U.S. food waste*. EPA 600-R21 171. U.S. Environmental Protection Agency.

U.S. EPA. 2021d. *LFG Energy Project Development Handbook*`. Washington, DC, USA: United States Environmental Protection Agency.

U.S. EPA. 2021e. *Persistent chemical contaminants*. Emerging Issues in Food Waste Management. Washington, DC, USA: United States Environmental Protection Agency.

U.S. EPA. 2021f. *Plastic contamination*. Emerging Issues in Food Waste Management. Washington, DC, USA: United States Environmental Protection Agency.

U.S. EPA. 2021g. Sustainable management of food: Food recovery hierarchy. United States Environmental Protection Agency. https://www.epa.gov/sustainable-management-food/food-recovery-hierarchy

U.S. EPA. 2021h. Types of Composting and Understanding the Process. United States Environmental Protection Agency.https://www.epa.gov/sustainable-management-food/types-composting-and-understanding-process. Accessed August 3, 2022.

U.S. EPA. 2022a. Landfill Methane Outreach Program Landfill and Project Database (version March 2022). United States Environmental Protection Agency. https://www.epa.gov/lmop/lmop-landfill-and-project-database.

U.S. EPA. 2022b. National overview: Facts and figures on materials, wastes and recycling. United States Environmental Protection Agency. https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/national-overview-facts-and-figures-materials. Accessed August 23, 2022.

U.S. EPA. 2022c. U.S. National Recycling Goal. United States Environmental Protection Agency. https://www.epa.gov/recyclingstrategy/us-national-recycling-goal. Accessed November 29, 2022.

U.S. EPA. 2023a. *2019 Wasted Food Report: Estimates of Generation and Management of Wasted Food in the United States in 2019*. United States Environmental Protection Agency.

U.S. EPA. 2023b. *Anaerobic digestion facilities processing food waste in the United States (2019)*. EPA 530-R-23-003. United States Environmental Protection Agency.

U.S. EPA. 2022c. Energy recovery from the combustion of municipal solid waste (MSW). United States Environmental Protection Agency. https://www.epa.gov/smm/energy-recovery-combustion-municipal-solid-waste-msw. Accessed February 13, 2023.

U.S. EPA. 2023d. Excess Food Opportunities Map. United States Environmental Protection Agency. https://www.epa.gov/sustainable-management-food/excess-food-opportunities-map. Accessed February 3, 2023.

U.S. FDA. 2022. Bovine Spongiform Encephalopathy. United States Food and Drug Administration. https://www.fda.gov/animal-veterinary/compliance-enforcement/bovine-spongiform-encephalopathy. Accessed April 18, 2023.

Usack, J. G., L. Gerber Van Doren, R. Posmanik, R. A. Labatut, J. W. Tester, and L. T. Angenent. 2018. An evaluation of anaerobic co-digestion implementation on New York State dairy farms using an environmental and economic life-cycle framework. *Applied Energy* 211: 28–40. doi:10.1016/j.apenergy.2017.11.032.

USDA. 2020. Swine: Other Programs. *USDA Animal and Plant Health Inspection Service*. United States Department of Agriculture. https://www.aphis.usda.gov/aphis/ourfocus/animalhealth/animal-disease-information/swine-disease-information/other-programs. Accessed April 17, 2023.

USDA. 2021. Soil health. United States Department of Agriculture. https://www.nrcs.usda.gov/wps/portal/nrcs/main/soils/health/. Accessed April 17, 2023.

Van Ewijk, S., and J. A. Stegemann. 2016. Limitations of the waste hierarchy for achieving absolute reductions in material throughput. *Journal of Cleaner Production* 132: 122–128. doi:https://doi.org/10.1016/j.jclepro.2014.11.051.

Vandermeersch, T., R. A. F. Alvarenga, P. Ragaert, and J. Dewulf. 2014. Environmental sustainability assessment of food waste valorization options. *Resources, Conservation and Recycling* 87: 57–64. doi:10.1016/j.resconrec.2014.03.008.

Vivaldi, G. A., D. Zaccaria, S. Camposeo, F. Pasanisi, F. P. Salcedo, and I. Portoghese. 2022. Appraising water and nutrient recovery for perennial crops irrigated with reclaimed water in Mediterranean areas through an index-based approach. *Science of The Total Environment* 820: 152890. doi:https://doi.org/10.1016/j.scitotenv.2021.152890.

Wallace, T., D. Gibbons, M. O'Dwyer, and T. P. Curran. 2017. International evolution of fat, oil and grease (FOG) waste management—A review. *Journal of Environmental Management* 187: 424–435. doi:10.1016/j.jenvman.2016.11.003.

Wang, S., and Y. Zeng. 2018. Ammonia emission mitigation in food waste composting: A review. *Bioresource Technology* (England) 248: 13–19. doi:10.1016/j.biortech.2017.07.050.

Wang, X., A. S. Nagpure, J. F. DeCarolis, and M. A. Barlaz. 2013. Using observed data to improve estimated methane collection from select U.S. landfills. *Environmental Science & Technology* 47: 3251–3257. doi:10.1021/es304565m.

Wang, X., Z. Bai, Y. Yao, B. Gao, D. Chadwick, Q. Chen, C. Hu, and L. Ma. 2018. Composting with negative pressure aeration for the mitigation of ammonia emissions and global warming potential. *Journal of Cleaner Production* 195: 448–457. doi:10.1016/j.jclepro.2018.05.146.

Wang, Y., Z. Yuan, and Y. Tang. 2021. Enhancing food security and environmental sustainability: A critical review of food loss and waste management. *Resources, Environment and Sustainability* 4: 100023. doi:10.1016/j.resenv.2021.100023.

Wang, Z., Y. Hu, S. Wang, G. Wu, and X. Zhan. 2023. A critical review on dry anaerobic digestion of organic waste: Characteristics, operational conditions, and improvement strategies. *Renewable and Sustainable Energy Reviews* 176: 113208. doi:10.1016/j.rser.2023.113208.

WEF. 2023. Circular Economy: Transformation Map. *World Economic Forum*.

van der Wiel, B. Z., J. Weijma, C. E. van Middelaar, M. Kleinke, C. J. N. Buisman, and F. Wichern. 2021. Restoring nutrient circularity in a nutrient-saturated area in Germany requires systemic change. *Nutrient Cycling in Agroecosystems* 121: 209–226. doi:10.1007/s10705-021-10172-3.

Wilkinson, A. D., and D. L. Meeker. 2021. How agricultural rendering supports sustainability and assists livestock's ability to contribute more than just food. *Animal Frontiers* 11: 24–34. doi:10.1093/af/vfab002.

Winans, K., E. Marvinney, A. Gillman, and E. Spang. 2020. An evaluation of on-farm food loss accounting in life-cycle assessment (LCA) of four California specialty crops. *Frontiers in Sustainable Food Systems* 4. doi:10.3389/fsufs.2020.00010.

Witzgall, K., A. Vidal, D. I. Schubert, C. Höschen, S. A. Schweizer, F. Buegger, V. Pouteau, C. Chenu, et al. 2021. Particulate organic matter as a functional soil component for persistent soil organic carbon. *Nature Communications* 12: 4115. doi:10.1038/s41467-021-24192-8.

Wolf, M.-A., R. Pant, K. Chomkhamsri, S. Sala, and D. Pennington. 2012. *The International Reference Life Cycle Data System (ILCD) handbook: Towards more sustainable production and consumption for a resource-efficient Europe*. EUR 24982 EN. Ispra, Italy: Joint Research Centre, Institute for Environment and Sustainability.

Woon, K. S., I. M. C. Lo, S. L. H. Chiu, and D. Y. S. Yan. 2016. Environmental assessment of food waste valorization in producing biogas for various types of energy use based on LCA approach. *Waste Management* 50: 290–299. doi:10.1016/j.wasman.2016.02.022.

World Health Organization. 2007. *Population health and waste management: scientific data and policy options*. Rome, Italy. https://www.euro.who.int/__data/assets/pdf_file/0012/91101/E91021.pdf.

WRAP. 2015. *Strategies to achieve economic and environmental gains by reducing food waste*. Banbury, England: Waste and Resources Action Programme.

WRAP. 2016. *Field experiments for quality digestate and compost in agriculture*. Waste and Resources Action Programme.

Wuest, S. B., and H. T. Gollany. 2013. Soil Organic Carbon and Nitrogen After Application of Nine Organic Amendments. *Soil Science Society of America Journal* 77: 237–245. doi:https://doi.org/10.2136/sssaj2012.0184.

Xiong, X., I. K. M. Yu, D. C. W. Tsang, N. S. Bolan, Y. Sik Ok, A. D. Igalavithana, M. B. Kirkham, K.-H. Kim, and K. Vikrant. 2019. Value-added chemicals from food supply chain wastes: State-of-the-art review and future prospects. *Chemical Engineering Journal* 375: 121983. doi:10.1016/j.cej.2019.121983.

Xu, C., W. Shi, J. Hong, F. Zhang, and W. Chen. 2015. Life cycle assessment of food waste-based biogas generation. *Renewable and Sustainable Energy Reviews* 49: 169–177. doi:10.1016/j.rser.2015.04.164.

Yan, M., H. Tian, S. Song, H. T. W. Tan, J. T. E. Lee, J. Zhang, P. Sharma, Y. W. Tiong, et al. 2023. Effects of digestate-encapsulated biochar on plant growth, soil microbiome and nitrogen leaching. *Journal of Environmental Management* 334: 117481. doi:https://doi.org/10.1016/j.jenvman.2023.117481.

Yang, F., G. X. Li, Q. Y. Yang, and W. H. Luo. 2013. Effect of bulking agents on maturity and gaseous emissions during kitchen waste composting. *Chemosphere* 93: 1393–1399. doi:10.1016/j.chemosphere.2013.07.002.

Yoshida, H., J. J. Gable, and J. K. Park. 2012. Evaluation of organic waste diversion alternatives for greenhouse gas reduction. *Resources, Conservation and Recycling* 60: 1–9. doi:10.1016/j.resconrec.2011.11.011.

Zan, F., J. Dai, F. Jiang, G. A. Ekama, and G. Chen. 2020. Ground food waste discharge to sewer enhances methane gas emission: A lab-scale investigation. *Water Research* (England) 174: 115616. doi:10.1016/j.watres.2020.115616.

van Zelm, R., P.-O. Roy, M. Z. Hauschild, and M. A. J. Huijbregts. 2015. Acidification. In *Life cycle impact assessment*, ed. M. Z. Hauschild and M. A. J. Huijbregts, 163–176. LCA Compendium—The Complete World of Life Cycle Assessment. Netherlands: Springer Science & Business Media.

Zhang, Z., W. Han, X. Chen, N. Yang, C. Lu, and Y. Wang. 2019. The life-cycle environmental impact of recycling of restaurant food waste in lanzhou, China. *Applied Sciences* 9: 3608. doi:10.3390/app9173608.

Zhao, Y., Y. Lou, W. Qin, J. Cai, P. Zhang, and B. Hu. 2022. Interval aeration improves degradation and humification by enhancing microbial interactions in the composting process. *Bioresource Technology* 358: 127296. doi:10.1016/j.biortech.2022.127296.

Zhou, Y., Y. Hu, A. J. Y. Chen, Z. Cheng, Z. Bi, R. Zhang, and Z. Lou. 2022. Environmental impacts and nutrient distribution routes for food waste separated disposal on large-scale anaerobic digestion/ composting plants. *Journal of Environmental Management* 318: 115624. doi:10.1016/j.jenvman.2022.115624.

APPENDIX F
ENDNOTES

[1] World Resources Institute (WRI), Consumer Goods Forum (CGF), Food and Agriculture Organization of the United Nations (FAO), United Nations Environment Programme (UNEP), Waste and Resources Action Programme (WRAP), World Business Council for Sustainable Development (WBCSD).

[2] The term ‘water resource recovery facility’, WRRF, is used in this report to refer to wastewater treatment plants that co-digest wasted food. The term ‘wastewater treatment plant’ is used more generally to refer to wastewater treatment facilities that might not recover resources or in cases where it is unclear. For example, within the sewer/WWT pathway not all facilities receiving wasted food will have anaerobic digestion, so the more general term is preferred. Other forms of energy or resource recovery can occur at WRRFs, that may qualify them as WRRFs, but these practices are outside the scope of this document.

[3] Damiani (2021) places the impact of beef production at more than 28 kg CO2e/kg, on average, whereas rice production results in one-tenth the GWP impacts of beef (2.7 kg CO2e/kg).

[4] Eriksson and Spångberg (2017)considered the conversion of nonmarketable fruits and vegetables into chutney. They looked at five different fruits and vegetables (apples, bananas, oranges, peppers, and tomatoes) and evaluated only GWP (kg CO2e/kg wasted food) and energy demand (MJ/kg wasted food). The analysis considered avoided production of fresh fruits and vegetables. Production intensities varied from 0.4 kg CO2e and 5 MJ for an apple to 1 kg CO2e and 15 MJ for a pepper, and thus the net impact was between -0.4 and -1 kg CO2e and between -5 and -15 MJ for the upcycling of the apple and pepper. Additionally, several processes were essentially cancelled by their opposites: processing and avoided processing, glass jar production and avoided glass jar production, transportation to supermarket and avoided transportation to supermarket.

[5] Eriksson et al. (2021) investigated changing broccoli harvesting practices to collect leaves and broccoli heads that fall outside the quality norms and are typically left behind. The leaves were considered for use in bread (displacing wheat flour) and broccoli soup (displacing imported broccoli powder), while the heads were considered for sliced broccoli. For all three scenarios, additional harvesting steps were negligible. Transportation was similar to that required for regular broccoli. The displacement of other products was a major contributor to overall impacts. In the case of bread, the displacement of wheat flour did not offset drying, so the net impact was 0.075 kg CO2e/kg broccoli. However, the soup and sliced broccoli displaced products with higher impacts, leading to net negative impact (-0.45 to -0.35 kg CO2eq/kg broccoli).

[6] For example, cows are often raised for milk and meat, but the leather produced from their hides is not treated as impact-free. In the case of upcycling, broccoli is generally cultivated for its florets, and the leaves are wasted food. As wasted food, broccoli leaf production is impact-free. However, if broccoli leaves were to become a regular substitute for wheat flour, they might cease to be classified as upcycled, and then they would no longer be considered impact-free. A study of broccoli leaves found that the leaves had higher environmental production impacts than wheat flour (Eriksson et al. 2021).

[7] Co-digestion of signficant quantities of wasted food with sludge from wastewater treatment operations may materially affect the composition and associated benefits and impacts of biosolids processing and use. The Biosolids Emissions Assessment Model (BEAM), was developed by the Canadian Council of

Ministers of the Environment, to estimate GHG emissions associated with biosolids management . Applicability of this model to biosolids generated from a co-digestion process is unknown.

[8] About 1,265 of the roughly 16,000 WRRFs in the United States process biosolids with AD, and roughly 10% of those facilities co-digest source-separated wasted food with biosolids (ASCE 2021; U.S. EPA 2023b). Co-digestion of food waste along with wastewater solids has been shown to increase biogas production, with fats, oils, and grease providing some of the greatest biogas benefits (Parry 2014).

[9] A U.S. EPA report of case studies of WRRFs adding wasted food (FOG, residential, commercial, and industrial) as co-digestate showed increases in biogas production ranging from 40% to 300% (U.S. EPA 2014b).

[10] The ratio of produced energy to input energy (termed the energy ratio) is sometimes used to quantify net energy production. Research cited by Khoshnevisan et al. (2018) reports that biogas plants typically have an energy ratio of between 2 and 6. In their own analysis, energy ratios varied from 3.1 to 5.2.

[11] A review of sustainable approaches to wasted food management summarizes nine studies that provide evidence of improved methane yields, enhanced process stability, or increased volatile solids reduction when wasted food is co-digested with green waste, biosolids, animal manure, and other organic materials (Paritosh et al. 2017).

[12] Many reviewed studies specify either single- or two-stage digestion processes, but do not go into detail on why a specific reactor type was chosen. Ahamed et al. (2016) state clearly that “conventional single-stage reactors are not suitable for food waste,” and the study relies instead on a two-stage technology that is expected to increase process stability and yield more methane. Ahamed’s analysis looks at digestion of pure wasted food, while many other studies co-digest wasted food with other organic materials. Tiwary et al. (2015) go into greater detail, describing that methanogenic organisms struggle to deal with the rapid drop in pH that accompanies hydrolysis when highly degradable materials such as wasted food are introduced directly into single-stage reactors. Separation of these reactions within multi-stage reactors can increase biological stability and process performance. Alternatively, co-digestion of highly degradable materials with more stable organic materials can circumvent this issue but limits the quantity of wasted food that can be processed in each facility.

[13] Instances in which it may not be appropriate to assign an avoided burden credit for the full quantity of energy produced include, for example, situations where there is no market for produced heat or where biogas has historically been used to provide heat for AD (i.e., baseline is not natural gas combustion).

[14] Al-Rumaihi et al. (2020) showed that GWP results are sensitive to changes in CHP output, with a 30% increase in heat and electricity production leading to a greater than 100% decrease in impact potential (i.e., a net benefit).

[15] A few studies examine disposal of digestate by incineration (Chiu and Lo 2018; Tong et al. 2018) or in a landfill (Righi et al. 2013).

[16] One study states that 30% to 50% of total nitrogen can be lost as ammonia within six hours of application, though incorporation into soil may reduce losses by 85%. Nitrate leaching is also of concern, with losses of up to 15% of applied nitrogen from autumn applications (Nicholson et al. 2017).

[17] Slorach et al. (2019a) report that there is not a significant difference in the production of biogas between mesophilic and thermophilic reactors, but that thermophilic reactors can have twice the energy consumption. Tiwary et al. (2015) come to a different conclusion, finding that thermophilic reactors do have greater biogas production.

[18] A consequential analysis of animal feed, composting, and AD found that the lower eutrophication, toxicity, and acidification impacts of wet and dry feed are largely due to avoiding conventional animal feed production (Salemdeeb et al. 2017).

[19] Albizzati et al. (2021b)consider avoided reference products for their wet and protein-concentrated animal feed pathways, which are maize grain and soybean-based, respectively. When compared to the gross impact of animal feed production and alternative management pathways, the reference product avoids considerable environmental burdens for several impact categories including human toxicity, cancer, particulate matter formation, global warming, ionizing radiation, photochemical ozone formation, eutrophication, and freshwater ecotoxicity.

[20] Thakali et al. (2022) studied the effect of source and regulatory requirements on the type and amount of contamination in food wastes intended for composting. Source separated food wastes generated from locations that were required by regulation to separate food waste from packaging and other non-food contaminants had lower levels of physical contamination than food wastes from locations that voluntarily separated. Chemical contaminates across both regulated and unregulated sources include zinc, copper, halogenated organics, and PFAS.

[21] A review of methane emissions found that typically between 1% and 4% of carbon entering a compost pile is released as methane (Ermolaev et al. 2014). A separate review of laboratory and field studies found that 2.7% of carbon is released as methane (Pardo et al. 2015). A review by Cerda et al. (2018) identified that between 0.03 and 71.4 kg methane are released per ton wasted food during composting. Accurate measurement of compost gaseous emissions can be difficult, given emission gradients at different locations in the pile (Büyüksönmez 2012).

[22] Levis and Barlaz (2011) performed a sensitivity analysis that looked at the fraction of carbon in compost feedstock that was liberated during biodegradation. The analysis considered loss of between 40% and 83% of initial carbon, with strong linear increases in GWP across all considered composting technologies as the fraction of carbon loss increased. Increasing carbon loss during composting increases process methane emissions while decreasing long-term carbon storage.

[23] An analysis of high and low emission scenarios for home composting systems (Quirós et al. 2014) illustrates the variable impacts of compost process emissions. In the high emission scenario, process emissions dominate all other operational impacts (energy, tools, and collection) for the acidification, eutrophication, and GWP categories. Energy-related inventory categories and ozone depletion are not influenced by compost process emissions. In the low emissions scenarios, process emissions still dominate for GWP, but the magnitude of impact is reduced. The contribution of process emissions to acidification and eutrophication is reduced dramatically in the low emission scenario and is similar in magnitude to impacts associated with infrastructure and energy consumption.

[24] The interplay of the various bacterial communities that compost organic waste is complex, and thus it can be straightforward to optimize for a single emission but difficult to optimize across emissions. For example, a review of strategies to minimize NH3 emissions identified several approaches: 1) adjusting pH, 2) adjusting C:N ratios (e.g., through initial adjustment or co-composting a green waste that has a high C:N ratio), 3) adding absorbents (e.g., zeolite or biochar), 4) struvite precipitation (with additional adjustment of salinity), 5) adjusting nitrification, or 6) adjusting aeration rates and pressures (Wang et al. 2018; Wang and Zeng 2018).

[25] The studies that explicitly state their assumptions related to carbon sequestration assume that between 10% and 15% of land-applied carbon is sequestered (Levis and Barlaz 2011; Yoshida et al. 2012). Whether or not the carbon in compost is assumed to be sequestered has a strong impact on results,

potentially shifting net GWP emissions from positive to negative (Levis and Barlaz 2011; Silver et al. 2018).

[26] Wasted food is a good source of nitrogen and phosphorus in composting systems, can help establish ideal C/N, ratios in compost systems and leads to finished compost products with benefits both as a soil conditioner and as a replacement for chemical fertilizers.

[27] Energy use has been found to be a major contributor of gross emissions of nitrogen oxides and sulfur oxides in ASP and bioreactor compost systems (Levis and Barlaz 2011). The analysis shows considerably less gross impact associated with windrow composting energy demand, which is dominated by windrow turning.

[28] Carbon dioxide, ammonia, methane, nitrous oxide, and VOCs constitute 99% of total emissions; further detail is provided in Quirós et al. (2014).

[29] Increasing aeration through pile turning and bulking agents decreased emissions of both methane and nitrogen dioxide, according to a review of management approaches (Pardo et al. 2015) and laboratory-scale experiments (Yang et al. 2013). Experiments by the California Natural Resources Agency also confirm the importance of maintaining aeration to limit methanogenesis (Silver et al. 2018). A study of GHG emissions from home-scale composting systems came to an alternate conclusion: that fewer mixing events led to reduced methane emissions, while increased temperature led to increased emissions (Ermolaev et al. 2014). Other authors, cited in the Ermolaev study, had a similar finding and attributed their result partly to the release during turning events of methane trapped within the pile. Oxidation of methane to carbon dioxide near the pile's surface, by methanogenic bacteria, could also explain part of the finding that reduced mixing leads to less GHG emissions (Jäckel et al. 2005).

[30] An analysis of a community-scale composting system showed a moderate to major (10% to >30%) reduction in gross impact for GWP, acidification, smog formation, and ozone depletion as a result of avoided nitrogen fertilizer production. Carcinogenic toxicity impact most strongly benefited from the avoided fertilizer production, with the avoided impact of fertilizer production leading to nearly net zero impact (Keng et al. 2020). Results for a wasted food aerobic composting system showed moderate to major reductions in gross GWP and minor to moderate reductions in gross energy demand due to avoided fertilizer production (Morris et al. 2017). An analysis of wasted food composting that included windrow, ASP, Gore and bioreactor compost systems identified moderate reductions in gross GWP for all systems as a result of avoided fertilizer production (Levis and Barlaz 2011). Other authors have identified only minor environmental benefits associated with avoided fertilizer production for GWP (Righi et al. 2013).

[31] The rate of compost application should be calibrated to consider all other nutrient sources such as legumes and chemical fertilizers, to avoid over-fertilization. Spreading equipment should be capable of metering compost application to avoid over-applying. Compost should not be applied when soil is frozen, and other best management practices related to organic amendment applications, such as manure, should be followed (NRCS and Agricultural BMP Task Force 2017).

[32] Food waste is a high-moisture waste product. It should be mixed with a dry, high-carbon material to absorb moisture and provide structure to the compost pile necessary for aeration and odor control. The ideal C:N ratio at the time of pile formation is 30:1 with a moisture content of 50% to 70% (Risse and Faucette 2017).

[33] See Trabold and Babbitt (2018) or Mayer et al.(2019)for more information.

[34] Some food seasonings have high chlorine levels, which are related to dioxin production potential.

[35] This statement refers to net impacts, which include olive production and olive oil processing in this analysis.

[36] Oily food waste is high in carbon (C:N ratio of 90:1) and can immobilize nitrogen in the soil, affecting crop yields if sufficient nitrogen is not available.

[37] The high carbon content of oily food waste can immobilize nitrate in the soil, prevent leaching, and make that nitrogen available for subsequent crops through re-mineralization, thus avoiding the need for additional fertilization.

[38] Of the reviewed studies that investigate land-applied wasted food, most indicate that direct application can benefit soil health, provide nutrients, and boost crop yields, but only if carefully managed temporally and co-applied with other organic wastes or inorganic fertilizers (Kumar et al. 2009; San Miguel et al. 2012).

[39] wasted food "must be applied at or below agronomic rates, and nuisance conditions (such as odors and flies) and negative impacts on surface and ground waters must be minimized" (Belcher and Aldrich 2006).

[40] Batuecas et al. (2019) commented on the risk that groundwater and surface water pollution could result from application of olive waste to agricultural fields, and a separate study on applying palm oil processing residues raised similar concerns (Embrandiri et al. 2012).

[41] "Landfill gas contains many different gases. Methane and carbon dioxide make up 90% to 98% of landfill gas. The remaining 2% to 10% includes nitrogen, oxygen, ammonia, sulfides, hydrogen and various other gases." (NYSDOH 2019).

[42] U.S. EPA's Waste Reduction Model (WARM) suggests that under "moderate" moisture conditions, an appropriate decay rate for food waste is 0.14 yr-1. For paper and wood products, under similar conditions, decay rates vary between 0.02 and 0.12 yr-1, with higher values indicating more rapid decomposition. Yard trimmings have an average decay rate under moderate moisture conditions of 0.2 yr-1.

[43] For example, lipids were calculated to contribute 59% to 70% of methane potential for a variety of food wastes in anaerobic environments, though there is a potential for lipids to inhibit methanogens (Lopez et al. 2016). The authors note the need for further research to clarify the role of lipids in wasted food decomposition.

[44] Reasonable ranges for decay rate and oxidation factor selection may lead to wasted food emissions varying from 0.23 to 0.71 MTCO2e (metric tons of CO2 equivalent) per ton of food waste (CARB 2017).

[45] Some studies suggest that typical EPA decay rates are too low (Wang et al. 2013), which would lead to underestimation of the portion of methane produced early in the landfill's life, before the installation of robust LFG capture systems.

[46] The carbon storage potentials of other commonly landfilled organic materials such as cardboard, newspaper, grass, and dimensional lumber were estimated to be 55%, 84%, 53%, and 88%, respectively (U.S. EPA 2020b).

[47] One study that modeled the landfill as "a 'dry tomb' where microbial activity is suppressed" assumed that 43% of landfilled carbon was sequestered (Yoshida et al. 2012). This value is considerably higher than the 16% estimate provided by WARM and is unlikely to apply in moist regions. Additional research has shown that landfill degradation is more limited in arid regions (Jain et al. 2021).

[48] Landfill leachate is the predominant contributor to eutrophication potential in several studies (Edwards et al. 2017; Lee et al. 2020; Slorach et al. 2020).

[49] Lee et al. (2017) found that when their LCA model was altered to increase LFG collection in the early stages of cell development, GHG emissions could be reduced by 27% compared to a less aggressive gas collection strategy.

[50] The authors noted that use of these factors was uncertain but was based on the best available information at the time.

[51] Zan et al. (2020) suggested that just 6% of the total COD in wasted food is degraded over a 4-hour sewer residence time. For comparison, direct GHG emissions from landfilling wasted food are approximately 16–30+ times greater than these marginal additional sewer emissions from added wasted food (Zan et al. 2020). In the U.S. context, this 6% figure may overestimate actual sewer emissions, given that a GIS-based analysis simulating the residence time of 3,422 U.S. sewer systems found that the median residence time of wastewater is approximately 3.3 hours (as opposed to Zan et al.'s 4-hour basis), with larger municipalities having longer residence times (Kapo et al. 2017).

[52] About 1,265 of the roughly 16,000 WRRFs in the United States process biosolids with AD (ASCE 2021; U.S. EPA 2023b).

[53] Concern that additional wasted food in sewers will drive solids deposition and clogging has not been universally borne out (Bolzonella et al. 2003). Further study and decision support tools are needed to navigate potential trade-offs between wasted food conveyance and sewer performance. For instance, various studies of long-term impacts have shown that, when high-specific-gravity wasted food (e.g., bones and egg shells) and/or FOG-rich wasted food are excluded, marginal additional sewer deposits from wasted food are small and have a minor impact on sewer performance (Mattsson et al. 2015). Mattsson et al. (2014) used cameras to monitor the effects of food waste disposers on small-diameter residential pipes across three Swedish municipalities. Despite disposer usage correlating with more deposits, especially in pipe segments with sags or gentle slopes, these deposits were small and appeared to have a minor impact on sewer performance (Mattsson et al. 2014).

[54] Source separation requires separate collection and transportation infrastructure that is not currently available in all communities, and sewage collection avoids hygiene and odor problems that can arise during other forms of collection and transportation (Kaur et al. 2019).

[55] A survey of local municipal authorities and waste management operators revealed that organizations providing organic feedstocks to AD operators can expect to meet contamination requirements of less than 5% on a total weight basis (Ogden et al. 2022).

[56] The animal feed pathway can address contamination concerns through preprocessing. Vermicomposting can recover the value of the food waste (in this case nutritional value) by transferring it to a different carrier (like fly larvae), avoiding the issue of feeding animals food waste with physical contaminants (Nick Hacheney and Sally Brown 2017).

[57] Heterogenous wasted food feedstocks can be difficult to formulate for appropriate animal nutrition.

[58] In a 2019 survey of composters in California, 48% reported that contaminated waste streams limit their ability to accept wasted food (CalRecycle 2019). Regulations governing commercial composting facilities in Washington State limit physical contamination in compost piles to less than 1% of wet weight with a plastic content of not more than 0.25% (State of Washington 2019).

[59] https://www.epa.gov/air-emissions-factors-and-quantification/emissions-estimation-tools

[60] Their reported values range from -4300, -3090, -162 and +536 kg CO_2e when preventing (source reduction), donating, digesting and landfilling 1 metric ton of wasted food, respectively. The source

reduction pathway receives a credit for avoiding food production, as does food donation. The AD and landfilling options do not receive this credit. Stated differently, the wasted food collected for donation comes with a negative carbon footprint, so that additional transportation and other processing may produce some GHG emissions but leaves the overall pathway with a large, net negative carbon footprint. Food waste enters the AD and landfilling pathways with a neutral (zero) carbon footprint, given its status as a waste product. These modeling options describe the convention, used in this report, to compare wasted food pathways to source reduction, wherein source reduction/donation have a virtual net negative impact while wasted food enters other pathways burden free (i.e. neutral environmental impact).

[61] An analysis of AD, composting, controlled combustion, and landfilling of wasted food found that management pathways have small GWP impacts (and potentials to reduce impact) compared to food production (Slorach et al. 2020).

[62] In 2021, over 65% of landfilled waste was sent to facilities with active energy recovery projects, and nearly 50% of collected LFG was sent for energy recovery (U.S. EPA 2022a).

[63] Including 4% used as alternative daily cover for landfills

[64] Excluding scenarios where cover crops are harvested for bio-energy.