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Sustainable Manufacturing and the Circular Economy

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Disclaimer

Authors

The authors of this report are:

- Kristina Armstrong, Oak Ridge National Laboratory (ORNL)
- Greg Avery, National Renewable Energy Laboratory (NREL)
- Arpit Bhatt, National Renewable Energy Laboratory (NREL)
- Robin Burton, National Renewable Energy Laboratory (NREL)
- Alberta Carpenter, National Renewable Energy Laboratory (NREL)
- Joe Cresko, Department of Energy (DOE)
- Caroline Dollinger, Energetics
- Wenquan Dong, University of Tennessee, Knoxville
- Chukwunwike Iloeje, Argonne National Laboratory (ANL)
- Diane Graziano, Argonne National Laboratory (ANL)
- Jeffery Greenblatt, Emerging Futures, LLC
- Colin McMillan, National Renewable Energy Laboratory (NREL)
- William Morrow, Lawrence Berkeley National Laboratory (LBNL)
- Scott Nicholson, National Renewable Energy Laboratory (NREL)
- Sachin Nimbalkar, Oak Ridge National Laboratory (ORNL)
- Darlene Steward, National Renewable Energy Laboratory (NREL)
- Sarang Supekar, Argonne National Laboratory (ANL)
- Kiran Thirumaran, Oak Ridge National Laboratory (ORNL)
- Shubhankar Upasani, National Renewable Energy Laboratory (NREL)

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List of Acronyms

AD	abiotic depletion, anaerobic digestion
ADP	abiotic depletion potential
ADF	abiotic depletion factor
AMMTO	Advanced Materials and Manufacturing Technologies Office
AMO	Advanced Manufacturing Office, U.S. Department of Energy
ASOP	absolute surplus ore potential
BAU	business as usual
BEV	battery electric vehicle
Btu	British thermal unit
CAGR	compound average growth rate
CBECS	Commercial Buildings Energy Consumption Survey
CCUS	carbon capture, utilization, and storage
CD&D	construction demolition and debris
CMI	Critical Materials Institute
DOE	U.S. Department of Energy
EAF	electric arc furnace
ECRA	European Cement Research Academy
EIA	U.S. Energy Information Administration
ESC	electrode-separator compound
EV	electric vehicle
EOL	end of life
EU REACH	European Union Registration, Evaluation, Authorization, and Restriction of Chemicals
FFP	fossil fuel potential
FLW	food loss and waste
FRP	fiber reinforced polymer
FSC	food supply chain
GDP	gross domestic product
GFRP	glass fiber reinforced polymer
GHG	greenhouse gas
GJ	gigajoule
GREET	Greenhouse gases, Regulated Emissions, and Energy use in Technologies Model
GT	metric gigatons
GWP	global warming potential
HDPE	high-density polyethylene
HEV	hybrid electric vehicle
IEDO	Industrial Efficiency and Decarbonization Office
LHV	lower heating value
LiB	lithium-ion battery
Li-ion	lithium-ion
LCA	life cycle analysis
LCI	life cycle inventory
LCIA	life cycle impact assessment
LCO	lithium cobalt oxide
LFP	lithium, iron, phosphate
LiB	lithium-ion battery
LIBRA	Lithium-ion Battery Recycling Analysis Model
LLDPE	linear low-density polyethylene
LMO	lithium, manganese oxide

LPDE	low-density polyethylene
LWI	landfill, wastewater treatment, and incineration
MAT	microwave-assisted technology
MF	material footprint
MF/cap	material footprint per capita
MJ	megajoules
MME	magnetic-manipulated electrode
MT	metric tons
MMT	million metric tons
MRF	materials recovery facility
MSW	municipal solid waste
NCA	nickel, cobalt, aluminum
NCM	nickel, cobalt, manganese
NEPA	National Environmental Policy Act
NiMH	nickel-metal hydride
NMC	nickel, manganese, cobalt
NMCA	nickel manganese cobalt aluminum
NREL	National Renewable Energy Laboratory
ODP	ozone depletion potential
PDF	probability density function
PET	polyethylene terephthalate
PP	polypropylene
PPP	purchasing power parity
PS	polystyrene
PVC	polyvinyl chloride
PVDF	polyvinylidene fluoride
QR	quick response code
REE	rare earth elements
RELOG	Reverse Logistics Model
RFID	radio-frequency identification
R&R	recycle or recover
TBtu	trillion British thermal unit
UNEP	United Nations Environmental Programme
U.S. EPA	U.S. Environmental Protection Agency
W&R	Wholesale and Retail
xEV	any electric vehicle

Executive Summary

An Approach for Evaluating Sustainable Manufacturing

The manufacturing industry predominantly follows a linear model of production in which raw materials are mined or grown, transformed into manufactured goods, used for a period, and then discarded. However, this once-through flow of materials to manufactured products is not sustainable. Many high-grade ore bodies have been consumed, increasing the energy and material impacts per ton of mined material. At the other end of the cycle, there are challenges with the efficiency and effectiveness of our waste disposal system.

In this report, we apply an approach for evaluating the sustainability of different circular economy strategies that quantifies projected material consumption and disposal but also evaluates associated life cycle environmental impacts to be able to understand the trade-offs. This approach allows the U.S. Department of Energy’s Advanced Manufacturing Office (now the Industrial Efficiency and Decarbonization Office and the Advanced Materials and Manufacturing Technologies Office) to understand the current state of knowledge about the technology, where data and knowledge gaps are, and what the research needs are to help improve material efficiency.

The concept of the circular economy first appeared as early as 1966, and the term first appearing in 1988 (Kneese 1988) as a response to the implications of natural resource limitations with the economic growth that was occurring (Winans 2017). The Ellen MacArthur Foundation popularized the concept, developing definitions and goals of a circular economy that were initially material-centric but have been extended to also mitigate climate change, biodiversity loss, waste, and pollution problems.[1] More recent research has looked at how a circular economy can be a significant driver to reduce the impacts of climate change (Circle Economy 2021) (Hertwich 2020).

In 2008, the United States had the second-highest global material footprint, consuming 8.26 metric gigatons (GT) of raw materials compared to China’s 16.32 GT. From 2013 to 2018, U.S. municipal solid waste (MSW) and construction demolition and debris (CD&D) grew from 750 to 840 million metric tons (MMT), which represents an average compound annual growth rate (CAGR) of 2.26%. The left side of Figure ES 1 shows the relative contribution of materials to the total MSW and C&D waste generated by the United States in 2018. These statistics do not include industrial waste generation.

In 2012, the United States produced 526 MMT of recycled goods, generating 681,000 jobs, $37.8 billion in wages, and $5.5 billion in tax revenue. Despite this potential, today, 75% of material going to landfills is recyclable (Recover, Inc. 2017). The portion of waste recycled varies enormously by type of material. As shown on the right side of Figure ES 1, reclaimed asphalt pavement and concrete have the highest recycled rates, and plastics have the lowest recycled rate.

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[1] “What is a circular economy?” Ellen MacArthur Foundation, https://ellenmacarthurfoundation.org/topics/circular-economy-introduction/overview

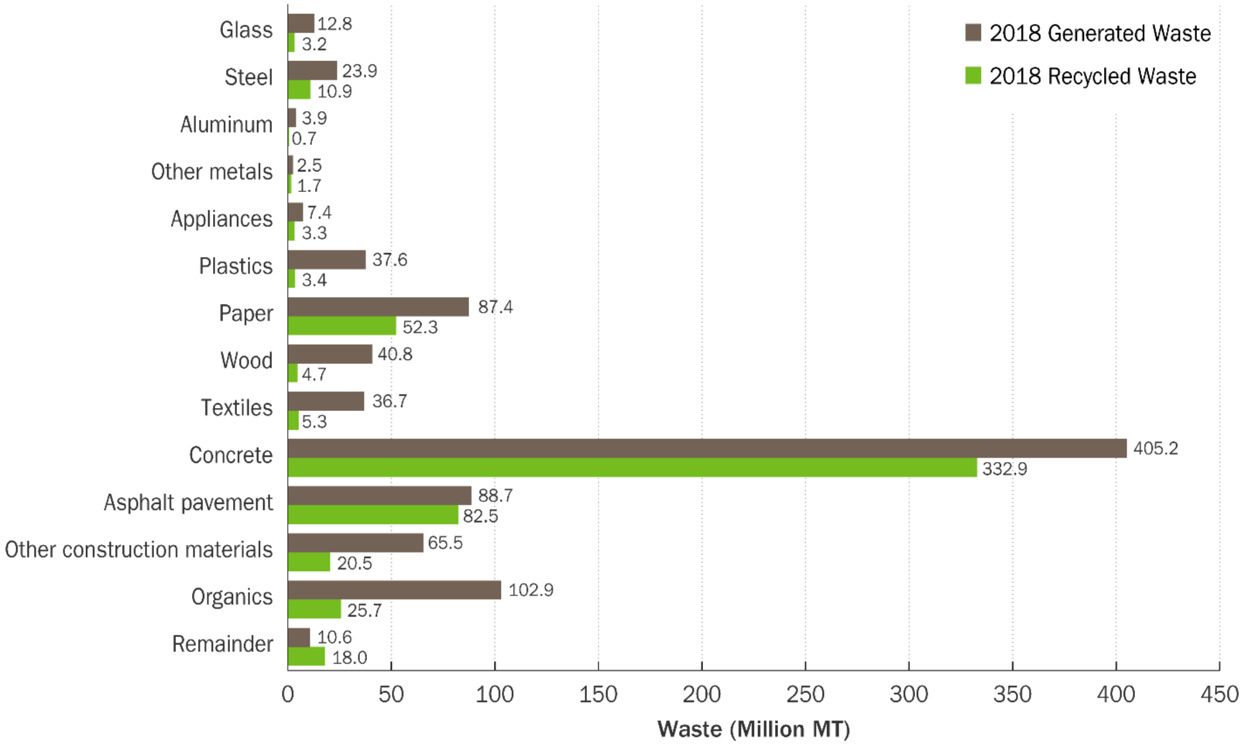

Figure ES 1. 2018 quantities of U.S. waste generated and recycled by material category in million metric tons

Calculations in figure based on data in *Advancing Sustainable Materials Management: 2018 Fact Sheet* (U.S. EPA 2020).

Given the Earth's finite resources and capacity to transform wastes, manufacturing must adopt a sustainable approach to survive in the long term and to maintain a healthy symbiotic relationship with the society and ecosystem it serves. Sustainable manufacturing adopts a life cycle approach, which recognizes the sequential and interwoven stages of a product system, from acquiring or generating raw materials from natural resources to final disposal. This approach includes the generation, transportation, and use of energy, which is required for any material transformation process and has its own nonzero material flows.

The circular economy builds up the life cycle approach as an intentionally restorative or regenerative industrial system that aims to eliminate waste or material losses[2] through the superior design of materials, products, systems, and related business models. It replaces the linear economy end of life (EOL) thinking with restoration, shifts to renewable energy, and elimination of toxic chemicals that impair reuse. And the circular economy reframes waste to include both pre- and post-consumer materials, as a resource to be collected and refined ideally (although not realistically) an infinite number of times into useful forms, provided worse economic, social, and environmental impacts are not inflicted.

The transition to a circular economy represents a strategic opportunity for the United States to pioneer new technologies with international demand. These circular economy technologies can be used as tools

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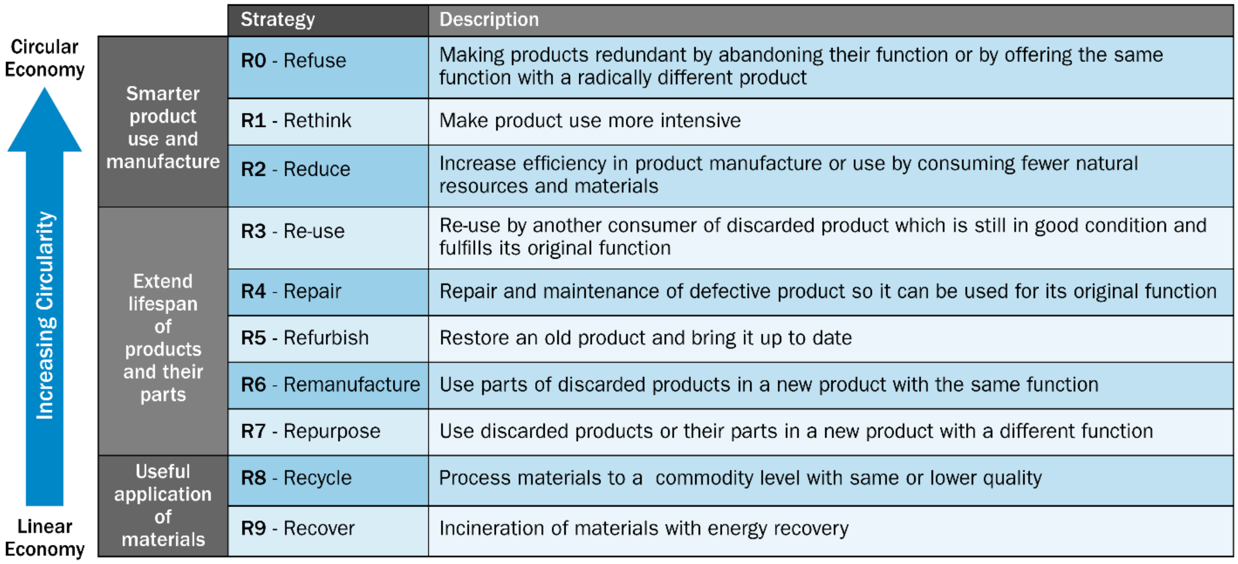
[2] Waste is generally considered to be resources that no longer are an asset and provide no value to the holder, but this definition does not adequately capture materials that are lost to the environment through emissions. Circularity, however, does include all resource losses to include materials, energy, water, emissions to air, water and land and material waste from end of life that goes to the landfills, incineration, or the environment.

to enable decarbonization, increase resource productivity, ensure sustained access to scarce resources, and extend the economic value of materials and products. Collectively, the circular economy can provide competitive advantages for domestic manufacturers and strengthen domestic supply chains ranging from critical materials to commodities.

Circular Economy Strategies

Companies employ several strategies to improve the sustainability of their products and increase circularity in their supply chains. While there are different perspectives on what is included in circularity, the literature currently points to 10 different strategies, collectively referred to as Re-X (Figure ES 2); the case studies in this report evaluate different known technologies and approaches that relate to some of these strategies.



	Strategy	Description
Smarter product use and manufacture	**R0** - Refuse	Making products redundant by abandoning their function or by offering the same function with a radically different product
	R1 - Rethink	Make product use more intensive
	R2 - Reduce	Increase efficiency in product manufacture or use by consuming fewer natural resources and materials
Extend lifespan of products and their parts	**R3** - Re-use	Re-use by another consumer of discarded product which is still in good condition and fulfills its original function
	R4 - Repair	Repair and maintenance of defective product so it can be used for its original function
	R5 - Refurbish	Restore an old product and bring it up to date
	R6 - Remanufacture	Use parts of discarded products in a new product with the same function
	R7 - Repurpose	Use discarded products or their parts in a new product with a different function
Useful application of materials	**R8** - Recycle	Process materials to a commodity level with same or lower quality
	R9 - Recover	Incineration of materials with energy recovery

Figure ES 2. Circular economy strategies (collectively Re-X) with descriptions and circularity ranking

After Potting et al. (2017), which is based on Rli (2015).

Some of the strategies are focused on product design and increasing the efficiency of material and energy use in manufacturing. Material efficiency can be achieved by reducing the amount of material used to manufacture a product or producing less waste, either through more material-efficient manufacturing or by using waste materials (both pre- and post-consumer)[3] in another product or process. These strategies include waste reduction, light-weighting, volume reduction, reduction of fossil fuel consumption, emissions reduction, and industrial symbiosis.

Other strategies look to prolong the in-use life or improve the use efficiency of products and components, including reuse, repair, refurbishment, remanufacturing, and repurposing of products that would

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[3] Pre-consumer waste is waste that is produced during the manufacturing process, sometimes referred to as manufacturing waste or in-plant waste. Post-consumer is the waste generated at the end of life of products. (ISO 2016)

otherwise have been discarded. Designing more products for circularity targeting one or more of these ReX strategies is needed to ensure materials can be retained in the economy.

Potting et al. (2017) indicates that recycling and energy recovery are the least circular of the ReX strategies, however they are the most widely recognized circular economy strategies. Energy and material recovery from waste streams includes approaches such as waste incineration and anaerobic digestion of MSW and wastewater to produce methane. Currently, 86 facilities in the United States recover energy from MSW. Combined, they produce 2,720 megawatts (MW) of power and process more than 28 MMT of waste per year.

A lateral sustainability strategy uses more environmentally benign materials in manufacturing processes and products, which impacts the entire life cycle of products. This type of strategy facilitates application of some of the other Re-X strategies that might be hindered by inclusion of toxic or hazardous materials. For example, reducing toxic materials in manufacturing can reduce the environmental impacts of product supply chains, improve worker safety and health, reduce manufacturing waste disposal costs, and improve the products' recyclability. Using environmentally preferable materials can also impact the energy consumed and carbon emissions during product manufacturing, use, and EOL management.

Case Studies

To demonstrate this material accounting and trade-off framework approach, in this report we present five manufacturing case studies that examine relevant pathways and strategies for moving toward the sustainable manufacturing goal of conserving resources by minimizing material waste and nonrenewable resource extraction. The case studies apply circular economy strategies to understand not only the material impacts from extraction and disposal but also the associated impacts on energy, greenhouse gas (GHG) emissions, and, where possible, the economic cost of those strategies.

The case studies employ three main analytical approaches:

- **Material Consumption and Waste Generation:** Analyze how the technologies, processes, or strategies evaluated in the case studies can reduce primary material consumption and reduce waste generation.
- **Trade-Offs:** Analyze manufactured product stocks and flows to provide estimates of the technical potential to reduce material consumption across manufacturing subsectors. Compare benefits, as well as energy, carbon, and other environmental impact trade-offs of alternative manufacturing options to inform transparent research and development (R&D) decision-making.
- **Adoption:** Analyze how adopting strategies or technologies can help minimize waste generation and resource extraction and understand the associated life cycle impacts.

We use two methods to quantify the potential reductions in resource consumption and waste generation achieved by the technologies and strategies and identify the trade-offs:

- *Wedge analysis* is a conceptual framework used to quantitatively describe economic activities that impact greenhouse gas (GHG) emissions, energy, and material and monetary flow relative to a reference trajectory.

- *Life cycle impact assessment* (LCIA) metrics are used to evaluate and understand the magnitude and significance of the environmental emissions and resources used for a product system throughout the product, process, or service life cycle.

Figure ES-2 summarizes the manufacturing sectors and circular economy strategies covered by the case studies.

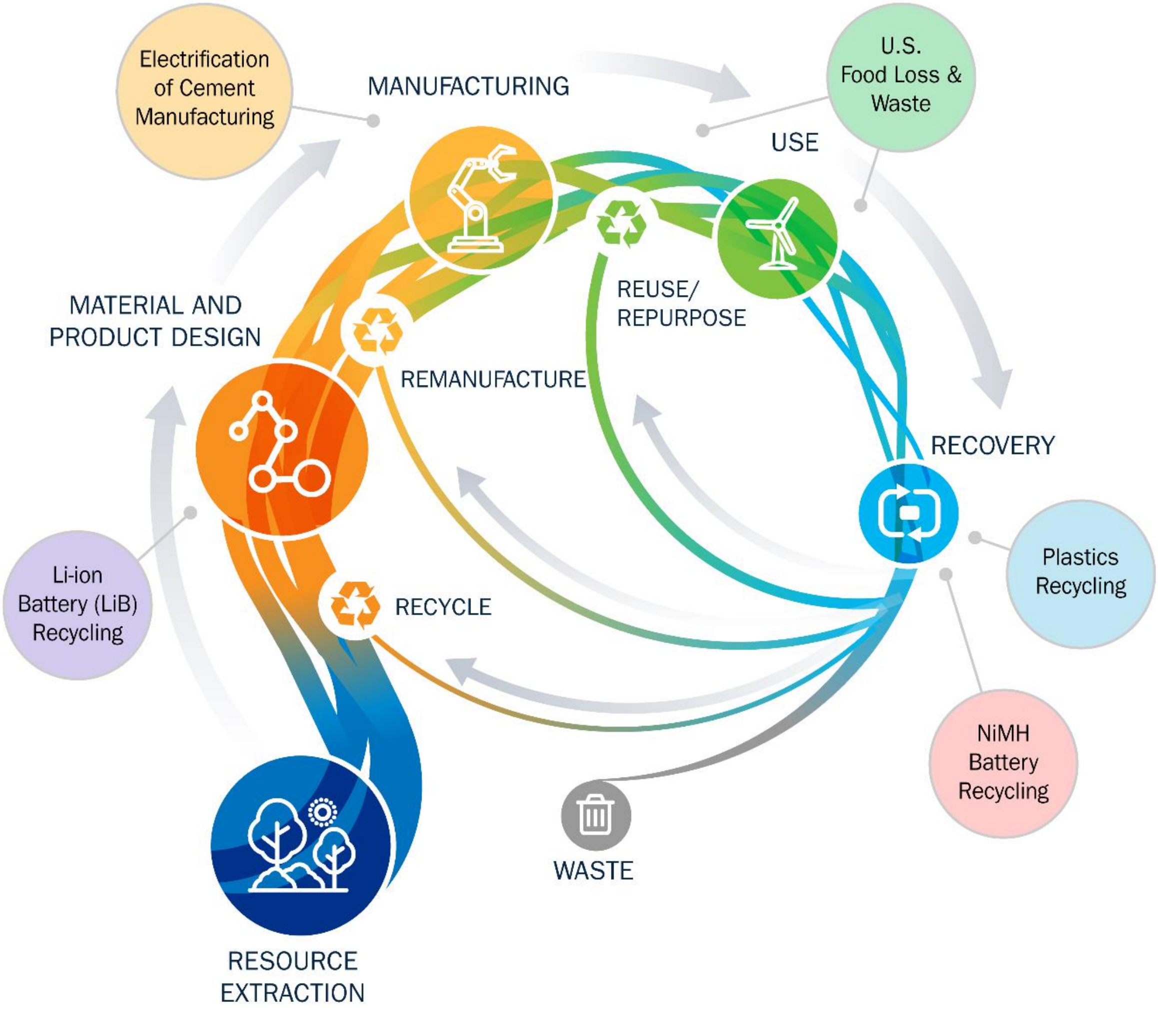

Figure ES 3. Sectors and circular economy strategies covered by the case studies in this report.

Reducing Food Loss and Waste

Food harvesting, manufacturing, distribution, and preparation contribute significantly to climate change. The entire food supply chain contributes 30% of global energy consumption and accounts for up to 29% of GHG emissions (Fabiola and Dalila 2016, U.S. EPA 2020). The expected growth of food demand will add more pressure on natural resources and intensify environmental impacts. While electrification and greener energy sources are viable paths to reducing energy use and GHG emissions, increasing the efficiency of the food supply chain (FSC) without further compromising ecosystems' quality and biodiversity is essential to reducing the global impact of the FSC.

The FSC has five stages: on-farm production, manufacturing, distribution, wholesale and retail (W&R), and consumption. Our study analyzed the amount of U.S. food loss and waste (FLW), GHG emissions, and energy consumption for each stage in 2016. This study also determined the amount of waste going to nine different EOL pathways, some of which are reuse pathways (food donation, animal feed, industrial uses, anaerobic digestion, composting, and land application) while the rest are considered waste (landfill, wastewater disposal, and incineration [LWI]). Overall, in 2016, the total U.S. FSC required 4,787 trillion British thermal units (TBtu) of on-site energy (site-electricity, fuel, and manufacturing energy of fertilizers, chemicals, and animal feed) or 7,258 TBtu of source energy, which converts on-site electricity use to grid-level energy consumption. This results in 984 MMT CO_2 equivalent[4] (CO_2 eq) emissions from energy consumption and other on-farm activities (e.g., enteric fermentation, manure management, soil management, and field burning).

On-farm production includes agricultural activities at the farm and those occurring before products reach the manufacturer's gate. This stage results in the third least amount of FLW, but the most energy use (site or source) and by far the most GHG emissions (because of the high non-energy GHG emissions).

Manufacturing food, which includes processing, manufacturing, and packaging, was the largest contributor to FLW. However, most of the FLW was reused for other industrial sectors or land application, leaving the least amount of waste landfilled or incinerated. This stage uses the third-most energy (site or source) and has the third-largest GHG emission.

Distribution, which includes shipping food products from manufacturers to wholesalers and retailers, generated the least FLW. However, over one-half of the FLW was disposed of as waste. This stage requires the least energy and has the fewest GHG emissions.

W&R, including warehousing and grocery stores, generated the second-lowest amount of FLW, primarily from consumer rejection and overstocking, with over one-half going to waste. Because this stage only keeps food in stable conditions without any preparation or transportation activities, it has the second-lowest energy demand and GHG contribution.

Consumption, which includes households, food services (e.g., restaurants and cafeterias), and food banks, contributed the third-largest amount of FLW. However, very little of this food waste is reused, resulting in the highest mass of waste of all stages. Because of the cooking and refrigeration activities in this stage, it is the second-largest user of energy and contributor to GHG emissions.

This case study identifies several technologies and practices that could be adopted to reduce FLW along the FSC. The case study then narrows its focus to examine the impact of actions that can be taken in the manufacturing stage that could result in FLW reductions in the later stages (distribution, W&R, and especially consumption). We found that extending perishable product shelf life and introducing standardized labeling can result in annual reductions of FLW (37.2 MMT), food demand (52.1 MMT), on-site energy consumption (159 TBtu), and GHG emissions (62.2 MMT CO_2 eq) by 2050. Moreover, this study demonstrates that although extending a product's shelf life by additional days could reduce FLW and food demand, the marginal impact of extending a product's shelf life is concave in terms of the days extended. According to our analysis, extending the shelf life by 2 days will bring more marginal

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[4] In this case study, CO_2 eq refers to the equivalent amount of CO_2 emitted that would result in the same global warming impact. Emission rates of non-CO_2 GHGs are multiplied by a global warming potential factor to estimate the equivalent CO_2 impact over 100 years.

impact (2.1% increased savings per additional day of shelf life) than prolonging the shelf life by 1 or 3 days (1.8% and 1.4% per additional day, respectively). The trade-offs between waste reduction and the costs of new technologies need to be understood before new technologies are adopted.

Recycling Plastics

Of the 32 MMT of plastic waste present in U.S. MSW in 2017, an estimated 74% was sent to the landfill. In this work, we analyze the potential for novel plastic separation and upcycling technologies to reduce landfilled plastic waste streams, and estimate material, energy, and GHG impacts.

The plastics considered in this analysis include low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), high-density polyethylene (HDPE), polyethylene terephthalate (PET), polypropylene (PP) and poly-vinyl chloride (PVC). Combined, these plastics accounted for approximately 79% of plastics in MSW for the United States in 2017.

Plastic waste-handling technologies for this case study are concentrated in two broad categories: separation and chemical recycling/upcycling. The first methodology evaluated is triboelectrostatic separation, a technology that uses an electric charge to separate plastic flake streams of similar densities or varying colors. This process allows for automated separation of the aggregate polyethylene stream into its specific types and removes polyvinyl chloride (PVC) impurities from the PET stream.

A second technology is PVC solvent separation, which is a recycling technology that applies specifically to waste PVC plastic. The solvent process extracts and precipitates PVC from the ground-up plastic waste stream, filters out any residual impurities from the solution, and then precipitates the PVC back out as freshly recovered polymer.

A third technology involves chemical recycling, where the plastic polymer molecules are broken down into their constituent monomers, which are then repolymerized. For this third technology, we consider a process known as PET deconstruction via glycolization.[5] This technology forms an unsaturated polyester that can be used to create new PET products and can also be used for a composite material known as fiber-reinforced polymer (FRP). This material is used in high-performance applications, such as wind turbine blades and snowboards, making it a higher-value product than the original PET on a per-mass basis.

We devise two scenarios corresponding to two assumptions about the rate of plastic waste collection over the forecast period of 2020–2050: a Baseline scenario and an Aggressive scenario. In both scenarios, plastic waste diversion from the landfill is the primary metric calculated by the model; we then use these results to derive estimated energy and emissions impacts.

The Baseline scenario assumes current collection rates of the polymers of interest (HDPE, LDPE, LLDPE, PET, and PVC) continue unchanged for entire forecast period. Overall, the estimated combined diversion from polyethylene terephthalate (PET) chemical recycling, HDPE separation, and PP separation technologies reaches 3 MMT per year by 2050.

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[5] Glycolization can handle some but not a lot of impurities in the feedstock. This analysis assumes a pure stream of PET for glycolization.

The Aggressive scenario assumes a much more ambitious plastic waste collection scheme unfolds by 2050: an 80% collection rate for PET and a uniform 50% collection rate across all non-PET plastics. In this scenario, recycled plastic increases substantially, reaching 18 MMT per year by 2050.

This study applied estimates of material quantity diversions from the adoption of these separation and upcycling methods to estimate energy and GHG impacts relative to conventional (fossil-derived) polymer production. From these estimates, we analyzed trade-offs between increased material recovery versus additional energy input and GHG emissions. Ultimately, a better understanding of such trade-offs will inform discussions about the transition from traditional linear consumption models to circular economies of material use and reuse.

We find that waste reduction and energy requirement reductions align for the PET upcycling and solvent separation technologies considered in this case study analysis. In economic terms, the marginal benefit of each additional quantity of waste diverted by these methods includes a reduction in energy use and GHG emissions.

In contrast, we estimate triboelectrostatic separation to have an inverse relationship between waste reduction and energy and GHG reductions. For each additional metric ton of plastic separated using the triboelectrostatic method, we estimate an additional 50 gigajoules (GJ) of energy requirement and approximately 21 kilograms (kg) CO_2 eq per kg relative to manufacturing from conventional fossil-based methods.

Recycling Electric Vehicle Lithium-Ion Batteries

Numerous studies and market analyses project that the market for electric vehicle (EV) lithium-ion batteries (LiBs) will continue its current upward trend for the foreseeable future. The rapid increase in demand coupled with supply chain bottlenecks has created a business opportunity for recycling EOL LiBs. Recovered high-value materials would have a ready market in battery manufacturers eager to reduce costs and supply chain uncertainty.

However, the rapid evolution of the LiB market also poses threats to recyclers. Building new recycling facilities or repurposing existing facilities requires recyclers to make significant investments in anticipation of a supply of raw material, in the form of EOL LiBs that does not yet exist at the scale, likely needed for a profitable operation. In addition, the accelerated turnover in LiB technology and changes in chemistry might reduce the demand for raw materials that are currently in short supply. This eventuality would reduce the pressure on manufacturers to find alternative supplies and accept recycled material. In the longer term, as advanced batteries reach their EOL, reduced concentrations of target materials in the battery waste stream might make recycling more expensive, and recovering materials may be unprofitable even if prices remain relatively high.

In this case study, we compare three whole life cycle scenarios to elucidate the material, economic, and life cycle environmental impact of potential future changes in EV LiB design and EOL management. The modeled scenarios represent variations from a baseline of vehicle battery material demand growth and static battery chemistry, which provides a consistent basis of comparison for the three scenarios.

In the Design for Disassembly scenario, EV LiBs are designed to facilitate EOL battery diagnostics, testing, dismantling, and refurbishment (e.g., through replacement of wiring). In this scenario, batteries that are not suitable for refurbishment and batteries that have reached the end of their “second life” are recycled to recover cathode metals via pyrometallurgy. Together, the sales of refurbished batteries and pyrometallurgical recycling result in a reduction in demand for virgin materials of more than 40% in 2040 and over 70% in 2050 in comparison to the baseline demand projection without recycling.

In the Design for Recycling scenario, LiBs are recycled using a direct recycling process that recovers LiB cathode material in its original compound structure, which can be reused without costly decomposition to elemental metals. By 2050, the supply of recovered cathode reduces the demand for virgin cathode metals by 83% compared to the baseline.

The Material Substitution scenario explores the implications of continuing the current trend toward reduced cobalt content in EV LiBs. In this scenario, the substitution of more common materials (nickel and aluminum) and the reduction in the concentration of valuable cobalt in the EOL battery recycling stream drive recyclers to improve the economics of recycling by recovering more materials. By 2050, the demand for virgin cathode metals (cobalt, nickel, lithium, and manganese) tracked in this study is reduced by 85% compared to the baseline.

The Design for Disassembly scenario preserves robust profits despite large variations in refurbished battery and recovered metals prices. However, this strategy did not reduce demand for virgin materials as much as the other two scenarios.

The Design for Recycling scenario had the highest potential profit because of the large fraction of batteries recycled using the direct recycling process and the high value of recovered cathode material in comparison to the cost of the recycling process. However, this scenario also had the greatest uncertainty in revenue because of the potential for a large price reduction for cathode material and the possibility that recyclers would recover cathode materials that are outdated and no longer in demand. The direct recycling technology envisioned in the Design for Recycling Scenario is not yet commercial and recovery of function has only been demonstrated experimentally and for certain chemistries. Therefore, the scenario model of full adoption of direct recycling, but without any capability for post-recovery changes to the chemistry of the cathode material, is still a very aggressive assumption.

The Material Substitution scenario had the lowest potential profit and significant uncertainty in profits because of the reduction in the cobalt concentration in the EOL battery stream and considerable uncertainty in the future prices of recovered metals. However, this strategy had the greatest reduction in demand for virgin materials tracked in this study.

The potential impact of evolving battery chemistries on the salability of recovered intact cathode material (i.e., the impact of Scenario 3 on Scenario 2) was modeled to assess the potential economic impact of recovering cathode material that is no longer in demand for the EV market. If high-Co batteries with obsolete chemistries were recycled using direct recycling starting in 2030, the potential lost value from being unable to sell the recovered cathode material would exceed $1 billion at baseline cathode prices. This is an example of the way in which combinations of innovations could have both positive and negative impacts for the environment and end-of-life process economic viability.

Life cycle assessment for high-cobalt chemistry (nickel-manganese-cobalt in a 1:1:1 molar ratio, respectively [NMC111]) and low-cobalt chemistry (NMC811) batteries, with and without recycling, revealed that recycling significantly reduced environmental impacts in all categories for both chemistries. Transitioning to lower cobalt concentrations reduced environmental impacts for all categories except for human noncarcinogenic toxicity and freshwater ecotoxicity, which were higher for NMC811 than for

NMC111. In all cases, the impact reduction with recycling was much greater than the impacts of changing chemistries.

The case study showed that none of the strategies was clearly superior in every metric. All three design-to-EOL scenarios reduced the demand for virgin materials by larger percentages than the baseline demand did. However, the potential profit and revenue uncertainty for recyclers varied widely between the scenarios. Generally, recovering more materials and preserving intact materials (i.e., intact batteries in the Design for Disassembly scenario and intact cathode in the Design for Recycling scenario) improved profitability for EOL management. A combination of design and EOL strategies (e.g., refurbishing as many batteries as possible and recycling the remaining batteries using hydrometallurgy) is likely the most promising method for ensuring long-term economic viability for recycling of EV LiBs.

Recycling Nickel-Metal Hydride Batteries

Nickel-metal hydride (NiMH) batteries contain nickel and rare-earth elements (REE) that, if recovered, could support domestic manufacturing of emerging clean energy and transportation technologies. Nickel is an important element in LiB cathodes. Didymium is an alloy of REEs and is used to produce magnets for energy-efficient motors in EVs and wind turbines. Other important materials present in NiMH batteries include cerium, lanthanum, and hydroxides of REEs and cobalt. At the time this report was written, the United States had limited capacity to produce nickel or didymium. Systematic recovery of these materials from EOL hybrid electric vehicle (HEV) NiMH batteries could offset the U.S. reliance on imports and contribute to securing their long-term supply. Recycling is a key strategy for securing critical material supplies and is a pillar of the circular economy.

However, recovering materials from EOL products faces challenges with economics, logistics, and environmental impacts. In the private sector, recovered materials compete in existing markets with materials sourced from primary sources, effectively constraining the capital and operating costs for recycling facilities. While materials recycling and recovery would reduce depletion of non-renewable mineral resources from which those materials are extracted, most commercially-established recycling technologies rely on processes that are not too dissimilar from those used to extract virgin mineral resources. As such, they create trade-offs with other environmental impacts through use of other natural resources and emissions of harmful compounds into the air, water, and land. Logistics are challenged by EOL products that are geographically dispersed, variable in quantity and quality over time, and difficult to collect. For these reasons, systematic analysis of diverse EOL technologies and reverse supply chain challenges will be essential to achieve circular economy goals.

This study applies an integrated analysis framework to explore the viability of NiMH HEV battery recycling in the United States. The framework comprises five main components: recycling process techno-economics, EOL battery scrappage and distribution, materials substitution wedge analysis, reverse logistics optimization, and environmental impact assessment. It considers two potential future scenarios for EOL battery availability: conservative and optimistic. The conservative scenario projects a decline in NiMH HEV batteries and represents the conservative limit on the current trend of LiBs replacing NiMH batteries in HEVs. The alternative optimistic scenario fixes the fraction of NiMH batteries at the 2019 value and maintains this proportion going forward. This scenario represents the optimistic possibility in which economic considerations or evolving technology keep NiMH batteries competitive with substitute technologies. While unlikely, it represents a useful upper bound for this analysis. For both cases, we assume only 50% of EOL batteries become available for recycling and the rest goes to competing pathways, such as remanufacturing (restoring to “as new” quality) and refurbishment (restoring to “like new” quality) for secondary use (including outside the transportation sector), export (via used car market), and simple disposal. This assumption keeps our overall projections for both scenarios on the conservative side.

We find that spent NiMH batteries could be a significant source of recycled nickel and REEs such as didymium, and cerium and lanthanum, producing 4.3–12.6 thousand MT of REE product and 5.7–16.8 thousand MT of nickel in the analysis time horizon of 2025–2050. The ranges reflect conservative and optimistic scenarios for the use of NiMH chemistry in HEVs in the future. Depending on which scenario for NiMH batteries is realized in the future, two or three recycling plants with a recycling capacity of up to 5 thousand MT/year of spent NiMH batteries would be sufficient to recycle all spent NiMH batteries, assuming 50% of spent batteries were recycled.

Compared to current total U.S. demands for didymium in the U.S. [<160 MT/year; (Castillous 2016)] and primary nickel [(~100,000 MT/year (USGS 2021)], the recycling complex does not provide nearly enough product to meet U.S. demand. Yet, given that the domestic production of this material remains insignificant, recycled didymium and nickel can still make a difference. For didymium, considering the plan to resume production of sintered NdFeB magnets in North Carolina (Lombardo 2020), recycling can provide a portion of the required didymium feedstock. At a reported capacity of roughly 2,000-MT/year NdFeB, and assuming 30% didymium content, demand for the didymium alloy for such a plant would be on the order of 600 MT/year. In this case, NiMH battery recycling could meet approximately 8–16% of the magnet facility’s didymium demand (Iloeje, et al. 2022).

However, transporting and recycling 1,000 kg of NiMH batteries leads to a fossil resource depletion potential of 385 kg oil equivalent, global warming potential of 1,295 kg CO_2 eq, and an abiotic mineral resource depletion of 3.75 kg copper equivalent. In all three categories of environmental impacts examined, chemical and reagent use in anode and cathode recycling and solvent extraction are the predominant contributors.

Since reagent use is the biggest driver of environmental impacts, one could look at acid regeneration to reduce acid consumption and the hydroxide consumption for neutralization of the spent solvent. However, such regeneration processes require additional chemicals such as sulfuric acid and consume additional energy (McKinley and Ghahreman, 2018), which would add to the environmental impacts and cost of the overall process. As such, a cost-benefit analysis of acid recovery measures is advised to ascertain its merits in reducing overall reagent use and environmental impacts of the battery recycling process. Viable solvent-free processes for leaching and solvent recovery would need to be developed to address this trade-off.

Important factors to consider in subsequent studies include uncertainty in EOL availability, technical and economic feasibility of a mixed-feed comprised of other battery chemistries to mitigate risks and uncertainty in feedstock availability, regional regulatory constraints, and existing infrastructure. Further, the study should be extended to account for uncertain markets for recovered product, cash flow, and conditional value of risk analysis.

Electrifying Cement Manufacturing

After water, concrete is the most-consumed material in the world (Gagg 2014). The United States is the world’s third-largest producer of cement—the most essential component and main binder for concrete.

In addition to cement being an extensively used resource, cement production is highly intensive in energy consumption and emissions. Manufacturing cement requires heating raw materials to high temperatures in a kiln in a fuel-intensive process known as pyroprocessing. The process creates what is known as clinker: small lumps of stony residue that are ground to a powder and combined with other ingredients to produce cement. The clinker production step accounts for 90% of the total energy consumed in the cement manufacturing process. Further, the decomposition of calcium carbonate that takes place during clinker production is associated with significant carbon dioxide (CO_2) emissions, 58% of total CO_2 emissions from the U.S. cement industry.

This case study evaluates the energy, emissions, and material impacts of applying electrification to the cement/clinker manufacturing processes and provides a valuable example of electrifying thermal processes in industry. These thermal processes typically derive heat from burning fossil fuel, resulting in the “leakage” of material from the economy in the form of unrecoverable fossil fuels. The fossil fuels must also be mined, processed, and transported to the facility for use, adding to the energy and emissions footprint. Using low-carbon electricity for process heating tasks in the manufacturing sector can contribute to achieving material efficiency by reducing the use of coal and petroleum-based materials. Further, by replacing fossil resources with low-carbon electricity generated from infrastructure reliant on highly recyclable materials, better circularity of the materials used in the cement life cycle is achieved.

This case study aims to understand the environmental impacts of implementing electric heating solutions in the clinker production process. Other thermal heating solutions that don’t directly impact the electric grid infrastructure and the associated material needs have not been considered. The study explores two pathways for the electrification of pyroprocessing:

- Electric calcining and conventional sintering: a hybrid system that uses electric preheaters in combination with traditional fuel-fired rotary kilns to achieve clinkerization
- Electric calcining and electric sintering: a completely electric solution for the thermal heating steps.

Our analysis found that deploying the two electrification pathways with existing renewable grid technologies reduces fossil energy consumption and emissions. However, the more material-intensive renewable electric generation infrastructure leads to additional depletion of metals and other critical materials. The analysis quantifies the trade-off between increased material depletion versus GHG emissions for different grid scenarios. While most electrification studies detail impacts on emission and energy use, its impact on life cycle material use is not commonly studied.

While not ideal from a circular economy perspective, the trade-off between emissions driven by non-recoverable fossil resources and recyclable metals/minerals is inevitable. The overall material circularity can be improved, however, under the electrified scenario by implementing proper material recycling practices and technological improvements. A more material-efficient renewable energy infrastructure with greater material reuse and/or increased generation efficiency would substantially reduce material consumption in the life cycle of cement. Thus, a crucial step in qualifying electrification as truly sustainable is to develop suitable methods to recycle and reuse renewable energy equipment at EOL. Furthermore, developing more efficient electrotechnology can lead to changes in the dynamics of the impacts estimated.

Conclusion

The case studies evaluated in this report examined the challenges and opportunities for adopting different circular economy strategies. Most of the case studies demonstrated that applying circular economy strategies can have decarbonization benefits, but trade-offs are clearly associated with abiotic resource depletion and other environmental impacts. Uncertainties are associated with each case study around (unknowable) future conditions; it is vital not only to recognize these uncertainties but to also interpret and use the information from the analyses with these uncertainties in mind.

The analysis also helped identify some of the challenges in evaluating the impacts of adopting circular economy approaches and technologies. Though the circularity benefit to making products last longer is clear, this strategy can sometimes conflict with realized energy savings, particularly for industries experiencing rapid technological change, such as consumer electronics.

Adopting circular economy strategies also presents significant challenges in forecasting future markets and infrastructure, and these challenges can make it difficult for business owners to defend new infrastructure investments.

Because of the nature of the circular economy, impact analysis needs to capture the complexity and economic, environmental, and logistic challenges of EOL and material efficiency approaches. The approaches and strategies are interconnected with other industries and nonindustrial activities in the United States and the global economy (as most industries rely on global supply chains). Manufacturers need to be familiar with the several EOL pathways, stakeholders, and the multiple opportunities and markets that can create demand or supply for the EOL materials on both the front and back ends of the value chain.

The case studies in this report also helped identify aspects of the circular economy that require additional focus and effort. Design solutions are needed for managing and recovering products and materials at EOL and will improve with products innovatively designed for reuse, design for remanufacturing, design for recycling, and other sustainable design strategies. A more comprehensive understanding of the impacts of landfilling (beyond energy and economics) would provide a more holistic view of the full environmental and social impacts.

For any circular economy strategy or technology to have an impact, it must be adopted and deployed in the market. The case studies in this report do not cover the impacts of policy or standards, nor do they include the broader private sector challenges around technology adoption, but they should be incorporated into analysis for future studies. Future analysis might consider how implementing extended producer responsibilities will impact EOL management of materials, what barriers manufacturers will face, and the options for overcoming them. Researchers should explore the incentives and disincentives that could increase industrial material efficiency and decarbonization. Additionally, future analysis could provide insight on how markets might affect the scale and rate of technology adoption.

Additional study is also needed to explore the social dimensions of circular economy approaches, including employment; health and safety of employees, customers, and communities; customer satisfaction; product responsibility; and social and environmental justice. Broad adoption of a circular economy will likely result in growth or shrinking of different sectors, new facilities, and operations in different parts of the country, all of which will impact the affordability of essential products; employee, customer, and community health and safety; environmental injustices associated with local air and water pollution; and job opportunities and accessibility for disadvantaged persons.

The authors have attempted to shed light on the important topic of material circularity as it applies to sustainable manufacturing in the United States. We hope readers find this report valuable.

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1 Introduction

1.1 Sustainable Manufacturing

Sustainability is defined globally as "meeting the needs of the present without compromising the well-being of future generations" (United Nations General Assembly 1987). For the United States, sustainable development means a commitment "to create and maintain conditions under which humans and nature can exist in productive harmony, that permit fulfilling the social, economic and other requirements of present and future generations" (U.S. NEPA 1969).

Sustainable manufacturing is the "creation of manufactured products through economically sound processes that minimize negative environmental impacts while conserving energy and natural resources" (EPA 2021). The Department of Commerce extends that definition to require safety for employees, communities, and consumers (Howard n.d.). Sustainable manufacturing is critical to achieving sustainable development. Like most sustainability strategies, sustainable manufacturing requires a life cycle approach. A life cycle is the "consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal." Life cycle assessment is the "compilation and evaluation of the inputs, outputs, and the potential environmental impacts of a product system throughout its life cycle" (ISO 2006a).

Thus, to be sustainable, manufacturers innovate to achieve continual improvement in life cycle resource efficiency (e.g., energy, materials, water) while maintaining economic viability, avoiding environmental externalities (e.g., pollution, carbon generation), and improving environmental justice (e.g., not locating polluting plants in underprivileged communities). They also seek to expand technology innovation and adoption and increase safety for employees, communities, and consumers, within a full life cycle context that includes all time horizons (present, short-term future, and long-term future).

The circular economy is an emerging topic of sustainable manufacturing and a primary focus of this report. Kirchherr, Reike, and Hekkert (2017) defined the circular economy as an industrial system that is restorative or regenerative by intention and design, replacing the end-of-life (EOL) concept with restoration, shifting to renewable energy, and eliminating toxic chemicals, which impair reuse. The circular economy aims to eliminate waste through the superior design of materials, products, systems, and related business models. The circular economy concept has raised awareness of the impact of materials and how they are used in manufacturing and the economy more broadly. The circular economy can be considered a tool to support sustainability, but as with many strategies and technologies, it must be considered holistically, accounting for all the potential environmental, social, and economic benefits as well as trade-offs. In other words, not all circular systems are more sustainable than the equivalent linear system.

Currently, manufacturing industries predominantly follow an essentially linear path in which raw materials are mined or grown; these materials are transformed into manufactured goods, used for a time, and then discarded. It has become increasingly clear that a once-through flow of materials through the economy is not sustainable. Much of the current metals and minerals being extracted are coming from low-grade ore bodies (J. West 2011), increasing the energy and material impacts per ton of newly mined material. At the other end of the product life cycle, our waste disposal challenges are increasing, with insufficient technologies to be able to fully recover materials at EOL and limited access to landfill space in high-population areas.

To maintain and restore the natural environment and enhance the well-being of society, manufacturing and the complex worldwide supply chain that feeds production must be transformed from a linear to a circular model. We must ensure manufacturing moves away from the linear model of production by reducing material demand; introducing more sustainable materials and processes into the supply chain; and reusing, remanufacturing, and recycling manufactured goods.

1.2 Zero Waste, Material Circularity, and the Cost of Recovery

Extracting materials has both economic and environmental costs; sustainable manufacturing should prioritize minimizing both types of costs and ensuring that materials can be maintained to provide service to the economy indefinitely, or at least for as long as possible. In an idealized world with constant demand (no population growth and no increase in product demand), complete circularity would imply working toward both zero-resource extraction and zero-waste generation while eliminating, or at least minimizing, environmental impact.

Zero waste implies an idealization of a concept that may never be achievable in real-world practice. Walter (2019) found expert opinions divided on whether true material circularity (zero waste) is theoretically achievable. He quoted from the landmark book by Nicholas Georgescu-Roegen (1971), *The Entropy Law and the Economic Process*: “It is theoretically impossible to recycle everything (and hence the foundation of a circular economy is questioned).”

Walter (2019) also showcases a paper by Ayres (1999), who argues that, in principle, “mining” waste materials is the same as mining raw resources, although material concentrations and required energy inputs may differ. Thus, waste is simply another type of resource that can be mined and refined an infinite number of times into useful forms, provided that sufficient energy and a large enough resource base exist. Walter concludes that it is an academic question whether recycling everything is theoretically possible: “In practice it will definitely not be possible.”

The circular economic model is driven by the cost of material recovery from waste compared with the cost of extraction from raw resources. Here, *cost* is defined broadly to include environmental and social costs. In general, it is less expensive today to extract at least a portion of resources from ore materials than to recover them entirely from wastes, especially when considering the quality of the material recovered from waste (Cooper, et al. 2020) (Reck and Graedel 2012). However, as high-grade ores become depleted and the economic cost and externalities of procuring those resources increase, waste recovery will increasingly become the cheaper option.

Indeed, higher concentrations of some materials in waste streams could make recovery more competitive. For example, a battery recycling company led by Tesla cofounder J. B. Straubel “envisions a process so efficient that batteries coming from the mountain of electric cars being retired in coming years could be quickly stripped down, recycled for their core materials, and used to rebuild new power cells, creating a closed loop where hardly any materials are lost” (Higgins 2020). Other examples include the high recycling rates of structural steel (98%) (USGS 2022) and lead in batteries (99.7%) (Recycling Today 2017).

Work by Material Economics[6] has shown that for some commodity materials, including steel, aluminum, concrete, and, in some cases, plastics, it is far less carbon-intensive to recycle materials than to produce them from virgin ores—sometimes by a factor of 10 or more (Enkvist and Klevnan 2018, Exhibit 1.9). However, challenges include copper contamination in steel, limited ability for easier disassembly and poor sorting of plastic and aluminum materials (particularly with respect to grade), and lower collection fractions overall. Different materials have different levels of recyclability and ability to be recycled multiple times. Some abiotic materials (e.g., glass, steel, and aluminum) tend to be able to be recycled many times, but level of trapped contaminants increases over time. The newer chemical recycling processes being developed for plastics can provide virgin equivalent quality materials, but there is loss occurring in those processes, so they are not providing 100% recyclability. Some biotic materials (e.g., paper and cardboard) can be recovered but are very susceptible to contamination, but also have the option of being composted and returning the nutrients to the soil.

Moreover, some secondary impacts of a zero-waste strategy must be considered:

- Zero waste targeted to a specific subset material stream might have undesirable secondary effects on associated streams outside that subset, such as increased energy or other material use.
- Achieving a zero-waste goal may have deleterious economic, environmental, and energy impacts, especially if energy supply is not renewable. The marginal energy and economic requirements at the limit of zero waste are likely to increase substantially.

As a result, consuming less resources in the first place typically results in lower overall impacts.

Several other factors need to be considered beyond the immediate economics. The time dimension is critical as we try to retain materials in the economy in a functional manner for as long as possible and understand how quickly society adopts the technology or strategy. In total, wide-ranging challenges need to be overcome to achieve a zero-waste society. The scope of the problem is very broad and must include all aspects of the environment and society to ensure there are not unintended negative consequences.

1.3 The Opportunity for U.S. Manufacturing

The transition to a circular economy represents a strategic opportunity for the United States to pioneer new material recovery technologies with international demand. These technologies can enable increased resource productivity, ensure sustained access to scarce resources, and extend the economic value of materials and products. Collectively, the circular economy can provide competitive advantages for domestic manufacturers and strengthen domestic supply chains ranging from critical materials to commodities.

The U.S. Department of Energy's (DOE's) Office of Energy Efficiency and Renewable Energy has a vision and mission that supports a strong and prosperous United States powered by clean, affordable, and secure energy where America leads the transition to a global clean energy economy.[7] The DOE is well-positioned to lead the research, development, demonstration, and deployment (RDD&D) of new technologies that can increase energy productivity and competitiveness advantages for U.S. manufacturers. The series of ongoing RDD&D programs across the Office of Energy Efficiency and

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[6] Material Economics is a management consulting firm advising companies on the link between sustainability and business performance.

[7] DOE EERE Mission.

Renewable Energy and other DOE offices can advance reuse, repair, remanufacturing, refurbishment, recycling, and other circular processes (collectively Re-X) by developing cost-effective solutions to reduce energy and materials in the manufacture of goods and products and recover resources at EOL. Additionally, DOE recognizes that it needs to look beyond energy efficiency and account for carbon intensity to achieve the administration's decarbonization goals (Enkvist and Klevnan 2018).

Such innovations could be valuable for several reasons. Many clean energy technologies are relatively new and developing rapidly; making improvements today can yield greater benefits in the future as these technologies reach obsolescence (e.g., recovery of critical materials). Secondly, it will be necessary not only to power direct operations using clean energy technologies but also to ensure that these clean energy technologies are themselves manufactured according to circular principles. Finally, both clean energy technologies and other products today rely on imported materials, including critical materials, which make supply chains vulnerable.

Over the past decade, material criticality has attracted increased and focused attention. DOE (2011) and the U.S. Geological Survey (USGS) (Fortier, et al. 2018, 2019) have identified critical materials essential to energy technologies. Of the 35 minerals or mineral groups identified as critical by the U.S. Department of the Interior, 33 are used in energy applications, including batteries, light-weighting alloys, thermal barrier coatings, catalysts, solar cells, fuel cells, and magnets for energy-efficient motors. Among these critical materials are rare earths (including neodymium, praseodymium, and dysprosium), lithium (Li), cobalt (Co), manganese (Mn), and platinum-group metals (U.S. DOE 2011).

Despite having vast mineral resources, the United States has limited operational mining capacity for many materials critical to clean energy technologies, leading to U.S. manufacturers' dependency on foreign supplies. For example, in 2019, U.S. mines only produced a little over 12% of the total materials needed for the manufacture of rechargeable batteries (calculated as a percentage of global supplies): 0.4% of Co, 12.0% of rare-earth minerals (concentrate ore from the California (U.S. DOE 2011) Mountain Pass Mine processed in China), and negligible quantities of Mn and Li (USGS 2020).

Consequently, in the United States, companies are turning to other mitigating strategies. These include developing sustainable conventional and unconventional material resources; improving manufacturing efficiency; reducing material consumption via process innovation, material substitution, and product design; and recovering materials from products after their useful life (Eggert, et al. 2016, Gaustad, et al. 2018, Fortier, et al. 2019).

Cobalt exemplifies the need for mitigating strategies. The United States is the leading consumer of refined cobalt, which is predominantly used for rechargeable batteries. As Shedd, McCullough and Bleiwas (2017) and Amnesty International (2016) explain, increasing cobalt production to meet increasing demand is constrained by several factors:

- As a byproduct of other more abundant metals, demand for these other metals drives production of cobalt.
- The Democratic Republic of Congo, the leading Co mine producer, has a history of political instability and is considered by many to be a high-risk business environment.
- The Democratic Republic of Congo's unregulated artisanal mining sector relies on tens of thousands of poor miners, including children, who suffer environmental and health impacts.

Such challenges are the focus of the DOE-sponsored Critical Materials Institute (CMI), an Energy Innovation Hub led by Ames National Laboratory. Its research and development objectives are to diversify supply, develop substitutes, and drive reuse and recycling of critical materials. Developing circular processes can better use, recover, or replace these materials to ensure robust supply chains and foster further economic growth. More work is needed beyond the early-stage R&D to facilitate deployment and adoption of these technologies into the marketplace.

Historically, DOE and its Advanced Manufacturing Office (AMO) (now the Industrial Efficiency and Decarbonization Office (IEDO) and the Advanced Materials and Manufacturing Technologies Office (AMMTO) have focused on the energy intensity and energy efficiency of manufacturing processes. In the last decade, a focus has also developed on the carbon intensity and decarbonization of manufacturing. Now, a focus on the circular economy brings to the forefront efforts to improve use intensity, resource conservation, and material efficiency, as illustrated in Figure 1.

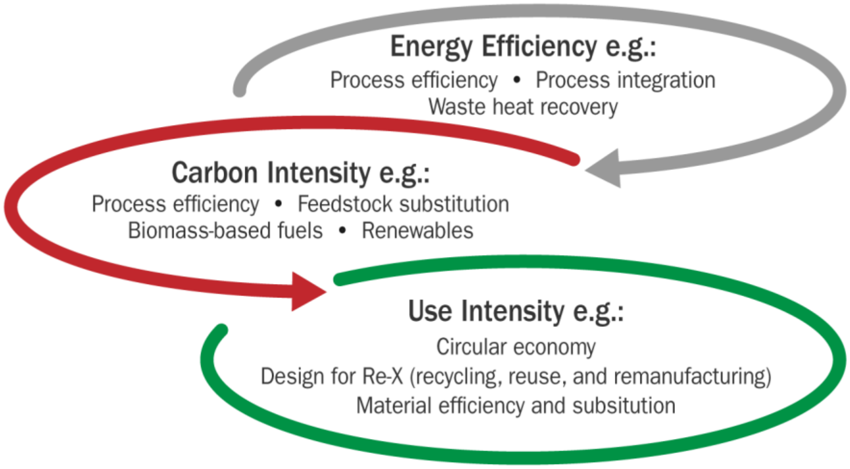

Figure 1. DOE and its IEDO and AMMTO sustainable manufacturing focus areas

1.4 The Challenge for U.S. Manufacturing

U.S. manufacturing is highly interconnected not only within the country but also around the globe. Manufacturing supply chains frequently cross international borders multiple times; very few are purely domestic, and raw materials are often sourced globally with primary and/or secondary processing taking place in intermediate international facilities. Moreover, depending on the product, different aspects of the assembly process may take place both domestically and internationally. Distribution to retailers is the only part of the process that must occur domestically. Manufacturers need to look across the full life cycle of their products and ensure that their processes are sustainable. Short of pulling as much manufacturing as possible back into the domestic manufacturing space (at a considerable expense often not viable economically), manufacturers need to have broader awareness of the supply chain and life cycle of their product(s), require certain environmental performance of their suppliers (and hold them accountable), and employ RDD&D and innovation within their domestic facilities that enhances sustainability.

1.5 Analysis Approach

Transitioning the U.S. manufacturing sector toward a more sustainable stewardship of natural resources requires a near-, mid-, and long-term perspective and coordination of investments across the RDD&D continuum. This report provides the analysis needed to quantify the potential for manufacturing to adopt materials, technologies, and practices that can potentially overcome the barriers to fully implement a sustainable manufacturing ecosystem and evaluate the trade-offs. The goal is to improve the sustainability of U.S. manufacturing, where material efficiency and other circular economy strategies are tools to this end.

We approach this work by presenting five manufacturing case studies to examine pathways and strategies for moving toward the sustainable manufacturing goal of conserving resources by minimizing material waste and nonrenewable resource extraction.

Before presenting the case studies, we first examine the current state of circularity in the United States, referencing the global state for relative comparisons. Next, we describe a set of circular economy strategies and identify those that are applied in the case studies. We then explain the two main methodologies used in the case studies—wedge analysis and life cycle impact assessment analysis.

The five case studies that follow discuss specific activities and opportunities within a subset of U.S. manufacturing, covering both the challenges and opportunities:

- The *Food Loss and Waste (FLW) Study* provides a detailed and comprehensive picture of the U.S. food supply chain (FSC), with a focus on food waste generation, disposal, and environmental impacts.
- The *Plastics Case Study* examines the potential for several technologies to handle the growing landfilled plastic waste stream. We estimate material, energy, and greenhouse gas (GHG) emissions impacts from several separation, recycling, and upcycling technologies.
- The *Lithium-ion Battery (LiB) Case Study* examines the role that innovations in electric vehicle (EV) LiBs and EV LiB EOL strategies could play in the economics of recycling and the future availability of recycled and recovered EV LiB materials.
- The *Nickel-Metal Hydride (NiMH) Battery Case Study* explores the structure and features of a large-scale infrastructure network for EV NiMH battery recycling.
- The *Cement Case Study* evaluates the energy, emissions, and material impacts of applying electrification to the cement/clinker manufacturing processes.

The case studies apply various circular economy strategies to understand not only the material impacts from extraction and disposal but also the associated impacts on energy, GHG emissions, and, where possible, the economic cost of those strategies. In evaluating these case studies, we quantify the reduction in resource consumption and waste generation potentially achieved by these technologies and strategies and identify the trade-offs.

The case studies use three main analytical approaches:

- Material consumption: Analyze how the technologies, processes, or strategies evaluated in the case studies can reduce primary material consumption and reduce waste generation.

- Trade-offs: Analyze manufactured product stocks and flows to provide estimates of the potential across manufacturing subsectors and compare benefits and trade-offs between alternative manufacturing options to transparently inform RDD&D decision-making.
- Adoption: Analyze how adopting strategies or technologies can help minimize waste generation and resource extraction and understand the associated life cycle energy consumption and GHG generation.

The scale of the challenge to U.S. manufacturing is immense, and, as the case studies demonstrate, it varies for different industrial sectors and even different product types. Through the case studies, the report identifies the scope of sustainable manufacturing opportunities in industry, as well as the research, innovation, and policy questions tailored to the specific needs of different industrial sectors. A concluding section summarizes our key findings.

2 Circularity of the U.S. Economy

Circle Economy's latest circularity gap report (Circle Economy 2021) shows a decline in the circularity of the global economy—the share of total material inputs that are recirculated, or "cycled"—from 9.1% in 2015 to 8.6% in 2017. While data are insufficient to indicate a clear trend, what data do exist are not encouraging. The drivers for this decrease in circularity, be it from increased consumption or decreased recycling, are also not clear, but the message is still the same – we need to improve how we are utilizing materials in the economy. The 2021 report also examined how circularity can support decarbonization and found that combining climate mitigation with circular economy strategies can put the world on a path by 2032 that limits global warming to below 2 degrees Celsius (°C).

2.1 Current State

In 2008, the United States had the second-highest global material footprint[8] in absolute terms, consuming 8.26 metric gigatons (GT) of raw materials compared to China's 16.32 GT (Wiedmann, et al. 2015); Figure 2a shows the material footprint breakdown by country. Notably, the U.S. domestic material extraction rate was only 4.82 GT, or 58% of its total consumption; subtracting exported materials of 1.47 GT, the remaining demand (4.92 GT) was imported from other nations. Developed countries tend to reduce their domestic material extraction in favor of importing materials, which hides a larger material footprint when revealing only domestic extraction.

Wiedmann et al. (2015) found that "two-fifths of all global raw materials were extracted and used just to enable exports of goods and services to other countries. This is far more than the 10 GT of direct physical trade of materials and products". In other words, trade itself has a huge impact on material consumption, primarily in the form of fossil fuel extraction.

In terms of per capita consumption, the United States is not an outlier. As Figure 2b shows, the United States' per capita consumption at 27.2 metric tons (MT) is comparable to that of Japan (28.5 MT) and considerably lower than Hong Kong, Singapore, and Luxembourg (all >70 MT). To some extent, these consumption rankings reflect a country's per capita wealth. Figure 2c shows the material import per capita for the 12 countries; again, while the United States is not at the top of the list, it is on par with other developed nations (i.e., Japan, the United Kingdom, and Australia). Thus, it is important to examine both domestic extraction and imported raw materials to fully understand the material footprint.

However, in terms of material consumption per unit of gross domestic product (GDP) (Figure 2d), wealthier countries tend to rank lowest; for example, the United States consumed approximately 0.65 kg/$, making it the ninth-lowest consuming country on this metric. Countries with much lower per capita GDP (Worldometer n.d.) tend to rank much higher (e.g., India, with a GDP per capita of only 6% of the U.S. GDP per capita, had a material consumption of 1.38 kg/$), although there are exceptions.[9] This is likely due to a higher proportion of a country's wealth derived from services rather than material goods. The data used by Wiedmann et al. (2015) is over a decade old and it would be valuable to have the analysis updated to understand the trajectory of U.S. material consumption.

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[8] Material footprint is defined by Wiedmann et. al (2015) as the full material requirement of a nation from a resource consumption perspective.

[9] Calculated from data in Wieidmann et al. (2015).

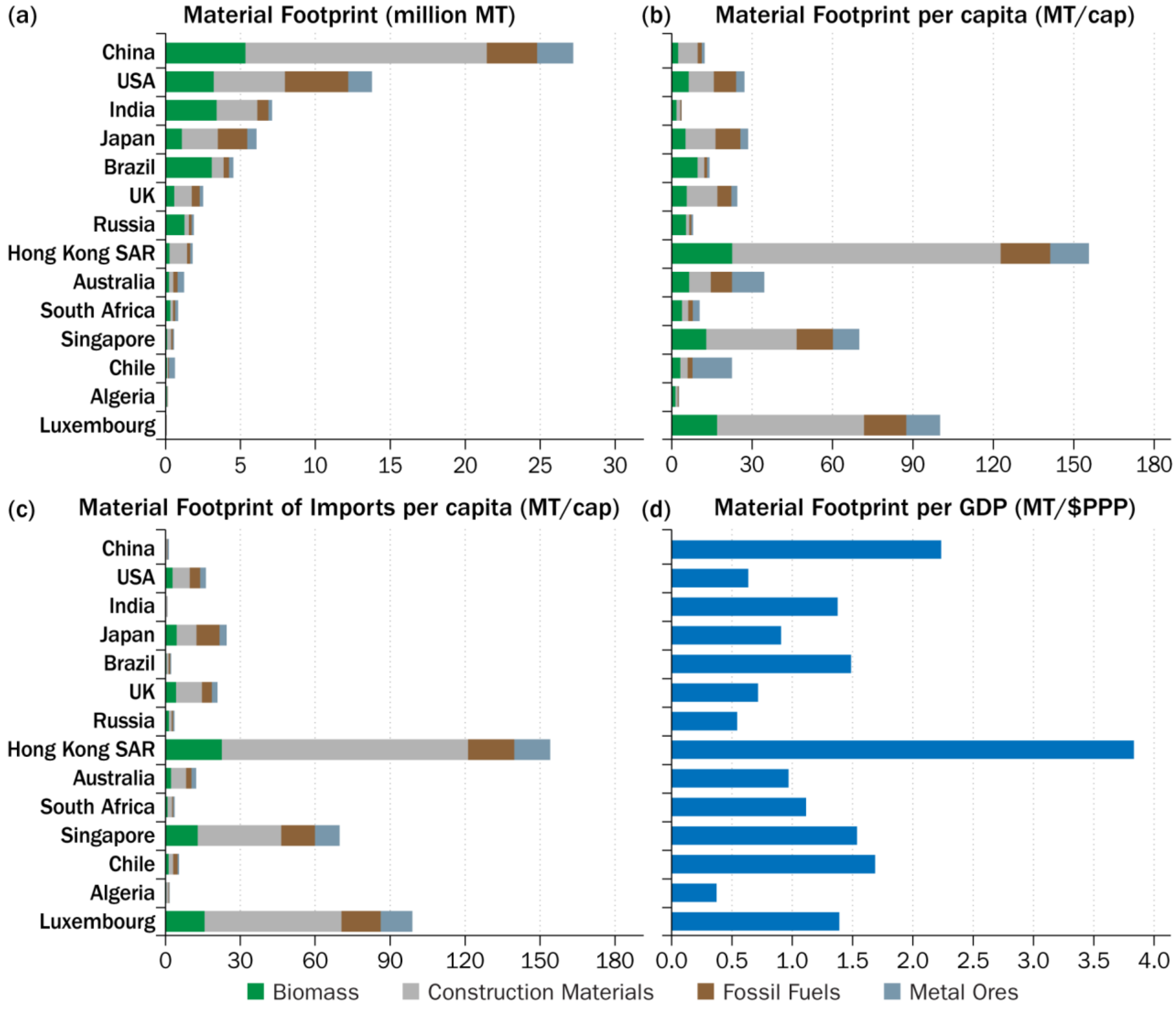
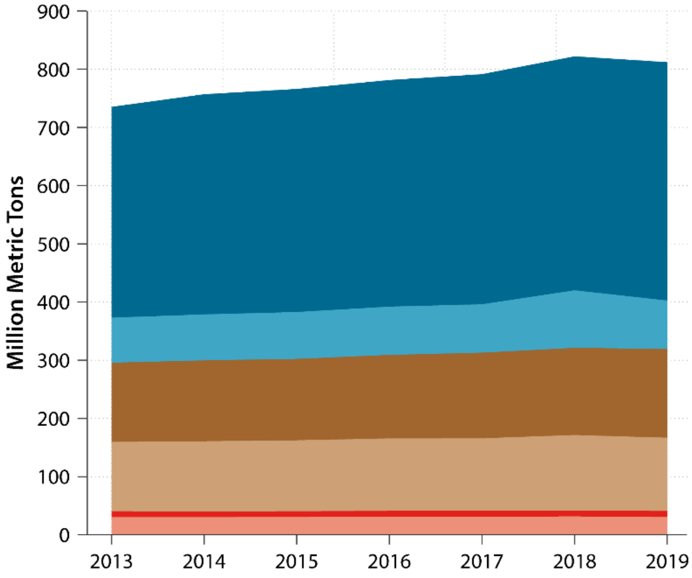


Figure 2. (a) Material footprint (billion MT), (b) material footprint per capita (MT/cap), (c) material footprint of imports per capita (MT/cap), and (d) material footprint per GDP of 12 high material footprint countries for the year 2008

Data Source: Weidmann et al. (2015), Figure 1

In terms of waste generation, from 2013 to 2018, U.S. municipal solid waste (MSW) and construction demolition and debris (CD&D) grew from 750 to 840 million metric tons (MMT), representing an average compound annual growth rate (CAGR) of 2.26%. These statistics do not include industrial waste which is not well documented in the United States; studies have estimated that industrial waste to around 36 times larger than MSW (Krones 2016). In addition, as Figure 3 shows, the percentages of recycled, landfilled, and combusted waste remained nearly the same, with recycling as the major stream (61.0%).



	2013	2014	2015	2016	2017	2018	2019
Recycled - CD&D	370.5	387.0	392.1	398.3	404.5	411.2	419.0
Recycled - MSW	79.3	80.9	82.5	85.0	85.2	101.3	85.2
Landfilled - CD&D	139.1	142.1	143.0	146.7	150.4	153.1	156.0
Landfilled - MSW	122.0	123.4	124.8	127.3	127.4	132.6	128.7
Combusted - CD&D	10.3	10.0	10.0	10.2	10.3	10.5	10.7
Combusted - MSW	30.1	30.1	30.4	31.0	31.0	31.4	30.3

Figure 3. Total U.S. municipal solid waste and construction demolition and debris (2013–2018)

Based on MSW trend data (EPA 2020b) and CD&D data from multiple annual MSW reports (EPA 2021); methodology for estimating CD&D found in (U.S. EPA 2018).

In 2018, U.S. MSW constituted 292 MMT while CD&D constituted 634 MMT. Combining these two categories to analyze waste by type of material, we find that the single largest contributor, by far, is concrete (43.8%), followed by organics (food waste, yard trimmings, and land clearing debris, 11.1%), reclaimed asphalt pavement (9.6%), paper (9.4%), and other construction materials (7.1%). The remaining mass flows constitute 26.1% in aggregate. See the breakdown in Figure 4.

The portion of waste recycled varied enormously by type of material: more than 80% for asphalt pavement and concrete, 67% for metals other than steel or aluminum, 60% for paper, and lower amounts for other materials, including significant stream categories such as organics (25%) and other construction materials (31%). Plastics have the lowest recycling rate (5% per Milbrandt, et al. (2022)) among all material classes listed, followed closely by wood (11%) and textiles (14%) (see Figure 4).

Surprisingly, the recycling rates of glass and aluminum, usually thought of as highly recycled waste streams, were 25% and 17%, respectively. In comparison, globally, up to 90% of aluminum is recycled in transport and construction applications; rates for aluminum beverage containers are 60% globally, and more than 90% in some countries (The International Aluminium Institute 2018). In 2018, the U.S. recycling rate for aluminum beverage containers, food containers, and other packaging was 35% (U.S. EPA 2020), but those rates were lower in aggregate. Likewise, glass containers achieve recycling rates above 90% in some countries, but the overall global rate is only 32% (Harder 2018), so U.S. recycling rates are on par with this global average. For the United States, U.S. Environmental Protection Agency (EPA) statistics reveal that 75% of material going to the landfills is recyclable; only 1% of plastic products and 1% of aluminum products are recycled. Put another way, 25 million plastic bottles are tossed in the trash every hour (Recover, Inc. 2017).

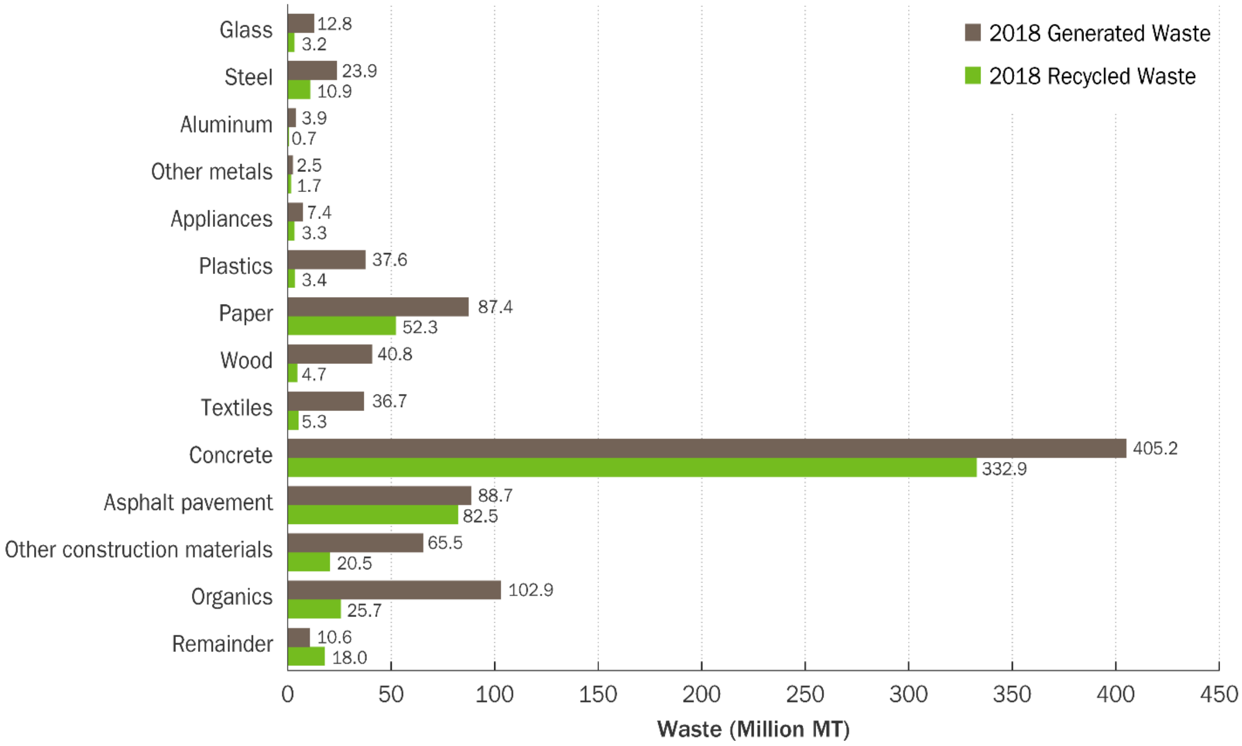


Figure 4. 2018 Quantities of U.S. waste generated and recycled by category in million metric tons

Calculations in figure based on data in "Advancing Sustainable Materials Management: 2018 Fact Sheet" (U.S. EPA 2020).

In 2012, the United States produced 526 MMT of recycled goods. Counting both direct and indirect economic impacts, the EPA reports that this activity generated 681,000 additional jobs, $37.8 billion in additional wages, and $5.5 billion in additional tax revenue compared to a world in which these goods were not recycled (U.S. EPA 2020). This activity represents 0.5%, 0.6%, and 0.8% of jobs, wages, and taxes, respectively, indicating an opportunity for growth.

2.2 Circular Economy Initiatives

A global movement is advancing toward the adoption of circular economy initiatives by industry. More than 800 companies across multiple sectors have adopted the Ellen MacArthur Foundation Circulytics measurement method to track their progress (Ellen MacArthur Foundation 2021). Moreover, more than 250 business have signed onto the Ellen MacArthur Foundation and United Nations Environmental Program (UNEP) New Plastics Economy Global Commitment to address plastic waste and pollution at its source, starting with packaging (Ellen MacArthur Foundation 2020).

U.S. industry is slowly starting to adopt various circular economy initiatives and business strategies. Twenty industrial partners are participating in the DOE Better Plants Program (U.S. Department of Energy 2020) waste reduction pilot. The pilot aims to establish waste reduction goals, track progress, identify and overcome barriers to reducing waste, and document best practices. Additionally, several large companies and industry associations have adopted circular economy strategies. The Alliance to End

Plastic Waste[10] was founded in 2019 with the mission of ending plastic waste through innovative and impactful solutions deployed at scale. The Alliance now has more than 80 member companies participating, including Dow Chemical, Chevron-Phillips, and P&G.

Armstrong has been looking at how to achieve zero-impact products since 1994[11]; the Armstrong Flooring's On&On Recycling Program provides a closed-loop process, recycling reclaimed materials into new Armstrong flooring.[12] Microsoft has a 2030 goal to achieve zero waste in its facilities and data centers (Microsoft 2020), 100% recyclable Surface devices in Organisation for Economic Co-operation and Development countries, and 100% recyclable packaging (Microsoft 2020).

Patagonia and Eileen Fisher have programs for refurbishing and reusing clothing as well as recycling clothing at EOL.[13] This concept is also being pursued by startup companies such as Circos, which offers a subscription-based clothing model for children under 2 years old (Ellen MacArthur Foundation 2020).

Other companies have adopted a products-as-service business model, allowing for tighter control of the products and increased use intensities. Xerox has provided printing services for several years, replacing the purchase of copy machines with leasing and providing maintenance. More recently, Signify (formerly Philips Lighting) began to offer customers lighting services that cover installation, maintenance, and management of the lighting product throughout its life cycle, without investing in new hardware up front.[14] In addition, other businesses offer zero or reusable packaging, either through bulk purchase options or providing packaging designed to be returned and reused.[15,16]

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[10] The Alliance to End Plastic Waste

[11] https://www.finchandbeak.com/1325/how-interface-makes-the-circular-economy.htm

[12] Armstrong Flooring On&On™ Recycling Program, https://www.armstrongflooring.com/commercial/en-us/sustainability/floor-recycling-program.html

[13] Patagonia Worn Wear and Eileen Fisher Renew, https://wornwear.patagonia.com/

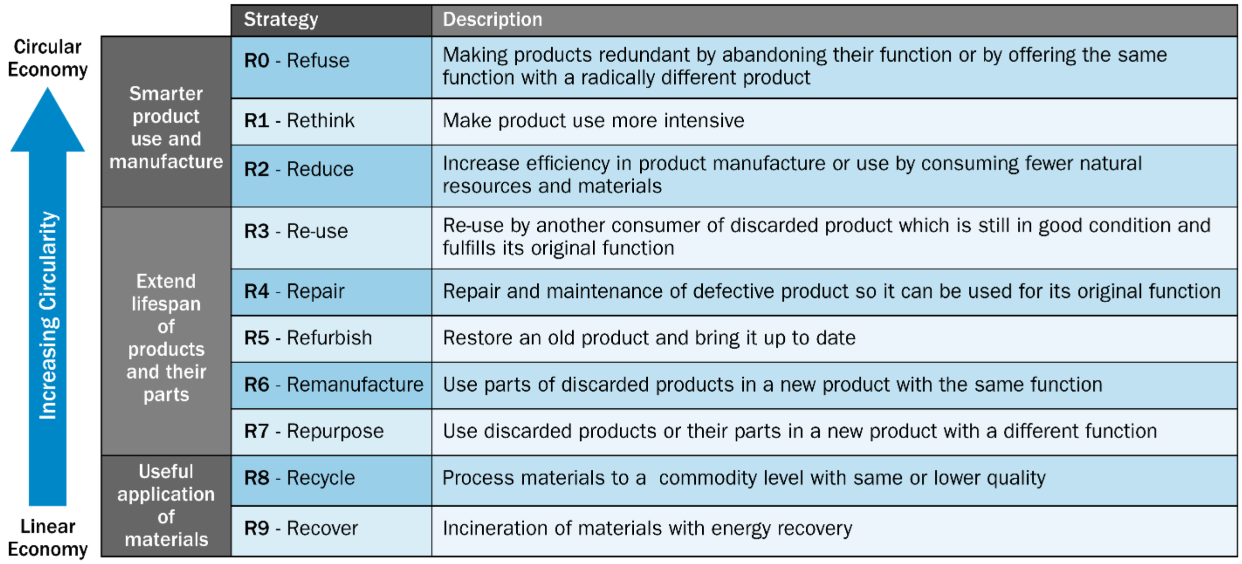
[14] Signify Life Cycle Services

[15] Litterless Zero Waste Grocery Guide

[16] Loop Designs Reusable Packaging For Everyday Products

3 Circular Economy Strategies

Linear materials and product life cycles are deeply entwined in the global economy. Thoughtful, coordinated, and sustained commitments throughout society will be needed to transition to a more circular model of product design, manufacture, use, and EOL management. However, companies that design and manufacture products stand at the center of efforts to reimagine the life cycle of materials. They may employ a wide variety of strategies to improve the sustainability of their products and increase circularity in their supply chains. The literature currently points to 10 different strategies, collectively referred to as Re-X, which are bucketed into three groups, as described in Figure 5.



	Strategy	Description
Smarter product use and manufacture	**R0** - Refuse	Making products redundant by abandoning their function or by offering the same function with a radically different product
	R1 - Rethink	Make product use more intensive
	R2 - Reduce	Increase efficiency in product manufacture or use by consuming fewer natural resources and materials
Extend lifespan of products and their parts	**R3** - Re-use	Re-use by another consumer of discarded product which is still in good condition and fulfills its original function
	R4 - Repair	Repair and maintenance of defective product so it can be used for its original function
	R5 - Refurbish	Restore an old product and bring it up to date
	R6 - Remanufacture	Use parts of discarded products in a new product with the same function
	R7 - Repurpose	Use discarded products or their parts in a new product with a different function
Useful application of materials	**R8** - Recycle	Process materials to a commodity level with same or lower quality
	R9 - Recover	Incineration of materials with energy recovery

Figure 5. Circular economy strategies (collectively Re-X) with descriptions and circularity ranking

After Potting et al. (2017), which is based on Rli (2015).

Olivetti and Cullen (2018) used four primary groups: the three listed above, plus one additional group, which are discussed in the sections that follow:

- Strategies that increase the efficiency of material and energy use in manufacturing (related to smarter product use and manufacture, R0-R2)
- Strategies that prolong the in-use life and/or improve the use efficiency of products and/or components (related to extended lifespan of products and its parts, R3-R7)
- Recovery of material and energy from products at the end of their useful life (related to useful application of materials, R8-R9)
- Use of environmentally preferable materials and technologies.

Many of these strategies can be complementary; for example, converting from a gasoline engine to an electric motor prolongs the in-use life of the motor and reduces the amount of maintenance materials used (e.g., oil). Further, an electric motor is smaller and lighter than an equivalent gasoline engine, combining the first three strategy groups.

However, implementing some strategies can preclude or reduce the effectiveness of others. For example, the use of biodegradable materials can reduce the environmental impacts of packaging but may hinder recycling. Certain attributes of products, primarily the product's in-use lifespan and technology turnover, can also determine whether a particular strategy is a good fit for a product.

Implementing sustainability and circular economy strategies usually involves key design innovations that improve the product's life cycle performance and/or enable new EOL options. Eco-design is a key enabler for sustainable and circular economy strategies and can address a variety of goals that typically fit into one or more of the following categories (in no specific order):

- Design for disassembly
- Design for increased durability, lifetime and/or performance
- Design for better performance and improved efficiency
- Design for biodegradation
- Reduction of complexity
- Reduction of toxicity and/or persistence of toxic substances in the environment
- Design for recycling, reuse and remanufacturing

The *Food Loss and Waste Case Study* in Section 5 evaluates advanced packaging technologies and practices to extend the life of food products. The *Lithium-ion Battery Case Study* in Section 7 explores the life cycle impacts of design innovations for disassembling, refurbishing, and recycling vehicle LiBs.

3.1 Strategies that Improve Material and Energy Efficiency in Manufacturing

Strategies that reduce the amount of material used to manufacture a product or produce less waste, either through more material-efficient manufacturing or through use of waste materials in another product or process, include:

- Waste reduction (e.g., additive manufacturing, reducing/reusing scrap materials from manufacturing and/or remanufacturing)
- Light-weighting and/or volume reduction (e.g., more efficient packing)
- Decarbonization to reduce fossil fuel consumption
- Industrial symbiosis (using the secondary products or waste from one process as the input into another process).

The *Cement Case Study* in Section 9 presents a life cycle scenario analysis of decarbonizing the clinker-making process in cement manufacturing, identifying challenges and solutions to improving energy and fossil fuel efficiencies. While not covered in any case study in this report, industrial symbiosis provides an interesting example of efficiency of material use that spans more than one company and can have regional benefits beyond the processes themselves.

3.2 Strategies that Prolong the In-Use Life of Products and/or Components

Strategies that prolong the life of products and/or components (hereafter, “products”) include refuse, rethink, reduce, reuse, repair, refurbishment, remanufacturing, repurpose that would have otherwise been discarded. The rethink strategy includes designing products for increased durability and reliability to ensure longer lifetimes. Remanufacturing, which is distinct from repair and refurbishment, restores the product to its original quality and performance specifications.

Historically, reuse, repair, and refurbishment have been applied to vehicles and, high-value equipment and electronics as well as for relatively low-tech products that can be repaired or refurbished using inexpensive processes. The Maytag repair man was an indicator of this approach. Additionally, the automotive industry provides a clear example of strategies and pathways for keeping vehicles in the economy for as long as possible through repair and refurbishment. The electronics industry in some cases has also employed strategies for extending the lifetime of products through remote updating as well as some refurbishment activities. An example of a relatively low-tech application is the collection and reuse of glass bottles, which was a common practice before World War II. However, innovations in plastic manufacturing and efforts to reduce the weight of bottles by manufacturing thinner and more fragile glass have largely replaced bottle collection in favor of single-use containers (Thierry, et al. 1995). While the average recycling rate for glass containers in Europe is currently estimated at 54%, the reuse rate is only 7% (Ramirez-Camperos, Nacheva and Tapia 2004). Refilling glass bottles reduces the overall need for glass and uses significantly less energy than manufacturing new bottles, either from virgin raw materials or recycled glass cullet (Ponstein, Meyer-Aurich and Prochnow 2019, 4R Sustainability 2009). However, heavier, more durable bottles are required, and transportation of the empty bottles can negate the energy benefits of reuse (Trombly and Fortier 2019).

Remanufacturing provides various sustainability and circular economy benefits, including energy savings, diversion of material from landfills, and creating a market for skilled labor. Parts that can be re-machined or components that can be replaced to restore the original performance and manufactured tolerances are good candidates for remanufacturing. Strong markets exist for remanufactured products, especially in the automotive industry, and evidence suggests that remanufacturing may be more profitable than original equipment manufacturing (Charter and Gray 2008).

3.3 Recovery of Materials and Energy from Waste Streams and End-of-Life Products

Recycling is the most widely recognized strategy for recovering materials from products at EOL. Energy can also be recovered from waste streams. Common strategies include waste incineration and anaerobic digestion of MSW and wastewater to produce methane. Currently, 86 facilities in the United States recover energy from MSW. Combined, they produce 2,720 megawatts (MW) of power and process more than 28 MMT of waste per year. Material and energy recovery from waste streams is also a common strategy in industrial symbiosis (as mentioned in Section 3.1).

Recycling strategies are applied in several case studies: In Section 6, the *Plastic Case Study* examines plastic waste recycling. In Section 7, the *Lithium-Ion Battery Case Study* explores the life cycle impacts of design innovations for recovering critical materials by recycling EV LiBs. The *Nickel-Metal Hydride Battery Case Study* in Section 8 evaluates the recovery of critical materials from NiMH vehicle batteries.

Figure 6 illustrates the sectors and strategies covered by the case studies.

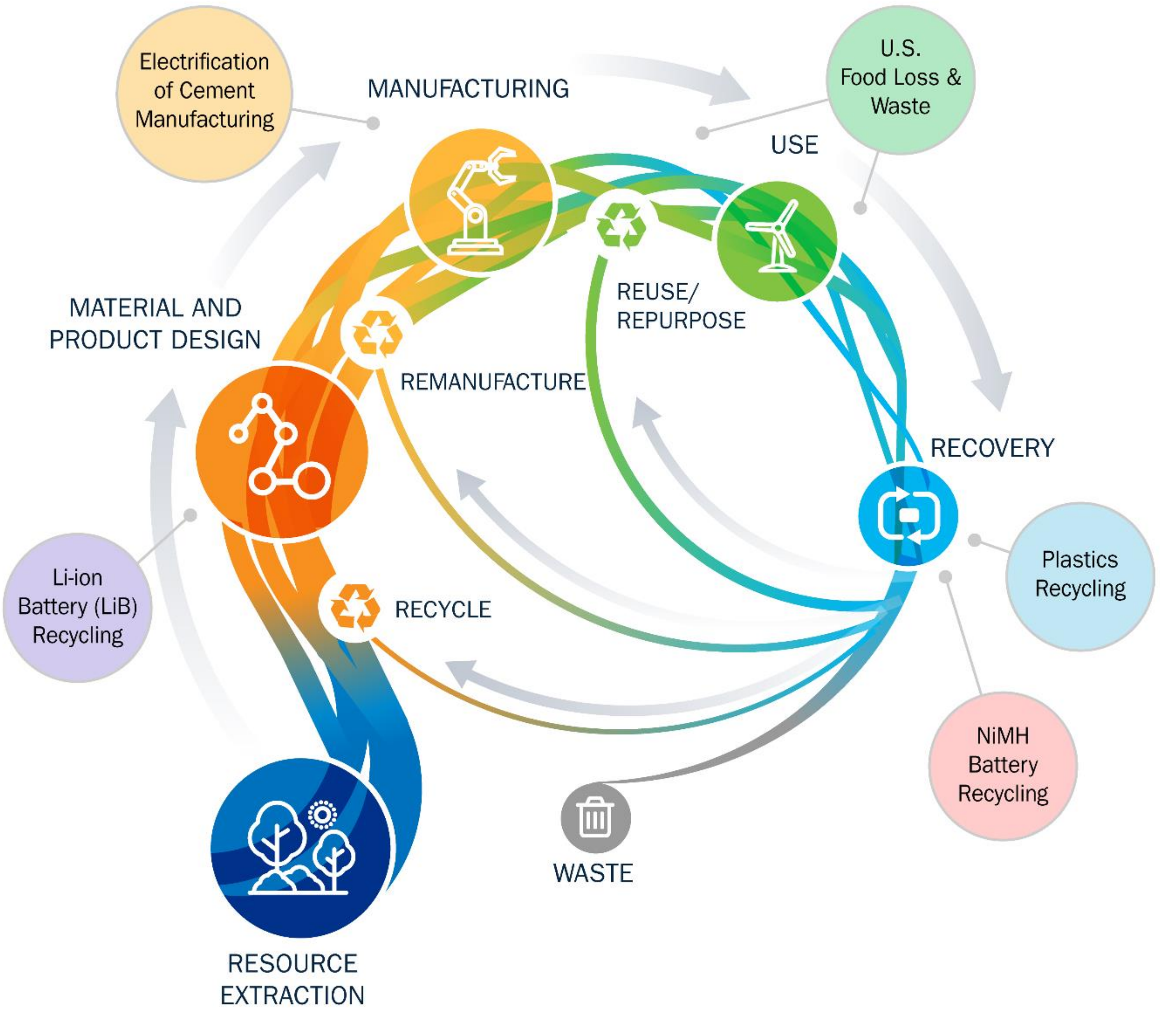


Figure 6. Sectors and circular economy strategies covered by the case studies in this report

3.4 Use Of Environmentally Preferable Materials and Technologies

Redesigning and rethinking products to incorporate more environmentally benign materials is a sustainability strategy that impacts the entire life cycle of products. Reducing toxic materials in manufacturing can reduce the environmental impacts of product supply chains, improve worker safety and health, reduce manufacturing waste disposal costs, and improve the products' recyclability. Using environmentally preferable materials can also impact the energy consumed during product manufacturing, use, and EOL management.

The construction industry consumes 50% of global steel production and accounts for more than 3 billion tons of global raw material use per year (State of Green, Danish Cleantech Hub and Yost 2018). The cement industry alone accounted for approximately 1.4 $GTCO_2$, about 70% of industrial emissions in 2014 (OECD 2017). Recent research into processing natural wood into a structural material as strong as steel (Song, et al. 2018) offers possibilities for replacing steel with a lightweight, abundant, inexpensive, renewable, and high-performance alternative. Replacing steel would also radically reduce the environmental and GHG emissions in the construction industry. However, mitigating flammability remains a barrier (Bartlett, et al. 2016).

4 Case Study Methodologies

This section describes the two methods used in the case studies to measure the benefits and trade-offs of applying circular economy strategies to specific industries.

4.1 Wedge Analysis

4.1.1 Introduction to Greenhouse Gas Wedges

Pacala and Socolow (2004) introduced the world to "wedges," a conceptual framework for quantitatively describing economic activities that reduce CO_2 emissions, or, more broadly, GHG emissions, relative to a reference trajectory. Wedges are idealized triangles with two dimensions: a length (time duration, typically 50 years) and height (reduction in GHG emissions rate after 50 years, measured in billions of metric tons of carbon, or GTC,[17] per year). Wedges are also sometimes expressed in terms of their area. The area bounded by a wedge constitutes the GHG emissions reduction over time, measured in GTC. The area can be computed by integration, which also allows for quantitatively comparing real-world activities that do not conform to the perfect triangle dimensions (see Figure 7).

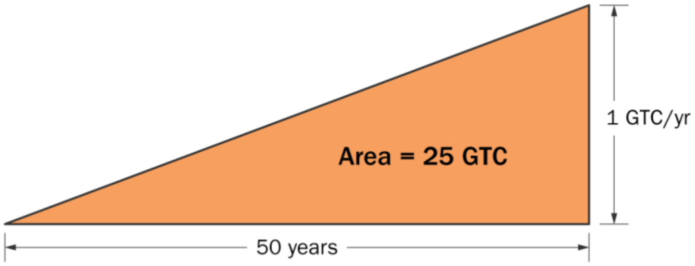


Figure 7. Global GHG wedge definition

Based on data in Pacala and Socolow (2004).

Next, several GHG wedges are stacked atop one another to create a "stabilization triangle": an integrated set of activities that together significantly decrease global GHG emissions. Relative to a reference or business-as-usual (BAU) trajectory, the lower emissions trajectory that results from subtracting these wedges represents a target GHG emissions pathway.

For the example shown in Figure 8, eight wedges construct a stabilization triangle over 2005–2055 that brings down global GHG emissions from a doubling trend (from 8 to 16 GTC/year over the period) to a "flat path" of constant emissions at 8 GTC/year. Note that, since its publication in 2004, the majority of climate scientists have concluded that a much more aggressive path is necessary to minimize climate

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[17] Most carbon accounting conventions since the publication of Pacala and Socolow (2004) have adopted units of billion metric tons of CO_2 ($GTCO_2$) rather than billion metric tons of carbon (GTC); the conversion factor is 1 GTC = 3.67 $GTCO_2$.

change; the IPCC's Sixth Assessment Report found that reducing greenhouse gas emissions to less than 3 GTC/year by 2050 is required to limit warming to 1.5°C (IPCC 2022).

Crucially, the scale of one wedge was chosen to be large enough to make a substantive impact on global GHG emissions, yet small enough to allow the scale of the activity to be feasible, at least over a 50-year time horizon. Moreover, by defining wedges in terms of GHG reduction only, vastly different types of activities can be compared in a direct way that might otherwise not be obvious or straightforward. Examples of such disparate global wedge activities include:

- Displacing coal-based electricity with 700 gigawatts (GW) of nuclear, 1,400 GW of natural gas, or 2,000 GW of renewable electricity[18]
- Doubling the fuel efficiency of two billion gasoline-powered personal vehicles (the projected global fleet size in 2055)
- Reforesting 500 million hectares of tropical forest (double the then-current rate of deforestation).

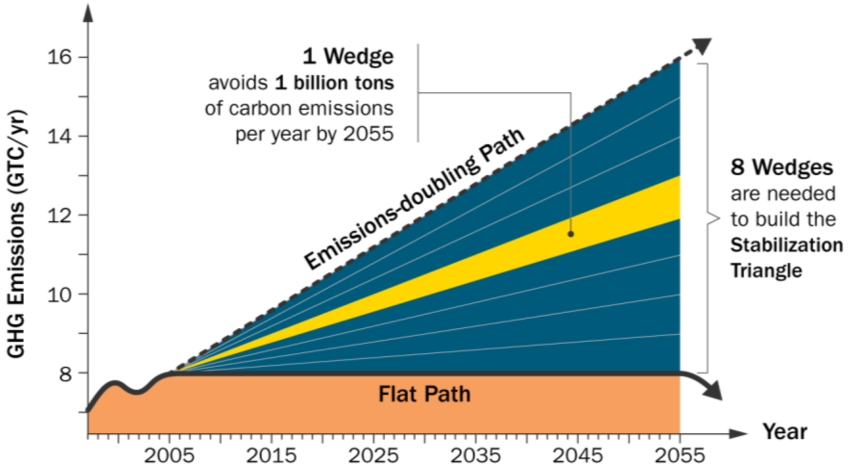

Figure 8. GHG stabilization triangle

Based on Socolow et al. (2004).

Notably, such wedges may represent activities with distinct natural units (GW of generation capacity, numbers of cars, hectares of forest); yet through the wedge definition, they may be compared and even combined. Since its publication, many enhancements and variations have been made to this basic framework, including:

- Revising the number of wedges required to maintain a flat emissions path (from 7 to 9 GTC/yr) (Pacala and Socolow 2004, Socolow, et al. 2004, Davis, et al. 2013)
- Downscaling to other geographic, temporal, and/or sectoral scales (Lashof 2006, Mui, et al. 2007)

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[18] Pacala and Socolow (2004) assumed different capacity factors and CO_2 emissions rates for different types of plants: about 30% capacity factor for renewables versus 90% for other generation technologies; no CO_2 emissions from nuclear and renewable electricity; and one-half of the CO_2 emissions per unit electricity produced from natural gas versus coal.

- Identifying additional wedges (e.g., new carbon-reducing technologies) as well as “hidden” wedges (additional reduction activities implied by efficiency measures embedded in the reference baseline) (Davis, et al. 2013)
- Adding non-CO_2 GHG emissions, such as CH_4, N_2O, and SF_6.[19] [e.g., Mui et al. (2007)]
- Exploring non-GHG wedges (Balbus, et al. 2014).

In each of these cases, the emissions rate quantity and/or time period has often been revised to make a more convenient “unit of currency” for comparison.

For instance, including all GHG sources, Greenblatt and Wei (2016) estimated that the United States would emit 6.1 GT CO_2 eq[20] in 2018 and grow to approximately 7.7 GT CO_2 eq in 2050. Because the United States’ energy-related CO_2 emissions in 2018 were 16% of global emissions (U.S. EIA 2019), an appropriately sized GHG wedge for this country might be approximately 0.16 GTC eq in 2050, which corresponds to 0.6 GT CO_2 eq. Over the 32-year horizon defined by the 2 reference years, the integrated GHG emissions reduction of this wedge is 9.6 GT CO_2 eq (see Figure 9).

Using this definition, keeping emissions flat over the time period would require 2.7 wedges, whereas reducing emissions to 80% below the 2018 level (to 1.2 GT CO_2 eq/year) in 2050 would require approximately 11 wedges.

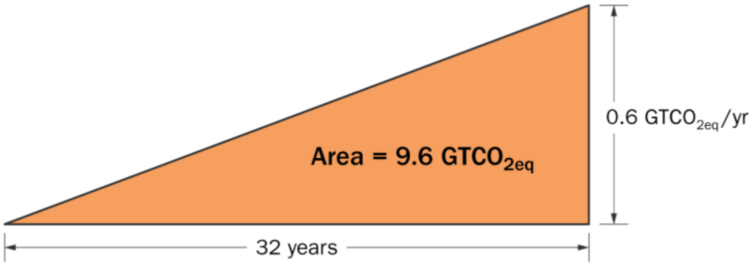

Figure 9. Example of U.S. GHG wedge definition

While idealized wedges are always depicted as triangles spanning the entire time horizon, real-world wedges can take on different shapes and may begin or end at different times. Examples of such wedges are shown in Figure 10, all of which have the same areas.

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[19] Non-CO_2 emissions are typically converted into “CO_2-equivalent” mass units (e.g., $MTCO_2$ eq) using 100-year GWP factors published by the IPCC (Myhre, et al. 2013). Note that GWP factors have evolved as the science has improved (e.g., Greenhouse Gas Protocol (2016)).

[20] CO_2 eq refers to the equivalent amount of CO_2 emitted that would result in the same global warming impact over a 100-year period; conversion ratios are typically expressed in GWP units taken from the International Panel on Climate Change (IPCC) Fourth Assessment Report (https://www.ipcc.ch/assessment-report/ar4/). The GWP for 1 kg of methane, for instance, is 25 kg CO_2 eq.

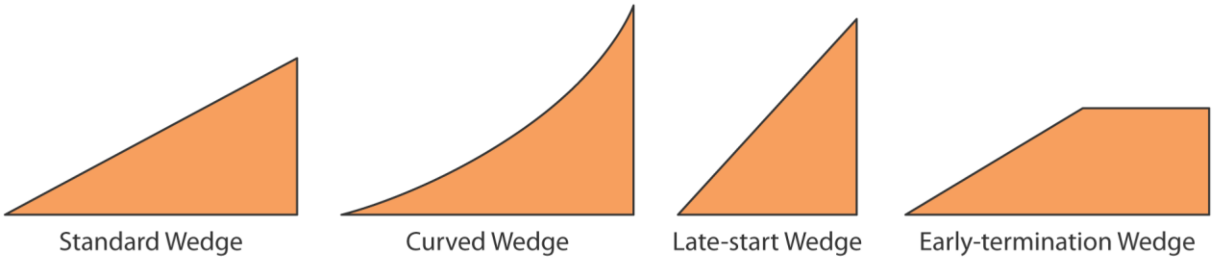

Figure 10. Real-world wedge examples (standard, curved, late start, and early termination)

The way to reconcile these various wedges is to compare them with a relevant metric of interest. For GHG wedges, this is usually integrated as avoided GHG emissions ("avoided carbon"), as climate change effects are more directly correlated to cumulative rather than annual emissions in a particular year.

4.1.2 From GHG to Materials Wedges

We can use this flexible wedge framework to describe changes over time of other quantities besides GHG emissions. Energy, material flow, or even monetary wedges can be easily defined with appropriate units and used to compare dissimilar activities producing changes in quantities of interest.

For example, Table 1 presents four illustrative U.S. wedges spanning energy, materials, and monetary flows. When graphed, the height (*y*-axis) quantity is an appropriate metric (e.g., metric tons of steel consumed annually). The length (*x*-axis) in each case is time, ranging from 2018 (the reference year) to 2050. If desired, one can further define a single wedge as a fraction of the reference year flow rate or an integrated quantity over time; however, such definitions are not mandatory for using non-GHG wedges for analysis. Table 1 shows examples of these definitions.

Table 1. Example of Non-GHG Wedge Definitions for the United States

			Possible Wedge Definition	
Quantity	**Flow Metric**	**Flow Rate in 2018**	**Flow Rate in 2050**	**Area (2018–2050)**
Primary energy [1]	Quads[21]/ yr	100	10	160 quads
Steel demand [2]	MMT/yr	102	10	160 MMT
Aluminum demand [3]	MMT/yr	4.9	0.5	8 MMT
Gross Domestic Product (GDP) [4]	U.S.$ trillion/yr	20.5	2.0	U.S.$32 trillion

Data Sources: 1: (IEA 2019); 2 and 3: (U.S. Geological Survey 2021); 4: (U.S. BEA 2019)

Constructing a set of wedges ("reduction" triangles) for a given quantity requires first generating a reference or BAU scenario and then subtracting wedges from the scenario to obtain a target pathway. For instance, if primary steel consumption is projected to double from 102 to 204 MMT/year by 2050 (unrealistic, but used here as an example), and the material reduction goal is to reduce consumption to 53% of the 2018 level or 54 MMT/year, a reduction triangle would consist of 150 MMT/year in 2050, or

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[21] Quad, unit of energy equal to 1 quadrillion of (10^{15}) British thermal units (Btus)

15 steel wedges (as defined in Table 1 and depicted in Figure 11). Each wedge might represent a different technology or strategy yet result in the same amount of material flow reduction.

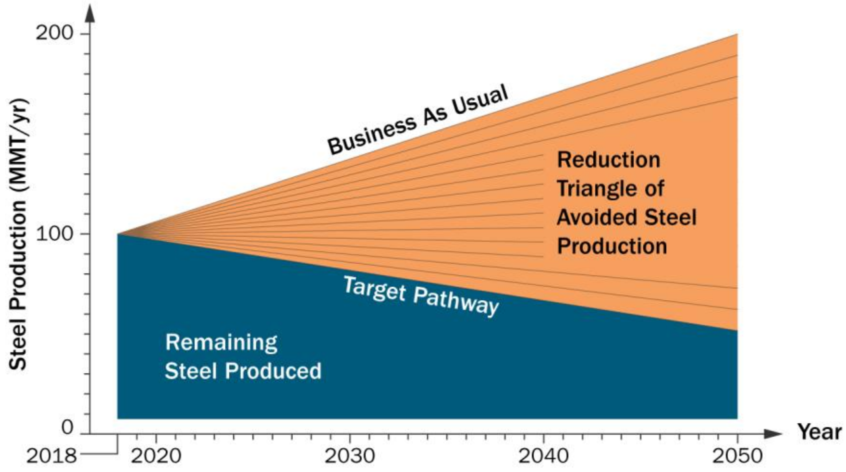

Figure 11. Example of steel wedge reduction triangle

Wedges can be categorized by various material reduction strategies that elaborate on circular economy strategies introduced in Section 3. While not an all-inclusive list, Table 2, based on data in Donati et al. (2020) and Allwood and Cullen (2015), represents useful starting points.

Table 2. Material Reduction Strategies

Category	Material Reduction Strategy
Resource efficiency	Scrap diversion Yield loss reduction Process improvements Design improvements Use intensification within industry Use intensification among end-use consumers Sharing within industry Sharing among end-use consumers Reducing service demand (e.g., behavior change) Resource substitution (e.g., wood for steel)
Product lifetime extension	Reuse, repair, and remanufacturing Delayed product replacement
Closing supply chains (e.g., Re-X)	Recycling Material recovery
Residual waste management	Landfill mining

Based on data in Donati et al. (2020) and Allwood and Cullen (2015).

Table 3 lists some possible materials worth tracking and divides them into two broad categories: commodity materials and critical materials. We define commodity materials as any material used in significant quantities throughout the world and produced widely in many countries. A critical material, by contrast, is a “non-fuel mineral, element, substance or material” that “has a high risk of a supply chain disruption” and “serves an essential function in 1 or more energy technologies, including technologies that produce, transmit, store, and conserve energy” or is otherwise defined as critical by the United States Geological Survey (USGS) (U.S. Congress 2020). As mentioned in Section 1.4, many of these materials are used in increasing amounts in clean energy technologies. They are also not usually produced in nearly as large of quantities as commodity materials. Table 3 identifies the materials we examine in the case studies that follow with an asterisk.

Note that each material normally consists of multiple subcategories regardless of type. For instance, for iron/steel, many incompatible alloys must be separately tracked when considering material reduction potential. These include cast iron; carbon steel; tool steel; high-strength, low-alloy steel; and other materials (Allwood and Cullen 2015). While in many cases, they can be summed up to an aggregate “iron/steel” category, not all steels are interchangeable. Likewise, other materials may consist of distinct chemical structures (e.g., plastics) or even different elements altogether (lead, tin, and zinc metals; or nitrogen, phosphorus, and potassium fertilizers). They are nonetheless considered in the same family of materials (defined perhaps by similar chemical or physical properties, or end uses), and are thus aggregated together. For example, the case study in Section 7 considers the cathode metals recovered together from LiBs: Co, Ni, Li, and Mn.

Table 3. Materials of Potential Interest for a Circular Economy

Commodity Materials		Critical Materials	
Iron/steel	Wood	Neodymium*	Cobalt*
Aluminum	Pulp/paper	Europium	Gallium
Copper	Plastics*	Terbium	Indium
Lead, tin, and zinc	Glass	Dysprosium	Manganese
Noncritical precious metals	Cement and other construction materials	Yttrium	Platinum group metals
Other nonferrous materials*	Food*	Lithium*	Vanadium
Energy carriers (e.g., hydrogen, methane, and petroleum)*	Fertilizers (nitrogen, phosphorus, potassium)	Tellurium	Battery-quality graphite
Fresh water	Municipal solid waste		

Data Sources: DOE (2020), Wiebe et al. (2019), and authors’ inputs.

* Material examined in this report.

An important distinction must be made between primary (virgin) and secondary (recovered) materials, and depending on the context, one or both may be of interest. For instance, one may care about reducing iron ore mining, in which case primary iron/steel production is the important quantity to display. On the other hand, if technologies, strategies, or policy goals aim to reduce the total amount of material consumed, regardless of source, total steel demand will be the relevant metric. In general, however, our focus in this report is on reducing primary (virgin) material extraction.

Another consideration is that some materials are produced as byproducts of mining other, higher-valued materials. This is often the case with critical materials; for instance, gallium is produced as a byproduct of both bauxite (aluminum) mining and zinc processing (Foley and Jaskula 2013) and thus are constrained by the production of the primary mined minerals. In these cases, reducing demand for a byproduct material may not have much impact on primary extraction activities, as the economics are driven by demand for a different material.

In the case studies that follow, each material wedge has energy and GHG wedges associated with it and potentially other material wedges if the strategy under consideration consumes or produces materials of its own (such as materials substitution). The magnitude of each associated wedge generally varies depending on the process involved. In some cases, the magnitude can even be negative, like an increase in an associated quantity when there is a decrease in the material, or, for example, a critical material for which a reduction in material dependency could increase energy or GHG but be economically or politically beneficial. Therefore, combining equally sized material wedges will typically result in energy, GHG (Agrawal, Singh and Murtaza 2015., Ahn, et al. 2020), or other wedges of different sizes. Figure 12 illustrates this concept.

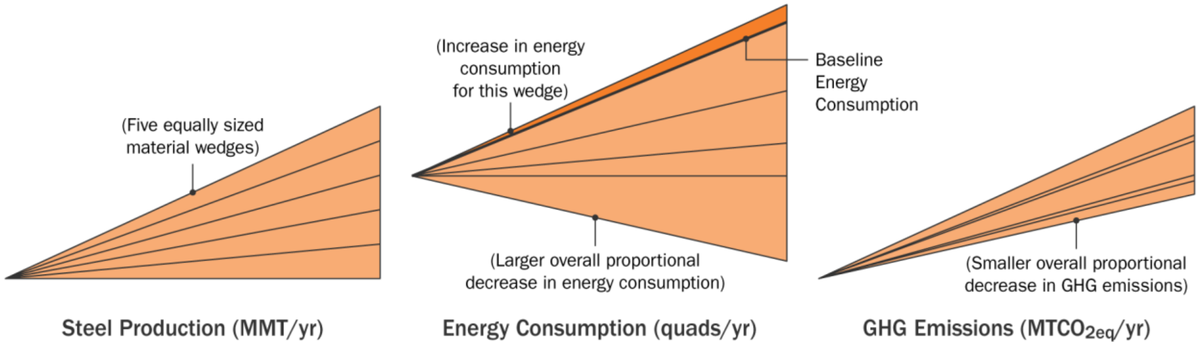

Figure 12. Example of material, energy, and GHG wedges

4.2 Life Cycle Impact Assessment

As noted in Section 1, life cycle assessment (LCA) is a framework that addresses the environmental impacts throughout the life cycle of a product, process, or service from raw material acquisition through production, use, EOL treatment, recycling, and final disposal. Life cycle impact assessment (LCIA) aims to evaluate and understand the magnitude and significance of the environmental emissions and resource use for a product system throughout the life cycle of the product, process, or service.

All the case studies in this report apply LCIA methods to analyze environmental impacts of the technologies examined and circular economy strategies employed. This section explains how the case studies apply LCIA methods.

4.2.1 Material Resource Use

Material resources are classified as either biotic or abiotic resources. While both types of resources have finite total stocks,[22] biotic resources are intrinsically renewable because of the continual biological activity that regenerates them over short time cycles. In contrast, abiotic resources are not generally renewable without deliberate human input because of the lengthy geologic time scales required for their

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[22] Stocks of a resource in this report encompass the amount of the resource currently in anthropogenic and/or natural use. For instance, the number of vehicles already on the road in a time period constitutes the “stock” of vehicles in that period.

spontaneous formation.[23] Biotic material resources in the economy include those from agriculture, forestry, fisheries, and hunting. Abiotic material resources in the economy include metals; nonmetals and their mineral ores; bulk minerals, such as those used in industry and construction; and fossil energy carriers (Giljum, et al. 2011).[24] Because the primary goal of the case studies presented in this report is to minimize abiotic resource consumption and waste, the methodologies and metrics described in this section mainly focus on abiotic material resource use. Certain thermodynamics-based methods also account for renewable resources used, although the renewable and nonrenewable resources are reported separately.

The consumption of abiotic material resources is often referred to as "depletion," as consumption in a given period of time implies reducing the availability of the resource for future periods. The term "availability" can encompass different physical, economic, technological, and social concepts, as illustrated in Figure 13. In the context of environmental analysis, and more specifically LCA of manufacturing technologies, systems, and supply chains presented in this report, abiotic material depletion is therefore characterized using different definitions as reported by various LCIA methods.

Irrespective of the LCIA method used, abiotic material depletion metrics support an aggregated and/or notional representation of several abiotic materials that may be used over the life cycle of a product or process. The goal is to provide a relevant metric for interpreting the LCA results.

This single-metric goal is identical to other midpoint impact metrics, such as global warming potential (GWP), used in LCAs. However, abiotic depletion metrics have strong economic underpinnings and almost always include social value judgments about the present extraction of resources vis-à-vis the availability of those resources for future generations (B. A. Steen 2006, Van Oers and Guinée 2016). As such, abiotic depletion metrics are not purely based on the physical, chemical, or biological relationships that exist between contributors to other midpoint impact metrics, such as GWP, wherein any uncertainty or variability in the values of the metrics emanates from uncertainties in physical measurements.

For instance, uncertainties in measuring methane's radiative forcing and atmospheric lifetime could result in uncertainties in its GWP relative to CO_2, but there is consensus on the definitions of radiative forcing and atmospheric lifetime. In contrast, abiotic depletion metrics lack consensus because some level of subjective, social judgement is used to define them, so variability in the values of these metric emanates from variability in judgement calls. For example, should the "reserves" of a particular resource be defined as the economically extractable amount of the resource, given current technology and prices? Alternatively, should reserve be defined as the total amount that resource present in the Earth's crust (which is a somewhat theoretical construct)? Is the current rate of extraction of a given resource relevant

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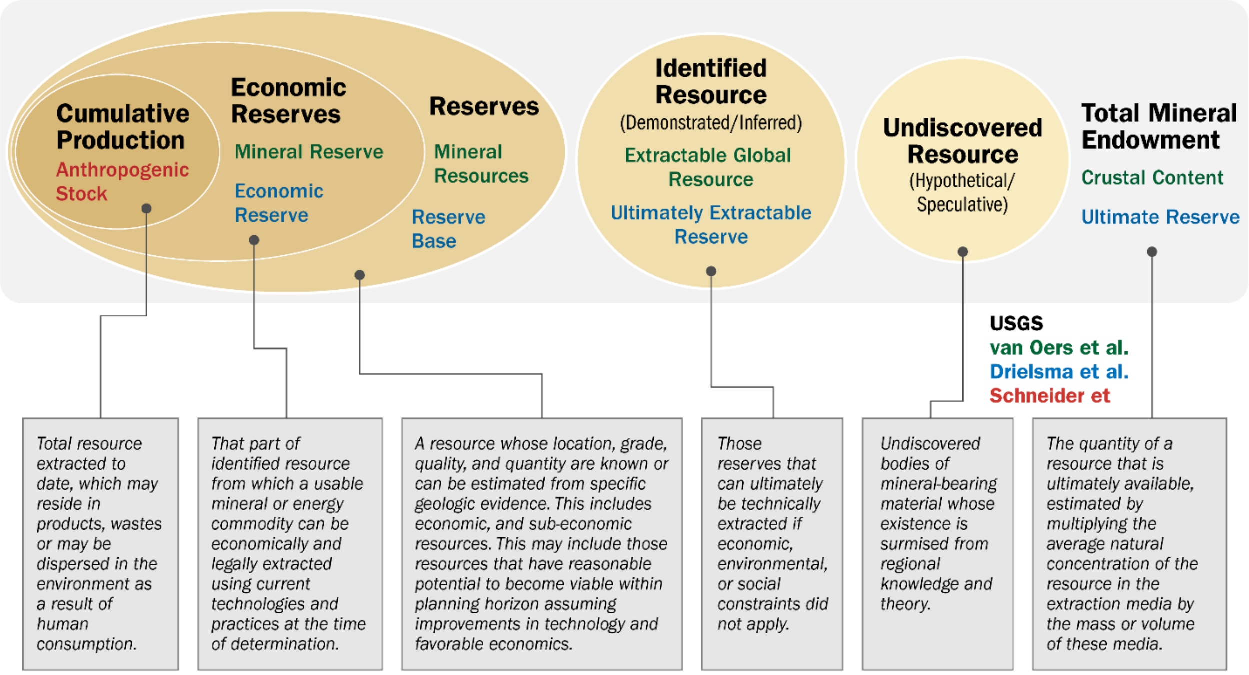
[23] Some abiotic resources are renewed through naturally occurring chemical, physical, or biological mechanisms (e.g., weathering, mineralization, and accumulation). However, the timescales at which these natural phenomena occur are too long to be meaningful in the context of sustainable human consumption.

[24] Water (specifically, fresh water) represents a "middle ground" that is not the (main) product of biological systems; it is produced at important levels by these systems, and the timescales at which abiotic phenomena (e.g., evaporation, transport, precipitation, and filtering through mineral systems) replenish it is very meaningful for human consumption, unlike other abiotic resources. Moreover, the LCA community does not yet use an elementary flow nomenclature for tracking water that is standard for other abiotic resources. Therefore, water is typically handled differently from other resources, although some efforts have looked to address this. For example, see Berger and Finkbeiner (2010) and Boulay et al. (2018).

to abiotic depletion? Should the amount of a resource already extracted and in use by humans be considered an “available” resource?

In the absence of a consensus or the possibility of a consensus on which abiotic depletion metric(s) are most appropriate for life cycle assessment, we examine several abiotic depletion metrics commonly used in the impact assessment literature. Table 4 summarizes these metrics and their definitions. The case studies presented in this report use one or more metrics for discussion of abiotic depletion in the context of the goals and scope of the LCA conducted within each case study.

Table 4 presents metrics and definitions of abiotic depletion (*AD*) as expressed in commonly used LCIA metrics and used in this report. Table 4 is compiled from sources listed in the table and review papers by Klinglmair et al. (2014) and Sonderegger et al. (2020). Appendix A presents a complete table of available methods from the literature.



Total resource extracted to date, which may reside in products, wastes or may be dispersed in the environment as a result of human consumption.

That part of identified resource from which a usable mineral or energy commodity can be economically and legally extracted using current technologies and practices at the time of determination.

A resource whose location, grade, quality, and quantity are known or can be estimated from specific geologic evidence. This includes economic, and sub-economic resources. This may include those resources that have reasonable potential to become viable within planning horizon assuming improvements in technology and favorable economics.

Those reserves that can ultimately be technically extracted if economic, environmental, or social constraints did not apply.

Undiscovered bodies of mineral-bearing material whose existence is surmised from regional knowledge and theory.

The quantity of a resource that is ultimately available, estimated by multiplying the average natural concentration of the resource in the extraction media by the mass or volume of these media.

Figure 13. Terms frequently used to describe the availability of abiotic materials for human consumption.

Terms labeled in black, green, blue, and red are respectively obtained from USGS (1980); van Oers, Guinée, and Heijungs (2020); Drielsma et al. (2016); and Schneider, Berger, and Finkbeiner (2015).

Table 4. Metrics and Definitions of Abiotic Depletion (*AD*) as Expressed in the Life Cycle Impact Assessment Metrics Used in This Report[25]

LCIA Method*	Sources	Abiotic Depletion Calculation Approach	Abiotic Depletion Units	Abiotic Depletion Metric and Characterization Factor
CML 2001	(Guinée and Heijungs 1995, J. Guinée 2002)	• Considers metallic and nonmetallic elements, metallic and nonmetallic mineral ores, and fossil resources. • ADP = Abiotic depletion potential.	kg Sb eq [26]	$AD_i = X_i \times ADP_{i,ref}$ $= X_i \times \left(\frac{P_i / R_i^2}{P_{ref} / R_{ref}^2} \right)$ ref = Antimony (Sb) R = Ultimate reserves
ILCD 2011	(Chomkhamsri, Wolf and Pant 2011, EC-IES-JRC 2012)	• Considers metallic and nonmetallic elements and fossil resources. • ADP = Abiotic depletion potential. • Based on CML 2001.	kg Sb eq	$AD_i = X_i \times ADP_{i,ref}$ $= X_i \times \left(\frac{P_i / R_i^2}{P_{ref} / R_{ref}^2} \right)$ ref = Antimony (Sb) R = Ultimate reserves
ReCiPe-2016	(Huijbregts, et al. 2016, Vieira, et al. 2017)	• Considers metallic and nonmetallic elements, bulk resources, and fossil resources. • SOP (Surplus ore potential function) is calculated with the basis that each unit of extraction of a mineral leads to a reduction in the ore grade, *g*.	kg Cu eq	$AD_i = X_i \cdot ADP_{i,ref}$ $X_i \cdot \left(\frac{SOP_i \left(R_i, g_i, CRE_i \right)}{SOP_{ref} \left(R_{ref}, g_{ref}, CRE_{ref} \right)} \right)$ ref = Copper (Cu) R = Ultimately extractable reserve (ReCiPe-E) OR Economic reserve (ReCiPe-H)
EDIP 2003 (Resources Only)	(Hauschild and Wenzel 1998)	• Considers metallic elements and fossil resources. • Characterization factor of 1 allows representation of abiotic mineral depletion as resources consumed without consideration for availability, historical extraction, or extraction energy needed.	kg	$AD_i = X_i$
Cumulative Exergy Demand (CExD)	(Boesch, Koehler and Hellweg 2009, Bösch, et al. 2006)	• Considers metallic and nonmetallic elements, metallic and nonmetallic mineral ores, bulk resources, fossil, nuclear, and renewable resources. • *CExD* is the cumulative exergy stored in a resource in the form of chemical, thermal, kinetic, potential, nuclear and radiative energy.	MJex	$CExD = \sum_i X_i \times Ex_{(ch),i} + \sum_j n_j \times r_{ex-e(k,p,n,r,t),j}$ Where $Ex_{(ch),i}$ = exergy per kg of substance *i* n_j = amount of energy from energy carrier *i* $r_{ex-e(k,p,n,r,t),j}$ = exergy to energy ratio of energy carrier *j*

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[25] *P*, *R*, *g*, and *CRE*, respectively, represent the annual extraction rate, resource availability, ore grade, and cumulative resource extracted of an abiotic mineral, and *X* represents the extraction of the resource *i* for which the resulting abiotic depletion is calculated.

[26] As explained in van Oers, Guinée, and Heijungs (2020), antimony was chosen because of the relative stability in estimates of *P* and *R* compared to other minerals at the time the metric was developed. Mathematically, the choice of reference material has no bearing on the relative abiotic depletion (*AD)* of a given materials to that of another.

*CML - Centrum voor Milieuwetenschappen (Institute of Environmental Science) at Universiteit Leiden, ILCD - International reference Life Cycle Data developed by the EU Joint Research Council, ReCiPe is a method for the impact assessment (LCIA) in a LCA developed by the Netherlands' National Institute for Public Heather and the Environment, EDIP – Danish developed LCIA methodology.

4.2.2 Assessing Environmental Trade-Offs

As discussed previously, the pursuit of minimizing abiotic material depletion and waste could lead to trade-offs with other environmental impact metrics, such as fossil energy use or climate GWP. Therefore, several case studies present the abiotic mineral depletion results in the form of a "quad chart," in which abiotic mineral depletion and another relevant environmental impact metric, such as global warming, for a given technology are shown on orthogonal axes in a two-dimensional scatter plot. This is demonstrated in Figure 14, where Technology D has reduced fossil resource depletion and reduced mineral resource depletion but has a higher carbon intensity and cost, whereas Technology E has reduced fossil depletion but slightly increased mineral resource depletion with a lower carbon intensity and lower cost than Technology D. Such a representation allows us to discern the extent to which changes in abiotic material depletion, as a result of a technology or other intervention, affect other environmental impacts, highlighting any potential trade-offs. The quad chart also allows us to represent additional environmental impacts obtained from a comprehensive life cycle assessment and/or economic metrics, using other attributes such as the size and color of the points on the scatter plot.

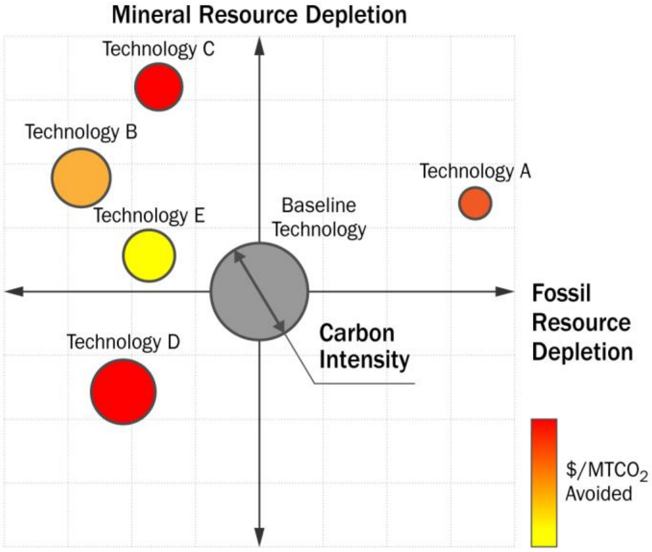
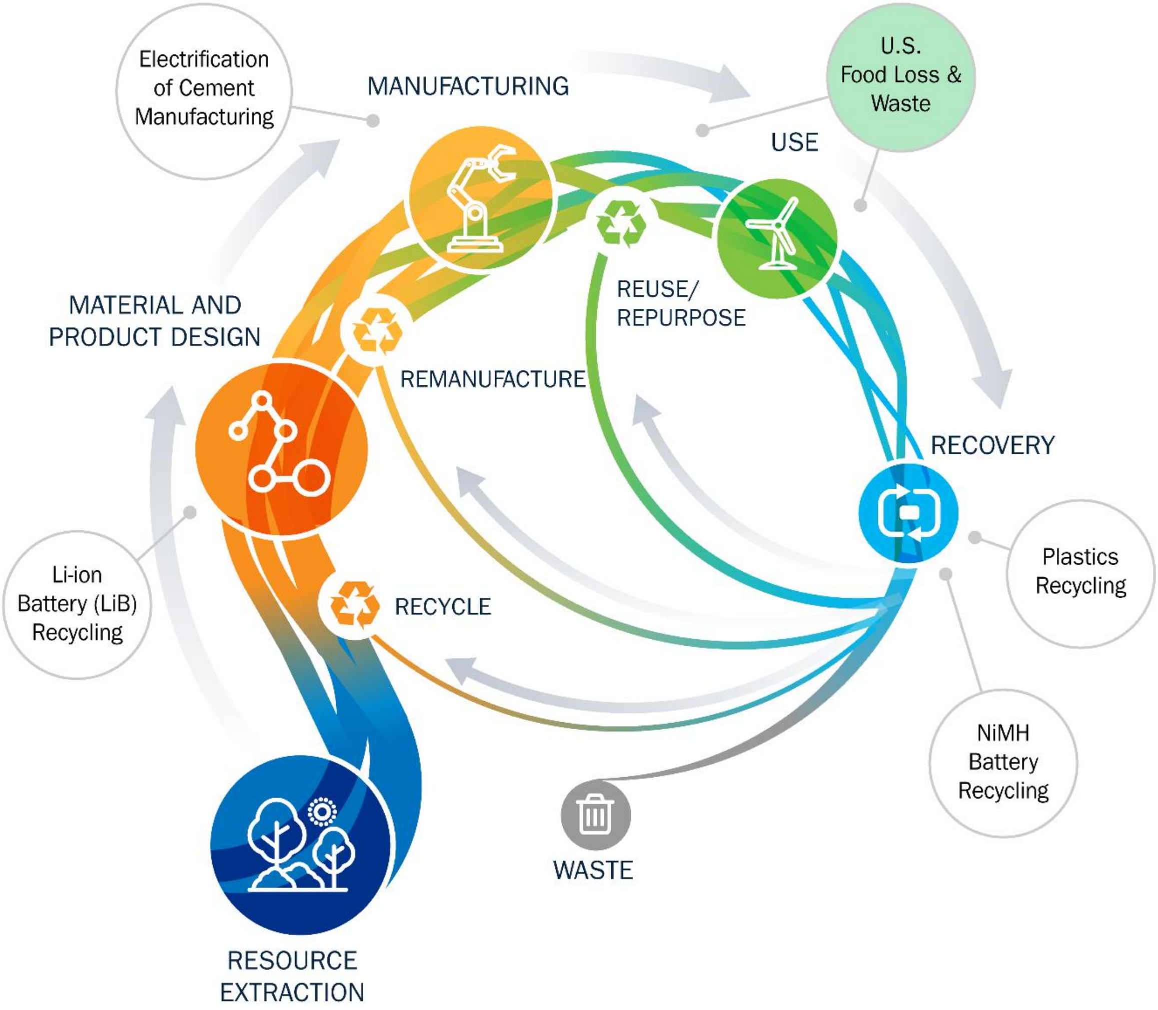


Figure 14. Example of a quad chart with different impacts being reflected on the *x* and *y* axes and the size and color of the dots representing the carbon intensity and cost of carbon avoided, respectively.

Note: Quad chart shows abiotic mineral depletion and other environmental, economic, and relevant quantitative or qualitative metrics for the baseline technology analyzed (gray) and for the technologies or approaches investigated as alternatives to the baseline with the goal of reducing material waste.

5 Case Study: Reducing Food Loss and Waste (FLW)



5.1 Introduction

Food-related activities (growth, manufacturing, packaging, distribution, and preparation) contribute significantly to the problems of dwindling natural resources, increasing environmental degradation, and climate change (Hall, et al. 2009, IPCC 2019). Food production and the rest of the supply chain are estimated to contribute 30% of global energy consumption (Fabiola and De Rosa 2016) and account for 19%–29% of GHG emissions (U.S. EPA 2020). The expected growth of global food demand will add more pressure on natural resources and intensify environmental problems (Parfitt, Barthel and MacNaughton 2010, Schanes, Dobernig and Gözet 2018). While electrification and low-carbon energy sources are viable paths to reducing energy use and GHG emissions (U.S. DOE 2022), increasing the efficiency of the FSC without further compromising ecosystems' quality and biodiversity is essential to reducing the global impact of the FSC.

Reducing FLW is a highly effective impact reduction strategy:

- Reducing FLW at all stages of the FSC directly reduces resources and environmental problems with food production.
- Reducing FLW avoids the consumption of resources and environmental problems with FLW management.
- Directing edible FLW to food banks will immediately help households with food insecurity (EPA 2017a).

In alignment with Target 12.3 of the United Nations Sustainable Development Goals to halve per capita global FLW at the retail and consumer levels by 2030, the USDA and EPA have set a joint national goal to reduce 50% of U.S. FLW going to landfills and incinerators by 2030. They are developing and implementing efforts and initiatives to achieve this goal. In addition, the DOE has set goals to promote innovative new technologies and solutions that can be implemented in the manufacturing stage to significantly reduce FLW throughout the U.S. FSC.

This study supports these goals: it quantifies human food mass flow and FLW at each FSC stage and estimates the energy consumption and GHG emissions of food-related activities. This study also identifies multiple technologies and strategies that can be adopted to reduce energy consumption and GHG emissions. A case study demonstrates the impact of advanced packaging solutions on energy consumption and FLW and GHG generation.

5.2 Overview of U.S. Food Supply Chain

5.2.1 U.S. FSC Mass Flow and FLW

Food is lost at every FSC stage for numerous reasons. For instance, at the on-farm production stage, crops and raised animals may go unharvested because of labor shortages, overplanting, pests, weather, and unplanned animal disease or death (Minor, et al. 2020). Manufacturers may reject harvested materials if they fail to meet market requirements. Food can be lost in the manufacturing stage from trimming, removing inedible parts, and inefficient manufacturing processes. The manufactured food products may also be discarded at the downstream FSC stages because of quality issues, overstocking, and over-preparing (i.e., preparing too much food). Dong et al. (2022) and Figure 15 describe additional FLW drivers by FSC stage.

What is Food Loss and Waste?

Food intended for human consumption that has left the FSC

Does Include

- Edible and inedible parts
- Donated food
- FLW for animal feed
- Water completely removed during manufacturing

Does NOT Include

- Food grown for animal feed
- Food grown for seed
- Food grown for other uses

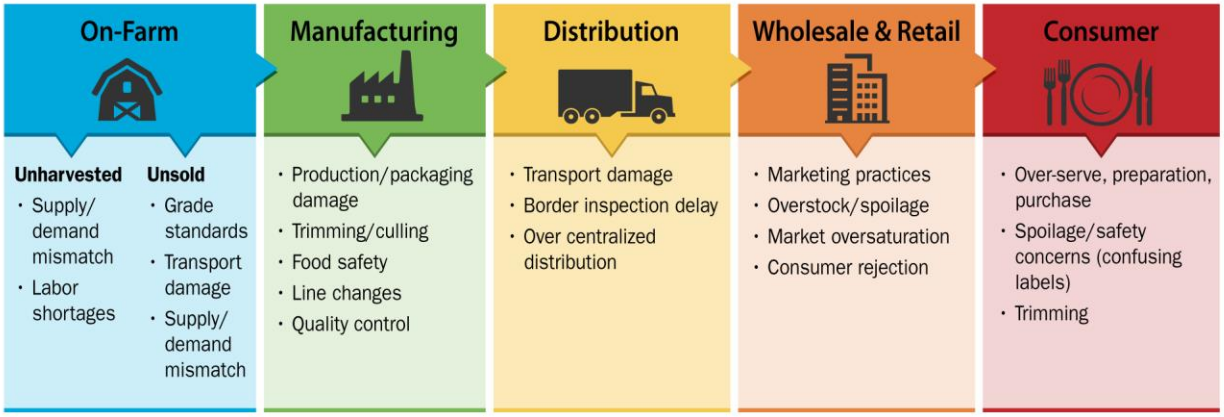


Figure 15. Major FLW drivers along the U.S. FSC

To better understand the U.S. FSC, we estimate the mass flow and FLW at each FSC stage for 2016 (Dong et al. 2022). In this analysis, we measure only the mass flow of agricultural products: we do not consider water, food packaging, and other materials entering the FSC. Moreover, this study defines FLW as food intended for human consumption that leaves the FSC, which includes both edible and inedible parts and food that will be donated to people with low food security. Food grown specifically for animal feed is not included in the mass flow or FLW analysis, as that focuses on the mass flow of human food with animal weight starting as harvested weight. However, it is included in the energy consumption and GHG emissions analyses as it represents a sizeable contribution to the total energy demand of animal products. Food that is grown for human consumption but then becomes animal feed via FLW (byproducts of human food manufacturing) is part of the human FSC as an EOL management pathway. FLW is managed through nine different EOL management pathways. We estimate the distribution of FLW across these EOL pathways primarily using the EPA's methodology (EPA 2020f). The data sources and scope for this analysis are as follows:

- **Major data sources:** FAOSTAT 2020; USDA (2019a, 2019b, 2019c, 2019d, 2019e, 2019f); EPA 2020f; U.S. Grains Council 2018.
- **Food commodities considered:** Grain (including corn sweeteners), fruit, vegetables, sugar, oils (e.g., soybean, cottonseed, sunflower seed, canola seed, and flaxseed), dairy (including dairy fat), meat and poultry (including animal fat), seafood, eggs, and nuts.
- **FLW EOL pathways considered:** Donations to feed people with low food security, animal feed (i.e., diverting FLW, directly or after processing, to feed animals), composting, anaerobic digestion (AD), land application, biochemical processing, landfills, incineration, and wastewater treatment (i.e., in-sink disposal). Agricultural materials grown specifically for nonhuman food use, such as grain grown to feed animals or produce biofuel, are not included in these pathways.

According to our earlier analysis (W. Dong, et al. 2022),[27] of the 572.9 MMT of food grown or raised for human consumption, more than one-half (343.6 MMT) left the FSC in 2016, as water (18.6%); reused or repurposed (R&R, 56.4%); or landfill, wastewater treatment, and incineration (LWI) (25%). Figure 16 is

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[27] The authors of this case study also authored Dong et al. (2022).

a Sankey diagram of the FSC starting with agriculture harvest. It illustrates several key points about the FSC.

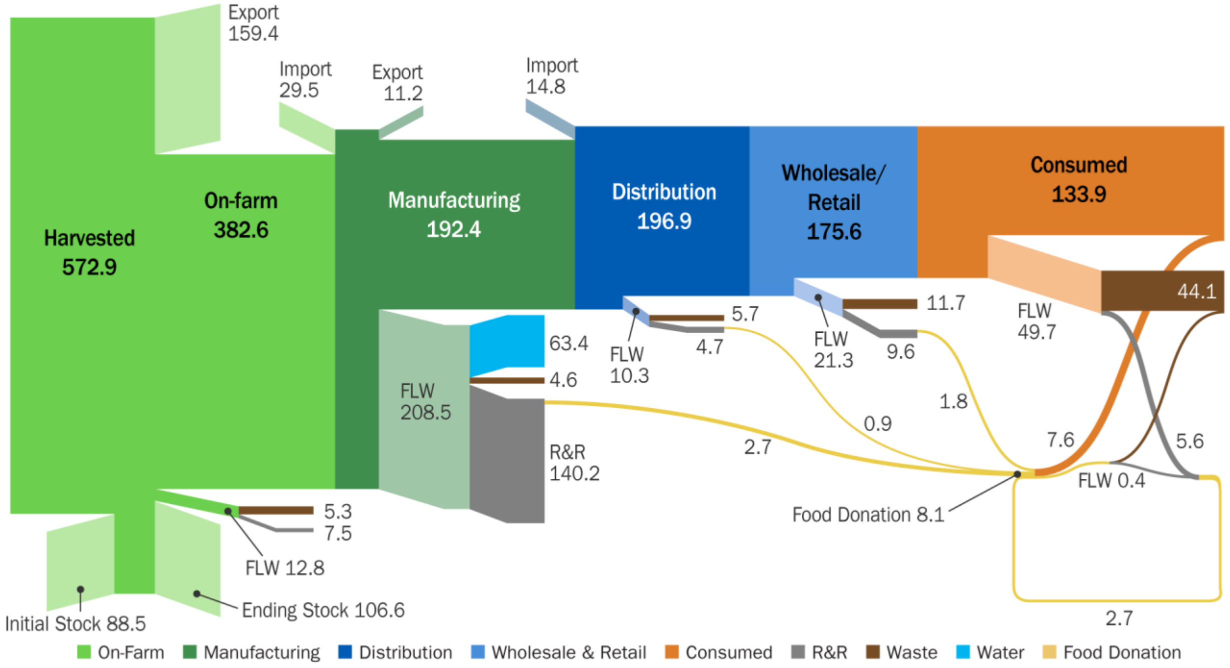

Figure 16. Sankey diagram of U.S. food supply chain (MMT of food)

- *Harvested* is any food harvested, grown, or raised for human consumption. It excludes food grown for animal feed, seeding, on-farm consumption, and industrial uses (e.g., biofuels and bioplastics). It does include food that is exported (although not foodstuffs imported) and does include the change in stock (88.5 MMT at the beginning of 2016 and 106.6 MMT at the end of 2016). Unharvested food (40.8 MMT; 11.9% of total FLW, not shown in Figure 16) occurs when expected and actual demand do not match, during labor shortages, and with a decrease in production as a result of disease, pests, or weather.

- *On-farm production* includes agricultural activities at the farm and those occurring before products reach the manufacturer's gate, such as preharvesting, harvesting, on-farm handling and storage, and shipment to manufacturers. On-farm food production has two different FLW streams: unharvested (discussed in the bullet above) and unsold food. Unsold food (12.8 MMT; 3.7% of total FLW) is food that was harvested but unsold to manufacturing entities: products damaged during improper harvesting activity or from spoilage during inadequate, improper, or overly long shipment or storage. From these two streams, 11.2 MMT (8.7 MMT and 5.3 MMT), or 21% of on-farm FLW, ends up in a landfill or incinerated.

- *The food manufacturing stage*, which includes food processing, manufacturing, and packaging, is the largest contributor to FLW, with a total FLW of approximately 208.5 MMT (61%). Additionally, 140.2 MMT (67%) is repurposed for use in other industrial sectors (such as animal food and AD) or for land application, and only 4.6 MMT (2%) is landfilled or incinerated. The remaining 63.7 MMT (31% of the manufacturing FLW) represents the water removed during the processing of certain foods (e.g., dried fruit and vegetables, sugar, dried and condensed dairy products) and not lost or wasted food. This water is inherently included in mass flows before the manufacturing stage but should not be allocated to the EOL management pathways discussed in this analysis, and is instead estimated as water removed via evaporation, drying, or similar activities. This may not be all the water removed from food and does not include water that is discarded with FLW or lost during other manufacturing activities (e.g., grain milling), storage, or transportation, but is an attempt to harmonize the mass balance approach with estimates for manufacturing FLW.

- *The distribution stage*, which includes shipping food products from manufacturers to wholesalers and retailers, generated the least FLW (10.3 MMT, 3.0% of total FLW). Food is lost at this stage from damage and spoilage during transport and poor matching of supply and demand. This loss is mitigated primarily by modern logistics technology and management (e.g., cold chain), contributing to the low amount of FLW. However, almost 55% (5.7 MMT) of the FLW was disposed of as waste, and 4.7 MMT (45%) was R&R.
- *The wholesale and retail (W&R) stage*, which includes warehousing and grocery stores, generated 21.3 MMT (6.2%) of FLW from consumer rejection (e.g., not buying because of appearance or damage) and overstocking, with 11.7 MMT (59%) going to waste and the rest to R&R (9.6 MMT; 41%).
- *The consumption stage*, which includes household, food services (e.g., restaurants and cafeterias), and food banks, also contributed a significant amount of FLW—49.7 MMT (14.5%). This amount includes 0.4 MMT FLW from food banks, as not all donated food can be distributed before spoiling. This stage is also the largest contributor to wasted food (LWI), contributing 44.1 MMT (89% of consumption FLW, 57% of wasted food), with very little recycled (5.6 MMT; 11% of consumer FLW).

We group these FLW EOL pathways according to the EPA's Food Recovery Hierarchy (Figure 17), which illustrates the preference for different food EOL management pathways. The Food Recovery Hierarchy emphasizes reduction at the source (i.e., loss prevention), directly reducing the environmental impact of the FSC on, for example, land, fertilizer, and water use. Food donation is the most preferred pathway to R&R FLW, followed by animal feeding, industrial use (i.e., anaerobic digestion and biochemical processing), and composting. While the EPA's hierarchy does not cover land application, this work discusses it and considers land application analogous to composting. Wastewater treatment is also not covered by the EPA's Food Recovery Hierarchy, but our analysis considers it to be a "least-preferred" stream with landfill and incineration (LWI).

The U.S. food demand and FLW generated from each FSC stage is projected to 2050 based on the food demand growth factors derived from Alexandratos and Bruinsma (2012). The fraction of FLW generation at each stage is assumed to remain constant. The 2016 mass flows and FLW estimates at each stage, discussed above, are used as the baseline year. Overall, while the U.S. food market is expected to grow because of population increases, experts do not expect any large, sudden changes in the market, with all commodities expected to grow 0.3%–1.0% per year through 2050 (Alexandratos and Bruinsma 2012).

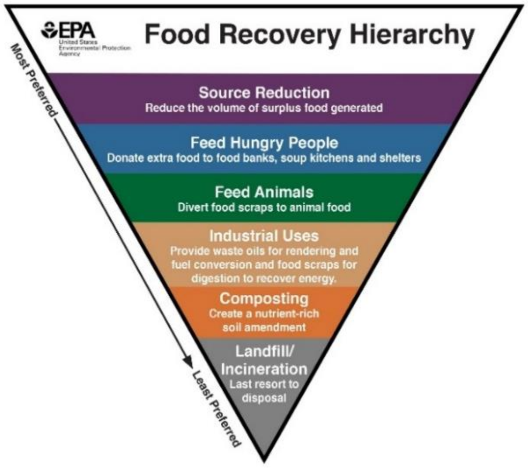

Figure 17. EPA's food recovery hierarchy

Source: "Food Recovery Hierarchy," U.S. Environmental Protection Agency, https://www.epa.gov/sustainable-management-food/food-recovery-hierarchy

5.2.2 Energy Consumption and GHG Emission Along U.S. FSC

FLW reduction will also reduce food demand—the food that needs to be produced—and all food flows through the FSC, helping reduce the FSC energy consumption and GHG emissions. In this study, the energy consumption and GHG emissions of U.S. FSC are based on or adjusted to 2016 data, and the FSC is divided into the same five stages to match the mass flow analysis in Section 5.2.1. We also include the energy consumption and GHG emissions of the on-farm production of animal feed, animal feed manufacturing, and transportation within the on-farm stage. The data for this analysis are taken from the EIA, the Freight Analysis Framework, USDA, and many other data sources (U.S. EIA 2015, EIA 2020a, U.S. EIA 2021, U.S. FHWA 2019, J. J. Brogan, et al. 2013, USDA 2020, Poore and Nemecek 2018, MNI 2017, U.S. EPA 2018). This study reports both site energy (energy used on site) and source energy (which breaks electrictiy down to the amount of fuels used to generate the site-electricity).

Overall, the total FSC requires 4,787 TBtu of on-site energy (site electricity, fuel, and manufacturing energy of fertilizers and chemicals and the growth and manufacturing energy of animal feed) or 7,258 TBtu of source energy.[28] This results in 984 MMT CO_2 eq emissions from energy consumption, fugitive emissions from refrigeration, and other on-farm activities. Figure 18, Figure 20, and Figure 22 show the breakdown of energy consumption and GHG emissions by contributor and stage. Figure 18 and Figure 19 illustrate the energy use in the U.S. FSC, broken down by the source of the energy contribution. Figure 18 details the on-site energy use, with the manufacturing energy of fertilizers and pesticides (referred to as Chemical Manuf. in the figures) and on-site electricity use shown as distinct entities, while Figure 19 distributes the energy for creating the electricity to the applicable energy sources. Figure 20 shows a similar diagram for GHG emissions by source. For these three figures, petroleum used for transportation from farm to manufacturing (within the on-farm stage) and from manufacturing to wholesale and retail (the distribution stage) are highlighted with diagonal lines. In Figure 21, the

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[28] The conversion of site to source energy for grid electricity was conducted using state-level energy use data (when available) and using each state's electricity profile for 2016 (https://www.eia.gov/electricity/state/archive/2016/). This allowed for source allocation, accounting for grid losses, and calculating GHG emissions. The calculation for the source energy of fertilizers and chemicals is detailed further in this section.

manufacturing site and source energy are broken down for the production of pesticides and fertilizer. Figure 22 presents a set of waffle charts aggregating site and source energy use and GHG emissions by energy source and FSC stage.

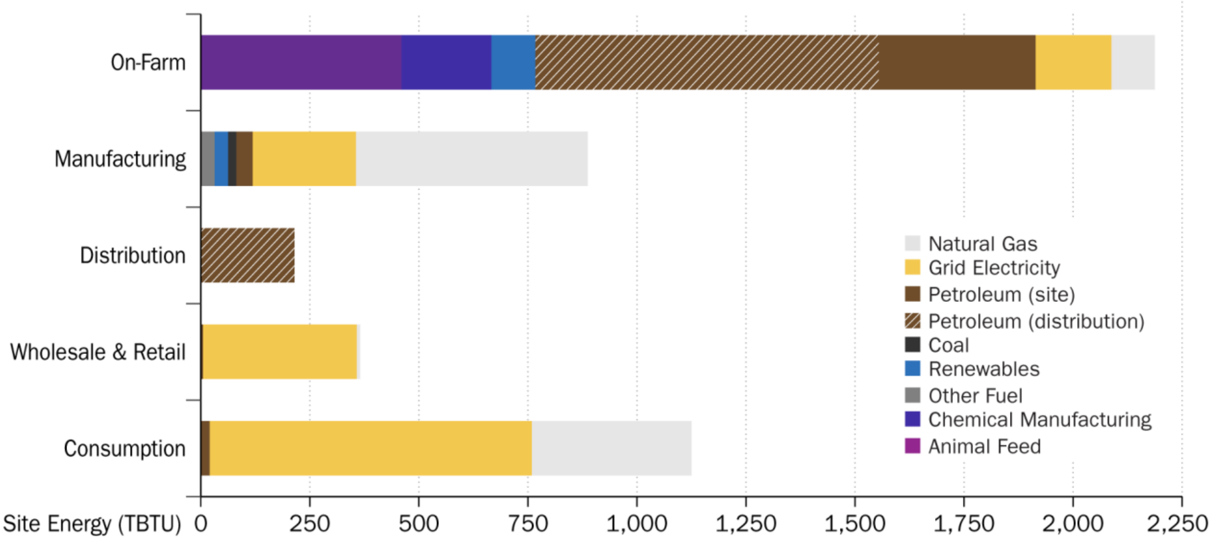

Figure 18. On-site energy use within the stages of the U.S. food supply chain by energy source (TBtu)

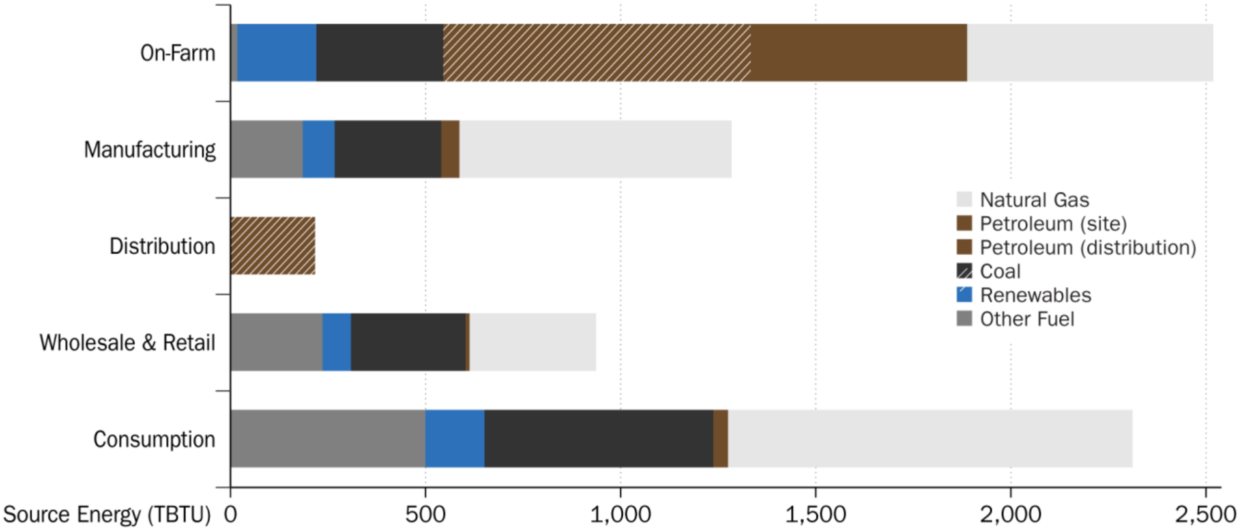

Figure 19. Source energy demand for each stage in the U.S. food supply chain by energy source (TBtu)

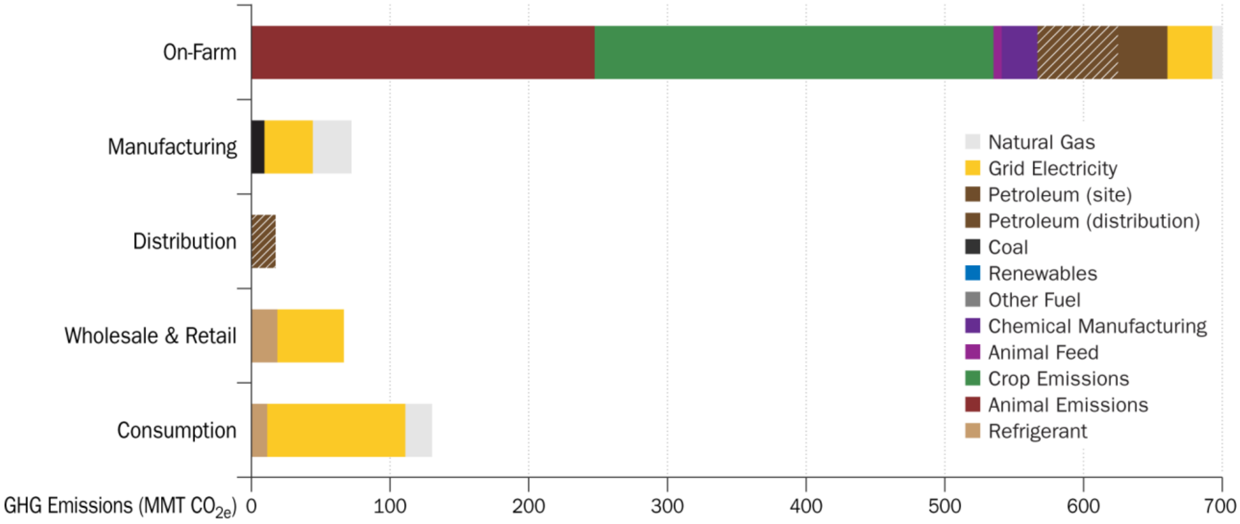

Figure 20. GHG emissions for each stage in the U.S. food supply chain by energy or emission source (MMT CO_2 eq)

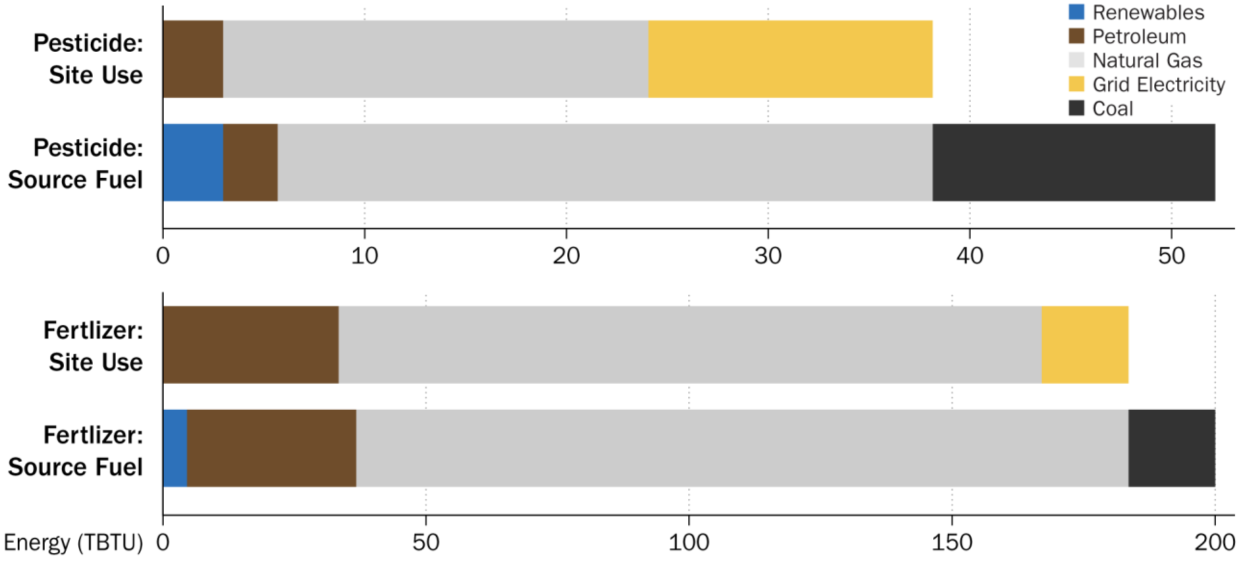

Figure 21. Manufacturing site and source energy breakdown for pesticides and fertilizer

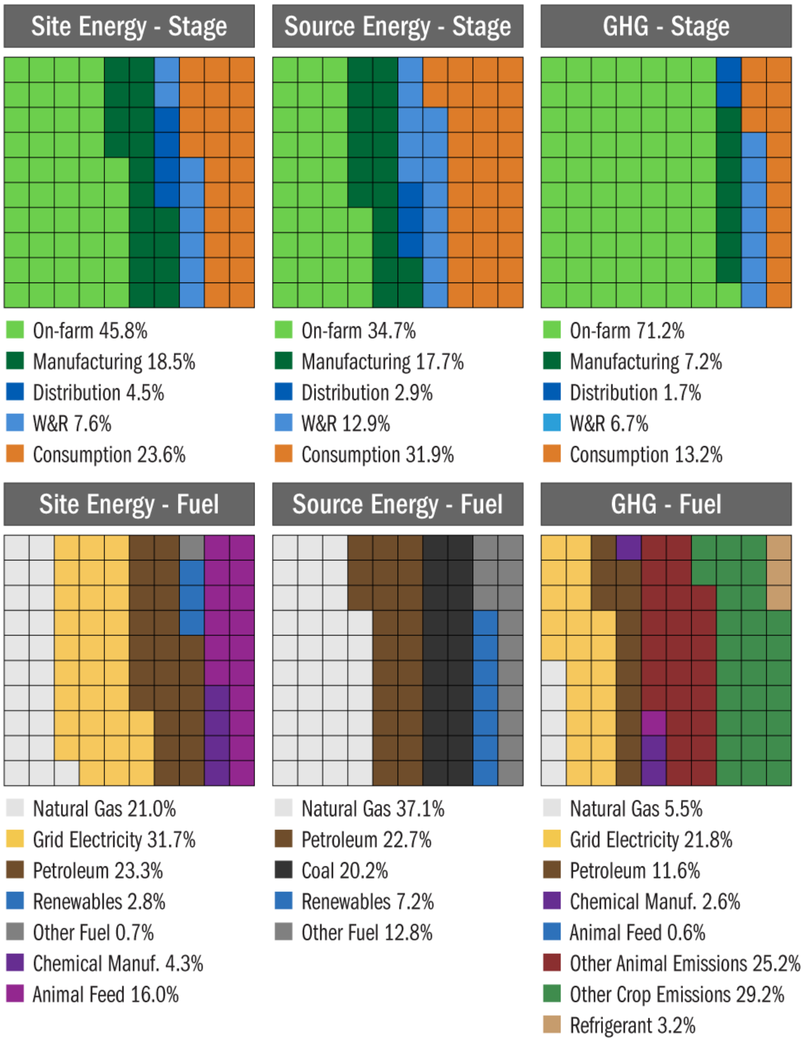

Figure 22. Breakdown of 2016 FSC on-site and source energy use and GHG emissions by contributing FSC stage and fuel type used.

5.2.2.1 On-Farm Energy

On-site energy consumption for on-farm production activity was 2,195 TBtu energy (46% of the FSC), the second-largest energy demand of all the stages. Of that amount, more than 735 TBtu was for fuel, electricity, or renewable energy consumed on the farm for human food production, and an additional 768 TBtu for animal feed production (growth, chemicals, transportation, and manufacturing). Energy associated with manufacturing fertilizers (nitrogen, potash, phosphate, and sulfur) and pesticides (herbicides, fungicides, and insecticides) applied to human food crops at this stage add 205 TBtu for human food crops.[29]

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[29] This represents only the manufacturing and transportation energy for pesticides and fertilizers. The available data for energy demand of agricultural chemicals is dated, with even more recent studies using energy data from the 1980s. For pesticides, older energy data were used for the energy use for individual chemicals (E. Audsley, K. Stacey and D. Parsons, et al. 2009), but an updated market mix (U.S. EPA 2017b) was used for a weighted average energy intensity. Fertilizer energy use data were from West and Marland (2002) without an updated market mix. Electricity was converted to source energy using the average U.S. grid mix (U.S. EIA 2020b).

Additionally, transporting food and feed between the farm and manufacturing required approximately 487 TBtu of energy, which is highlighted in Figure 18, Figure 19, and Figure 20 with diagonal legend coloring (MNI 2017, W. Dong, et al. 2022, U.S. FHWA 2019, AFIA 2020, U.S. EPA 2018, J. Brogan, et al. 2013). If the source energy for fertilizer, pesticides, and electricity is considered, this stage requires 2,517 TBtu, 35% of source energy for the FSC. Energy consumed for raising agricultural products for other uses—for example, consumer alcohol and industrial ethanol—is excluded from this analysis.

5.2.2.2 On-Farm GHGs

On-farm production activities generated 700 MMT CO_2 eq (71% of the FSC), by far the highest of all the stages. Of this amount, only 74.8 MMT CO_2 eq came from on-farm fuel and electricity use, 15.3 MMT CO_2 eq from animal feed use, and 25.3 MMT CO_2 eq from fertilizer and pesticide manufacturing for both human food and animal feed. However, there are several unique sources of GHG emissions at this stage, categorized in the Figure 22 above as other animal and other crop emissions. Animal digestion releases methane from enteric fermentation and methane and nitrous oxide (N_2O) from manure management. Human food and animal feed crop production releases methane from methanogenic bacteria in rice cultivation, N_2O release from soil, emissions from field burning, urea fertilizer CO_2 release, and other agricultural emissions. These contributed 248 MMT CO_2 eq directly from animals and approximately 287 MMT CO_2 eq from food and feed crops (U.S. EPA 2021). On-farm, non-fuel-related emissions are by far the largest contributor to the FSC GHG emissions, representing over 75% of the total FSC emissions. This does not include any land use contributions, which were relatively small in 2016: cropland staying as cropland was a sink for 22.7 MMT and land converted to cropland created an emission of 54.4 MMT, with a net emission of 31.7 MMT (U.S. EPA 2021).

5.2.2.3 Manufacturing Energy and GHGs

Manufacturing in the FSC required approximately 866 TBtu (19% of FSC energy) of on-site energy from fuels and electricity, or 1,283 TBtu (18%) of source energy (EIA 2020a, MNI 2017). This energy use, plus fugitive emissions from refrigeration, contributed approximately 71.2 MMT CO_2 eq (7.2%) of total GHG emission. Overall, manufacturing has the third-largest energy requirements and highest GHG contribution, although manufacturing does have a lower GHG emission than energy use because of its high natural gas use compared to other energy sources.

5.2.2.4 Distribution Energy and GHGs

The energy for the distribution stage covers transporting processed meat and seafood, milled grains, other agricultural products,[30] and other processed foodstuffs. The distribution stage required 213 TBtu of fuels (4.5% of FSC on-site energy) and contributed 16.2 MMT CO_2 eq (1.7%) (MNI 2017, W. Dong, et al. n.d., U.S. FHWA 2019, U.S. EPA 2018, J. Brogan, et al. 2013).

5.2.2.5 Warehouse and Retail Energy and GHGs

In total, W&R required 364 TBtu of on-site energy or 933 TBtu of source energy. As activities for this stage only include keeping food in stable conditions, without any preparation or transportation activities, it makes sense that this stage has the second-lowest energy demand (7.6%) and GHG contribution (6.8%).

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[30] While this includes processed fruits and vegetables, it also includes fresh produce, some of which may be misallocated to distribution from farm to manufacturing.

Its high GHG emissions to energy use ratio is caused by the high use of electricity and the 2016 electric grid composition.

Warehouses: USDA (2018) estimated that there are 166 million square feet (ft^2) of public and 48 million ft^2 of private cold storage food warehouses in the United States (USDA 2018), and CRBE (2019) estimated an additional 48 million ft^2 of private dry storage. While the energy intensity for warehouse storage varies by location and storage type (cold or dry), it has been calculated in the Commercial Buildings Energy Consumption Survey (CBECS) (U.S. EIA 2015). The U.S. average areal energy intensities for cold storage are 94,146 Btu/ft^2 for electricity and 21,153 Btu/ft^2 for natural gas. Similarly, U.S. average dry storage uses 18,071 Btu/ft^2 and 15,544 Btu/ft^2 of electricity and natural gas, respectively. A small amount of fuel oil is also used by both cold and dry storage. Overall, cooling and operating these warehouses required 32.8 TBtu of electricity, natural gas, and fuel oil, contributing 4.9 MMT CO_2 eq; while a small contribution of fugitive GHGs comes from refrigerants, it was estimated to be less than 0.01 MMT CO_2 eq.

Retail: According to the EIA (2020a), food retail required 332 TBtu of electricity, contributing 44.3 MMT CO_2 eq, plus an additional 17 MMT CO_2 eq from fugitive refrigerants in 2016.

5.2.2.6 Consumption Energy and GHGs

The consumption stage contains both food services and households. Food services' energy use is estimated from the commercial sector's energy for cooking (natural gas and electricity) and refrigeration, resulting in 582 site-TBtu, 1,087 source-TBtu, and 55.9 MMT CO_2 eq (plus 9.2 MMT CO_2 eq from refrigerant emissions). Households used energy for cooking (natural gas, propane, and electricity) and refrigeration and freezers, resulting in 545 site-TBtu, 1,224 source-TBtu, and 63.6 MMT CO_2 eq (plus 1.3 MMT CO_2 eq from refrigerant emissions). While the two segments of the consumption stage only use 32.7% of the FSC's on-site energy, they account for 43.1% of the FSC's source energy because of their high electricity use (EIA 2020a).

5.2.2.7 End-of-Life Energy and GHGs

Overall, disposing of FLW contributes very little new energy to the full food supply chain. Most of the energy for EOL disposal via food donation, animal feed, and land application/composting is already accounted for in the energy analysis, as the FSC currently includes them. However, the transportation from generation to use does contribute a small amount of energy: 0.15 TBtu/MMT for food donation and animal feed, and 0.81 TBtu/MMT for composting and land application. EOL disposal via LWI and industrial uses (e.g., anaerobic digestion, biofuel, bioplastic manufacturing) and industrial composting (e.g., utilizing large-scale turners) require more energy input unaccounted for in the FSC (U.S. EPA 2019, Oregon DEQ 2014, Tsang, et al. 2019). Figure 23 shows the additional energy demand and GHG emissions required for processing FLW via these EOL pathways that is not already accounted for in the FSC analysis (144 TBtu and 70 MMT CO_2 eq). The black bar represents the net energy demand, as some of the pathways also generate electricity or fuels (e.g., combustion of landfill gas or gas produced from AD to produce electricity, which offsets site electricity use or source fuel use). The graphs on the right side of Figure 23 also show the additional on-site energy and GHG emissions (in green) as a portion of the total stage energy and GHG emissions (in orange as illustrated previously in Figure 18 and Figure 20). The figure shows that for most stages, the additional EOL energy contributions are quite small (with on-site energy ranging from 1.3% to 4.9% of the total stage energy use), although because of the large fugitive emissions of landfilling, GHG emissions are a relatively larger portion of the total impact (2.1%–25.5% of the total stage emissions).

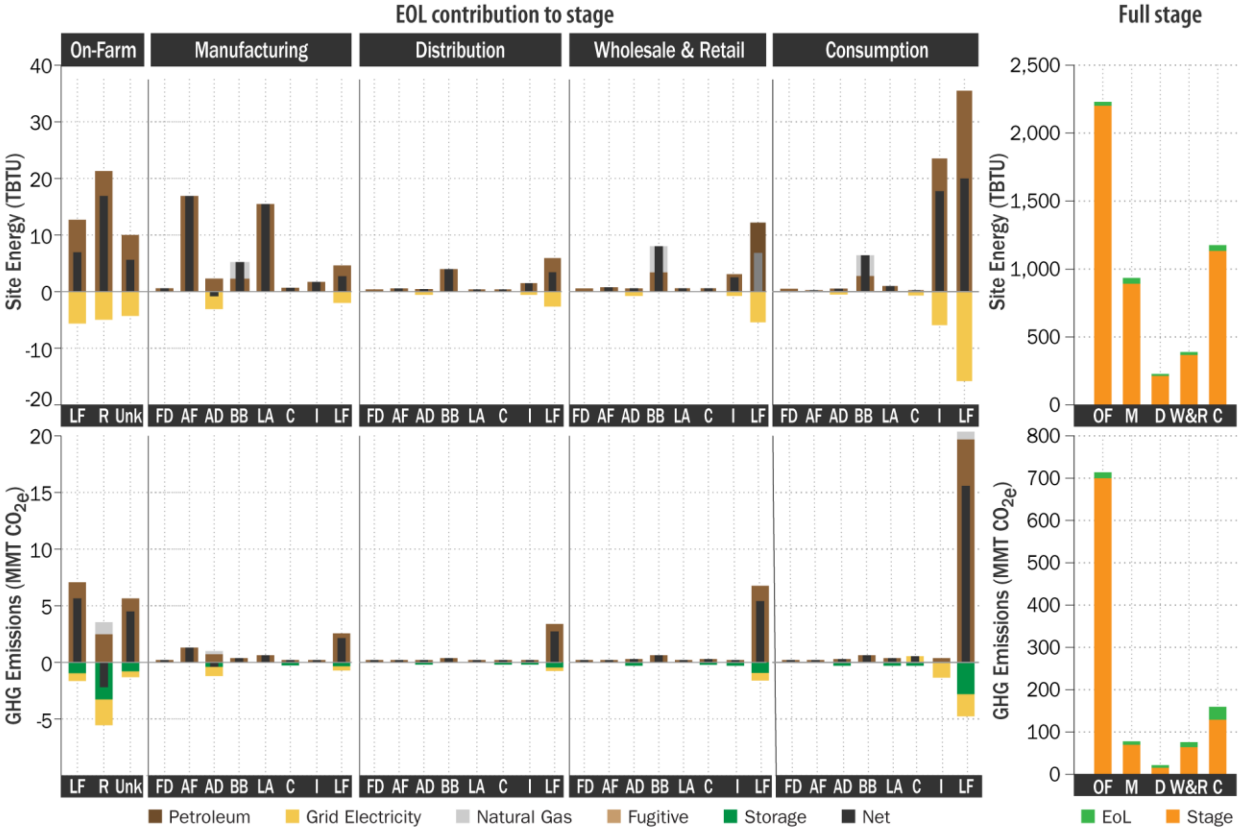


Figure 23. Breakdown of EOL contribution to full stage site-energy requirements and GHG emissions for several EOL FLW pathways by source of contribution and stage (left) and relative size of EOL energy use and GHG contribution as part of the total stage contribution (right)

Note: Pathways covered include animal feed (AF), anaerobic digestion (AD), bio-based/bio-chemical (BB), composting (C), wastewater (WW), incineration (I), landfill (LF), unknown reuse method (R), and unknown, non-reuse, method (Unk).

For brevity, on the right, the stages are abbreviated as OF (on-farm), M (manufacturing), D (distribution), W&R (warehouse and retail), and C (consumption).

In addition to producing energy, some of these pathways have benefits already included in this analysis as the FSC currently includes them. These include contributing human food back to the FSC, generating animal feed, and producing fertilizer. But, if the allocation of FLW changes, additional offsets and burdens will be provided. The impact of each FLW EOL pathway is different for each commodity studied, as shown in Figure 24. However, the pathways are not universally appropriate. For example, only certain types of FLW can go to food donation; animal feed and anaerobic digestion (AD) must maintain certain ratios of commodities.

Figure 24 shows the energy impact per pound of FLW of the commodity to be disposed. For commodity-disposal pathway combinations that are not used or able to be used (e.g., animal products cannot be used for land application), the impact is zero. If the combination is possible for only a portion of the FLW (e.g., only the whey from dairy can go to animal feed; the remaining dairy FLW must be disposed of via other methods), the impact is multiplied by the fraction of FLW that can be disposed of via that pathway. This modified impact allows comparisons of the actual impact of FLW disposal method between commodity groups (e.g., animal feed is a better disposal method than anaerobic digestion for animal products, but not dairy products).

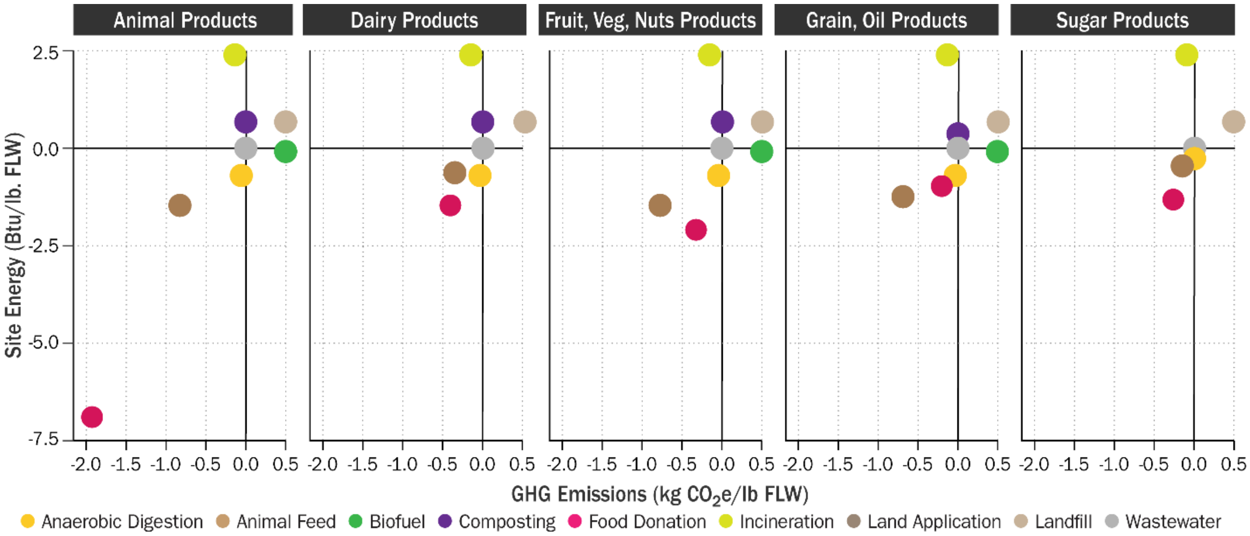


Figure 24. Energy intensity potential impacts and offsets of FLW EOL pathways

Note: Pathways evaluated include animal feed (AF), anaerobic digestion (AD), wastewater (WW), biofuels (BF), landfill (LF), composting (C), and incineration (I).

For most commodities, food donation is the most preferred pathway for EOL disposal, even after applying a multiplier for the fraction of food that is acceptable to donate (e.g., not already prepared food, scraps, spoiled food). Repurposing food via donation can offset 477 to 3,441 Btu per pound (lb) of all FLW (or 4,150–20,200 Btu per lb of appropriate FLW), as that represents food that does not have to be created, cared for, transported, and prepared. The only exception is grain and oil products, because of their low suitability for food donation (8%, as they tend to spoil the most quickly). Animal feed is the next most beneficial EOL pathway, again because it replaces primary food growth, offsetting 102–726 Btu/lb of FLW (or 726 Btu/lb of appropriate FLW). As most products are suitable for anaerobic digestion, it is the next most beneficial EOL pathway, followed by either wastewater treatment, land application, or biofuels. Composting was shown to have a net energetic burden as a disposal method because of the industrial stirrers used when bulk composting; smaller community or at-home composting would be more similar to land application. Overall, the worst two EOL methods were incineration and landfill; even with their potential electricity offsets, they are still a substantial energetic burden on the FSC, although as incineration produces more electricity than landfilling and does not have as much fugitive emissions, it has a net negative GHG emission.

Further analysis is needed to better understand the impacts of reallocating FLW to different EOL pathways and EOL GHG impacts of the EOL pathways. The EPA's Waste Reduction Model (WARM) (U.S. EPA 2019) helps quantify the EOL emissions of food waste, showing that some pathways can have significant emissions (landfilling emits 500 kg CO_2 eq per short ton of waste) and others can be net GHG sinks (e.g., composting, anaerobic digestion). While the initial analysis is complete, further analysis of the total 2016 FLW EOL emissions and the emission intensity of all the pathways will help improve our understanding of the full impacts of FLW and the best ways to dispose of FLW.

5.2.2.8 FSC Energy and GHG Projection

As shown in Figure 25, this analysis also projects the on-site energy use and GHG emissions for the U.S. FSC through 2050, based on projections from the original sources and the estimated changes to the FSC and excluding energy use and GHG emissions from FLW EOL pathways. Overall, we expect both to increase from the 2020 EIA Annual Energy Outlook (AEO) Reference case (EIA 2020a), which did not include aggressive decarbonization strategies. However, with increasing renewable energy and electrification in the energy generation portfolios, GHG emissions will dip before increasing again as food demand increases. This analysis is a simpler estimation of the current and future U.S. FSC, based more directly on the EIA's AEO. Projections for fertilizers, pesticides, and on-farm crop and animal emissions were estimated based on the increase in animal and crop demands (fugitive emissions were not included). As expected from the slow growth of the food industry, the required energy also increases over the next 30 years, GHG emissions dip slightly because of expected grid improvements and similar activities.

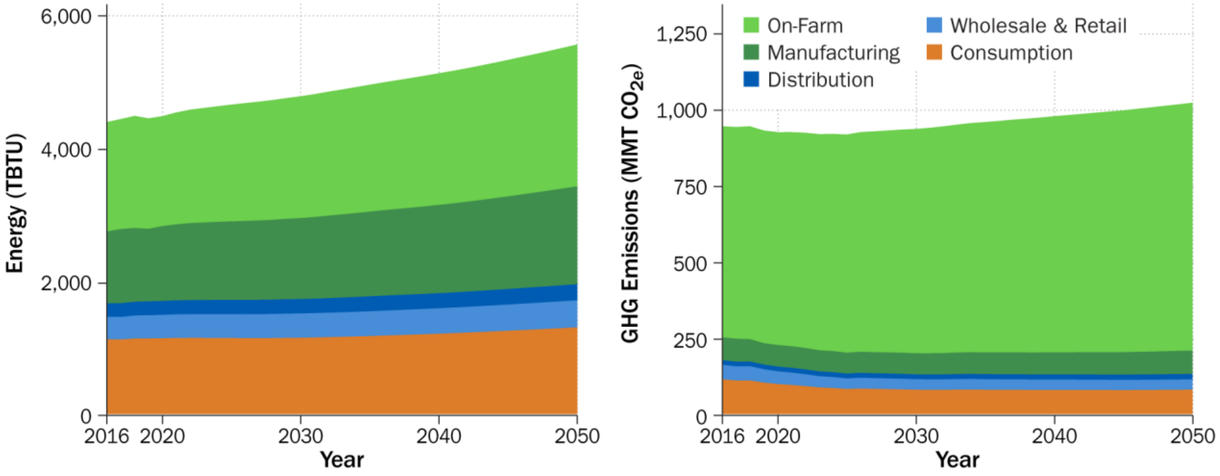


Figure 25. FSC site-energy and GHG projections through 2050

5.2.3 Technologies and Practices for Sustainable FSC

Numerous technologies and practices for the various FSC stages have been proposed to reduce FLW generation (see Table 5). However, this study focuses only on actions and technologies implemented in the manufacturing stage as manufacturer improvements may reduce FLW from subsequent stages and on-farm production (as a result of demand reduction). Through a comprehensive literature review, we have identified several technologies and practices that food manufactures can adopt to reduce the U.S. FLW. These include, for example, nonthermal processing, membrane separation, and changing product grading standards and packaging to prevent food products from spoilage.

We divide these technologies and practices into two groups:

- Solutions to reduce FLW during food manufacturing
- Solutions to reduce FLW at the W&R and consumption stages with prolonged product shelf life and improved packaging alternatives.

Given the high level of FLW recycling in the manufacturing sector and high waste in the later FSC sectors, this study focuses on the impact of the second group of solutions. Specifically, we focus on advanced packaging technology and practices, which are considered some of the most economically and environmentally impactful strategies for preventing FLW downstream in the W&R and consumption stages [(The Rockefeller Foundation 2017, American Institute for Packaging and the Environment 2018, ReFED 2016, Lipinski, et al. 2016, S. Kaza, et al. 2018).

Table 5. Examples of Solutions of FLW Reduction Beyond Packaging and Shelf-Life Improvements

FLW Reduction Technologies or Practices	Applicable Stage	Description	Potential Benefits
Mobile service to farmers with tailored data	On-farm production	Provide farmers with tailored data (seed / input selection and availability, better weather forecasts), best practices, and improve negotiation leverage at the time of sale	Reach 275–350 million farms worldwide (by 2030): 250–500 MMT more food production 20–65 MMT less FLW[1]
Off-grid renewable energy generation and storage for access to electricity	On-farm production	Off-grid renewable electrification that uses solar energy provides a sustainable alternative for diesel- and gasoline/petrol-fueled mini-grid and off-grid generation systems	Install on 50–75% of farms (worldwide) lacking access to electricity, 10–15 MMT less FLW due to added cold storage[1]
Manufacturing line optimization	Food manufacturing	Identify opportunities to reduce FLW from manufacturing / processing operations and product line changeovers	5–15% of industry-wide FLW reduction [2]
Sensing technology/smart packaging	Wholesale and retail, and household	Sensors or smart packaging to monitor quality and safety of food.	Reaching 30–50% of consumers, 10–20 MMT fewer FLW[1]
Dynamic pricing system	Wholesale and retail	Dynamically optimize pricing based on food quality and economic factors	50% less FLW[3]
Waste tracking analysis[2]	Food services	Providing restaurants and prepared-food providers with data on wasteful practices to inform behavior and operational changes	Reduces preconsumer / kitchen FLW by 20–40%[2]

Sources: [1] (Nayyar, De Cleene and Dreier 2018); [2] (ReFED 2016); [3] (Wasteless n.d.)

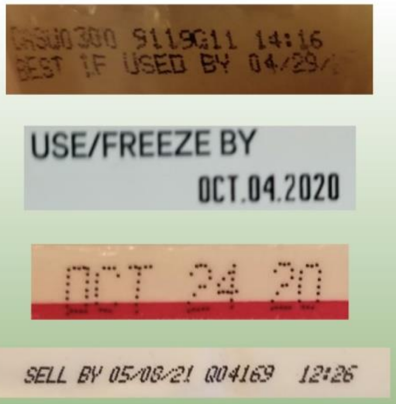
The next section investigates the packaging-related issues that contribute to the FLW at the W&R and Consumption stages and discusses the innovative packaging technologies and practices in terms of their impact on FLW reduction, implementation barriers, and R&D opportunities.

5.3 Advanced Packaging Solutions at the Food Manufacturing Stage

5.3.1 Packaging-Related Issues

The U.S. FSC generates 77.2 MMT of FLW disposed of via LWI. The consumption stage only accounts for 14.5% of total FLW but contributes 44.2 MMT (57.2%) of landfilled FLW. There are numerous causes of consumption FLW, such as overpreparing and trimming; however, most food thrown away at this stage is expired or assumed to be expired. Food discarded by households is overwhelmingly composed of products with a short shelf life, for example, fresh vegetables, fresh fruit, meat, eggs, dairy, and bakery products (Lyndhurst, B. 2011). Studies have also shown that consumers incorrectly assume that food date labels are a safety indicator and often describe them as one of the main drivers of household FLW (Aschemann-Witzel, et al. 2015, Williams, et al. 2012, Lyndhurst, B. 2011, NRDC 2013, Verghese, Lewis, et al. 2013, Verghese, Lewis, et al. 2015, The Rockefeller Foundation 2017). Dates printed on food and drink packaging ("Use By," "Sell By," and "Best Before") are intended to provide consumers with information regarding the freshness of foods. In the United States, these dates specifically refer to food quality or flavor, not to food safety. Only infant formulas have food safety dates that convey whether the food could potentially cause illness when eaten (Lipinski, et al. 2016). Further, there are no federal standards for manufacturers regarding their use (USDA 2019).

What is Wrong with Food Date Labels?



Food date labels (Best By, Use By, Sell By) can be confusing to consumers. While assumed to be a safety label, in the U.S. it only refers to quality or flavor retention. Standardizing labels could save up to 25% of the food thrown out at the Consumption stage.

However, because consumers tend to assume these labels relate to food safety, they question how long it is safe for them to store food and when they should dispose of uneaten items, resulting in throwing food out "just to be safe." While some of this food may actually be spoiled, studies suggest that household FLW can be reduced by 20–25% through standardized labeling and customer education (Williams, et al. 2012, ReFED 2016). Furthermore, Verghese et al. (2013) estimate that 5% of FLW at restaurants is related to date label confusion.

5.3.2 Potential Solutions

Packaging plays an essential role in creating sustainable food chains. It prevents FLW by delaying spoilage, avoiding product damage during transportation and storage, right-sizing food portions, offering closable packaging, and providing information for consumers—such as expiration date, nutrition, storage guidance, and branding.

While a large amount of food is discarded because of spoiling, most studies conducted cannot differentiate between actual or assumed spoilage. In either case, extending the length of time a food item has between harvest or packaging and spoilage (shelf life) can reduce this waste (Lee, Osborn and Whitehead 2015), (Yildirim, et al. 2018), (Zhang, et al. 2015), (S. Chen, et al. 2020), (American Institute for Packaging and the Environment 2018), (ReFED 2016). Lebersorger and Schneider (2014) explored the causes of food discarded at the W&R stage and found that the main drivers are similar to those in the consumption stage: product expiration and product/package damage related to packaging. Extending shelf life by preserving freshness and protecting food gives retailers more time to sell and consumers more time to consume the food before it spoils, and many technologies and activities at all levels of technology readiness aim to extend shelf life. For example, pasteurization and other sterilization technologies that have been used for decades greatly extend milk and juice shelf life (Putnik, et al. 2020). Cutting-edge technologies (e.g., active packaging[31] and edible coatings) can further extend the shelf lives of food commodities (E. Almenar 2020). Conversely, simple solutions such as reducing package portion size, such that all the food can be consumed before spoiling, can help reduce consumption FLW (Williams et al. 2012). While studies have investigated the reduction of FLW from novel packaging, the full life cycle implications of changing packaging are not yet well established. Novel packaging will likely involve increasing the relative amount of packaging materials, thus increasing the packaging waste, energy consumption, and GHG emissions, although some claim this increase will be offset by the reduction in FLW and demand (FPA 2014), (American Institute for Packaging and the Environment 2018).

In this study, we quantify the impact of the most popular packaging solutions and discuss implementation barriers and R&D opportunities.

5.3.2.1 Date Labeling System

Confusion about the date labeling system and product spoilage is more impactful than most other packaging-related causes of FLW and is considered a higher priority by many (NRDC 2013), (Williams, et al. 2012), (American Institute for Packaging and the Environment 2018). Consumers' confusion regarding the definitions of the labels they find in the market contributes to 20–25% of FLW from households. To address this issue, food manufacturers and government agencies should work together to

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[31] Active packaging materials are designed to extend the shelf life, maintain, or even improve the condition of packaged food. They incorporate components that release (e.g., antimicrobial compounds, ethylene) or absorb (e.g., moisture, carbon dioxide) substances into or from the packaged food or the environment surrounding the food. They can be placed inside the package along with the product or directly as part of the packaging (EC 2009) (E. Almenar 2020) (Yildirim, et al. 2018).

establish a comprehensive national date labeling system based on food safety considerations, with reliable, coherent, and uniform language that clearly communicates to consumers the meaning of dates as well as other safety and handling information. The federal Food and Drug Administration (FDA) and USDA have been authorized to regulate misleading labels and could develop a standardized, less arbitrary system of date labeling that consumers can understand (NRDC 2013). In 2016, the USDA issued updated information to manufacturers and retailers who want to apply a quality-based date, which recommended using “Best if Used By,” and updated their guidance to consumers as to what that means. In response, the Grocery Manufacturers Association and Food Marketing Institute suggested using “Best if Used By” for quality-based labels and “Use By” for safety-based labels, although the FDA is not addressing the use of “Use By” (U.S. FDA 2019). Additionally, several states have laws in conflict with these guidelines, often requiring more specific labels for certain commodities (e.g., “Packaged on” for eggs in Colorado and “Pasteurized on” in Montana) (FL&PC 2019). While these efforts are commendable, confusion is still inherent when different states have different requirements, especially when they conflict with the current guidance. Regardless, food manufacturers can reduce confusion about date labeling, and the resulting FLW, by removing unnecessary or confusing dates from consumer packages. Dates important only to the manufacturer or retailer (e.g., “Sell By”) could be hidden on bulk packaging, while dates relevant to consumers could be rephrased to improve clarity (e.g., implementing the FDA’s suggested language). These actions would only require small shifts and investments, making them easy to implement. Thankfully, the Grocery Manufacturers Association found that 87% of respondents on a member survey had implemented the “Best if Used By” and “Use By” labels as of December 2018 (Roe, et al. 2019), although this survey was not comprehensive for the entire industry, and there is often a delay before seeing these changes reflected on store shelves. Additionally, consumer education is essential to clarify the definitions of new manufacturers’ or standardized date labels and provide guidelines for storing foods and determining food safety without reference to date labeling (ReFED 2016).

5.3.2.2 Innovative Packaging Technologies

As mentioned in Section 5.2.1, many established and emerging food packaging technologies reduce food spoilage and extend shelf life. For example, active packaging materials is one of the technologies receiving significant attention from researchers. The European Commission (2009) defines active packaging materials as “materials and articles that are intended to extend the shelf-life or to maintain or improve the condition of packaged food; they are designed to deliberately incorporate components that would release or absorb substances into or from the packaged food or the environment surrounding the food.”

The large variety of active packaging materials developed fall into two types: active scavenger and active-releasing emitters. Scavengers remove undesired compounds, for example, moisture, carbon dioxide, and oxygen. Active-releasing emitters add compounds to the packaged food or the headspace, such as antimicrobial compounds, carbon dioxide, antioxidants, flavors, ethylene, or ethanol (EC 2009), (Yildirim, et al. 2018). The active compounds are applied by placing them inside the package with the product or making the active material part of the package (E. Almenar 2020). According to Yildirim et al. (2018), active packaging systems can extend food products’ shelf life by days, or even months (depending on the food product, active compounds applied, and storage conditions).

Another innovative packing technology, intelligent (smart) packaging systems, uses data carriers, indicators, and sensors to monitor the condition of packaged food or the surrounding environment. While these systems do not directly prolong product shelf life, they provide consumers with information on storing and consuming the food in time to avoid waste and losses (Muller and Schmid 2019), (S. Chen, et al. 2020). Other technologies developed include in-package cold plasma, which supports a nonthermal method for killing microorganisms after the food is packaged. In addition, films with permeability adjustable to temperature change can accommodate the respiratory needs of produce in a cold chain (E. Almenar 2020).

5.3.3 Quantifying the Value of Packaging Solutions

While numerous studies have emphasized the benefits of advanced packaging solutions for reducing FLW, few quantify their impact. Rethink Food Waste Through Economics and Data (ReFED) (2016) is the only study to reveal the impact of packaging solutions in terms of FLW reduction and cost-benefit analysis. However, that study considered only the landfilled FLW of several specific food commodities and failed to estimate the impacts on GHG emissions, life cycle energy consumption, or total FLW and food demand.

In this study, we examine the impact of introducing standardized date labeling and extending shelf life on FLW and food demand reduction and quantify the related savings for GHG emissions and energy consumption. To estimate the impact of extending product shelf life by 1, 2, and 3 days, regardless of the technology used to achieve this, we assume that all food that is thrown out because of spoilage (or assumed spoilage) 1 to 3 days past its “Best By” date is instead consumed. We also assume that this avoided FLW is generated by perishable, short-shelf-life food products between 3 and 12 days (W. Dong, et al. 2022) (NRDC 2013), where a shelf-life extension of 1 to 3 days will have the most impact. The FLW caused by spoilage for each food group and the potential reduction of FLW from extending shelf life are derived from the literature (Lyndhurst, B. 2011), (Lebersorger and Schneider 2014), (Lee, Osborn and Whitehead 2015), (NRDC 2013), (Williams, et al. 2012).

This method is agnostic to the cause of the shelf-life extension; waste reductions could be a result of the implementation of innovative packaging technologies in the manufacturing stage, or even a change in the harvest-manufacturing-distribution schema (e.g., distributed food production, four-season growing facilities) such that it takes less time to deliver food to consumers, who then have more time to eat it before it spoils. We assume a linear adoption rate of new technologies to prolong shelf life, setting the rate to 0% in 2016, when no new technologies had been implemented, and to 100% in 2050, when food is no longer thrown out within 1 to 3 days of the current shelf life. The adoption rate of a technology is usually nonlinear and should be estimated through comprehensive market research or observing the adoption pattern of similar, previously adopted technologies. However, data demonstrating the adoption rate of packaging technologies is very limited, and original market research is needed to estimate a better adoption rate.

This study also analyzes the benefit of standardizing date labels. We assume that standardized labeling (i.e., food labels with clear safety-related information) would reduce FLW in households by 20% and food services by 5%, based on literature values (Williams, et al. 2012) (NRDC 2013). We also assume that none of the FLW at the W&R stage results from misunderstanding labels.[32] Finally, we assume that

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[32] No data were found regarding the food label confusion at the W&R stage, so the impact of improved labels is unknown. We assumed that wholesalers and retailers are more familiar with the different labeling system definitions and therefore not prone to misunderstanding the label and causing FLW. However, there may be FLW at this stage caused by consumers not purchasing food near the label date, which could cause confusion-based FLW. Regardless, this assumption was chosen to represent a conservative estimate of the impact of improved labels on the full FSC.

the regulation of standardized labeling had been finalized in 2016 and would have taken 5 years to become effective (U.S. FDA 2020). Figure 26 and Figure 27 show the impact of extending shelf life and standardizing date labeling on the reduction of consumer FLW (and thus demand), total FLW (excluding unharvested FLW), demand (on-farm harvested for human food use), energy consumption, and GHG emissions.

Prolonging product shelf-life by 1, 2, and 3 days can reduce the 2050 FLW by 6.5 MMT, 14.0 MMT, and 18.9 MMT, respectively. Unsurprisingly, the benefits of increasing shelf life have diminishing returns. Extending shelf-life a single day reduces annual FLW 1.81% by 2050; adding a second day will reduce FLW a further 2.14%. However, adding a third day provides only an additional 1.41% reduction in FLW, as most food is thrown out within 1 or 2 days of its expiration date (only about 11% of food was thrown out 4 or more days after the expiration date in the original study) (Lee, Osborn and Whitehead 2015). This result implies that a detailed cost-benefit model for any new technology is required to determine if FLW and demand reductions are sufficient to justify the technology costs and other burdens.

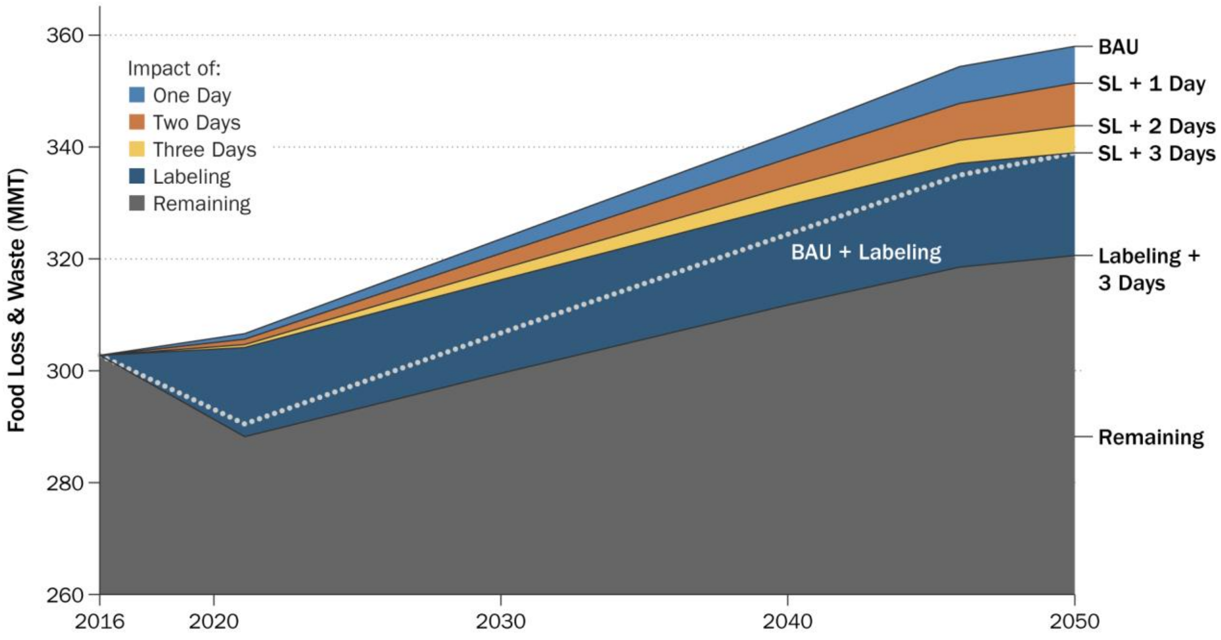


Figure 26. Reductions in FLW due to shelf life (SL) extension and implementation of standardized-safety-related date labeling (Labeling) on U.S. FLW

Note: Four scenarios are evaluated and include SL extension (1–3 days) and implementing safety-related label standards with a 3-day SL extension.

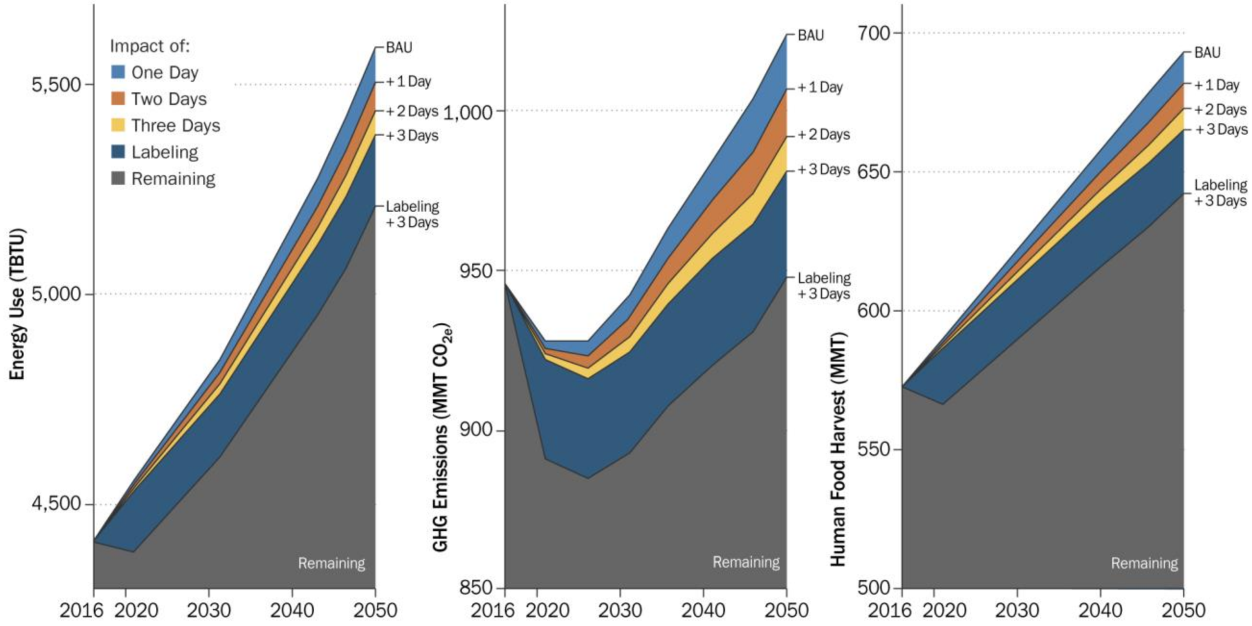


Figure 27. Impact of shelf-life extension and implementation of standardized-safety-related date labeling (Labeling) on energy consumption, GHG emissions, and food demand (required human food harvest)

Additionally, extending products' shelf life can reduce the demand—mass flow—at each stage, and by 2050, reduce the annual required food harvested for human food use by 11.5 MMT (1.7%) for a single-day extension, 20.9 MMT (3.1%) for 2 days, and 28 MMT (4.1%) %) for 3 days. As Figure 26 and Figure 27 also show, the impact of implementing standardized labeling alone is roughly equal to extending shelf life by 3 days (5.2%, 18.6 MMT) in terms of reducing 2050 FLW. Implementing standardized labeling and extending shelf life by 3 days could reduce the 2050 FSC FLW by 10.4% (37.2 MMT) and reduce demand by 7.4% (51 MMT).

Increasing the shelf life of unstable foods—such as fresh meat, produce, and bakery items—effectively reduces the production of these products at each stage, reducing direct energy use, fertilizer demand, and other crop/animal GHG emissions at each FSC stage. This study also analyzes savings on energy consumption and GHG emission achieved by demand reduction at each stage. Here, we demonstrate only the reduction of annual energy consumption and GHG emission along the FSC for 2050. Focusing on the different stages of the FSC (shown in Figure 28), extending product shelf life by 1, 2, and 3 days can reduce total 2050 energy consumption along FSC by 82 TBtu (1.3%), 151 TBtu (2.4%), and 204 TBtu (3.3%), respectively, for 2050, and reduce GHG emissions by 17.6 MMT CO_2 eq (1.6%), 32.4 MMT CO_2 eq (3.0%), 43.8 MMT CO_2 eq (4.1%), respectively.

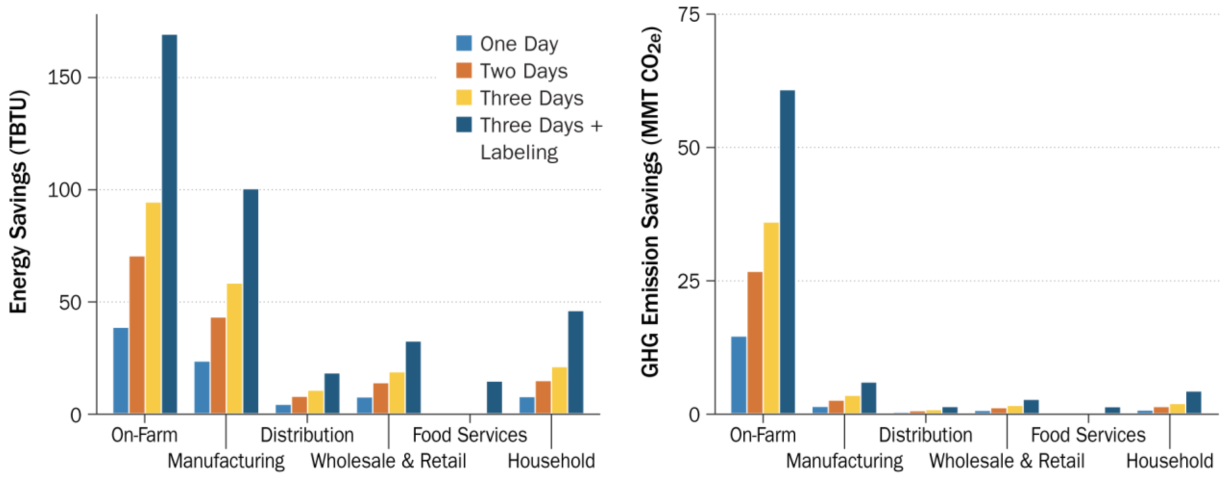


Figure 28. 2050 annual energy and GHG emission reductions because of avoided food production.

Note: Four scenarios evaluated include SL extension (1–3 days) and implementing safety-related label standards with a 3-day SL extension.

Overall, extending product shelf life by 2 days seems to have the best marginal impact. Adopting standardized labeling and extending products’ shelf life can achieve savings on 2050 energy consumption of 382 TBtu (6.2%) and GHG emission of 76.4 MMT CO_2 eq (7.1%).

These case studies do not account for the impact of these changes on the energy demand or GHG emissions relating to EOL disposal. Most household FLW (67%) is disposed of via landfill. Assuming that the FLW avoided via a 3-day shelf-life extension and standardized labeling would also have gone to landfill, an additional 21.2 MMT CO_2 eq would be avoided (U.S. EPA 2019).

Estimating the impacts on the energy use or GHG of the EOL pathways for the additional avoided FLW requires further analysis. For example, a large portion of FLW along the U.S. FSC is recycled through animal feeding and land application. Researchers have not yet analyzed the impact of reducing the recycled FLW—for example, growing more animal feed and producing or applying more fertilizer. Moreover, more analysis is necessary to quantify the impact on energy consumption and GHG emission of the new packaging technologies.

5.3.4 Barriers and Opportunities

Our analysis demonstrates the benefit of packaging solutions on reducing FLW, food demand, energy consumption, and GHG emission. However, although innovative packaging solutions are already in use, with new technologies and solutions under development, adoption is slow despite a low monetary investment requirement (e.g., standardized labeling) (The Rockefeller Foundation 2017).

For novel packaging technologies and shelf-life enhancements, the associated increase in costs and reduction of waste, emissions, and energy use are highly variable for each food commodity. Work must focus on the impact of adopting new packaging materials and practices for the major commodities: analyzing the benefits of adopting packaging, estimating the (likely) higher costs, and identifying the potential issues, such as environmentally harmful material and more packaging waste (ReFED 2016) (The Rockefeller Foundation 2017).

Although companies' food manufacturers and retailers may commit to reducing FLW to demonstrate corporate stewardship and present a green image, the uncertain environmental impacts of novel packaging materials prevent them from adopting a packaging solution. Moreover, there is no guarantee that standardized date labels would change consumer behavior, as many consumers throw food away when they *think* it has spoiled. This behavior and the difficulty in achieving a voluntary, consistent labeling system across different manufacturers are both significant deterrents to a manufacturer-led approach (ReFED 2016). Furthermore, consumers hesitate to purchase novel packaging materials, not trusting their safety (American Institute for Packaging and the Environment 2018), and thus thwart manufacturers' adoption of packaging innovations. In addition, consumers' willingness to pay more for products with a longer shelf life is untested, which may require retailers to bear the extra cost, further reducing adoption. Although FLW-reduction packaging solutions have been shown to reduce the overall environmental impact of food and its packaging (Silvenius, et al. 2011), many consumers hesitate to buy products with enhanced packaging, believing that this action will have a more negative environmental impact (American Institute for Packaging and the Environment 2018). Better consumer education is required to promote the adoption of advanced packaging technologies. Finally, the low monetary benefits do not incentivize manufacturers to adopt packaging solutions.

To overcome these barriers, researchers should focus on improving the reliability and efficiency of packaging technologies while reducing their costs. For example, they may need to develop cost-share models to distribute the cost to stakeholders—food and packaging manufacturers and consumers. Governments may also need to issue incentive policies to promote the adoption of packaging solutions. Finally, more studies are needed to quantify the benefits of adopting these packaging solutions from different perspectives, including cost-benefit analysis and FLW reduction. The results should be freely shared with customers to change their attitudes and ease their concerns.

5.4 Conclusion and Future Study

This work aims to provide a detailed and comprehensive picture of the U.S. FSC with a focus on food waste generation and disposal and its environmental impact. FLW reduction is considered a pathway toward sustainable FSC, and this study identifies multiple solutions for preventing FLW generation at each FSC stage. If the U.S. FSC continues current practices, we estimate the total FLW generated by 2050 to be 357.8 MMT, with 693 MMT of food entering the manufacturing stage (food demand). We estimate that FLW will consume 6,173 TBtu of energy and emit 1,075 MMT CO_2 eq annually by 2050.

While the U.S. FSC appears inefficient, as humans consume only 19.1% of the food that enters the manufacturing stage, according to Dong et al. (2022), most of the FLW in the FSC is not considered edible. FLW contains offal, bones, oil meal, eggshells, grain milling byproducts, and evaporated water. This inedible FLW is usually recycled for other users, such as for animal feed and land application (22.2% of food demand), and only 10.5% of the food demand is disposed of via LWI. However, while the consumption stage only contributes 14.5% of the total FLW, consumption FLW accounts for 57.2% of landfilled FLW, and recommendations for reducing FLW focus on this stage rather than the whole FSC (U.S. EPA 2012).

This case study estimates the impact of applying advanced packaging solutions at the food manufacturing stage on food demand and FLW and the corresponding effects on energy consumption and GHG emissions. It also discusses the barriers preventing the implementation of packaging-related solutions and identifies potential R&D opportunities.

The study finds that extending product shelf life and introducing standardized labeling can result in annual reductions of FLW (37.2 MMT), food demand (52.1 MMT), energy consumption (378 TBtu), and GHG emissions (76.5 MMT CO_2 eq) by 2050. Moreover, this study demonstrates that although extending product shelf life by additional days can reduce FLW and food demand, the marginal impact of extending product shelf life is concave in terms of the days extended. According to our analysis, extending shelf life by 2 days will bring the best marginal impact compared with prolonging it 1 or 3 days. Understanding the trade-offs between waste reduction and the costs of new technologies will be required before new technologies are adopted.

What is the impact of 76.5 MMT CO_{2e} and 52.1 MMT of food?

- **19** coal-fired power plants burning **84.5 Billion lb.** coal to fuel **13.9 Million** homes' annual electricity use
- **3.1 Billion** propane cylinders used for home barbeques
- **100 Million** additional people fed
- **3.1 Billion** propane cylinders used for home barbeques

Via https://www.epa.gov/energy/greenhouse-gas-equivalencies-calculator
Assuming about 0.5 MT per person, per year, based on this analysis.

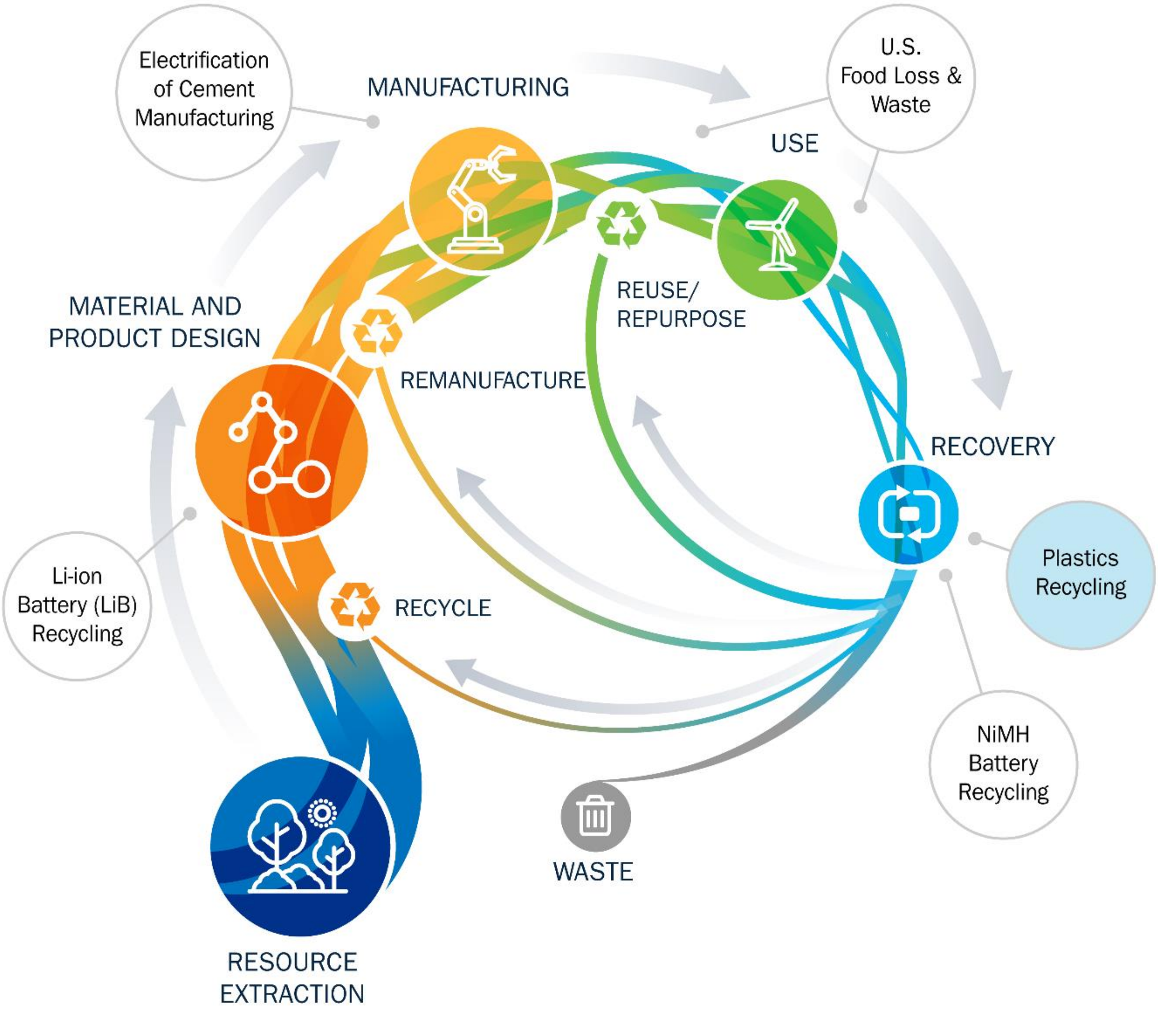
In general, advanced packaging technologies are a promising pathway for reducing FSC FLW generation, energy consumption, and GHG emissions. As discussed in Sections 5.2.2 and 5.2.3, active packaging technologies have received significant attention from researchers. However, instead of using shelf-life extension as a metric, most studies evaluate the active packaging materials according to product-specific metrics (e.g., product color, freshness, moisture level after several days). These metrics do not support manufacturers that want to extend their product's shelf-life by a specific amount; it is almost impossible to decide which, and how much, active packaging material to use. Therefore, more active materials should be explored with special consideration for the impact on shelf life, product safety, cost, and environmental impact.

Combining active packaging with smart packaging devices is another promising research direction. With smart packaging devices (i.e., sensors), the active packaging materials only need to emit or absorb compounds when necessary, maximizing the effect of active packaging materials and reducing the consumption of the active material. Active packaging material can also be mixed with novel packaging material (e.g., nanocompounds) to improve the strength of antimicrobial food packaging. Finally, combining active packaging material with biodegradable packaging should be explored to reduce the environmental impact of food packaging.

This study considers the impact of packaging solutions implemented only at the food manufacturing stage; however, many waste reducing technologies apply to other stages. In addition, this study only considers FLW, food demand, energy consumption, and GHG emissions as metrics of the impact of technologies on the U.S. FSC. However, more metrics should be included for a more exhaustive analysis. For example, 52% of U.S. land area is used for agricultural purposes (USDA 2019); land use change is a major driver of global environmental change and an important measurement of FSC sustainability (Lambin and Meyfroidt 2011), which this study does not address. Furthermore, previous studies show a significant dietary change in the Unites States (Liebman 2016), which could be another path toward a more sustainable FSC (Alexandratos and Bruinsma 2012). Future work could investigate the effects of diet change on food demand and FLW, and environmental impact metrics beyond energy use and GHG emissions (such as land use change) could be included. This work would assess how dietary changes could improve FSC sustainability while still meeting the nutrition demand.

Additionally, although most of the FLW generated is recycled through several EOL management pathways, according to the EPA Food Recovery Hierarchy, recycling is inefficient compared to FLW reduction. Therefore, demand reduction should be the goal. However, reducing the FLW means producing more raw material for the industries that use the FLW (e.g., fertilizer, animal feed) and may add more pressure to natural resources. Therefore, trade-offs between FLW reduction and different EOL management pathways need to be further explored. To understand this better, researchers must first analyze the components of FLW from each stage and, second, examine the suitability of each type of FLW for particular recycling methods. In addition, the analysis should compare the full life cycle impacts of each FLW recycling method.

6 Case Study: Recycling Plastics—Projecting Costs, Energy Use, and Emissions



6.1 Introduction

Of the 32 MMT of plastic waste present in U.S. MSW in 2017, an estimated 74% was sent to landfill, with the balance being either combusted for energy recovery (16%) or recycled (8%) (U.S. EPA 2019). In this work, we employ a wedge analysis framework to understand the potential for various technologies to handle the growing landfilled plastic waste stream. We estimate material, energy, and GHG impacts from several separation, recycling, and upcycling technologies[33], including triboelectrostatic separation (Silveira, et al. 2018), (Park, Jeon and Cho, et al. 2007), (Park, Jeon and Yu, et al. 2008); polyvinyl chloride (PVC) solvent extraction (VinylLoop 2021); and polyethylene terephthalate (PET) upcycling to composites (Rorrer, et al. 2019). We employ nonlinear technology adoption trajectories (Hanes, et al. 2019) unique to each of the referenced technologies. These plastics are low-density polyethylene (LDPE),

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[33] These technologies were selected based on availability of data and to try and represent a diverse set of technologies for different plastic types as well as different parts of the recycling system.

linear low-density polyethylene (LLDPE), high-density polyethylene (HDPE), PET, polypropylene (PP), and PVC. Figure 29 shows this breakdown of the MSW stream. The remaining 21% of plastic resins, comprising polystyrene (PS) and hundreds of smaller-volume polymers ("Other"), are not covered by this case study.

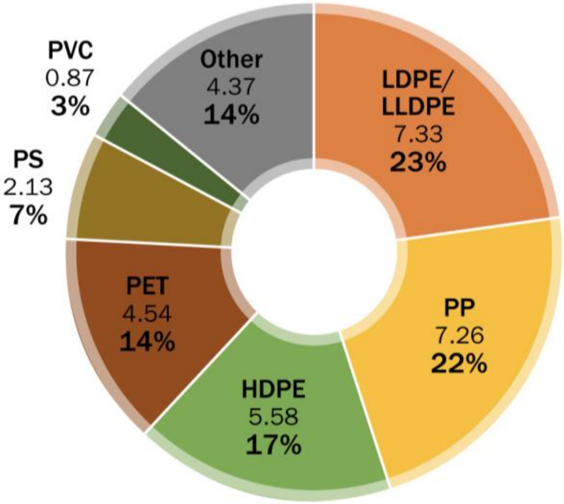

Figure 29. Composition of plastics in the U.S. MSW stream as of 2017

Note: (in thousands MT and as a percentage)

6.2 Methodology

6.2.1 Circular Economy Technologies Examined in this Case Study

Our preliminary literature search found that many technologies are being developed to address the high levels of plastic sent to U.S. landfills. Plastic waste handling research appears concentrated in two broad categories: separation and chemical recycling/upcycling. The plastics technologies included in this case study cover these main categories. The selected technologies are described in Table 6 and the following sections.

Table 6. Characteristics of Selected Plastics Waste Management and Treatment Technologies

	Triboelectrostatic Separation	PVC Solvent Separation	PET Upcycling
Which plastics does this technology apply to?	HDPE, LDPE/LLDPE, PP, PET	PVC	PET
What is the quality of the end product?	Same flake quality, but fewer impurities and higher yield	Same quality as original virgin PVC	Higher-quality composite product
What is the technology readiness level (TRL)?	Lab-scale	Commercialized in some countries (Italy, Japan)	Lab-scale
What category of waste plastic handling does this technology represent?	Improved separation technologies for plastics that are already appreciably recycled	Improved separation technologies for plastics that are already appreciably recycled	Technologies for repurposing plastic into higher-value materials.

6.2.2 Triboelectrostatic Separation

When a plastic waste stream containing many types of plastic is processed at a materials recovery facility (MRF), it undergoes several types of sorting, including flotation (Dodbiba and Fujita 2004). Existing sorting techniques can separate a collected plastic waste stream into the major groups of polyolefins (HDPE, LDPE/LLDPE, PP) and a stream of PET with trace PVC (contaminant). However, separation *within* these groups remains a challenge, as the densities of the plastic are similar. Approaches such as robotic vision sorting have been proposed (Peršak, et al. 2020). Another approach, known as triboelectrostatic separation, uses an electric charge to separate plastic flake streams of similar densities or differing colors. This method would allow for automated separation of the aggregate polyethylene stream into its specific types, as well as removal of PVC impurities from the PET stream. The latter of these applications would greatly enhance the quality of the recovered PET flake. Even trace amounts of PVC will lower the quality of recycled PET because of its chlorine content, which degrades the polymer upon heating. Triboelectrostatic separation has already found a commercial market in the mining sector, separating mineral ores in beneficiation processes (Bittner, et al. 2014). Figure 30 illustrates triboelectrostatic sorting equipment.

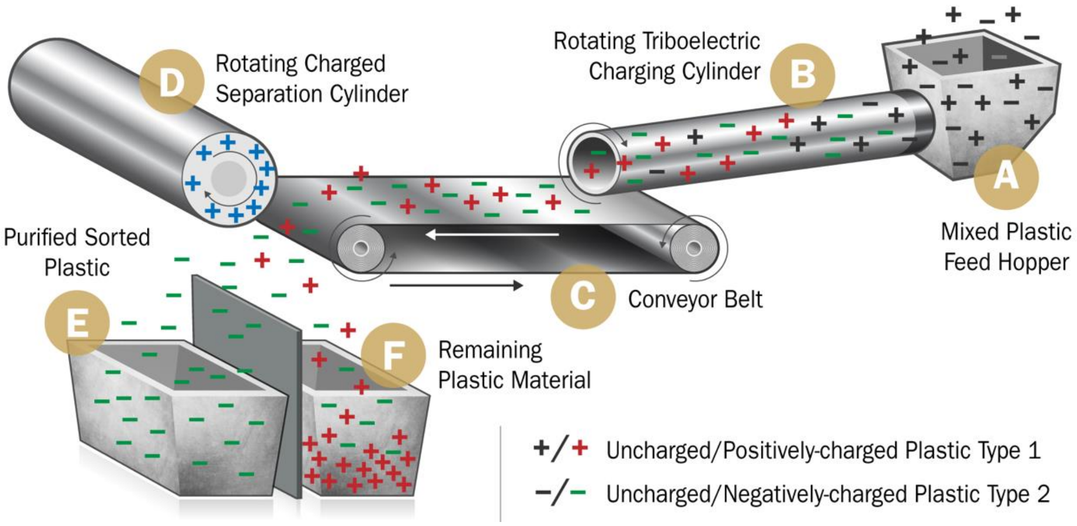

Figure 30. Triboelectrostatic separation mechanism

6.2.3 PVC Solvent Separation

Another plastic separation process considered in this case study, known as PVC solvent separation, applies specifically to waste PVC plastic, a notoriously difficult plastic to handle in the recycling process. The solvent process extracts and precipitates PVC from the ground-up plastic waste stream, filters out any residual impurities from the solution, and then precipitates the PVC back out as freshly recovered polymer (see Figure 31). This technology operated commercially in Italy from 2002 to 2018, and also in Japan (Inovyn 2018), (Sinclair 2002). The Italian operation shuttered because of an unacceptably high level (under European Union Registration, Evaluation, Authorization, and Restriction of Chemicals [REACH] regulations) of phthalate compounds in the recovered PVC, which were not economically feasible to remove.

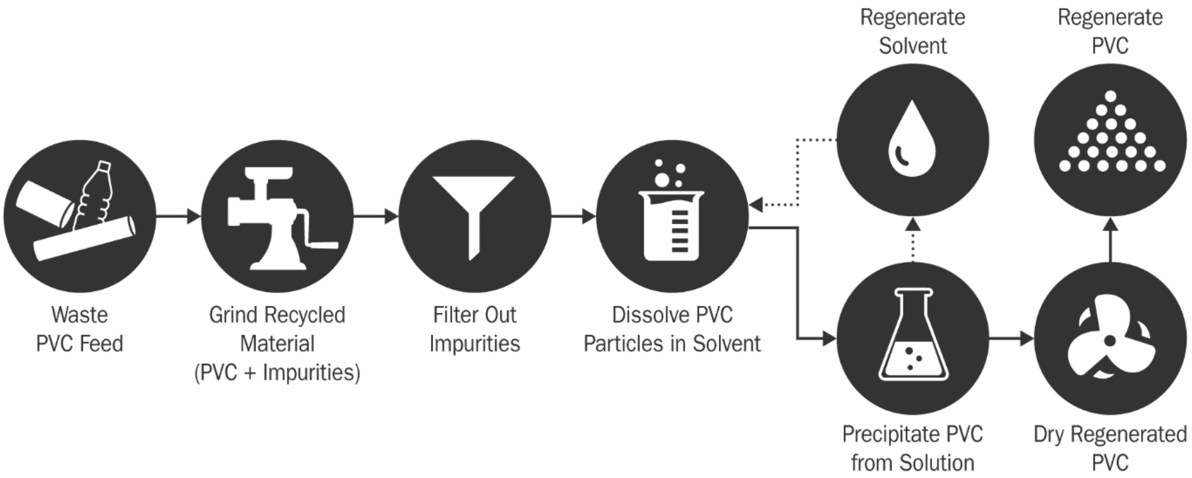

Figure 31. PVC solvent separation flow diagram

Note: Grinder and fan icons were created by Firza Alamsyah and Adrien Coquet, respectively. Used under the Creative Commons license (Noun Project).

6.2.4 PET Chemical Upcycling

The first two technologies covered in this case study are most closely characterized as physical or mechanical recycling methods. A third technology involves chemical recycling, where the plastic polymer molecules are themselves broken down into their constituent monomers, which are then repolymerized. For this third technology, we consider a process known as PET deconstruction via glycolization. One application of this technology was proposed by NREL researchers in 2019 (Rorrer, et al. 2019) and was specifically designed to handle PET waste plastic.

Another critical difference between this technology and the first two is that the resulting product from PET chemical upcycling is not functionally equivalent to the input waste material.[34] When the PET is deconstructed and then repolymerized with other comonomers derived from bio-based sources, it forms an unsaturated polyester that can be used to create a composite material known as fiber-reinforced polymer (FRP). This material is used in high-performance applications such as wind turbine blades and snowboards, making it a higher-value product than the original PET on a per-mass basis.

6.2.5 Technology Adoption

Given that these technologies are still in the demonstration phase and not yet commercialized (as of 2020), we need to make some assumptions about their future adoption to estimate their potential to divert plastic waste streams from landfill. In lieu of an overly simplistic linear increase in adoption of the technology, we opt for a more realistic S-shaped adoption paradigm based on an extension of the Bass diffusion parameter methods (Hanes, et al. 2019), (Bass 1969). Of course, any adoption scenario makes an implicit assumption that the technology is, in fact, commercialized, and this analysis is no exception. Assessing the broader adoption of these technologies is beyond the scope of this “what if” analysis, which takes a nonzero level of adoption essentially as a given to proceed.

In this case study, “adoption” of the technology is assumed to mean the implementation at a commercial scale by the relevant industrial firm(s). For example, we assume MRFs and associated firms would adopt

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[34] If not combined with a separate monomer, the end product is chemically identical to PET.

the triboelectrostatic separation of PVC solvent separation technologies (i.e., added as an additional waste processing option), whereas a composites manufacturer might adopt the PET upcycling technology. These assumptions dictate the upper bound for market penetration of such adoption. For MRF-oriented technologies, we assume adoption is limited by the amount of technology-relevant polymer that is collected and can be processed. For the composite manufacturing-oriented technology, we assume adoption is limited exogenously by the market size for glass fiber composites (i.e., further adoption of the PET upcycling technology does not, itself, increase the market for composites).

We estimated Bass diffusion technology adoption parameters for each plastic waste technology detailed above. The most important of these parameters are the coefficient of innovation (p) and the coefficient of imitation (q). In this work, p is estimated from technology characteristics. Following the methodology described in Hanes et al. (2019), the associated q parameter is estimated from a correlation derived from the historical relationship (from actual technologies adopted over time) between the q/p ratio and p. Individual adoption trajectory paths for each technology were calculated out to Year 2050, the horizon of this analysis.

6.3 Wedge Analysis Results

Early in our analysis of the overall impacts from the technologies considered, we found that the assumed plastic waste collection rate significantly impacted the results. Therefore, two scenarios were devised, corresponding to two assumptions about the rate of plastic waste collection. The first scenario, Baseline, assumes current collection rates of the polymers of interest (HDPE, LDPE, LLDPE, PET, and PVC) continue unchanged for the entirety of the forecast period, which is 2020–2050. A second scenario, Aggressive, assumes a much more ambitious plastic waste collection scheme unfolds by 2050: an 80% collection rate for PET and a uniform 50% collection rate across all non-PET plastics.[35] In both scenarios, plastic waste diversion from landfill is the primary metric calculated by the model; these results are then used to derive estimated energy and emissions wedge impacts. The following sections discuss these scenarios in detail.

6.3.1 Baseline Recycling Scenario

The Baseline scenario assumes the current plastic collection rates continue unchanged over the modeled period. Table 7 lists these assumed waste collection rates for each plastic along with the assumed growth rates in plastic waste generation. The assumed growth rates in plastic waste generation are equivalent to annual consumption increases in the United States for the period of 2020–2030, as estimated by the IHS (2020) *Chemical Economics Handbook*.

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[35] This Aggressive scenario assumes that the rate of collection for each plastic linearly increases from 2020 to 2050.

Table 7. Assumed Plastic Waste Collection Rates and Waste Generation Growth Rates in Baseline Scenario

Polymer	Baseline Scenario Recycling Rate	Waste Generation Growth Rate (Annual)
HDPE	30.4%	2.8%
PET	28.9%	0.7%
PP	17.0%	1.7%
LLPE/LLDPE	1.2%	1.2%
PVC	1.7%	2.0%

Figure 32 shows the estimated plastic waste diversion from landfill for the Baseline scenario. PET Chemical Recycling, HDPE Separation, and PP Separation have the largest estimated wedge contributions to the overall reduction in landfilled waste. Overall, the estimated combined diversion from these technologies reaches 3 MMT per year by 2050. This diversion is equivalent to an estimated 90% increase in aggregate mass of plastic waste diversion away from U.S. landfills and toward material recovery operations[36] compared to current recycled streams.

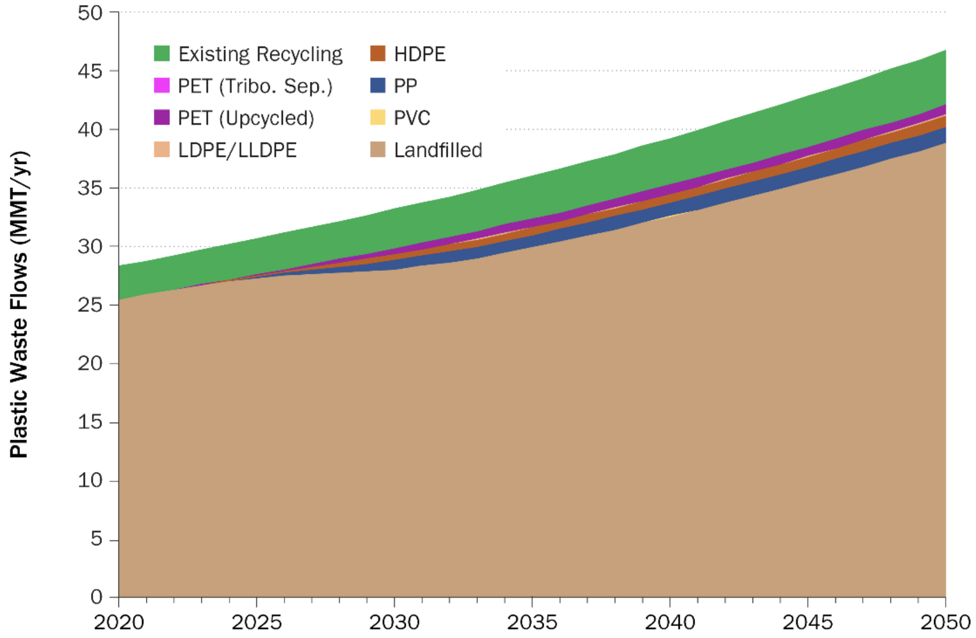

Figure 32. Wedge analysis results for estimated MSW plastic landfill diversion in the Baseline scenario, 2020–2050

6.3.2 Aggressive Recycling Scenario

To examine the potential impact of a more concerted effort to increase collection rates across all plastics, we consider a scenario in which PET collection rates reach 80% by 2050 and the other plastics considered reach 50% collection by 2050. For PET, this collection level falls between the current collection rates of PET bottles in Europe (64% as of 2019) (McGeough 2020) and Japan (92% as of 2017). For remaining

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[36] Note that energy recovery from combustion of plastic waste is not considered here.

plastics, 50% collection is comparable to Japan's 46.3% collection rate for plastic containers and packaging (Keidanren 2018). Given the representative technologies presented in this case study, potential impacts to the diversion of landfilled plastic appear substantial in this case. As shown in Figure 33, recycled plastic increases substantially, reaching 18 MMT per year by 2050, or an estimated 650% increase in the total mass of recycling diversion of waste plastic compared to 2020. The highest contributions come from increased PP and LDPE recycling, as these high-volume plastics are currently collected at very low rates.

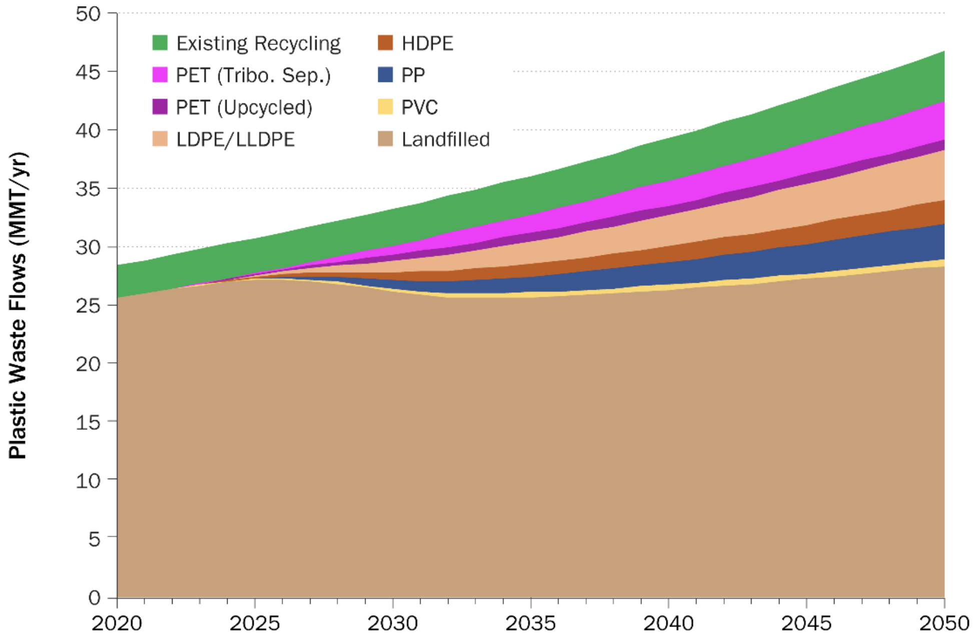


Figure 33. Plastic circular economy technology adoption wedge analysis: Estimated MSW plastic landfill diversion.

Note: Projections are for 2020–2050. The Aggressive scenario is presented with an 80% PET/50% (All Other) Recycle Rate by 2050.

Note that even with the very optimistic high collection rates, the amount of plastic sent to landfill increases from about 2035 onward because of the growth in overall demand for plastics. Our analysis also accounts for the assumed recovery yields (< 100%) from the technologies on top of the assumed adoption of that technology. These recovery yields range from 90.3% for triboelectrostatic separation to 95.9% for the PVC solvent separation process. The combined effect of multiplying these percentage factors together works to depress the overall recovery amount of each plastic. A key consideration in the analysis is whether and how these experimental yields for not-yet-commercialized technologies would hold up under a scale-up to industrial processing volumes.

6.3.3 Energy and GHG Wedges and Quad Chart Analysis

As an extension of the mass-based waste stream wedge analysis detailed above, we also determine accompanying energy and GHG emission wedge impacts for these plastic waste technologies. The data for evaluating these impacts are obtained either from literature (in the case of PET upcycling and PVC solvent separation) or estimated based on the available process specification (triboelectrostatic separation; see Appendix C).

We find that waste reduction and energy requirement reductions align for the PET upcycling and solvent separation technologies considered in this case study analysis. In economic terms, the marginal benefit of each additional quantity of waste diverted by these methods includes a reduction in energy use and GHG emissions. Figure 34 illustrates these harmonious impact directions in the quad chart depiction of the results presented in Table 8.

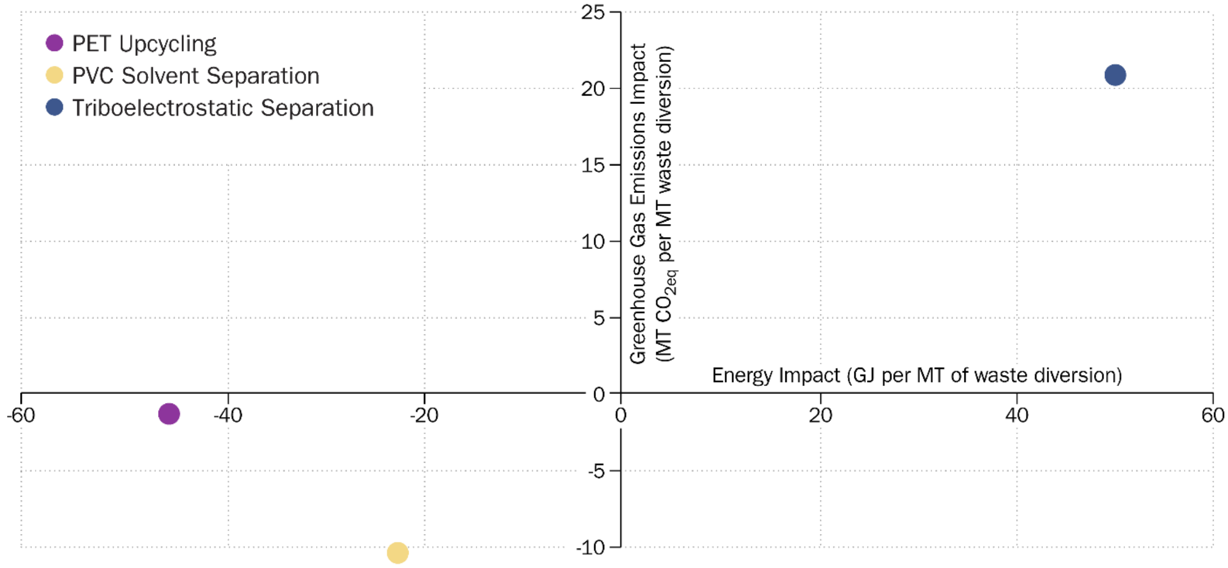


Figure 34. Quad chart analysis of energy and GHG emissions impacts for the technologies considered in this analysis.

As shown in Figure 34, in contrast to the PET upcycling and PVC solvent separation technologies, the triboelectrostatic separation is estimated, at least in its current lab-scale form, to have an inverse relationship between waste reduction and energy and GHG reductions. For each additional metric ton of plastic separated using the triboelectrostatic method, we estimate an additional 50 GJ of energy requirement relative to manufacturing the recovered plastic from conventional fossil-based methods. This is a rather high energy requirement, and the GHG trade-off is even higher at approximately 21 kg CO_2 eq per kg (much higher than the GHG emissions associated with producing virgin polymers from fossil feedstock sources) (Nicholson, et al. 2021).

Table 8. Changes in Energy and GHG Emissions Impacts of Selected Plastic Waste Technologies

	PET Upcycling to GFRP		PVC Solvent Separation		Triboelectrostatic Separation
	Per MT Impact	Total Impact in 2050	Per MT Impact	Total Impact in 2050	Per MT Impact
Energy	-45 GJ/MT/yr	$-32*10^6$ GJ/yr	-22 GJ/MT/yr	$-8*10^6$ GJ/yr by	+ 50 GJ/MT consumption
GHG Emissions	-1.4 MT CO_2 eq/MT	-11 MMT CO_2 eq/yr	-11.3 MT CO_2 eq/MT	-10.4 MMT CO_2 eq /yr	+ 21 MT CO_2 eq/MT GHG emissions
Relative to…	Conventional GFRP production		Conventional PVC production		Conventional production of various polymers (PE, PP, PET)

However, this analysis of triboelectrostatic separation impacts is a preliminary estimate using laboratory-scale parameters from the literature and does not account for assumed improvements in energy efficiency resulting from improved economies of scale at a hypothetical commercial operation.

6.4 Caveats and Assumptions

The assumptions that most dramatically impact these results concern the level of plastic waste collection from the MSW stream. The strength of these assumptions rests not necessarily in their feasibility (as they have been observed, or at least targeted, in other regions), but more firmly in their assumed independence from technology adoption rates. Endogenous effects on these plastic collection rates due to the adoption of plastic waste separation or repurposing technologies would require more rigorous partial equilibrium economic modeling, which is not considered in this study. However, such effects could be significant, especially if adoption of new technologies leads to the establishment of more robust markets for certain polymers (e.g., PVC). This is a potential area of future research.

Adoption scenarios in this case study assume the technologies reach a commercialized stage. Currently, only the PVC solvent separation technology has been commercialized; the other two technologies have only been demonstrated at lab-scale.

Plastic waste generation rates may vary considerably from the trajectories assumed for this analysis (obtained from IHS (2020) forecasts of U.S. plastic consumption growth trends).

The recovery yields reflected in this analysis for each technology are taken from the original literature sources that describe the technology and are not assumed to increase over the course of the adoption period (2020–2050). This assumption is intentionally conservative and applies across all waste collection scenarios.

6.5 Summary and Further Research

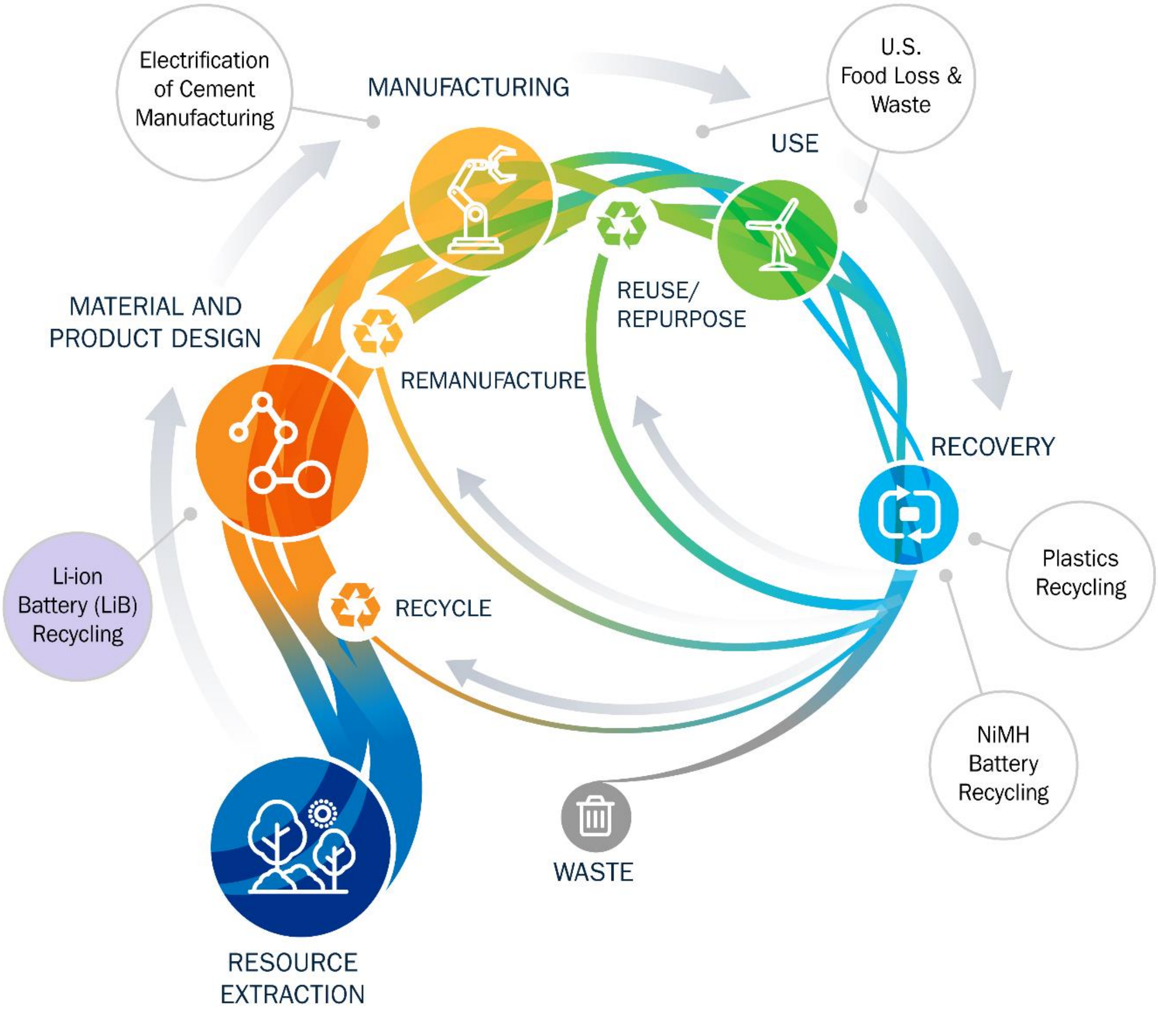
This case study analyzes the potential for novel plastic separation and upcycling technologies to reduce landfilled plastic waste streams, especially when combined with more ambitious plastic collection levels from consumers. The three technologies evaluated in this case study can be considered options for:

- Increasing recycling rates and yields for plastics with existing recycling infrastructure

- Establishing recycling systems for plastics not currently recycled in any significant capacity
- Deriving additional value from waste plastics through the production of upcycled materials.

This study applied estimates of material quantity diversions from the adoption of these separation and upcycling methods to estimate energy and GHG impacts relative to conventional (fossil-derived) polymer production. From these estimates, trade-offs between increased material recovery versus additional energy input and GHG emissions were analyzed. Ultimately, a better understanding of such trade-offs will inform discussions about the transition from traditional linear consumption models to circular economies of material use and reuse. While not discussed in this report, the quality of the recovered plastics is important and should be evaluated for any technology being considered.

7 Case Study: Recycling EV Lithium-Ion Batteries—Evaluating the Role of Innovation in Battery Economics and Circularity



7.1 Introduction

Numerous studies and market analyses project that the market for EV LiBs will continue its current upward trend for the foreseeable future (USDRIVE 2019), (McKerracher, et al. 2020). While consumer electronics make up a significant fraction of the current market, LiBs for EVs and large-scale energy storage are rapidly gaining market share. The rapid increase in demand coupled with supply chain bottlenecks has created a business opportunity for recycling EOL LiBs. Recovered high-value materials would have a ready market in battery manufacturers eager to reduce costs and supply chain uncertainty.

However, the rapid evolution of the LiB market also poses threats to recyclers. Building new recycling facilities or repurposing existing facilities requires recyclers to make significant investments in anticipation of a supply of raw material, in the form of EOL LiBs, that does not yet exist at the scale likely needed for a profitable operation. In addition, the accelerated turnover in LiB technology and changes in chemistry might reduce the demand for raw materials that are currently in short supply. This would reduce the pressure on manufacturers to find alternative supplies and accept recycled material. Reduced demand might also reduce prices, potentially making the recycling operations unprofitable. In the longer term, as advanced batteries reach their EOL, reduced concentrations of target materials in the battery waste stream might make recycling more expensive, and recovering materials may be unprofitable even if prices remain relatively high. Analysis is needed to identify and develop strategies for overcoming the barriers that EV battery recyclers are likely to face.

This case study elucidates where innovations in EV battery design (cells, modules, and packs), reverse supply chain, and recycling processes yield the greatest impacts on virgin material use and the value of recycled materials.

7.2 Methodology

This analysis uses a Microsoft Excel material flow model to assess the impacts of the three scenarios evaluated in this study. The model is based on inputs to the NREL LIBRA systems dynamic model (Weigl, et al. 2022), currently in development, and a simplified model of recycling based on Argonne National Laboratory's (ANL) BatPaC and EverBatt models.[37,38] These models rely on multiple assumptions and input values (e.g., EV adoption rates) that vary widely in the literature. Nonetheless, through a critical review of the literature, close collaboration with ongoing research projects at NREL and elsewhere (e.g., research on reduction of Co content), and sensitivity modeling, this case study yields valuable insights into the role of innovation in the circularity of LiBs in particular and high-technology ("energy-relevant") products in general.

This analysis explores the implications of the recycling economics of the following five innovation areas:

- Material reduction and substitution for EV LiBs to minimize the use of critical materials (specifically Co)[39]
- Battery in-use life extension and second use in a vehicle battery aftermarket
- Reverse supply chain innovations (e.g., removing electrolyte before transport, policies for collecting and recovering materials)
- Innovations in recycling processes (e.g., robotic disassembly, direct recycling of cathode material)
- Design for recycling (e.g., removing organic binders and replacing the glued assembly of battery modules with removable fasteners).

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[37] EverBatt is an Excel-based model that evaluates cost and environmental impacts for the various life cycle stages of a lithium-ion battery.

[38] BatPaC – A Spreadsheet Tool to Design a Lithium Ion Battery and Estimate Its Production Cost.

[39] Minimizing the use of Co helps address supply chain vulnerability and environmental, safety, and human rights concerns, as discussed in Section 1.3.

7.3 Scenarios and Results

Innovations in battery design or recycling processes may have little value if the EOL strategy does not take advantage of the new design. For example, designing batteries for ease of disassembly would have little value if, at their EOL, the batteries are simply ground up and fed to a pyrometallurgy recycling process. Therefore, the five innovations listed above have been combined into three whole-life scenarios that fully exploit the benefits of each innovation. The schematic in Figure 35 represents the three scenarios: Design for Disassembly, Design for Recycling, and Material Substitution.

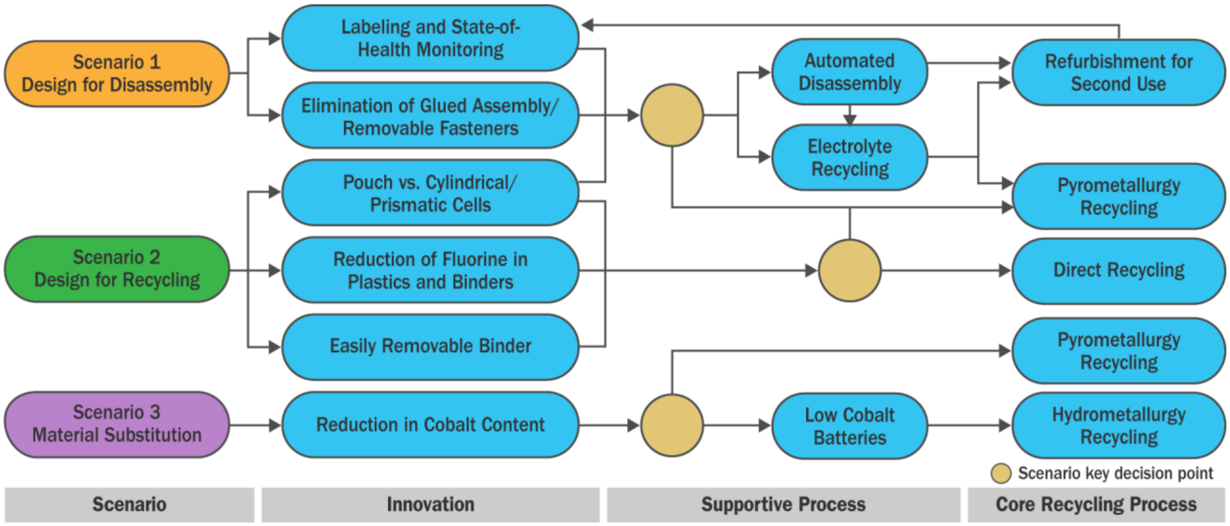

Figure 35. Three whole-life battery scenarios for evaluating the impact of innovation on LiB circularity

Scenario 1, Design for Disassembly, implements a key technology: labeling and state-of-health monitoring over the battery's lifetime that facilitates rapid screening at EOL. Two other changes in the design of cells could facilitate more extensive refurbishment of EOL batteries. Eliminating permanent joining (i.e., gluing) in assembly of modules would facilitate disassembly to remove individual cells without damaging them. Designing cells that support replacing electrolyte could provide a mechanism for rehabilitating a fraction of individual failed cells or restoring overall module performance in some cases. The key *decision point* is sorting EOL batteries into a stream that culminates in either selling refurbished batteries on a secondary market or routing batteries to pyrometallurgical recycling.

Scenario 2, Design for Recycling, evaluates direct recycling, in which the cathode material is recovered intact from EOL batteries. There are several potential enabling manufacturing technologies for direct recycling, such as developing an easily removable binder. However, the primary *decision point* is adopting the technology itself, which is not yet commercially available.

Scenario 3, Material Substitution, examines the implications of the ongoing trend toward lower-Co batteries. The *decision point* is whether to recover materials in addition to Co and Ni as the valuable Co becomes less concentrated in battery waste. Co and Ni are economically recoverable using pyrometallurgy, while other materials require a somewhat more complex hydrometallurgical process. Will recovery of other materials be a sufficient economic driver to justify the process?

7.3.1 Baseline Scenario

The baseline for the analysis incorporates industry and government best estimates of the demand-side evolution in LiB battery deployment. The baseline demand analysis includes the impacts of EV adoption rates and increases in battery capacity (kWh) on energy and carbon. It assumes the U.S. adoption of EVs and the evolution of battery capacity (kWh) over that period is the same for all three scenarios. Battery nominal lifespan is assumed to be 8 years for new vehicles sold between 2010 and 2015, transitioning from 8 to 10 years between 2015 and 2020 and 10 years thereafter. Figure 36 plots vehicle sales and retirements. Appendix D, Table D.1 provides a brief description of the data and references used to develop the baseline scenario.

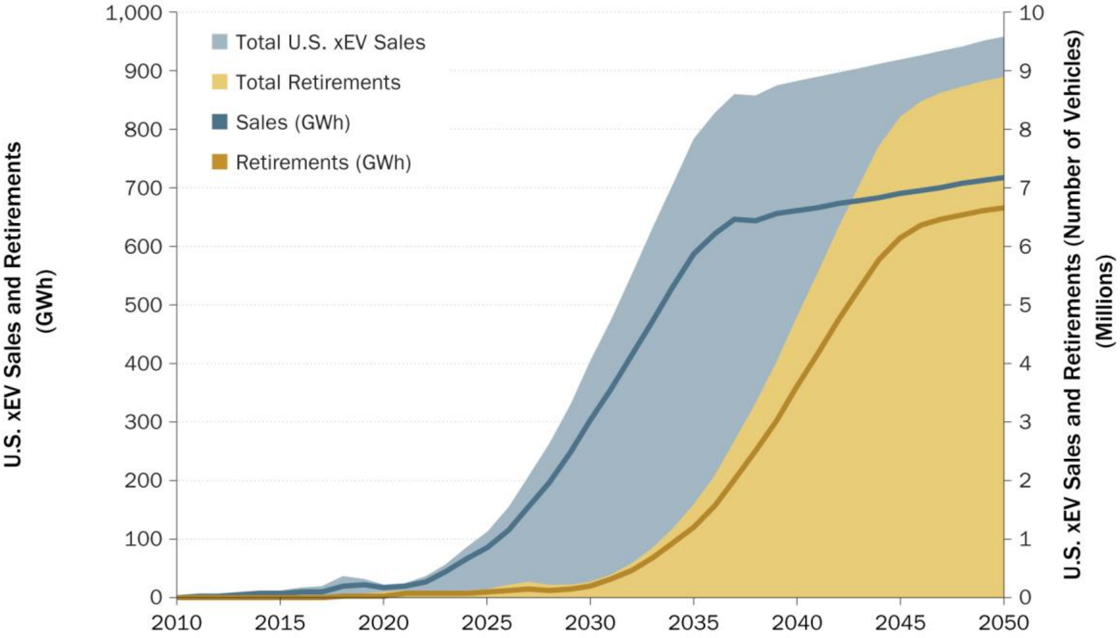

Figure 36. EV sales and retirements for all three scenarios

Source: McKerracher et al. (2020) and NREL analysis

The baseline assumes the current 2020 mix of battery cathode chemistries for EV passenger vehicle traction batteries is maintained until 2050 (McKerracher, et al. 2020). Throughout this analysis, nickel, manganese, cobalt chemistries are indicated by the first letter of the primary cathode elements Ni, Mn, Co is written as [NMC] followed by the molar ratios, respectively, of the elements (e.g., for 60% Ni, 20% Mn, and 20% Co, the nomenclature is [NMC622]). Currently, NMC622 represents nearly 60% of the current market. Tesla's Ni, Co, Al (NCA+) and NMC811 chemistries make up the bulk of the remainder.[40] The baseline of vehicle battery material demand growth and static battery chemistry provides a consistent basis of comparison for the three innovation scenarios. Scenario 3 (Material Substitution) examines the impact of changes in battery chemistry on the material flows and economics of recycling. Battery chemistry is held constant for the other two scenarios so that the impacts of the

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[40] https://www.tesla.com

design innovations in Scenarios 1 & 2 can be analyzed individually without the confounding influence of changes in battery chemistry, which is analyzed separately in Scenario 3.

Figure 37 plots the total EV demand for the predominant cathode metals of interest (Ni, Co, and Li) in the baseline. Appendix D, Table D.2 lists the chemistries and element ratios for the batteries included in this study. All three scenarios assume a 95% collection rate for EV batteries—high, but reasonable given their size and the value of battery materials.

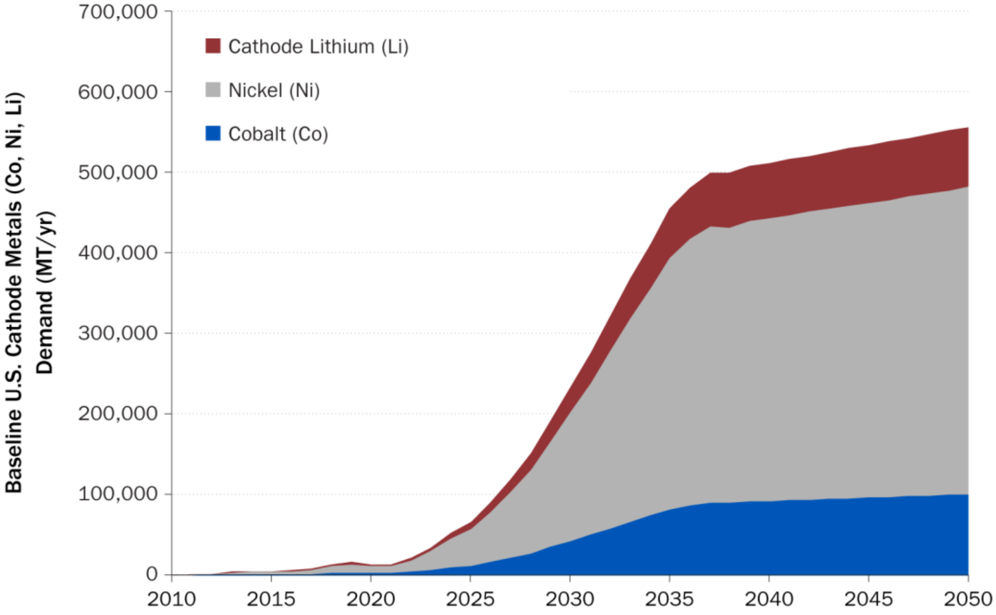

Figure 37. Baseline U.S. EV demand in metric tons/year for cathode metals for all three scenarios

Source: McKerracher et al. (2020), ANL BatPac4.0 19FEB2020,[41] and NREL analysis

7.3.2 Scenario 1: Design for Disassembly

In this scenario (Figure 38), LiBs are designed for ease of disassembly at the end of the initial use phase to enable life extension by refurbishment for reuse in vehicles,[42] or second-use by reengineering to repurpose in stationary energy storage applications (Foster, et al. 2014). For this case study, all refurbished batteries will be used for vehicles to maintain the focus of the study on U.S. vehicle batteries. Various strategies could be employed for refurbishing batteries. In the simplest strategy, batteries are dismantled to the module level, each module is tested, and modules that meet the end-use performance criteria are reconfigured into new battery packs.

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[41] ANL BatPac4.0 19FEB2020 Model.

[42] In Scenario 1, it is assumed that batteries will be disassembled to the module level, and modules that meet a certain standard of performance based on onboard diagnostics will be repacked into new batteries for sale back into the vehicle market. This strategy is somewhere between refurbishment and remanufacturing and will be referred to as refurbishment (World Business Council for Sustainable Development 2018).

Modules that do not meet the performance criteria for second use are routed to a recycling process for metals recovery.

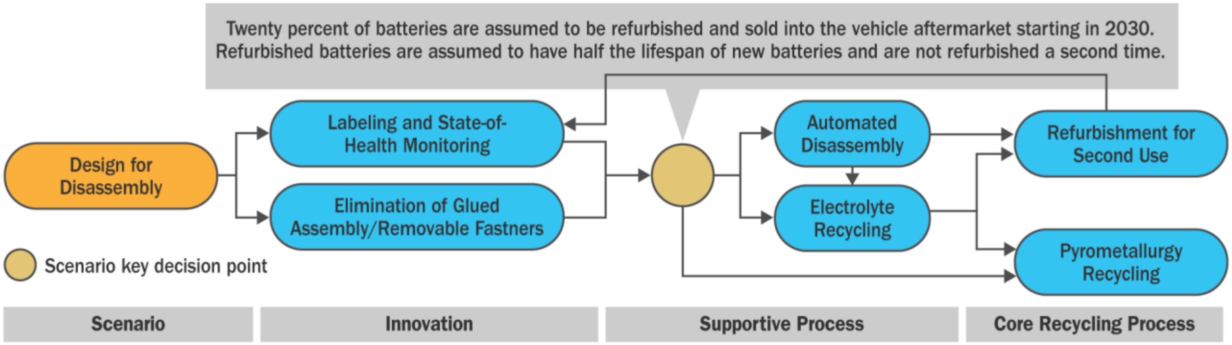

Figure 38. Scenario 1: Design for Disassembly

At the end of the final use phase for batteries, the batteries are routed to pyrometallurgical recycling, which recovers the Co and Ni to manufacture new batteries. Estimates of the initial lifespan of EV LiBs vary from 8 to 10 years based on distribution modeling (Richa, et al. 2014), which is consistent with current new vehicle warranties,[43] to as long as 15–20 years (Turcheniuk, et al. 2018). An EV LiB reaches the end of its initial use phase at around 70–80% of its original capacity, when it is no longer considered appropriate for traction purposes (Casals, Garcia and Canal 2019). However, the lifespan depends on factors such as charging and driving patterns and climatic conditions (Richa, et al. 2014), (Elwert, et al. 2018) (Neubauer, Brooker and Wood 2012) (Smith, Shi and Santhanagopalan 2015). A fraction of batteries would be removed from vehicles that are scrapped for reasons other than deterioration of the battery, and many of these batteries could be refurbished for continued use in vehicles. There is also a substantial market for used EVs. Used vehicles made up 35% of EV sales in 2017, according to Chargepoint (2017).

A key potential benefit of this scenario is the ability to perform some of the dismantling, testing, and recycling steps at or near the location where the batteries are first collected at EOL. The Canadian firm Li-Cycle has employed a "spoke and hub" model for its LiB recycling facility in Rochester, New York (Sylvia 2020). Spent batteries are collected at the spoke facilities, where a submerged shredding process produces "black mass" (a mixture of Li, Co, Mn, and Ni), which is then shipped to the hub facility for hydrometallurgical recycling. Li-Cycle recently announced the opening of their second spoke facility at Eastman Business Park in Rochester, New York (Green Car Congress 2020). The new facility can process 5,000 MT of EOL batteries per year, bringing its North American total capacity to 10,000 MT/year. Eventually, the hub facility is expected to annually process material from the equivalent of 60,000 MT of batteries, or about 120,000 EV batteries. The dismantling and battery testing facility for the Design for Disassembly scenario could use a similar hub and spoke strategy.

Vehicles and their batteries may reach EOL at different times. One of the primary enabling requirements for life extension or second-use LiB applications is the ability to monitor, preferably while still in service, the battery's state of health, or the extent to which it still meets specifications (Neubauer, et al. 2015). Advanced in-service diagnostics would serve as a prescreen for EOL batteries. Batteries (modules) for

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[43] Chevrolet Bolt EV 2020, 8 years/100,000 miles; Hyundai Kona EV 2020, 10 years/100,000 miles; Tesla 2020 models, 8 years/100,000–150,000 miles (varies by model) and 70% battery capacity retention

which diagnostics indicated that the module would fail testing could be immediately routed to recycling. The remaining modules would have a much higher probability of passing, significantly reducing the cost of testing. Neubauer et al. (2015) modeled the effects of advanced diagnostics by assuming two different cell fault rates for modules: a very low cell fault rate of 1 in 100,000 (0.001%) for the advanced diagnostics case, and a cell fault rate of 1 in 1,000 (0.1%) for the case in which less information was available for the batteries.

The number of cells per module also plays a role. The fewer cells per module, the greater the investment of time and labor required to test a given total capacity (kWh) of EOL batteries. The advantage of a small number of cells per module is that more modules, and a greater total kWh of battery capacity, may meet the specified performance criteria.[44] The average module size for the batteries in this study is

3.4 kWh/module.[45] For these batteries, battery design with advanced in-service diagnostics and EOL screening would save approximately 24% compared to the case where advanced diagnostics were not available (Neubauer, et al. 2015).

For the Design for Disassembly scenario, a high-throughput testing system would be installed at the collection (spoke) facility to identify batteries suitable for reuse or repurposing and separate them from EOL batteries to be recycled. Removing the battery management system, wiring, and casing from the battery packs at the point of collection reduces the weight of the remaining battery module by about 13%.[46] Removing the module casing and wiring would achieve another 4% reduction. The pack and module components could either be reused in refurbished batteries or recycled. In our analysis, in 2040, over 175,000 MT of aluminum (Al) and copper (Cu) could be recovered from pack and module materials, representing nearly $315 million in value and about 34% of the EV battery manufacturing demand for Al and Cu in that year.[47] These amounts would nearly double by 2050. For modules routed to recycling (EOL modules that do not meet standards for reuse and end-of-second-life batteries), producing black mass at the collection facility would reduce the weight of material shipped to the central (hub) recycling facility as well as support a nonhazardous classification for the materials. For a facility of the size envisioned for the Li-Cycle process (60,000 MT/year battery packs), and assuming a 100-mile radius from the recycling facility to the collection facilities, shipping black mass as a nonhazardous material versus shipping intact battery modules as hazardous material would save over $30 million per year in shipping costs.[48]

Labeling key battery parts with machine-readable QR or radio-frequency identification (RFID) tags that refer to external data in an accessible format also aids the disassembly process (Harper, et al. 2019). Once EOL batteries are categorized for reuse or repurpose, new assembly methods facilitate remanufacturing and disassembly. Cells in a battery pack age at different rates (Guo, et al. 2021); (Han, et al. 2019); (Lipu, et al. 2018) Further, cells in series can only deliver as much current as the weakest cell, which can be replaced during remanufacturing (Groenewald, Marco and Grandjean 2018). For some cells, replacing the electrolyte could restore some functionality. Replenishing the electrolyte could be facilitated by designing LiB cells with filling ports and controlled vent streams (Figure 39, left). Modifying jelly-roll packaging

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[44] Assuming that failure of individual cells is the primary cause of module failure (Neubauer et al. 2015)
[45] ANL BatPac4.0 19FEB2020 Model.
[46] ANL BatPac4.0 19FEB2020 Model.
[47] Prices are calculated using the ANL GREET Model.
[48] Shipping costs for 100 miles using an HD truck: $0.69/kg for hazardous, $0.02/kg for non-hazardous. Calculated using ANL EverBatt.

also enables easier replenishing of the electrolyte. This strategy has not been validated at scale; redesigning LiB cells to facilitate replacing electrolyte might be costly. Therefore, the case study does not envision replacing electrolyte. However, it does include recovery of some of the electrolyte using liquid CO_2 solvent (Grützke, et al. 2015).

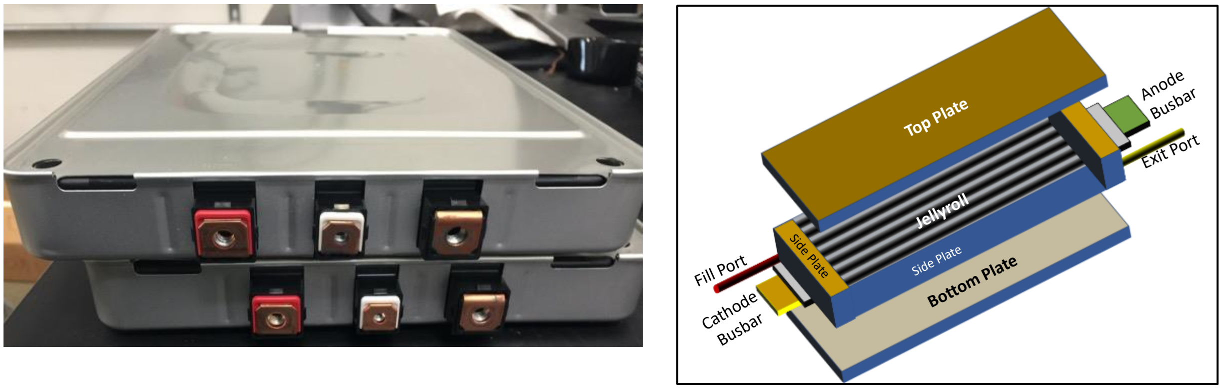

Figure 39. LiB cell with filling port (left) and LiB assembly with bolted terminals (right)

Note: Photo and graphic by Shriram Santhananagopal; used with permission.

An example of Design for Disassembly is using bolted, rather than welded, terminals, eliminating glued assembly, which would be critical for refurbishment at the cell level (Figure 39, right). However, as for designing for electrolyte replacement, it is unlikely that battery manufacturers would find these innovations economical. Gluing requires less labor and less expensive materials than assembly using removable fasteners. In addition, EV batteries are likely to be subjected to considerable vibration, which could loosen bolts or other removable fasteners.

Standardizing cell, module, and pack design also streamlines disassembly at EOL. The Design for Disassembly scenario assumes 20% of batteries are refurbished and sold into the vehicle aftermarket starting in 2030. The scenario envisions battery packs are disassembled to the module level, pack and module connectors are recycled, and the vehicle's advanced diagnostics data are used to screen modules for further testing. Tested and refurbished modules are reassembled and configured into batteries, which are assumed to have one-half the lifespan of new batteries and are not refurbished a second time (Hossain, et al. 2019).

Automated disassembly and electrolyte recovery using supercritical carbon dioxide (CO_2) (Grützke, et al. 2015) enable both reuse and recycling of LiBs. At EOL, mechanically separating battery components and materials before reuse or recycling separates materials into more homogeneous streams. Each stream can be routed to a recovery process that is tailored to that stream, which enhances the recoverability and purity of final recycled products. However, the value and extent of disassembly must be balanced against the labor costs and the potential for more efficient downstream separation processes to achieve the same ends (Wegener, Andrew, et al. 2014). While disassembly of the battery before pyrometallurgical recycling may be relatively minor (Ellis and Mirza 2017), more extensive disassembly to the cell level is necessary for hydrometallurgical and direct recycling (Scenario 2), as well as reuse and repurposing options (recycling processes discussed in Section 7.3.3).

Manual disassembly of an EV LiB requires time and specialized tools (Wegener, Chen, et al. 2015) and includes as many as 24 disassembly steps to obtain modules (Wegener, Andrew, et al. 2014). Disassembly time depends on the depth of disassembly; Schwarz et al. (2018) note that disassembly to the module level takes 25 minutes. Disassembly is also often the most expensive aspect of battery recycling because of labor costs and individual handling of each battery system (Schwarz, et al. 2018).

While automated disassembly could improve safety by assigning potentially dangerous tasks (e.g., discharging the battery and removing high-voltage wiring to a robot), many tasks would be challenging to automate. Damage and wear to the battery would be especially difficult to manage for automated systems. Partial automation with human-robot collaboration has been identified as a promising solution to LiB disassembly (Wegener, Andrew, et al. 2014), (Wegener, Chen, et al. 2015), (Cerdas, et al. 2018), (Gerbers, et al. 2018), where humans do complex tasks and troubleshooting.

7.3.2.1 Scenario 1 Barriers

The barriers to the Design for Disassembly scenario are primarily related to the rapid evolution of LiBs accompanied by steep reductions in the price of new batteries and potential purchasers' uncertainty surrounding refurbished battery performance and lifespan (BloombergNEF 2020). Continued price reductions in new LiB batteries make refurbished batteries less attractive economically. Based on the lowest battery repurposing cost found by Neubauer et al. (2015) of $20/kWh, it would cost $540 for refurbishment alone for the 27-kWh battery from a 2015 Kia Soul.[49]

Meanwhile, BloombergNEF (2020) recently reported EV battery prices below $100/kWh. Improvements in battery performance might have a similar dampening effect on the demand for refurbished batteries. For example, the Israeli company StoreDot[50] recently produced a battery on a standard assembly line that can be fully charged in 5 minutes on advanced charging equipment and can be charged for a 100-mile range on existing equipment in the same time period (Carrington 2021). In contrast, batteries that would reach their EOL in 2025 (a reasonable time frame for commercial availability of the StoreDot battery) would have been manufactured in 2015, with a range of approximately 100 miles, and would take 4–5 hours to fully charge at 220V and half an hour at 440V.[15]

7.3.2.2 Scenario 1 Results

In Scenario 1, EV LiBs are designed to facilitate EOL battery management, dismantling, and refurbishment. The refurbished batteries are assumed to be sold back into the vehicle market, thus effectively increasing the lifespan of the refurbished batteries in vehicles and reducing the overall demand for raw materials. This employs a somewhat different strategy from researchers who more typically assume that refurbished batteries would be used in less demanding stationary storage applications. However, Hossain et al. (2019) suggested that up to 35% of EOL EV batteries could be sold in the vehicle aftermarket after refurbishment. We chose to keep the batteries in the vehicle life cycle loop to simplify the analysis and better illustrate the impact of refurbishment on material flows. Likewise, we did not include consumer electronics in the baseline or scenario analyses because of the focus on innovation in vehicle batteries. Parameter values for Scenario 1 are summarized in Appendix D, Table D.5.

We performed a wedge chart analysis to illustrate the impact of the Design for Disassembly scenario on the demand for the four primary cathode metals: Co, Ni, Li, and Mn (Figure 40). In this scenario, EOL

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[49] 2015 Kia Soul EV specifications

[50] https://www.store-dot.com

EV batteries are screened for design features that facilitate disassembly and tested for their state of health. This scenario assumes that by 2030, 20% of all retired (EOL) EV batteries can be refurbished by testing, sorting, and repacking well-functioning modules into new battery packs. By 2040, about 11% of batteries sold are refurbished. The percentage rises to nearly 19% by 2050. The reduction in demand for virgin cathode metals from the sale of refurbished batteries is shown in blue in Figure 40.

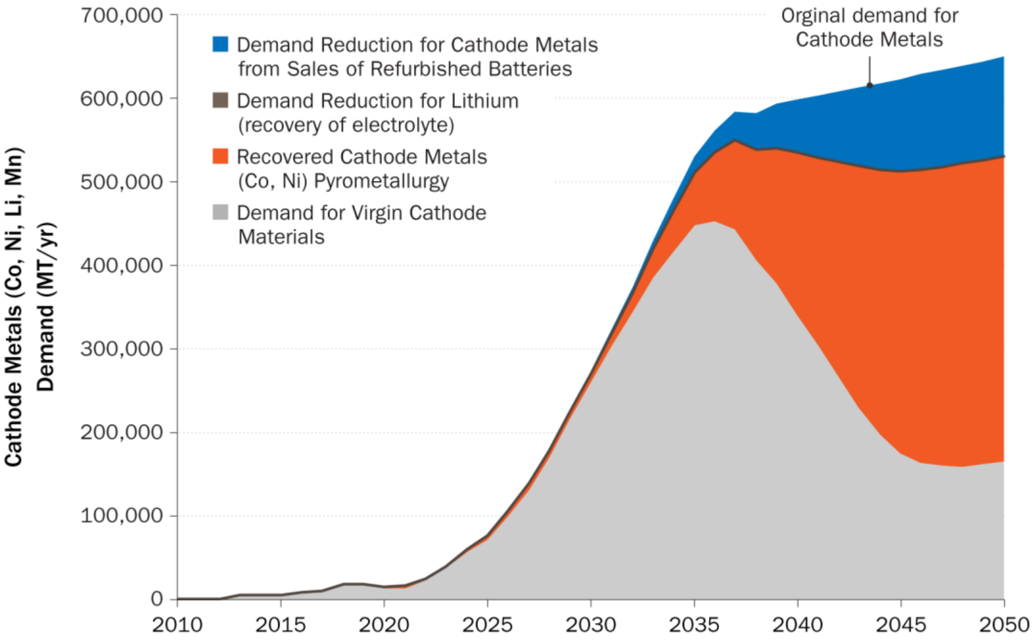

Figure 40. Wedge chart analysis of the reduction in cathode metals (Co, Ni, Li, Mn) demand of more than 70% by 2050 for the Design for Disassembly scenario

The following batteries are routed to a pyrometallurgical recycling process that recovers Co and Ni from the cathode:

- Batteries that are not designed to collect the appropriate state-of-health data throughout the battery's in-use life
- Batteries in too poor condition for refurbishment (i.e., on-board diagnostic data indicate the battery has reached its EOL or the battery fails testing)
- Refurbished batteries that have reached the end of their second life (assumed to be one-half the lifespan of a new battery).

The pyrometallurgical process is assumed to recover 98% of Co and Ni (see Appendix D, Table D.3). The recovery of metals in the recycling process, represented by the red wedge in Figure 40, results in a reduction in the need for virgin materials of more than 190,000 MT (20% Co and 80% Ni by weight) in 2040 and almost 365,000 MT in 2050. Together, the sales of refurbished batteries and pyrometallurgical recycling result in reducing demand for virgin materials of over 40% in 2040 and over 70% in 2050.

7.3.3 Scenario 2: Design for Recycling

In this scenario (Figure 41), LiBs are designed for recycling using a direct recycling process. LiB cathode material, graphite, electrolyte, metals, and possibly plastics are recovered in their original compound structure (Chen, et al. 2019), and cathode material is relithiated[51] without costly decomposition to elemental metals. Battery design features that facilitate direct recycling include:

- The use of pouch versus wound cells for easier access to electrolyte and cathode materials
- Easily removeable binder, such as magnetic binder (Liu, et al. 2015)
- Reduction of fluorine in plastics and binders, the removal of which requires use of toxic solvents or high heat.

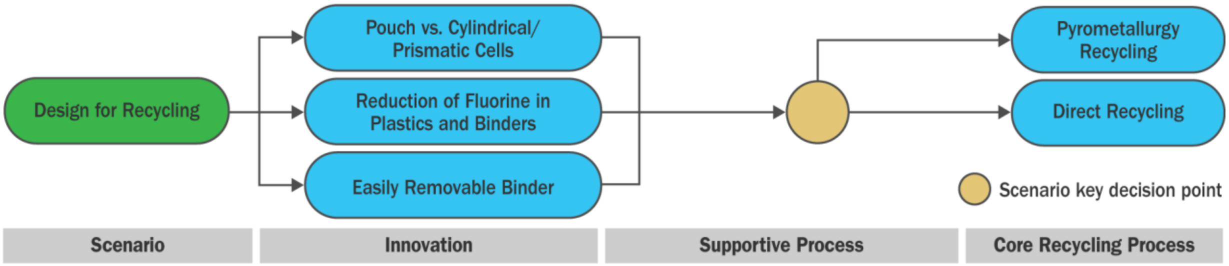

Figure 41. Scenario 2: Design for Recycling

LiB cell shapes are categorized as cylindrical, pouch, or prismatic. The housing material and electrode-separator compound (ESC) designs for each shape have advantages and disadvantages for manufacturing and disassembly (Li, et al. 2019).

In the pouch shape, a stamped aluminum film coated with polymer encapsulates the ESC. It is best suited for automated disassembly in preparing for direct recycling (Herrmann, et al. 2014), (Grützke, et al. 2015), (Li, et al. 2019)]. Similarly, a magnetic-manipulated electrode (MME) binder that uses Fe_3O_4 particles is easier to remove during disassembly than polyvinylidene fluoride (PVDF), as well as being less expensive and less toxic (Liu, et al. 2015).

7.3.3.1 Scenario 2 Results

The key technologies in the direct recycling process concentrate on maintaining the chemical structure of the cathode material and relithiating the cathode. The process is not yet commercialized but is the focus of intensive research efforts (U.S. Department of Energy 2020). We assumed a very aggressive adoption of direct recycling starting in 2030 for high-value cathode chemistries.[52] Figure 42 presents the wedge chart, and Appendix D, Table D.6 presents details of Scenario 2.

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[51] Battery capacity degrades when the crystalline structure of the cathode is degraded and lithium is lost from the cathode. Relithiation rejuvenates the cathode material by replacing the lithium through solid-state sintering or various combinations of thermal and chemical processes (Chen, et al. 2019).

[52] 25% adoption of direct recycling for high-cost cathode in 2030, 50% in 2036, and 100% in 2041 (high-cost is >= $20/kg in 2020); see Appendix B, Table B.2. ANL BatPac4.0 19FEB2020 Model.

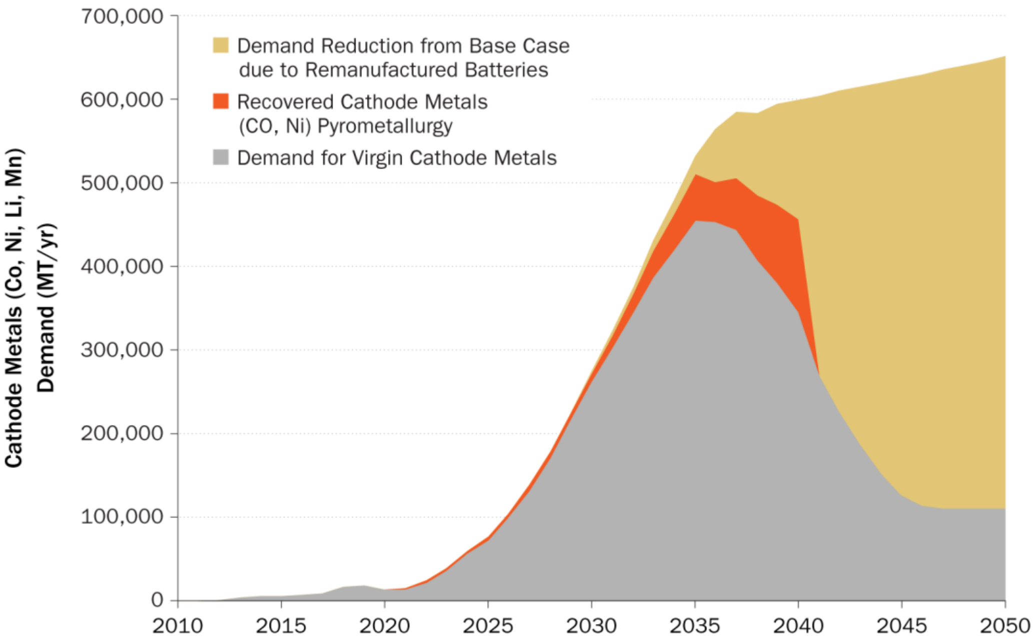

Figure 42. Wedge chart analysis of the reduction in cathode material demand of more than 80% by 2050 for Scenario 2

We termed using recovered cathode material in new batteries as “remanufacturing of batteries” to distinguish it from using separated recovered metals in the production of new cathode material. The reduction in demand for cathode metals resulting from the sale of remanufactured batteries is shown in ochre-yellow in Figure 42. Before 2040, EOL low-cost battery chemistries and high-cost chemistries not directly recycled are routed to pyrometallurgy. By 2042, direct recycling recovers all the retired batteries, which are higher-cost chemistries. By 2050, the demand for virgin cathode metals decreases by 83%, compared to the baseline demand (without recycling).

7.3.4 Scenario 3: Material Substitution

This scenario (Figure 43) identifies substitutions for critical materials in LiBs such as Co, Li, and Ni for reasons including cost, performance, supply chain vulnerability, and environmental or safety concerns. In the case of EV LiBs, manufacturers have also focused on reducing the Co content of batteries because of human rights issues (Jin et al. 2016; Igogo et al. 2019). Battery manufacturers have pursued Co supply and cost mitigation steps, such as developing long-term contracts (Ayre 2018) and designing LiB cathode chemistries with lower-Co content (Croy, Long and Balasubramanian 2019). Battery safety is also a primary concern for LiB and battery electric vehicle (BEV) manufacturers (Zaghib, et al. 2014). Chemical stability is one of the primary challenges in developing low- or no-Co cathodes, and there is a great deal of literature on the subject (Hou, et al. 2017), (Kim, et al. 2018), (Manthiram, Song and Li 2017), (Kerman, et al. 2017), (Nayak, et al. 2018). The DOE Vehicle Technologies Office also has initiated a $32-million, 3-year program to develop low-Co chemistries (U.S. Department of Energy 2018).

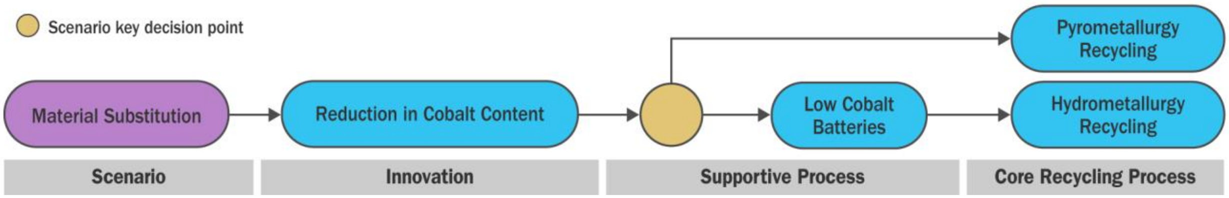

Figure 43. Scenario 3: Material Substitution

Despite the challenges, industry analysts predict that the Co content of automotive LiBs will decrease significantly in the coming decades. McKerracher et al. (2020) expect LiB battery chemistry trends for passenger EVs to continue to evolve from primarily NMC622 chemistry (with over 50% of the market in 2020) to a mixture of lower-Co chemistries; predominantly 90% Ni, 5% Mn, and 5% Co (NMC955, a further evolution of NMC811), and Ni, Mn, Co, aluminum (NMCA) chemistries by 2030. H. Li et al. (2019) found that Co can be removed from NCA without compromising performance. In this scenario, chemistries change rapidly between 2020 and 2035 but stabilize after 2035 as the market matures.

Increased demand for Ni presents its own challenges, and the potentially lower value of recovered metals could push recyclers to recover more materials (especially lithium) and improve the energy efficiency of recycling processes. Figure 44 outlines the contribution of various metals to the cost of LiB cathode material.

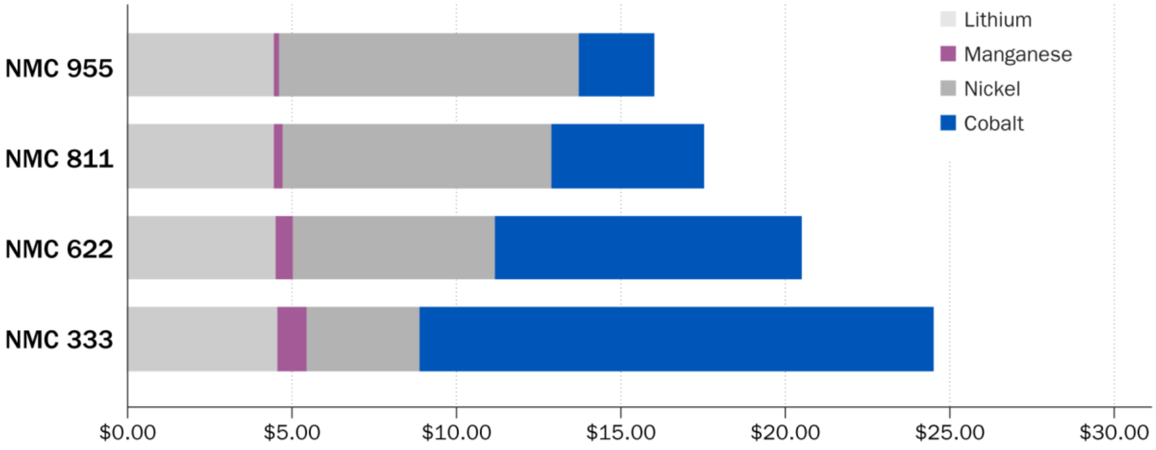

Figure 44. Cost breakdown per kilogram of LiB cathode material with NMC battery chemistries

Sources: McKerracher et al. (2020), GREET,[53] and NREL analysis.

In this scenario, battery chemistries with approximately 18% or higher concentration of Co in the cathode are routed to pyrometallurgical recycling. Low-Co EOL batteries are routed to a hydrometallurgy process, which recovers Co, Ni, Li, and Mn.[54] See Appendix D, Table D.2 for a more complete description of battery chemistries, and Table D.3 for recycling process recovered materials and recovery rates. Appendix

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[53] https://greet.es.anl.gov/index.php

[54] Aluminum and electrolyte organics are also recovered in the hydrometallurgy recycling process. However, they are not included in this analysis to maintain consistency between scenarios.

D, Table D.7 presents details of the Material Substitution Scenario. Figure 45 presents a wedge chart analysis for this scenario.

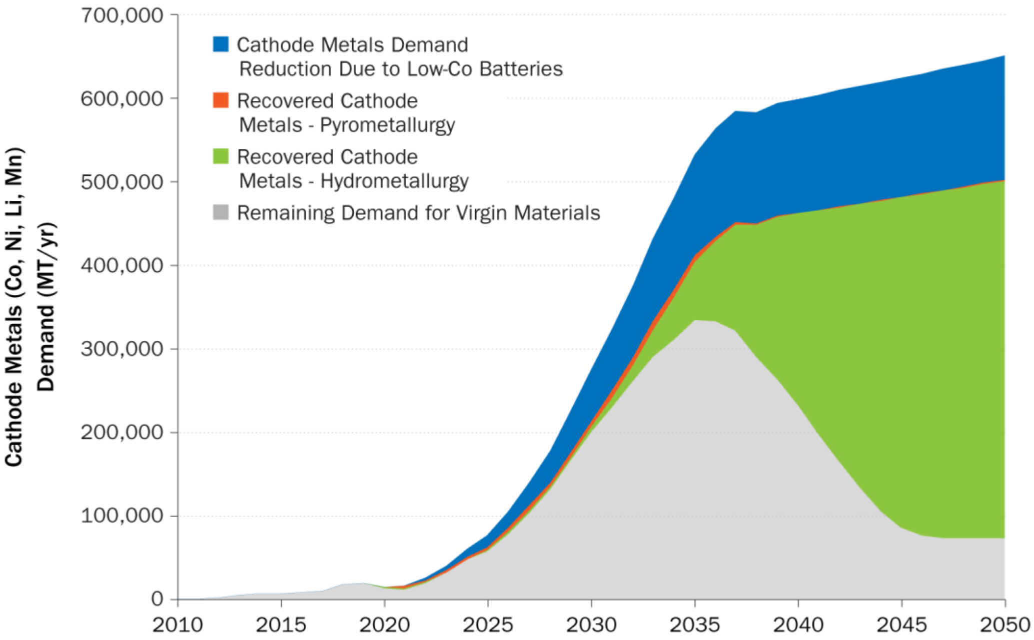

Figure 45. Wedge chart analysis of the reduction in cathode material demand by more than 85% by 2050 for Scenario 3

Adopting lower-Co battery chemistries, including migration to chemistries containing Al,[55] substantially impacts the demand for cathode metals. By 2050, the demand for the four metals included in this analysis has dropped by over 20% in comparison to the base case without low-Co batteries or recycling.[56] Recycling of cathode metals via hydrometallurgy accounts for an additional 65% of the reduction in demand. Pyrometallurgy—which, in this scenario, is used only for high-Co batteries (>18wt% of the cathode metals; NMC 333, 532, and 622), is almost completely phased out as low-Co batteries begin to dominate the EOL battery supply after 2032. In the low-cobalt scenario, lithium iron phosphate (LFP) batteries make up about 15% of the EV batteries sold after about 2025. Their use contributes to the reduction in demand for nickel and cobalt. However, materials recovered from recycling of LFP batteries are not tracked in this analysis.

7.3.5 Direct Recycling and Low-Co Batteries

Direct recycling preserves the structure of the cathode material from EOL batteries, and, after relithiation, produces “good as new” cathode material that can be used in new batteries. The expected rapid evolution of battery chemistries between 2025 and 2035 presents a challenge for direct recycling; by the time batteries reach their EOL after 10 or more years in service, the battery chemistry of the recovered cathode

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[55] NCA+ and NMCA chemistries make up over 30% of EOL batteries by 2040 compared to the chemistry mix in the baseline case, which includes about 9% battery chemistries with aluminum.

[56] Part of the decrease in demand can be attributed to the adoption of NMCA (Ni, Mn, Co, Al) batteries in the low-Co battery scenario. Aluminum is not included in the chart, and therefore adoption of these batteries is not fully represented.

material may not match the chemistry of recently manufactured batteries. We examined the potential magnitude of this problem by applying the direct recycling process from Scenario 2 to the evolution of battery chemistries depicted in Scenario 3. Figure 46 plots the value of recovered cathode material that exceeds demand for that chemistry by year.[57]

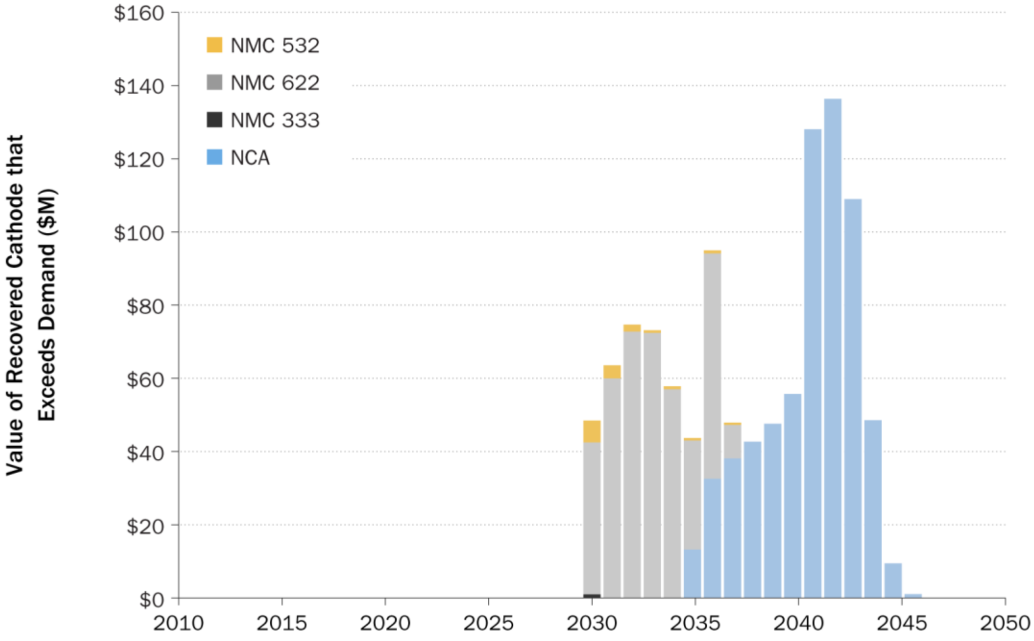

Figure 46. Recovered but unsaleable obsolete chemistry cathode material ($ millions).

In the Material Substitution scenario, the NMC 333 and 532 chemistries are phased out entirely by 2025, and the NMC 622 chemistry that dominates the current market is largely phased out by 2030. A few vehicles with NMC 532 chemistry would still reach their EOL as late as 2039. Most NMC 622 batteries being manufactured now would reach their EOL between 2030 and 2045—well after demand for this chemistry has fallen below 1% of the total demand in the scenario. In the Material Substitution scenario, NCA batteries are never a large percentage of the total, but sales peak much later (in 2034) when more vehicles are being sold. Sales of NCA batteries fall off abruptly, so many batteries are retired after demand has fallen. If all these high-Co batteries were recycled using direct recycling starting in 2030, the potential lost value from being unable to sell the recovered cathode material would exceed $1 billion at baseline cathode prices.

While this analysis represents a worst-case scenario, in which cathode material is recovered but cannot be sold, it provides insight into the value of research efforts directed at reconfiguring cathode material after recovery. However, even if research efforts result in a process for adjusting the cathode chemistry after recovery by direct recycling, such a process may be uneconomical compared to proven existing technologies, such as hydrometallurgy.

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[57] The value is based on the number of batteries of each chemistry recycled each year versus demand for each chemistry with 95% collection of retired batteries and 90% recovery for direct recycling (see Appendix B, Table B.8 for cathode cost).

7.3.6 Economic Considerations for EV LiB Recycling

7.3.6.1 Prices and Revenues for Recycled Materials

Figure 47 charts Ni prices over 2 years (2019–2021) and illustrates one of the primary challenges for LiB recycling: the potential for large price fluctuations for recycled metals. The impact of these fluctuations could be quite significant over time. Figure 48 presents the results of assuming revenue for recycled Ni for the material substitution scenario at low ($11.30/kg—baseline) and high ($17.80/kg—sensitivity) values of Ni prices over these 2 years.

Figure 47. Nickel prices ($/MT), 2019–2020

Note: Source of data for the graph: LME (2021)

The potential revenue from sales of recycled cathode metals, refurbished batteries, and recovered intact cathode for the three scenarios depends on the market value of the materials. We used “baseline” and “sensitivity” cases to explore the potential ranges of revenue and profits for the three scenarios. We derived the baseline values using assumptions based on current or near-past prices and based sensitivity values on near-future price projections (1 to 3 years). However, while the baseline and sensitivity prices roughly correspond to “current” and “future” prices, they are used in the case study to represent a reasonable range of values for sensitivity analysis rather than a projected evolution of prices over time. Prices for EV LiBs, which are currently about $137/kWh, are projected to decrease in price to $100/kWh by 2023 (an approximately 27% reduction in prices from current values) (BloombergNEF 2020).

The market price for refurbished batteries would be reduced in comparison to new batteries because they would have a reduced lifespan (we assumed a 50% reduction in lifespan) and reduced performance in comparison to new batteries (see Section 7.3.2.1, *Scenario 1 Barriers*). We assumed that refurbished batteries (Design for Disassembly Scenario) could be sold for one-half the prevailing price for new batteries (Neubauer et al. 2015). Thus, the baseline price for refurbished batteries is assumed to be $68.50/kWh and the sensitivity price is assumed to be $50.00/kWh.

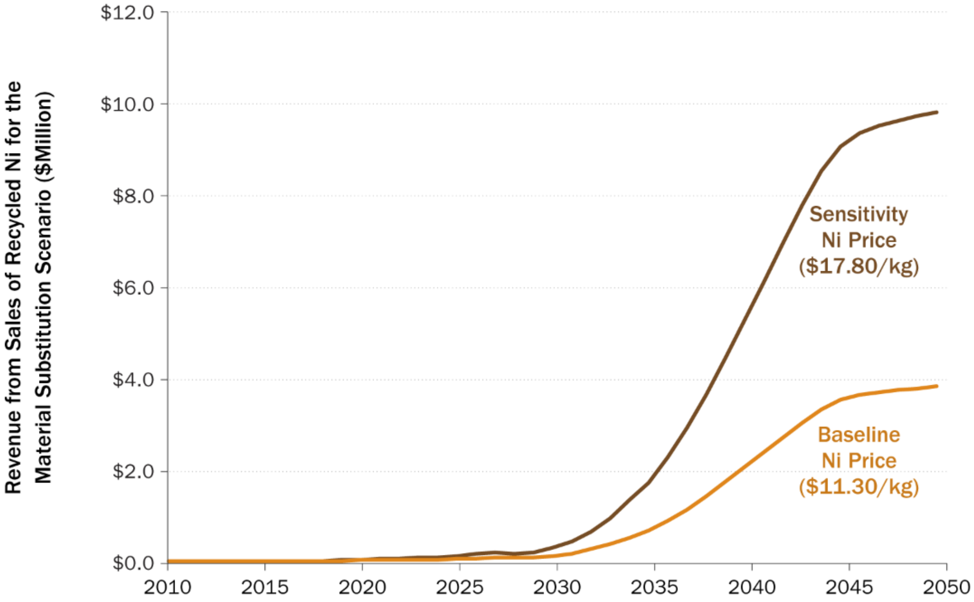

Figure 48. Revenue from recycled nickel for Scenario 3 for two values of nickel prices

In contrast, recovered intact cathode material for the Design for Recycling Scenario was assumed to be "good as new" and, therefore, would be priced the same as cathode material derived from virgin materials.[58] However, we assumed that both new and recovered cathode material prices would follow the same trend as battery prices, decreasing 27% for the sensitivity case. As illustrated in Figure 47, future metals prices are difficult to predict. Therefore, we assumed that baseline metals prices are represented by metals prices cited in GREET (see Appendix D, Table D.4) and sensitivity metals prices are based on recent prices from the London Metal Exchange.[59] Table 9 summarizes the baseline and sensitivity price cases.

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[58] Appendix D, Table D.2 lists the current prices of various cathode materials.

[59] https://www.lme.com/Metals

Ni	17,807	USD/MT (2/1/2021)	(17,800 USD/MT Ni assumed)
Co	41,240	USD/MT (2/1/2021)	(41,200 USD/MT Co assumed)
Li	9	USD/kg LiOH2O (1/28/2021)	(53,100 USD/MT Li assumed)

Table 9. Baseline and Sensitivity Prices for Recovered and Refurbished Materials

Recycled Material	Baseline Price Assumption	Sensitivity Price Assumption
Cobalt	\$51.30/kg	\$41.20/kg ↓ 20%
Nickel	\$11.30/kg	\$17.80/kg ↑ 58%
Lithium	\$42.10/kg	\$53.10/kg ↑ 26%
Cathode Material[a]	\$20-\$22/kg[b]	\$15-\$16/kg ↓ 27% [c]
Refurbished Batteries	\$68.50/kWh	\$50/kWh ↓27% [c]

a. Baseline prices in GREET

b. Chemistries containing Co

c. The sensitivity prices for cathode material and batteries are based on projected decreases in battery cost rather than projections of underlying metals costs

7.3.6.2 Recycling Costs

Figure 49 plots the recycling costs (\$/kg input cells) for the three recycling methods used in this case study. Recycling costs are identical for all battery chemistries,[60] for both the baseline and sensitivity cases. A thorough analysis of the optimum size (MT/year) for recycling facilities would account for the increasing supply of EOL batteries and changes in their geographic distribution over time to balance transportation costs with facility costs, but that analysis is beyond the scope of this study.

We chose a plant size of 5,000 MT cells per year for this analysis throughout the time period. At 5,000 MT/year, most economies-of-scale have been realized while maximizing the number of plants to reduce transport distances for EOL batteries. Collection and transport of EOL batteries is discussed in more detail in the Design for Disassembly section (Section 7.3.2). Values for recycling costs for the three technologies are listed in Appendix D, Table D.4. This case study does not include costs for EOL battery removal from vehicles, collection, storage, transport, or disassembly to the module or cell level for recycling. While these costs could be substantial, they are similar for the three scenarios and are beyond the scope of this analysis.

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[60] ANL EverBatt Model, 2019. https://www.anl.gov/amd/everbatt.

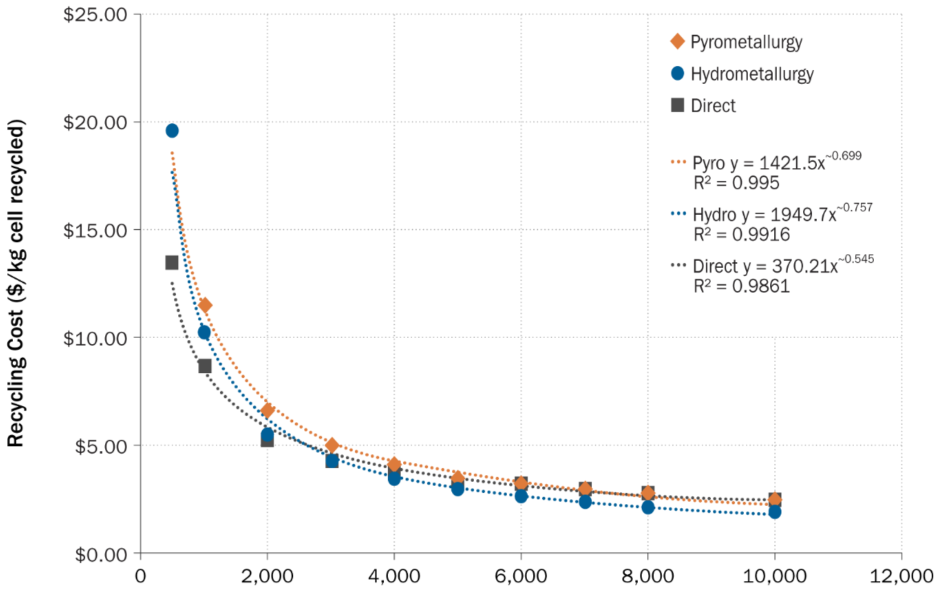

Figure 49. Recycling cost as a function of facility size (MT/yr) for pyrometallurgy, hydrometallurgy, and direct cathode recycling of LiB cells

Source: ANL EverBatt Model (05/23/2019)

7.3.6.3 Profit Ranges

Figure 50 plots profit ranges for the three scenarios under the baseline and sensitivity recovered material price assumptions and the recycling and refurbishment cost assumptions discussed above.

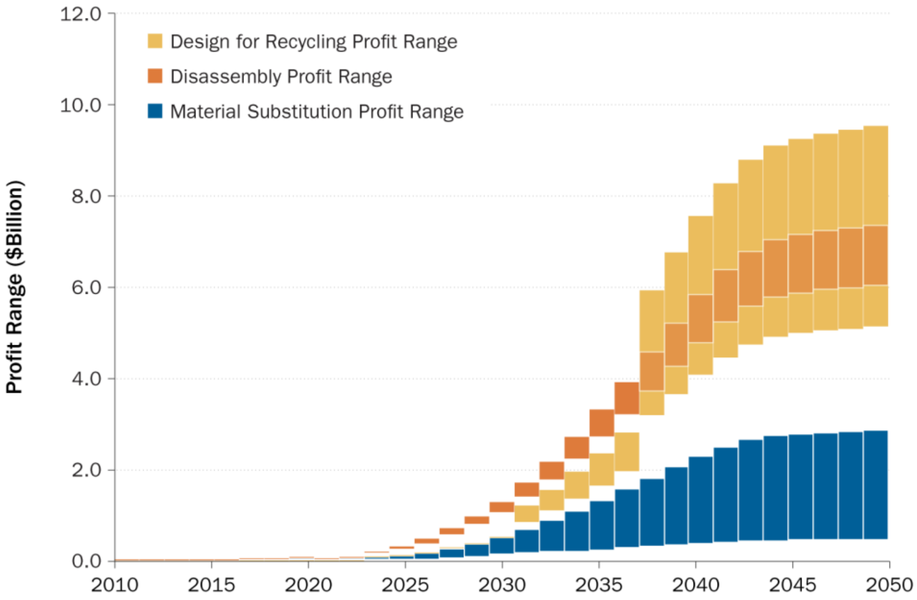

Figure 50. Profit ranges for the three scenarios using baseline and sensitivity assumptions for prices for recovered and reused materials.

For both Scenario 1, Design for Disassembly and Scenario 2, Design for Recycling, profits using the sensitivity price assumptions are lower than the profits using the baseline price assumptions. Both reductions are due to the assumption that prices for batteries will continue to decrease following recent trends. The profit range for the baseline versus sensitivity assumptions for the Design for Recycling Scenario is much larger than for the Design for Disassembly Scenario because adoption of the direct recycling technology over time results in virtually all EOL batteries being recycled to recover intact cathode material in the Design for Recycling Scenario by 2043. In comparison, only 20% of EOL batteries are refurbished in the Design for Disassembly Scenario. Thus, the reduction in revenue for recovered cathode for the sensitivity prices for the Design for Recycling Scenario has a much bigger impact on the overall revenue than the identical percentage reduction in revenue for refurbished batteries in the Design for Disassembly Scenario.

For the Material Substitution Scenario, a lower volume of cathode material is recovered from retirements of low-Co batteries, so the total profit for recovered materials is lower than for the other two scenarios. However, because Ni is substituted for Co in this scenario and, thus, the mass of Ni recovered is many times the mass of Co (projected to be 14x in 2050), the revenue is sensitive to the price of Ni (see Table 9 for price assumptions). Overall, the sensitivity case prices yield a higher total revenue and profit for the Material Substitution Scenario.

Figure 51, Figure 52, and Figure 53 plot the percent profit [(revenue - cost)/revenue] for the three scenarios.

In the Design for Disassembly Scenario (Figure 51), testing and refurbishment of batteries begin in 2030, resulting in a significant jump in profitability. Even for the sensitivity case, in which refurbished batteries are sold at a lower price ($50/kWh), refurbished batteries have a higher value than the recovered metals would have if the same batteries were recycled using pyrometallurgy. For example, in 2040, using the sensitivity price assumptions, the revenue for metals derived from pyrometallurgy recycling is about $1,190 per battery at a cost of about $900/battery, yielding a profit of $290/battery. Testing for refurbishment would cost about $1,500/battery (cost based on Neubauer et al. 2015, see Table D.4 for assumptions), earning a revenue of about $3,750/battery for the sensitivity case, or a profit of approximately $2,250/battery—almost an order of magnitude higher.

The Design for Recycling Scenario profitability (Figure 52) follows a similar pattern after recovery of cathode material and resale for batteries begins in 2030. For the sensitivity case with a 27% reduction in the price of recovered cathode material, the profit per battery for sale of recovered cathode is about $660/battery. The overall profitability of this scenario continues to increase as the direct recycling technology is adopted over time.

The Material Substitution Scenario is the least profitable of the scenarios analyzed (Figure 53). It is also the only scenario in which the sensitivity price scenario is more profitable than the baseline price scenario throughout the analysis period. Sales of recovered Li make up 25% of the revenue in 2040 for the sensitivity case for the Material Substitution Scenario, resulting in a profit of about $344/battery recycled via hydrometallurgy. If pyrometallurgy had been used to recover only Co and Ni from these batteries, applying the sensitivity price assumptions would result in a loss of about $72/battery.

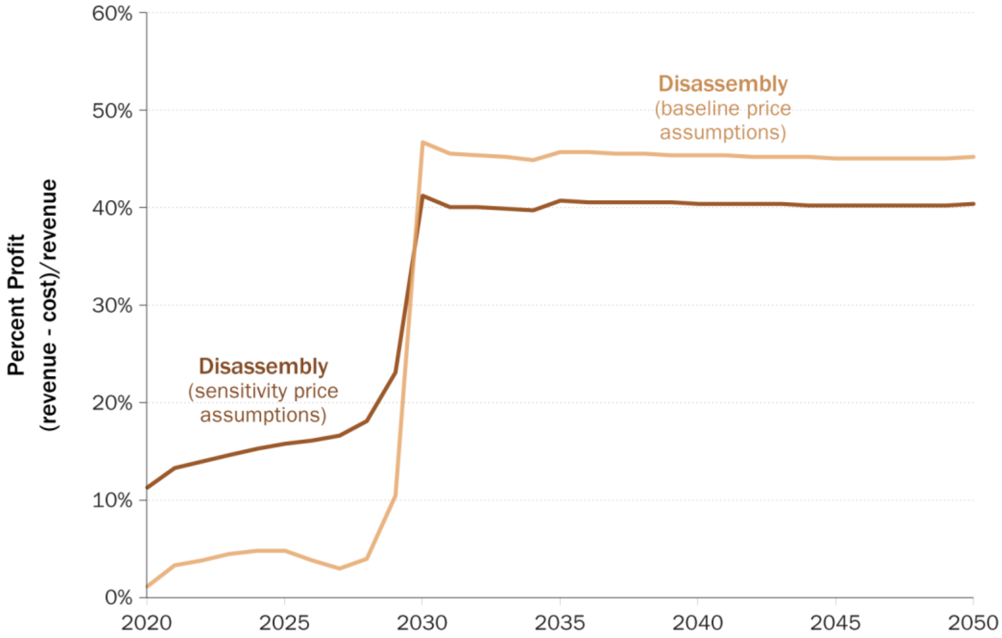

Figure 51. Percent profit ([revenue-costs]/revenue) for Scenario 1 under two price assumptions

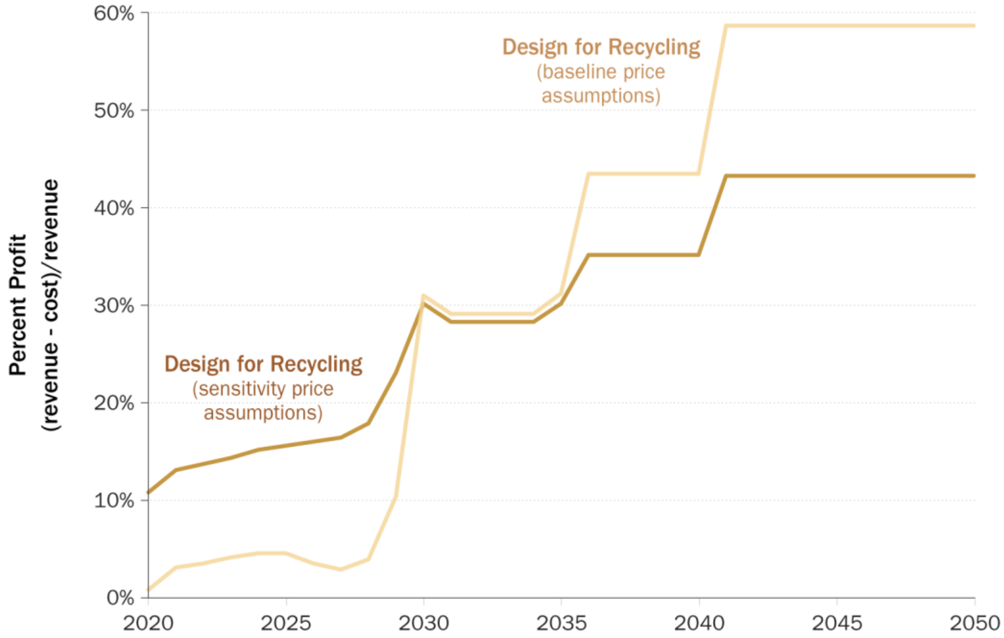

Figure 52. Percent profit for Scenario 2 under two price assumptions

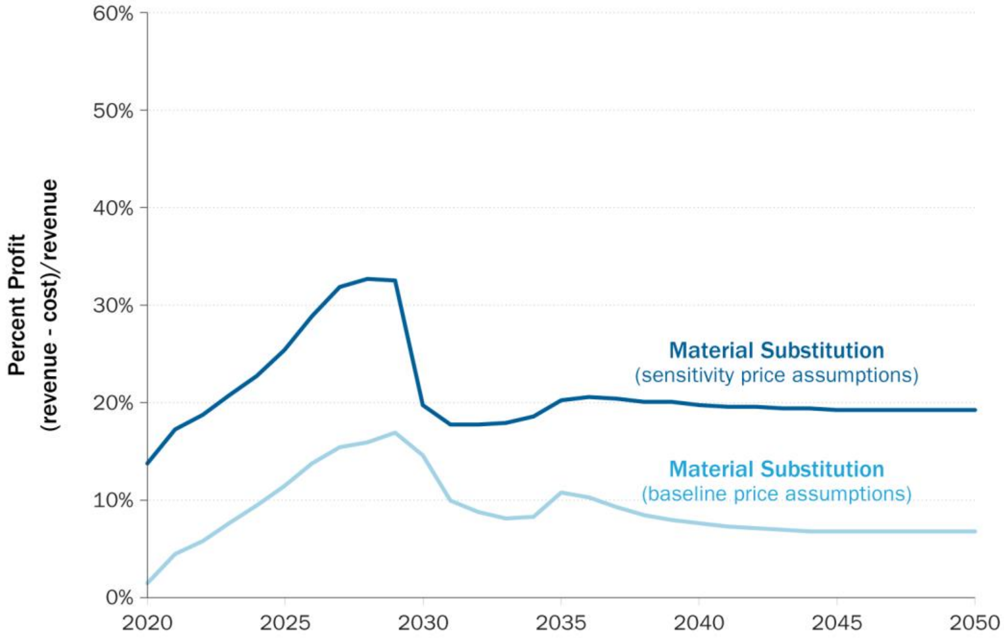

Figure 53. Percent profit for Scenario 3 under two price assumptions

7.3.7 Life Cycle Environmental Impacts of Lithium-Ion Batteries and Associated Materials

In this LCA analysis, the source of data for virgin metals is the life cycle inventories present in the ecoinvent database.[61] For the battery chemistry materials and the battery recycling process, the analysis uses data from the GREET modeling tool.[62]

7.3.8 LCA Need

Some have proposed that future advances in EV battery chemistry include reducing the Co content of batteries and replacing it with Ni. Although this substitution might have functional benefits, it could generate relatively higher environmental impacts because of the increased demand of Ni to replace Co. Therefore, it is critical to conduct environmental impact assessments, such as LCA, to develop insights into the comparative environmental benefits or penalties associated with changing battery chemistries and the subsequent demand for metals in the BEV sector.

Moreover, because metals in currently deployed batteries will become available over the next 30 years due to EOL recycling, it is essential, now, to consider the environmental burdens of recycling processes to extract Li, Co, and Ni from batteries for subsequent uses.

7.3.8.1 Scope and Functional Unit

This LCA compared the impacts for virgin and recycled battery cells for two bounding cases of battery cathode chemistries: LiNi1/3Mn1/3Co1/3O2 (NMC111) and LiNi0.8Mn0.1Co0.1O2 (NMC811). The analysis basis is 1 kWh of battery cell energy capacity. The overall lifecycle impacts of the three scenarios would depend on the total virgin material use and the amount of various materials that are recycled, which reflects the ratio of metals in each battery chemistry as well as the overall mass of battery

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[61] https://www.ecoinvent.org

[62] https://greet.es.anl.gov/index.php

materials and processes used to derive them (mining and recycling). However, the scope of this analysis is focused on comparing virgin material extraction to recycling, and all results are presented on a per kWh basis. The limited scope of the LCA precluded analysis of the total impacts for each of the scenarios, which are planned for follow-on work.

While recycling LCA impacts are largely independent of battery chemistry, the impacts of mining are highly dependent on the materials being mined and the processes used to purify and refine them. Therefore, two bounding cases of battery chemistry are used to capture the importance of materials in the LCA, The lower relative content of Co per unit cathode material for the NMC811 represents an upper limit scenario for substituting the C in NMC battery cathodes with Ni. The boundary of the LCA study for virgin battery cells is from cradle-to-gate and end-of-life disposal (i.e., from mining and extraction of raw materials, transport to the battery manufacturer, manufacturing of the cells, and subsequent landfilling). At the EOL of recycled battery cells, 95% of the cells go through a hydrometallurgical recycling process where the battery materials are then recovered and become available for further use. The use phase impacts of the battery are a function of the electricity mix and battery lifetime. Because this LCA aims to compare material substitution impacts for two battery chemistries, the use phase is outside the scope of this LCA. The available data on material recovery, additional inputs, and emissions for the recycling process is expressed as per kg cell. To enable a fair comparison between the different virgin and recycled battery chemistries and life cycles, the functional unit chosen for this LCA is 1 kilowatt hour of battery energy delivered, which accounts for the differing amounts of, and consequently environmental impacts associated with, materials required in each battery chemistry to meet the same functional requirements. Based on data available from the bill of materials in the GREET model (Dai, Kelly, et al. 2018), the pack energy of BEV LiBs is 23.5 kWh and the mass of total cell components amounts to 119.77 kg for the NMC111 chemistry and 112.37 kg for the NMC811 chemistry.

7.3.8.2 Data for the Virgin Battery Life Cycle

The bill of material for BEV NMC111 and NMC811 batteries on a cell basis is from GREET.[28] The process information for producing battery chemicals is modeled from several sources, as explained in this section.

Metals

Lithium carbonate is used in EV batteries to make cathode material. Virgin Li is the base material for producing lithium carbonate; it is available in the form of concentrated Li brine containing 6.7% Li by weight. The production site for lithium brine is in Chile, and all data originate from an environmental survey about the plant in Chile. This data set represents global Li brine production, as this plant produces 70% of the lithium carbonate worldwide. This data set includes transportation by truck to the end user of Li brine.[63]

Virgin cobalt production takes place primarily in Congo (mining) and China (refining). The ecoinvent database provides the global Co production process. Adjusted to represent only the share of virgin Co produced worldwide, it includes estimates of transport distances from the mine to the market (including battery producers).

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[63] The basis for the data is the ecoinvent data set ‘Lithium brine, 6.7 % Li {GLO}| lithium brine inspissation | Alloc Def, U’. This data set mainly refers to the lithium brine production in Chile and is based on values listed in the ecoinvent database via personal communication with the manufacturers.

Virgin nickel production represents the cradle-to-gate production of Ni mined, extracted, smelted, and refined globally from Ni ores and of Ni extracted as a coproduct of platinum metal group mining. The data set also includes a transportation mix of global Ni production from the mine to the market (including battery producers).

Other Battery Chemicals

The battery cells also contain the following elements:

- A graphite anode (assumed to be produced from pet coke and coal tar pitch by carbonation followed by graphitization; data exclude the final particle refinement step)
- A cathode depending on the specific chemistry (NMC111 versus NMC8111, in this case)
- A binder made from polyvinyl difluoride
- Electrolyte made from lithium hexafluorophosphate and an equal ratio of ethylene carbonate and dimethyl ether
- A separator made of approximately 80% polypropylene and 20% polyethylene
- Carbon black
- Copper
- Aluminum.

The production of NMC cathodes of either chemistry requires several upstream steps, such as producing lithium carbonate, manganese sulfate, nickel sulfate, and NMC precursor before the actual NMC cathode is made. The upstream energy and material requirements are common to both the NMC111 and NMC811 cathode chemistries. However, the manufacturing process of the two cathodes differs slightly in the materials and energy required. The life cycle inventory of the battery chemicals is modeled based on data available in GREET wherever possible. Where detailed LCI data are not available, the analysis uses global average market data sets for the chemicals from the ecoinvent database.

7.3.8.3 Data for the Recycling Process

The analysis assumes the spent Li-ion battery cells are recycled through a hydrometallurgical process using organic acid leaching to extract lithium carbonate, cobalt sulfate, nickel sulfate, manganese sulfate, graphite, electrolyte organics, copper, and aluminum. It further assumes a 95% collection rate for the recycling process, disposing the remaining spent cell waste stream in a landfill. The recycling process requires additional materials, such as citric acid, soda ash, hydrogen peroxide, and sulfuric acid, along with steam, natural gas, and electricity. The life cycle inventory information for the recycling process is based on data from GREET, and the upstream flows (i.e., inputs to the recycling process) are modeled using existing ecoinvent data sets.

7.3.8.4 Life Cycle Impact Assessment Methods

The impact assessment method ReCiPe 2016 Midpoint (I) (Huijbregts, et al. 2016) is used to quantify the potential environmental impacts of virgin and recycled NMC111 and NMC811 cells. The mineral resource scarcity is expressed using the Surplus Ore Potential (SOP) midpoint indicator for each mineral (SOP expressed in kg Cu eq), which expresses the average extra amount of ore produced in the future caused by the extraction of a mineral resource considering all future production of that mineral resource

Fossil resource scarcity is expressed as Fossil Fuel Potential (FFP, expressed in kg oil-eq), defined as the ratio between the higher heating value of a fossil resource and the energy content of crude oil. Climate change impact is expressed in global warming potential (GWP, expressed in kg CO_2 eq), which quantifies the integrated infrared radiative forcing increase of a greenhouse gas. The stratospheric ozone depletion is expressed using the ozone depletion potential midpoint indicator (ODP expressed in kg CFC-11 eq), which refers to a time-integrated decrease in stratospheric ozone concentration over an infinite time horizon. The Ionizing Radiation Potential (IRP, expressed in kBq Cobalt-60 eq to air) is used to quantify the ionizing radiation impact potential and is derived from the collective dose resulting from the emission of a radionuclide. Toxicity impacts on humans and ecosystems are calculated separately in ReCiPe as human carcinogenic toxicity, human noncarcinogenic toxicity, marine ecotoxicity, freshwater ecotoxicity, and terrestrial ecotoxicity, using kg 1,4 -dicholorobenzene eq as the unit basis. Huijbregts et al. (2016) present a detailed definition of all the other midpoint impact assessment categories included in the ReCiPe 2016 method.

7.3.8.5 Life Cycle Impact Assessment Results

Figure 54 summarizes the potential midpoint environmental impacts on a per-kilowatt-hour basis for the NMC11 and NMC811 cells, both for the virgin and recycled products. The production and recycling of NMC811 cells shows the lowest environmental impacts for these midpoint impact assessment categories: global warming, ionizing radiation, fossil resource scarcity, water consumption, freshwater eutrophication, ozone formation (terrestrial ecosystems), fine particulate matter formation, stratospheric ozone depletion, and ozone formation (human health). When comparing the virgin production life cycles of NMC111 and NMC811 (i.e., without recycling), the shift to a lower-Co content battery cathode (NMC811) still seems promising from an environmental perspective. Although substituting Co with Ni may increase human carcinogenic and noncarcinogenic toxicity and freshwater ecotoxicity due to a higher content of Ni and Ni-containing compounds, a lower amount of battery cathode materials required for the NMC811 chemistry to deliver a kilowatt hour of energy output offsets the aforementioned potential increase in the toxicity-related impacts. Among the impact assessment categories, NMC811 battery chemistry performs slightly worse than the NMC111 only in the case of terrestrial acidification, which can be directly attributed to the higher nickel content in the cathode leading to SO_2 emissions from nickel salts such as $NiSO_4$

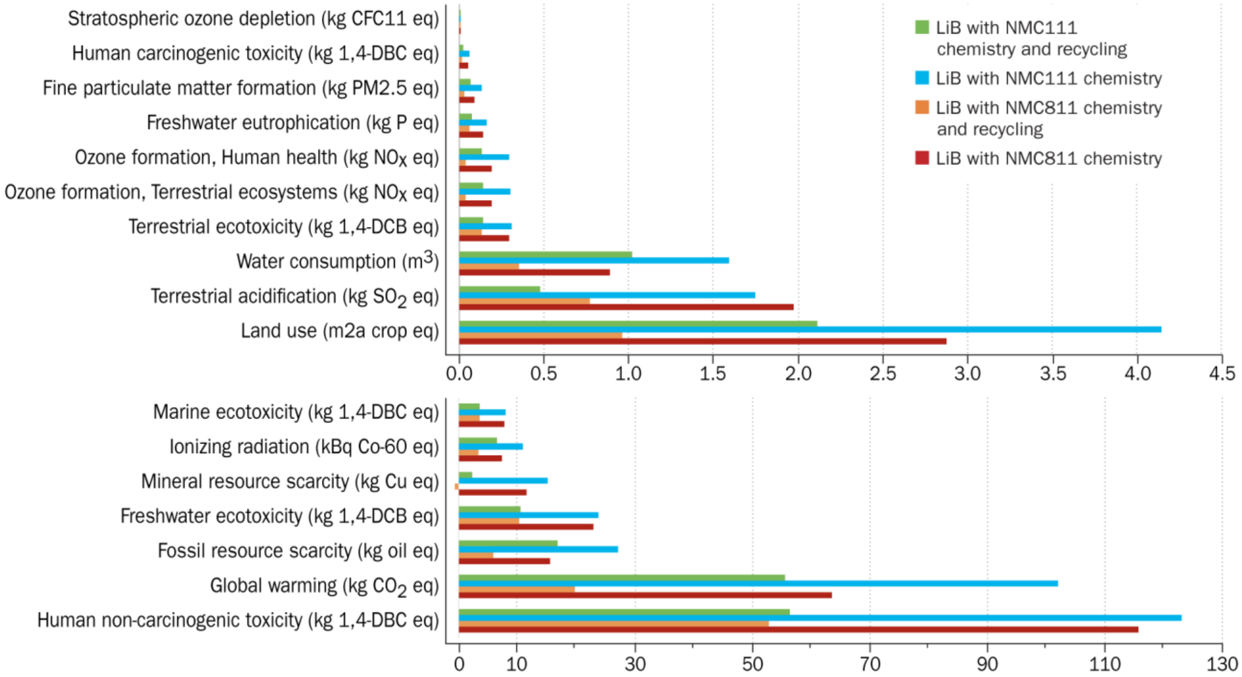

Figure 54. LCA comparison of NMC111 versus NMC 811 cells with and without recycling

Note: Comparison is on a per-kilowatt-hour basis using ReCiPe midpoint impact assessment method.

Figure 55 shows the impacts of mineral resource scarcity (shown on the *y* axis), fossil resource scarcity (shown on the *x* axis), and climate change (shown by the size of the bubbles and GWP in kg CO_2 eq values in the bubbles) for the different LiB scenarios considered in the case study relative to the baseline scenario of virgin NMC111 LiB without recycling (shown at the origin of the plot). From the figure, it can be inferred that the NMC811 LiB chemistry with the recycling scenario has about 21 units lesser fossil resource scarcity impact, 16 units lesser mineral resource scarcity impact, and more than five units lesser climate change impact as compared to the virgin NMC111 baseline. Comparing only the virgin production life cycles of NMC111 and NMC811, we can see that although the climate change impact for NMC811 is lesser than that of NMC111 by about 37%, there is only a slight improvement when it comes to mineral and fossil resource scarcity impacts. This highlights the significance of recycling in addition to cathode material substitution in LiBs for achieving the maximum environmental performance benefits.

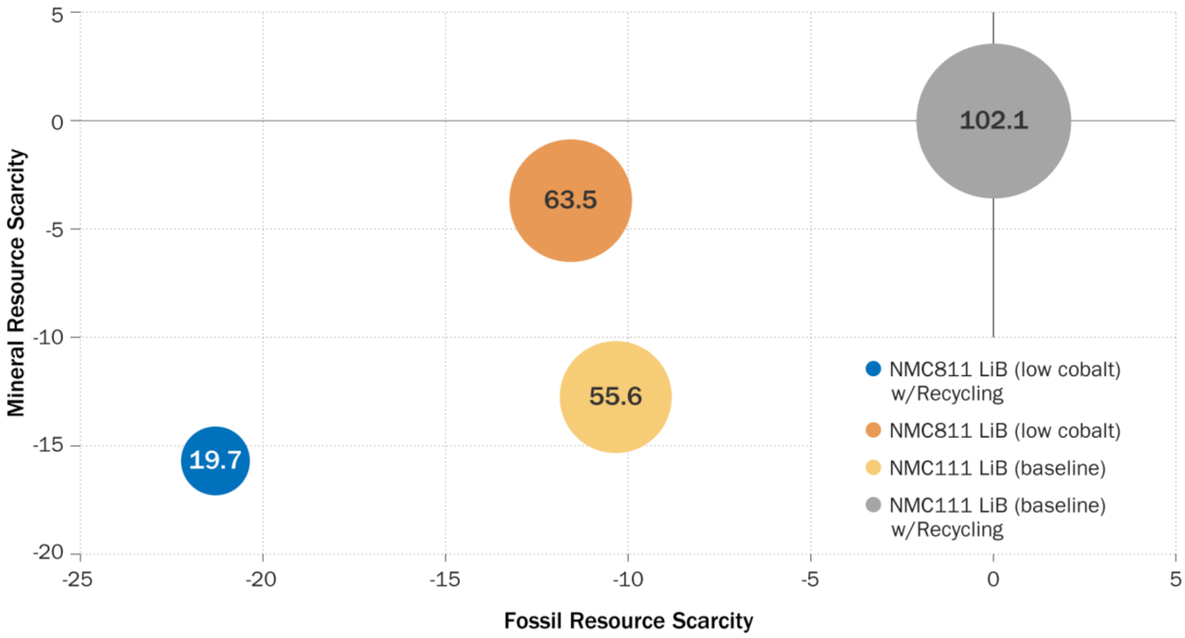

Figure 55. Mineral resource scarcity and fossil resource scarcity for the four LiB scenarios

Note: Results are presented relative to the baseline scenario (NMC111 LiB); GWP (bubble size and quantity in the bubbles in kg CO_2 eq).

Figures D.1 and D.2 in Appendix D highlight the relative contribution of the major processes involved in the life cycles of virgin NMC111 and NMC811 production, respectively, to the overall environmental impact. The impacts associated with battery cathode production play a significant role for many of the impact categories (e.g., global warming, terrestrial acidification, mineral resource scarcity, fossil resource scarcity, and water consumption). In addition, copper and graphite anode are major sources of the environmental impact associated with virgin battery production for both the cathode chemistries, indicating the need to recycle these materials and consequently improve recycling rates and efficiencies for the BEV sector.

7.4 Discussion

The three scenarios all result in dramatic decreases in the demand for virgin raw materials for EV batteries after about 2035. The sale of refurbished batteries (Scenario 1) and the change in battery chemistries represented by the Material Substitution Scenario (Scenario 3) both result in a reduction in the overall demand for materials for batteries.[64] Substituting Co with Ni in the battery cathode is represented by considering the NMC811 cathode, which has the lowest relative Co content compared to Ni. The LCA analysis, as shown in Figure 54, indicates that substituting Co with Ni in Scenario 3 (NMC811) has a lower environmental impact in most of the environmental impact categories as compared to the baseline cathode chemistry (NMC111). However, it might create unintended trade-offs due to higher impacts related to effects of Ni on human toxicity and ecotoxicity. The recycling of battery materials to produce new batteries shows a significant reduction in environmental impacts compared to their virgin

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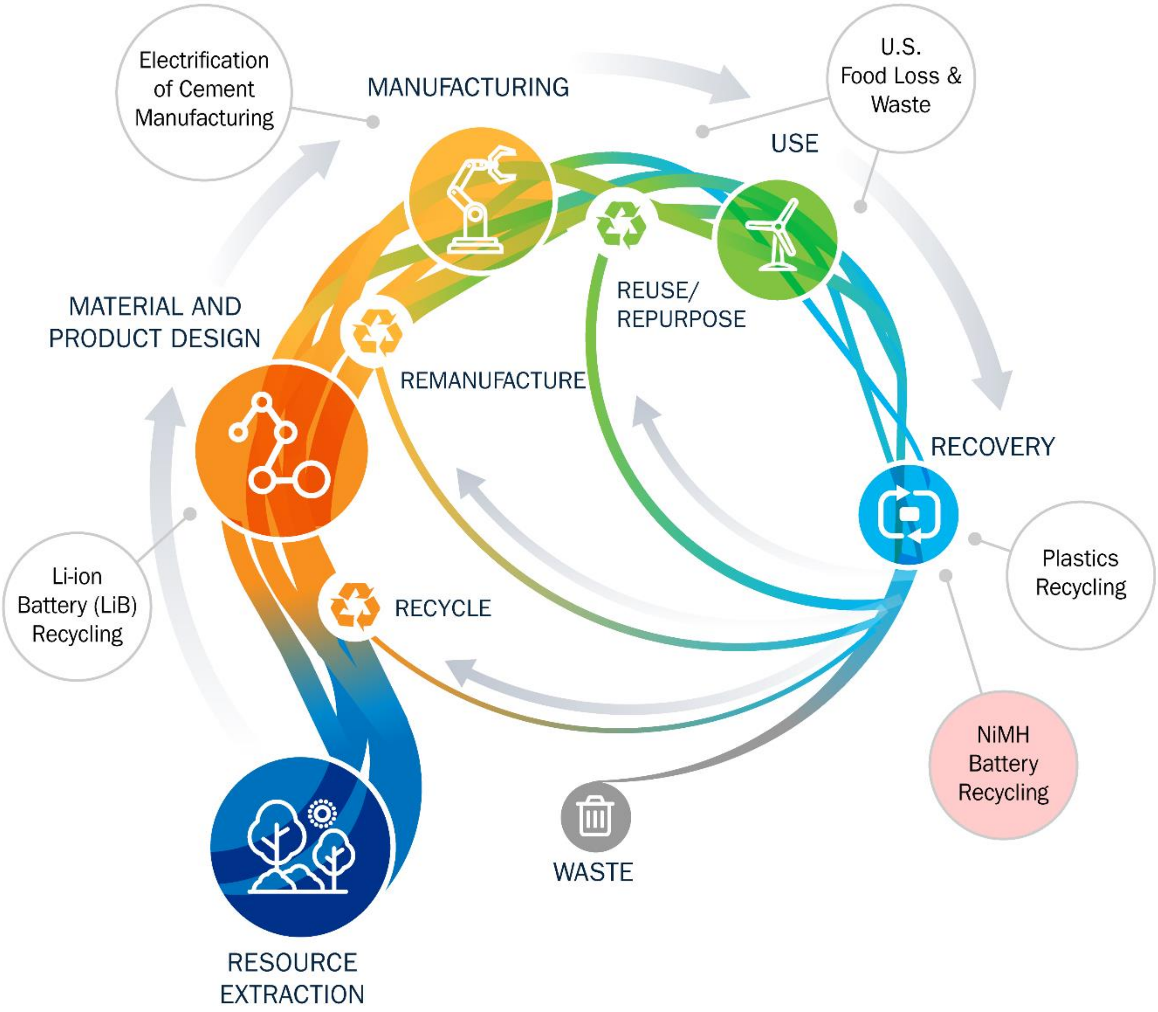
[64] In addition to substitution of Ni for Co, future battery chemistries envisioned in Scenario 3 also shift toward batteries with more Al. Because Al is not included in the study, the overall demand for cathode metals in later years is underrepresented in Scenario 3.

counterparts across all impact categories. From an environmental standpoint, increased recycling in combination with a shift to low-Co chemistries seems to be the most promising approach. Scenario 3, which shows both the largest reduction in total (combined) virgin material use of the three scenarios and the impacts reduction associated with transitioning to lower-cobalt chemistries, is the clear leader in reducing environmental impacts. However, it is worth noting that due to a lower volume of cathode materials recovered and price sensitivity of Ni under the Material Substitution Scenario, substituting Co with Ni has a lower profitability, which increases under the sensitivity price scenario, as shown in Section 7.3.6.3.

The analysis also highlights the benefits of recycling methods that recover more materials (i.e., direct recycling in Scenario 2 and hydrometallurgy for low-Co batteries in Scenario 3). Even though the Design for Disassembly Scenario (Scenario 1) substantially reduces the overall demand for materials, pyrometallurgy recycling does not recover enough materials from the final EOL batteries to make Scenario 1 the best strategy for reducing demand for virgin materials. Recovery of intact cathode material (Scenario 2) has similar material benefits to hydrometallurgy and the potential for a higher-value product. However, as discussed previously for direct recycling and the low-Co chemistry scenario, there is some risk that direct recycling would recover material that is no longer in high demand. Pyrometallurgy may not be profitable for low-Co batteries, especially if the price for Co drops in response to lower demand.

8 Case Study: Recovering Critical Materials from NiMH Batteries—Assessing Costs and Environmental Impacts



8.1 Introduction

NiMH batteries contain Ni and rare-earth elements (REEs) that, if recovered, could support domestic manufacturing of emerging clean energy and transportation technologies. Ni is an important element in LiB cathodes. Didymium (Di) (an alloy of neodymium and praseodymium, rare-earth elements identified as critical materials) is used to produce neodymium iron boron (NdFeB) magnets for energy-efficient electric machines in many applications, including EVs and wind turbines. At this writing, the United States does not have the capacity to produce refined Ni or Di. Systematic recovery of these materials from EOL hybrid vehicle NiMH batteries could offset the country’s reliance on imports and contribute to securing their long-term supply. Recycling is a key strategy for securing critical material supplies as well as being a pillar of the circular economy.

However, recovering materials from EOL products faces challenging economics, environmental impacts, and logistics. In the private sector, recovered materials compete in existing markets with materials sourced from primary sources, effectively constraining the capital and operating costs for recycling facilities. Additionally, recycling processes that cause more harm to the environment than primary material processing oppose circular economy sustainability goals. Logistics are challenged by EOL products that are geographically dispersed, variable in quantity and quality over time, and difficult to collect. For these reasons, systematic analysis of diverse EOL technologies and reverse supply chain challenges will be essential to achieve circular economy goals.

NiMH batteries deployed in hybrid electric vehicles (HEVs) are candidates for recovering critical materials from EOL recycling. Varied metal compositions for the NiMH anode have been deployed, based primarily on AB-type metal alloys (e.g., AB_5, AB_2, A_2B_7) containing REEs, Ni, and combinations of metals such as Mn, Co, and magnesium (Fetcenko, et al. 2007), (Ebin, Petranikova and Ekberg 2018). Besides being on the critical minerals list (Nasser and Fortier 2021), industry stakeholders are concerned about the ability of markets to meet future demand for the high-purity Ni required in Li-ion battery cathode production (U.S. DOE-EERE 2020). Unlike recycling consumer NiMH batteries, recycling vehicle batteries and recovery of rare-earth metals are relatively nascent endeavors (Gaines 2014). However, corporate and government interest in exploring and implementing reverse logistics solutions is growing in response to emergent trends in environmental legislation, public opinion, sustainability goals, and material criticalities (Gaustad, et al. 2018). Therefore, a comprehensive assessment of reverse logistics for battery-critical material recovery is essential to evaluating the costs and environmental impacts of recycling NiMH HEV batteries in the United States.

This study applies an integrated analysis framework built on RELOG, ANL's open-source reverse logistics model (Xavier and Illoeje 2020) to explore the viability of NiMH HEV battery recycling in the United States. It considers two potential future scenarios for EOL battery availability: conservative and optimistic. Specifically, the analysis explores the following questions:

- What are the key characteristics of a cost-minimized reverse supply chain infrastructure for recovering critical materials from EOL NiMH batteries?
- To what extent can a systematic reverse supply chain for critical materials from EOL NiMH batteries improve resource security by offsetting U.S. dependence on primary material imports?
- What are the cumulative costs, energy, and emissions associated with critical material recovery from such a large-scale vehicle NiMH battery-recycling infrastructure?
- What are the associated cradle-to-gate primary energy, material, and GHG impacts associated with the overall critical material reverse supply chain infrastructure?

8.2 Methods

To address the questions outlined in the previous section, we estimate the cost, energy, material flows, and GHG impacts associated with recycling EOL NiMH batteries to boost the United States supply security of critical (rare-earth) materials. Our approach accounts for stocks, time-resolved availability, and the logistics of procuring EOL NiMH batteries from battery collection to the final recovered materials. Figure 56 illustrates the analysis framework at a very high level, comprising three main components: the EOL battery distribution model, the recycling unit process models, and the RELOG model. We define a simulation window from 2025 to 2050, and we assume that all plants operate continuously for the entire period.

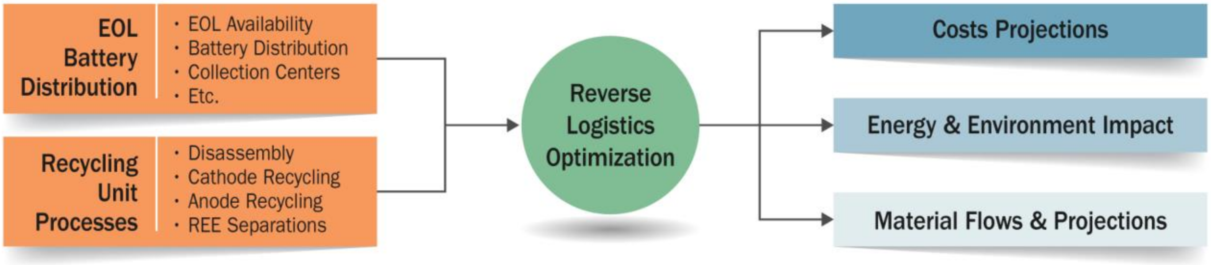


Figure 56. Analysis model framework for the NiMH battery-recycling case study

8.2.1 Recycling Process Model

Figure 57 presents a simple flowsheet of the different components of the recycling unit processes: disassembly, cathode material recovery, anode material recovery, and the REE separations.

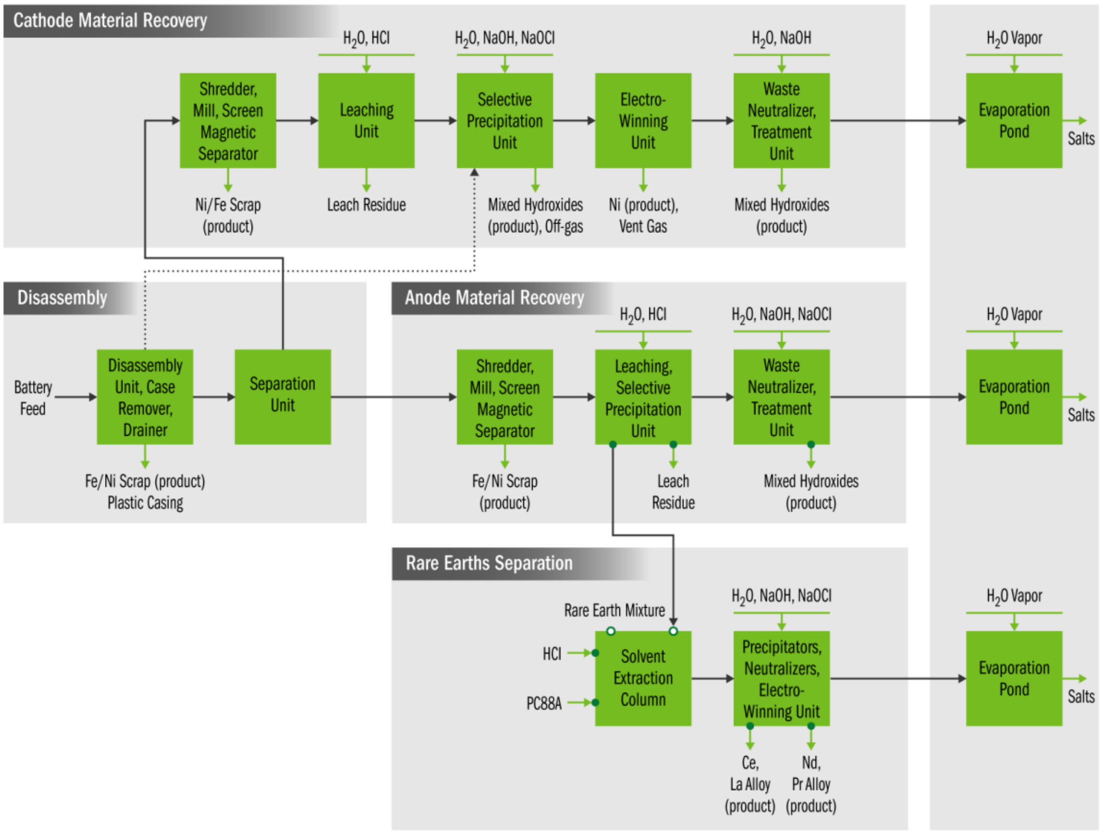


Figure 57. Flowsheet of overall battery recycling process model.

Process model shows the disassembly, cathode material recovery, anode material recovery, and REE separation components.

8.2.1.1 Process Model

The material balance model for the disassembly, cathode, and anode processing units is based on the "physical separation/chemical process" configuration described by Sabatini et al. (1994) and illustrated in Figure 57. Each solid box shown in Figure 57 represents a conceptual grouping of operations that may take place in a single piece, or in multiple pieces, of equipment. The disassembly unit accepts the NiMH battery and delivers cathode, anode, casing, and scrap material streams, as well as the electrolyte drain. In a collocated plant, the electrolyte potassium hydroxide (KOH) is reabsorbed downstream in the process to supplement sodium hydroxide (NaOH). The cathode processing unit contains several types of equipment ranging from magnetic separators to electrowinning tanks, recovering pure Ni, scrap metal, and some metallic hydroxide residues. The anode processing unit recovers more scrap metal, hydroxide residues, and an REE chloride mixture. Each of these units also includes equipment and infrastructure for waste treatment.

The REE separations facility houses the solvent extraction column and post-extraction equipment, including precipitators, electrowinning units, and waste treatment infrastructure. The solvent extraction column receives the REE mixture and separates it into individual or mixtures of REE(s), depending on the separation target. For this study, we consider separation into didymium (Nd-Pr) and cerium-lanthanum (Ce-La) clusters. The electrowinning units were added to convert the recovered REE chlorides into higher-value REE alloys. The solvent extraction process model used in this study is described in earlier work (C. O. Iloeje 2019), with an open-source implementation available on GitHub (C. O. Iloeje 2019).

8.2.1.2 Economic Model

The economic model applies a bottoms-up approach that uses the sizing information from the process models to evaluate corresponding equipment costs (C_{equip}), capital costs (C_{capex}), and operating costs (C_{opex}) for each facility. The overall economic model follows the approach described in Peters, Timmerhaus, and West (2022) for capital engineering projects. Equations 9-1, 9-2, and 9-3 summarize the economic formulation for the plant costs. For each facility unit,

$$C_{equip} = \sum_i \left(C_{ref_i} \left(\frac{m_i}{m_{ref_i}} \right)^n * N_i * PPI\left(\frac{Year_{curr}}{Year_{ref}} \right) \right) \tag{9-1}$$

$$C_{capex} = C_{equip}\left(1 + f_{delivery}\right)\left(1 + \sum f_{directs} + \sum f_{indirects}\right) \tag{9-2}$$

$$C_{opex} = \sum_k f_k C_k + \sum_j m_j C_j \tag{9-3}$$

Where C_{ref_i} is the reference cost of the corresponding equipment i (Sabatini et al. 1994); $\left(\frac{m}{m_{ref}}\right)^n$ is the scaling parameter (Peters, Timmerhaus and West 2002), (Williams Jr. 1947) based on the ratio of actual to reference facility feed mass flow rates; N_i represents the number of units of equipment i; and the $PPI()$ term uses the producer price index (U.S. BLS 2020) to convert equipment prices from the reference year to the project's currency year equivalent. We determine the capital cost by multiplying the total equipment cost with factors that capture contributions from direct costs (e.g., equipment delivery and installations) and from indirect costs, (e.g., supervision and contingency). Appendix E, Tables E.3 and E.4 summarize these direct and indirect factors.

Operating costs include the summed contributions of designated variable costs (e.g., utilities, raw materials, labor) and fixed costs (e.g., taxes, insurance) as represented in Eq. 9-3. For the variable cost component, m_j represents the relevant quantity (e.g., solvents, process water) and C_j represents the cost per unit quantity. For the fixed costs, f_k represents the multiplier factor for each cost item (e.g., taxes), and C_k is their associated cost basis (e.g., fixed capital). Appendix E, Tables E.5, E.6, and E.7 summarize the relevant parameter values and cost factors used in determining the operating cost items.

8.3 End-of-Life Battery Distribution Model

This section describes the overall battery recycling network and associated information, including the battery chemistry (component materials), availability and distribution, and assumptions about collection and transportation.

8.3.1 Battery Chemistry

Appendix E, Table E.8 shows the battery composition adopted in this study, which we derived from several literature sources. These sources typically reported different compositions, reflecting technology variation and evolution (e.g., changing from metallic to mostly plastic casing). We also estimated an average battery weight of 46 kg by combining battery specification data from various sources for different battery models (CEC 2015), (Kelleher Environmental 2019), (Young, et al. 2013), (Davis and Boundy 2021), (INL 2017). Of course, individual battery weights may differ, as they depend on battery model, capacity rating, and, sometimes, reporting source.

8.3.2 Battery Availability

We developed a battery EOL stock and flow model to estimate the cumulative number of batteries available for recycling for each year within the time horizon of the analysis. The formulation builds on HEV vehicle sales projection data and then applies a discard probability and scrappage rate formulation to determine the final battery quantities. We obtained data for vehicle sales from Gohlke and Zhou (2020) and annual battery type for vehicles from Oak Ridge National Laboratory transportation energy market report data (Davis and Boundy 2021). The latter data were used to estimate the NiMH fraction of total HEV batteries up to 2019. Beyond 2019, we assume that HEV demand (sales) increases monotonically, following the overall linear trend from extrapolating the underlying data.

For NiMH batteries, we consider two possible future scenarios: conservative and optimistic. The conservative scenario projects a decline in NiMH HEV batteries, using a sigmoid function to extrapolate the trend of the NiMH fraction of batteries. This scenario represents the conservative limit on the current trend of Li-ion batteries replacing NiMH batteries in HEVs. The alternative optimistic scenario fixes the fraction of NiMH batteries at the 2019 value and maintains this proportion going forward. This scenario represents the optimistic possibility in which economic considerations or evolving technology keep NiMH batteries competitive with substitute technologies. While unlikely, it represents a useful upper bound for this analysis. Figure 58 illustrates both projection scenarios. For both cases, we assume that only 50% of EOL batteries become available for recycling, the rest going to competing pathways like remanufacturing and refurbishment for secondary use (including outside the transportation sector), export (via used car market), and simple disposal. This assumption keeps our overall projections for both scenarios on the conservative side.

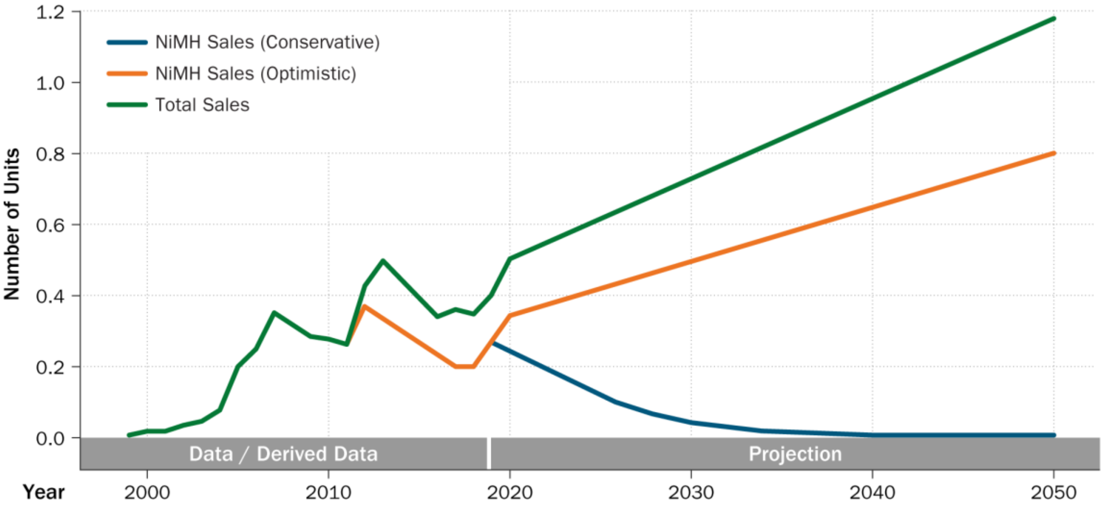


Figure 58. Battery sales data (to 2019) and projections (after 2019).

Note: Figure shows total HEV sales, conservative and optimistic projections for NiMH HEV battery demand/sales. Sales data from Gohlke and Zhou (2020).

8.3.3 End-of-Life Scrappage Flows

8.3.3.1 Scrappage Function

The vehicle scrappage probability density function (PDF), denoted by *f(t)*, where *t* is the age of the vehicle, is assumed to follow a logistic distribution based on similar formulations in the literature (Greene and Chen 1981), (Bandivadekar 2008), (Supekar and Skerlos 2017). The scrappage PDF takes the form shown in Eq. 9-4, wherein μ is the median lifespan of the vehicle in years, and *s* is a shape parameter proportional to the standard deviation of vehicle lifespans in years.

$$f(t) = \frac{e^{\frac{t-\mu}{s}}}{\sigma\left(1 + e^{\frac{t-\mu}{s}}\right)^2} \qquad (9\text{-}4)$$

The cumulative scrappage probability, *F(t)*, where *t* is the age of the vehicle, is therefore the integral of *f(t)*, as shown in Eq. 9-5.

$$F(t) = \int_{-\infty}^{t} \frac{e^{\frac{x-\mu}{s}}}{\sigma\left(1+e^{\frac{x-\mu}{s}}\right)^2} dx = \frac{1}{1+e^{-\left(\frac{t-\mu}{s}\right)}} \qquad (9\text{-}5)$$

Both μ and *s* are assumed to remain constant across the HEV model years in this study. The value of μ is set to 12 years following data from the U.S. Department of Transportation (DOT) (U.S. DOT 2019). The shape parameter *s* is set to 2 years to allow for a larger spread in the distribution following vehicle survivability data from the DOT (Lu 2006). Substituting these values for μ and *s* in Eq. 9-5, the cumulative scrappage probability is expressed in Eq. 9-6. Figure 59 shows the PDF and its corresponding cumulative distribution function.

$$F(t) = \frac{1}{1+e^{-\left(\frac{t-12}{2}\right)}} \qquad (9\text{-}6)$$

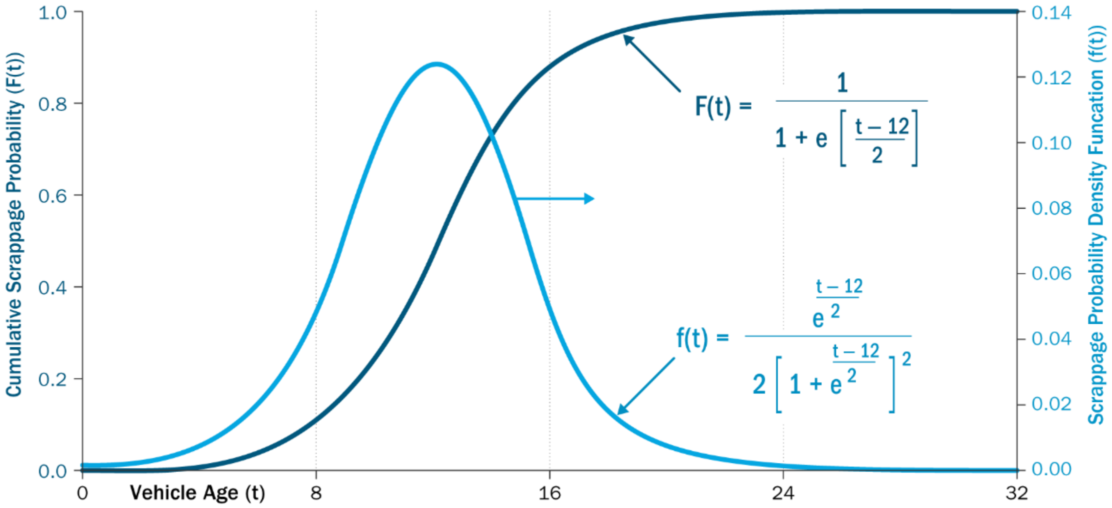


Figure 59. Battery scrappage (light blue) and cumulative scrappage (dark blue) probability density functions

Note: Scrappage PDF and cumulative scrappage PDF are used to calculate availability of spent batteries for recycling.

8.3.3.2 Vehicle and Battery Scrappage Quantities

Although NiMH batteries in HEVs typically have a warranty of 8–10 years, evidence is scant that the batteries are indeed replaced after 8–10 years. Reports from accelerated battery testing (Bernard and Lippert 2015) and from the field (O'Dell 2014), (Tatarevic 2016), (Hybrid and EV Battery Repair 2018) suggest that most NiMH batteries last the lifespan of the HEV, ranging from 150,000 to 200,000 miles. Following these reports, we assume the scrappage rate of NiMH batteries in this study to be the same as the scrappage rate of HEVs. This assumption results in a conservative estimate of the number of NiMH batteries that would be available for recycling at the end of their life, as it does not include batteries that may be discarded and replaced with new batteries before the vehicle's EOL.

Given the quantity, *S(k)*, of annual sales of vehicles, the total annual scrappage of vehicles in a given year, *d(k)*, can be calculated in increments of 1 year, as shown in Eq. 9-7, where *j* is the index for vehicle age and *F(j)* is given by Eq. 9-6. The index *k* refers to a specific year within the analysis time horizon of *Y* years, and thus $k \in \left[1, 2, \ldots, Y\right]$. We also define the practical lifespan of a vehicle, *T*, for which $F(T) \approx 1$. For the values of μ and *s* chosen in this study, *T* = 32 years captures 99.995% of all vehicle scrappage and is therefore chosen as a practical lifespan for all vehicles.

$$(k) = \begin{Bmatrix} S(k)\, F(j) & : k = 1 \\ \sum_{j=1}^{min(k,T)} S(k-j+1)\, F(j) - \sum_{j=1}^{min(k-1,T-1)} S(k-j) F(j) & : k > 1 \end{Bmatrix} \qquad (9\text{-}7)$$

Note that *d(k)* is not cumulative; that is, it represents scrappage of vehicles that have been scrapped between year *k* and its previous year, and does not represent the totality of vehicles scrapped until year *k*. Further, *d(k)* accounts for all vehicles of all ages between 0 and *T* in Year *k* following from the scrappage probability function. Figure 60 plots the resulting scrappage projections evaluated from Eq. 9-7, and Table 9 summarizes the model notation.

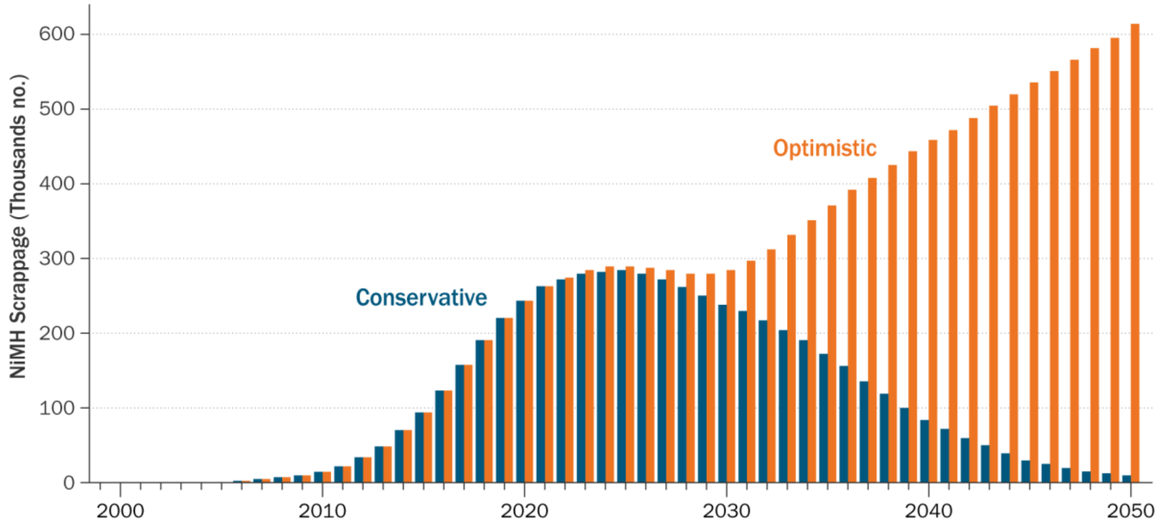


Figure 60. Projected NiMH EOL battery scrappage for EOL batteries.

Note: Subsequent analysis in this study assumes 50% of these quantities are available for recycling annually.

8.3.4 Battery Distribution

To estimate the quantities and distribution of EOL NiMH batteries across the continental United States at county resolution, we first multiply the annual scrappage projection from the stock-flow model with the normalized HEV sales data (equivalent HEV market share by state) to determine state-level quantities. Then, within each state, we use the major city/populated places data to approximate quantities of EOL batteries at the city level. Finally, county-level data are obtained by summing annual battery stocks from all the cities associated within the county. Figure 61 shows the geographical information system (GIS) plot of the resulting EOL NiMH battery distribution heat map, with counties in California indicating the highest EOL battery intensities. Figure 62 shows the 10 continental U.S. states with the highest EOL battery quantities. These Top 10 states account for 75% of the HEV market share, with California accounting for over 45%.

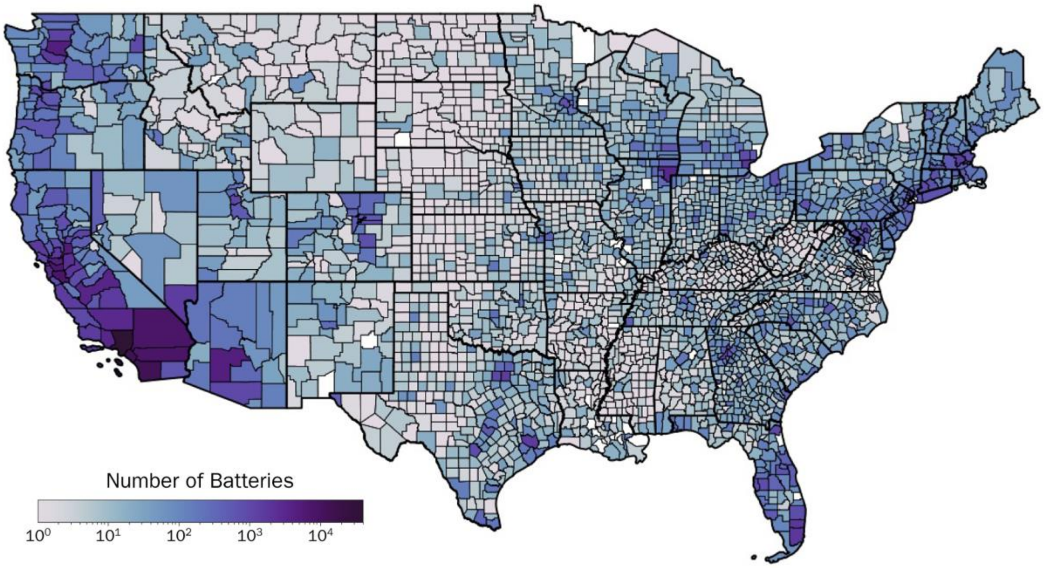


Figure 61. GIS plot of the distribution of EOL HEV batteries across the continental United States on a county-level resolution

Note: This information is provided as input to the logistics optimization model.

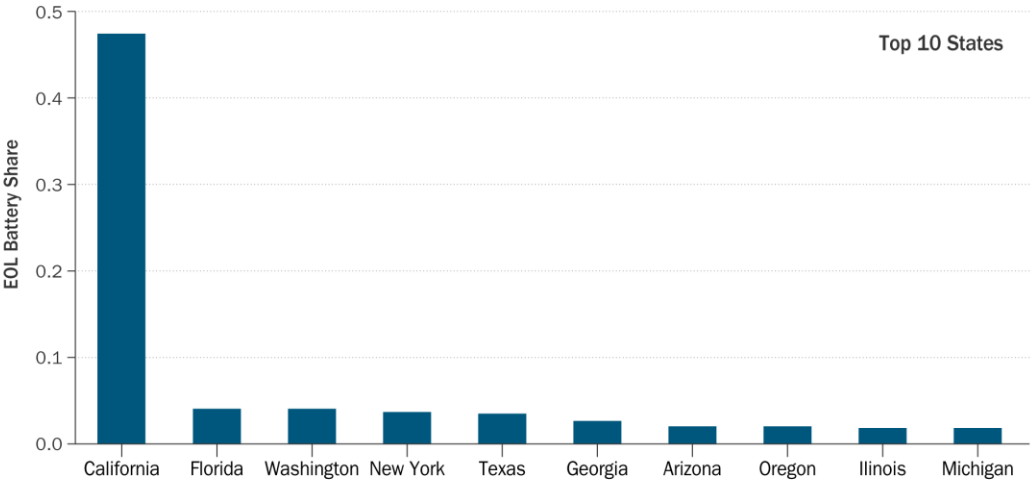


Figure 62. Share of EOL NiMH batteries for the Top 10 states

8.3.5 Collection and Transportation

The first step in the NiMH battery reverse logistics is collecting batteries. Based on the assumption that most HEVs are in urban areas, collection centers are placed in cities with populations greater than the set threshold (we used 10,000 people for this study). Collection centers may represent vehicle graveyards or mechanic shops or dealerships that collect EOL HEV batteries when individual consumers replace their battery or dispose of their old vehicles. For the reference case, we assume that only a fraction (β) of the EOL batteries are available for recycling. We estimate the amount of batteries arriving at each collection center using Eq. 9-8, where $N_{batt,i}$ is the number of batteries at each collection center i, $N_{tot\ batt}$ is the total number of available EOL batteries, S_{state} is the corresponding state-level HEV market share, Pop_i is the population of the given collection center's city, and $StatePop_{city}$ is the total population of all cities in the state. Figure 63 shows the resulting distribution of the battery collection centers, where the marker shade reflects the relative number of batteries at the collection center.

$$N_{batt,i} = \beta * N_{tot\ batt} * S_{state} * \frac{Pop_i}{StatePop_{city}} \tag{9-8}$$

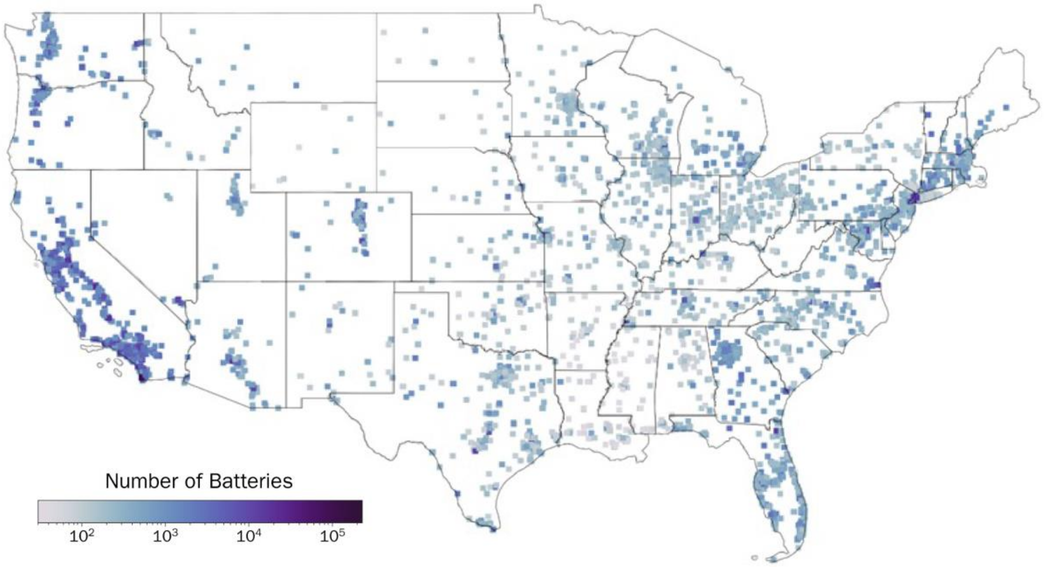


Figure 63. GIS plot of the distribution of HEV battery collection centers

Note: U.S. city population threshold is set at 10,000. The marker shade reflects the relative number of batteries in each location.

Because batteries available in the collection centers mirror the EOL battery distribution, collection centers in California handle the bulk of the available batteries, with 7 of the largest 10 collection centers in the state. The states with fewer collection centers correlate to states with lower HEV market shares and population densities.

8.3.6 Data Sources and Assumptions

For this study, we obtained geospatial data from various sources. These include GIS data for the U.S. highway network, 2010 census data for each U.S. city and state, statewide EV sales data to determine the proportional distribution of batteries by state, and location-based capital project cost factors (U.S. DOT 2020), (U.S. DOC 2020), (U.S. Census Bureau 2020) based on U.S. Department of Defense numbers for normalizing project costs across states (U.S. ACOE 2019).

8.4 Reverse Logistics Optimization

To determine the most cost-effective reverse logistics pipeline for recycling NiMH batteries, we solved a large-scale, mixed-integer, linear optimization problem. Our implementation has been made publicly available as RELOG. Although the package was developed during the course of this study, it can be used more broadly. It supports customized reverse logistics pipelines with multiple types of plants and products and multiple time periods, with storage and plant expansion. In this subsection, we describe the use of RELOG in this case study and the modeling assumptions, and we present a simplified mathematical formulation of the problem.

8.4.1 NiMH Recycling Pipeline

In RELOG, reverse logistics pipelines are described by two main model components: products and materials, and processing plants. In this case study, we have 14 types of products and materials: NiMH batteries; cathode; anode; nickel (Ni); nickel-iron (Ni-Fe) scrap; iron-nickel (Fe-Ni) scrap; mixed hydroxides; mixed rare-earth chlorides; didymium (Nd-Pr alloy); cerium-lanthanum (Ce-La) alloy; mischmetal;[65] leach residue; salt; and plastic pack. NiMH batteries are the source materials. Cathode, anode, and mixed rare-earth chlorides are intermediate products, and the remaining products are final products.

We have four types of processing plants: disassembly plants, anode material recycling plants, cathode material recycling plants, and rare-earth separation plants. In the RELOG, plants receive a single type of material and convert it into multiple other materials. Figure 64 illustrates the inputs and outputs for each type of plant in this case study. For example, disassembly plants receive NiMH batteries as input and produce anode, cathode, iron-nickel scrap, and plastic packs.

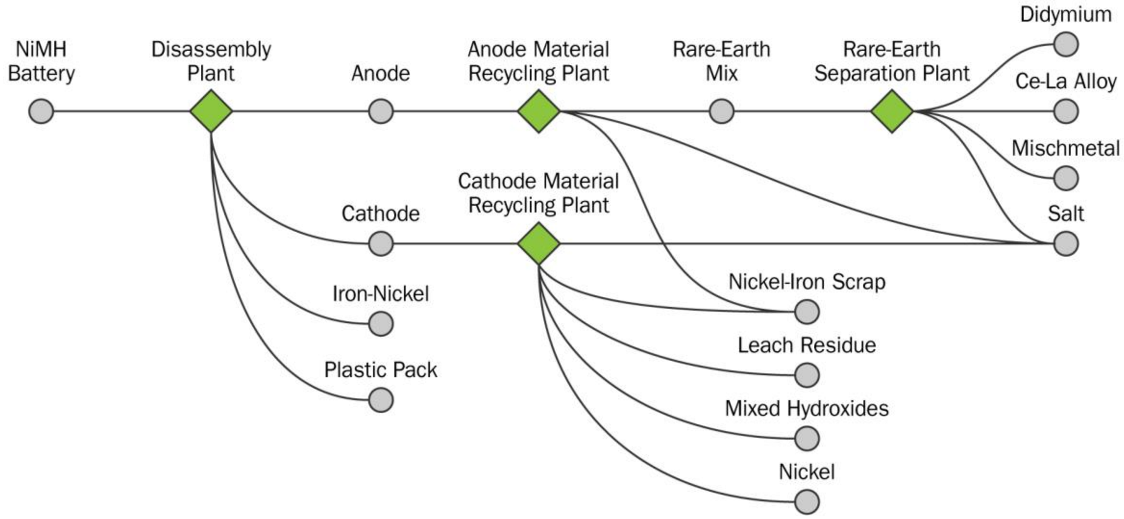


Figure 64. Reverse logistics pipeline for EOL NiMH battery recycling

8.4.1.1 Modeling Assumptions

RELOG assumes source materials (NiMH batteries, in this case study) are initially available at specific locations known as collection centers, described by their latitude, longitude, and the amount of material available per year, in metric tons. All remaining products become available only during the recycling process. Transporting products from one location to another incurs a transportation cost ($/km/MT), spends energy (Joule [J]/kilometer [km]/MT), and generates emissions (MT/MT). In this study, we considered only CO_2, but RELOG can track other types of emissions if data are provided.

To process each metric ton of material, the model assumes plants incur a variable operating cost ($/MT), spend energy (GJ/MT), and produce emissions (MT/MT). Plants also incur a fixed operating cost ($) per year regardless of the amount of material they process, as long as they are open.

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[65] Mischmetal is an alloy of rare-earth elements.

Plants can be built at specified candidate locations. Opening a plant incurs a one-time opening cost ($), which is region-specific. Plants also have a limited capacity (in metric tons), which indicates the maximum amount of input material they can process per year. For each candidate location, we have also specified the minimum and maximum capacities of the plants that can be built at that location. Different plant sizes have different opening costs and fixed operating costs. After a plant is built, it can be further expanded in the following years, up to its maximum capacity. See Appendix E, Figure E.1 for candidate locations considered for this analysis.

Products received by a plant can either be processed immediately or stored for later processing. Plants have a maximum storage capacity (MT) and incur storage costs ($/MT).

All products generated by a plant can be either sent to another plant for further processing or disposed of locally for a cost ($/MT). To model environmental regulations, it is also possible in RELOG to specify the maximum amount of each product that can be disposed of at each location; we did not use such functionality in this case study.

The detailed mathematical formulation can be found in RELOG's documentation on the GitHub repository (Xavier and Illoeje 2020).

8.5 System-Level Estimates

8.5.1 Wedge Analysis Model

The wedge analysis focuses on Nd/Pr and Ni metal saleable products from the recycling complex, for which the United States currently has no production capacity. (The results of the wedge analysis are presented in Section 8.6.2, Figure 68 and Figure 69.) The intent is to assess the extent to which the NiMH recycling complex could provide a domestic source of these metals that are critical for clean energy technologies. Table 10 provides a comparison of the maximum recovery (occurring in 2025) of these alloys from NiMH recycling with estimates of their U.S. production and demand.

The primary applications for Nd/Pr are for the manufacture of NdFeB magnets and other alloys. The United States currently has manufacturing capacity for bonded, but not for sintered, NdFeB magnets used in clean energy technologies. Bonded magnets are produced by injection molding, contain less Nd, and are less magnetic than sintered magnets, which are produced via powder metallurgy. Current U.S. demand for Nd/Pr is estimated from information in the Adamas report (Castilloux 2016). A step-up in U.S. demand could, with significant uncertainty, occur if the sintered NdFeB magnet manufacturing equipment formerly owned and operated by Hitachi Metals America, LTD, in North Carolina, and purchased by RareEarth LLC, is restarted (Althaus 2020). With a reported capacity of 2000 MT/year NdFeB and assuming a 30% Nd/Pr content, demand for the Nd/Pr alloy for such a plant would be on the order of 600 MT/year.

Applications for Ni metal include stainless steel, other steel alloys, superalloys, electroplating, catalysts, and chemicals (USGS 2020). Demand for Ni is expected to significantly increase as global production of LiBs for EVs steadily expands (Taylor 2019).

Table 10. Maximum Recovery of Ni and Nd/Pr Metals from NiMH Recycling and their Annual U.S. Production and Consumption

Alloy	Maximum Recovery (Year 2025, MT)	U.S. Production MT (year) (source)	U.S. Consumption MT (year) (source)
Nd/Pr	45—conservative 102—optimistic	No production of separated REO 43,000 MT REO concentrate production in 2021 (USGS 2022a); REO concentrates separated and refined in China (2019) Note: Mountain Pass Mine owner reports plans to build separation plant, with NdPr production capacity of 6,075 MT NdPr oxide (MP Materials 2022)	UNCERTAIN: 280 MT (2016) (Castilloux 2016)
Ni	420—conservative 940—optimistic	No production of separated Ni 18,000 MT of concentrate shipped to smelters in Canada and overseas for Ni production (2021) (USGS 2022b)	82,000 MT primary Ni + 110,000 MT recycled Ni, which includes Fe/Ni alloys (2021) (USGS 2022b)

Because the U.S. demand for the recovered products is significantly higher than the tonnages recoverable from the NiMH recycling complex, we instead compare the annual recovered tonnages to the tonnages of Nd/Pr and Ni metals in the United States sales of HEVs containing NiMH batteries.

8.5.2 Environmental Impact Assessment

We calculate the environmental impacts of battery collection and recycling LCA. The functional unit used for the analysis is 1,000 kg of NiMH battery scrap input (composition shown in Appendix E, Table E.8) to the recycling facility. Given the research focus of the study, we choose the ReCiPe-H (Huijbregts, et al. 2016) impact assessment method to determine the nonrenewable energy use (kg oil eq), GHG emissions (kg CO_2 eq), and abiotic material depletion (kg Cu eq) resulting from collecting and recycling 1,000 kg of spent NiMH batteries. Other impact metrics in the ReCiPe method associated with air and water pollution, ecotoxicity, human toxicity, and water use, while important in the context of battery recycling, fall outside the scope of this analysis and are therefore not presented.

Material and energy inputs to the recycling process (shown in Appendix E, Table E.9) are developed using the process flow described in Section 8.2.1 (Figure 57). Life cycle inventories for material and energy inputs are obtained from the ecoinvent 3.5 database.[66] Where data for exact or proprietary materials were unavailable, suitable substitutes were used (as shown in Appendix E, Table E.10). The electricity mix for locations of the recycling process is calculated as a recycling production-weighted average of the North American Electric Reliability Corporation (NERC) electricity mix for the locations of the recycling facilities. The RELOG model determined the recycling facility locations and their production quantities used to calculate this average mix. Note that the NERC regional electricity mix

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[66] https://www.ecoinvent.org

itself is an electricity production-weighted average of various fossil, nuclear, and renewable electric power sources within a particular NERC region for the year 2018 based on eGrid data (U.S. EPA 2020).

Transportation environmental impacts from moving spent NiMH batteries from collection centers to storage, and ultimately to the recycling facilities, are calculated assuming a 16–32 MT diesel-powered, long-haul freight truck and using the average distance traveled by 1,000 kg of spent battery scrap over the analysis time horizon. This average value is calculated by dividing the total km of transportation distance from 2020 to 2050 by the metric tons of spent batteries collected during that same period as determined by the RELOG model. Appendix E, Table E.14 summarizes the transportation intensity data.

For additional context and for comparison with other literature estimates for primary and recycled production of relevant materials, we also calculate the environmental impacts on an output basis using a revenue-based economic allocation method. We assume that all products with economic value that are generated from the recycling process are sold in their entirety at the prevailing price for those products (listed in Appendix E, Table E.2). The environmental impact, e_{ij}, in impact category *i* per kg of product *j* is then calculated using Eq. 9-9. E_i is the environmental impact per 1,000 kg of spent NiMH battery in impact category *i*; q_j is the amount of product *j* obtained per 1,000 kg of spent NiMH battery; and p_j is the market price of product *j*.

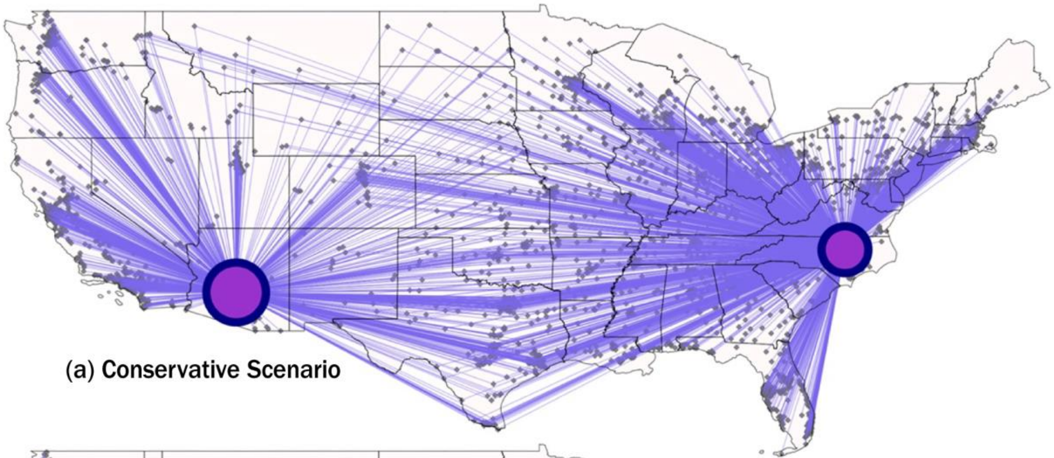
$$e_{ij} = E_i \left(\frac{p_j}{\sum_j p_j q_j} \right) \tag{9-9}$$

The ReCiPe-H assessment method captures two impacts of AMD. One is the relative abundance of a particular abiotic mineral resource in the earth's crust (ultimately a recoverable resource). The second is the impact of continued extraction of the mineral on its ore grade in a metric called the absolute surplus ore potential (ASOP). The ratio of the ASOP of a given mineral to the ASOP of Cu is then expressed as the mineral resource scarcity of that mineral in kg Cu eq. Although this metric provides useful quantitative insights into the consumption of an abiotic resource, it does not capture the supply vulnerability or economic importance of a particular mineral resource. Therefore, we supplement the mineral resource scarcity calculation with an estimate of the total mass of certain key groups of abiotic materials used during the life cycle of the spent NiMH battery recycling process. These groups include select critical minerals, select platinum group metals, and select ferrous and nonferrous minerals. The estimates of the total mass of these minerals are then normalized by their per capita U.S. consumption obtained from USGS data (U.S. Geological Survey 2019) for additional context.

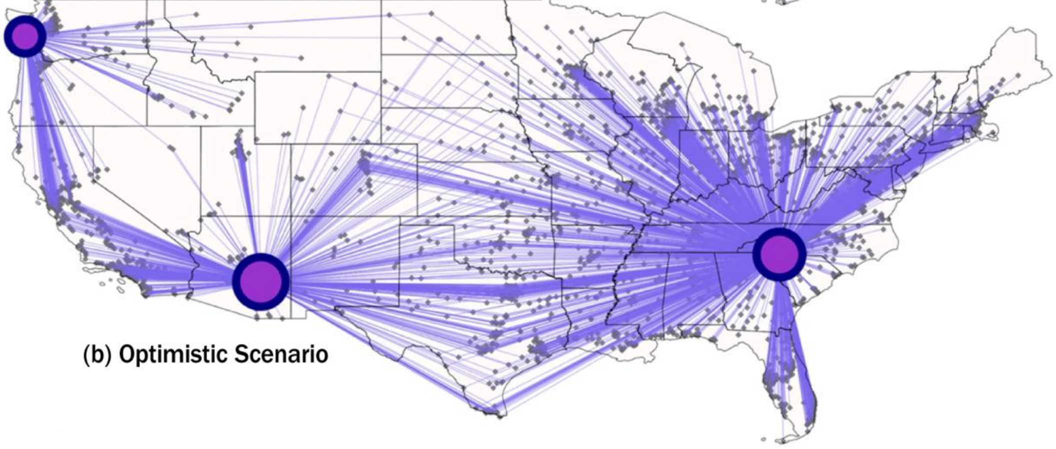
8.6 Results and Discussion

8.6.1 NiMH Battery End-of-Life Reverse Logistics Complex

We start this analysis by determining a cost-optimized network that describes the logistics of procuring and recycling EOL NiMH batteries, including the sizes and locations for processing facilities. This network informs the subsequent analysis to characterize the material flows, cost, energy, and GHG impacts and trade-offs for securing the supply of critical REE materials for the United States. Figure 65 shows outcomes for both the conservative and optimistic scrappage scenarios. As discussed in the methods section (Section 8.2), we assume for each case that only 50% of projected battery scrappage for each year is available for recycling. Figure 65 visualizes the resulting reverse logistics complex for NiMH battery recycling across the continental United States.



(a) Conservative Scenario



(b) Optimistic Scenario

Figure 65. GIS plant plot illustrating the optimized recycling facility locations for the Conservative Scenario (a) and the Optimistic Scenario (b).

Note: Each facility includes disassembly, recycling, and REE separation. The marker size reflects relative amounts of materials processed. The gray dots represent EOL battery collection centers, and the lines show battery transport to processing facilities.

The recycling complex in Figure 65(a) and (b) results from minimizing the total facility and transportation costs in the 2025–2050 analysis of the two selected scenarios. The transportation routes between the plants are straight-line approximations. In Figure 65(a), we see two major facilities on the West Coast (Arizona) and East Coast (North Carolina), with about 62% of batteries processed in the Arizona plant. For the optimistic case, a third plant is required to accommodate the increased material flow, with a resulting redistribution of material flows from collection centers to processing plants. Note that the specific processing facility locations depend on the assumptions and constraints prescribed for this study. Our objective is not to determine where plants ought to be built, but to base our economic and environmental impact analysis on a rationally designed battery recycling complex. Keep in mind that this infrastructure does not yet exist in the United States, and eventual build-out must consider alternative recycling technologies with smaller waste streams and land-use footprint.

8.6.2 Secondary Supply of Critical Materials from Spent NiMH Batteries

Besides critical Di metal and high-grade Ni metal, other significant recycled battery material-derived flows include mixed hydroxides (e.g., Co(OH$_2$), Ni(OH$_2$)), mixed metal scraps (Ni-Fe and Fe-Ni),[67] and waste streams: salts, leach residue, and plastics. The Sankey diagram in Figure 66 illustrates the material flow breakdown for the battery material recycling process. Figure 67 shows the potential market value associated with each material class from disassembly to rare-earth recovery. We calculated the potential market values by multiplying the total recycled amount for each material by the unit price found in literature sources. (See Appendix E, Table E.2 for reference data.)

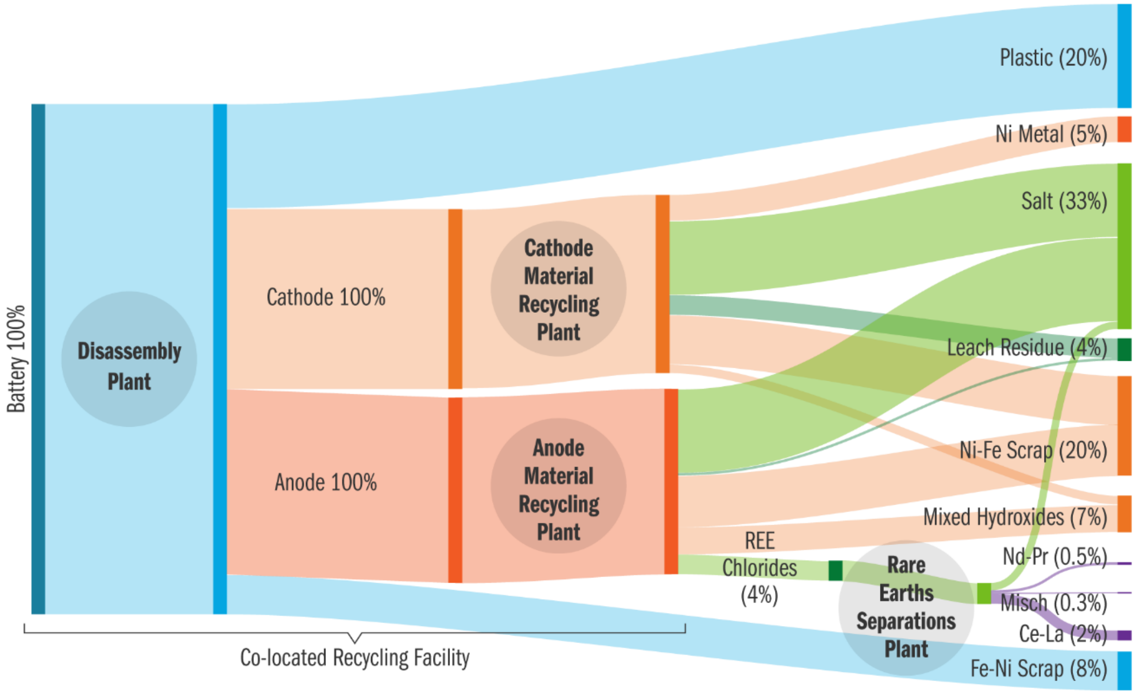


Figure 66. Sankey diagram of the material flows per unit battery (excluding electrolyte) through respective processing facilities for the HEV battery recycling process

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[67] Ni-Fe and Fe-Ni both represent mixed Ni-iron blends. Ni-Fe represents high-Ni blends, whereas Fe-Ni represents high-Fe blends.

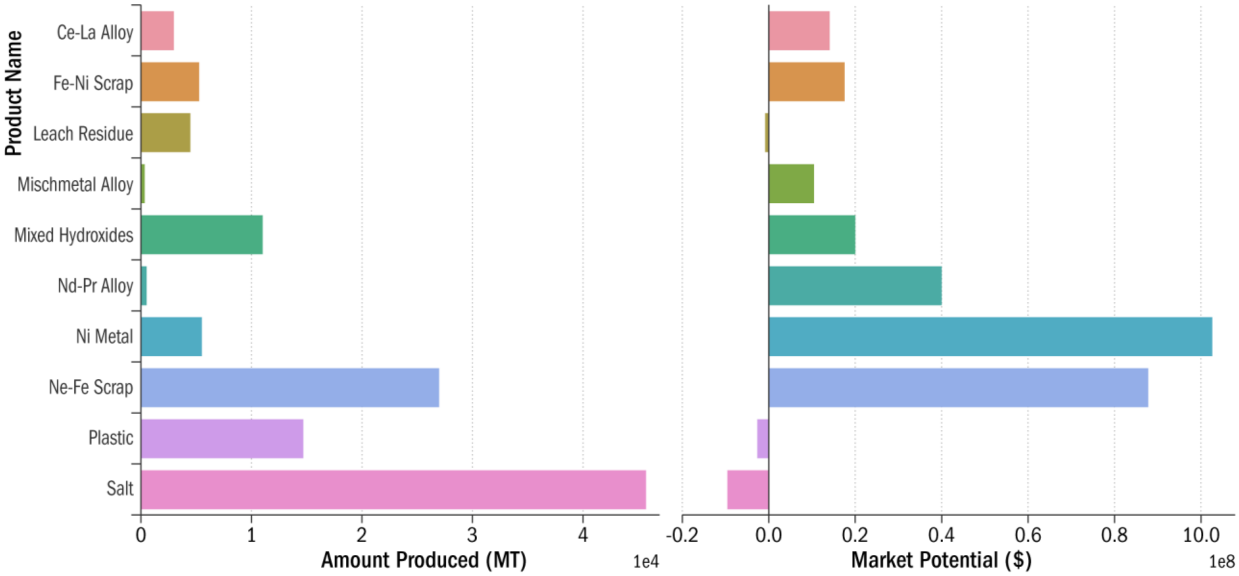


Figure 67. Recycled product amount and market potential cumulative over 2025–2050

Note: Landfilled materials accrue disposal costs (negative market potential).

While accounting for less than 1% of the total recovered materials, Di represents a disproportionate fraction of market potential (~15%). Similarly, Ni accounts for 5% of total flow and about 35% of potential market value. While the combined Ni-Fe and Fe-Ni scraps represent a large proportion of both total material flows (28%) and market potential (~30%), the value recovery from scrap markets is far less certain because of the oversupply of recycled scrap in the iron and steel industry. According to the USGS (Tuck 2020), iron and steel scrap are sourced from EOL automobiles, structural steel, appliances, rebar and reinforcement steel, and steel packaging, with current U.S. recycling rates of 100%, 98%, 88%, 71%, and 70%, respectively. Consequently, the recycling facility may have to sell these metals at below market value to compete and accommodate the costs associated with long-term storage. The two major waste streams—salts and plastic from casing—account for over 50% of total flows and underscore the importance of the associated environmental and land-use impacts (e.g., from evaporation ponds and landfills).

Figure 68 and Figure 69 show material wedge model results. These diagrams show the potential of recovered products from the NiMH recycling complex to displace the Di (Figure 68) and Ni (Figure 69) metals consumed in the production of NiMH batteries for HEVs sold in the United States. In the conservative scenario, Di and Ni metal recoveries from the recycling complex far exceed their contents in HEVs sold for most years. The opposite is true for the Optimistic Scenario reflecting the assumption of significantly higher United States sales of HEVs, including those with NiMH batteries.

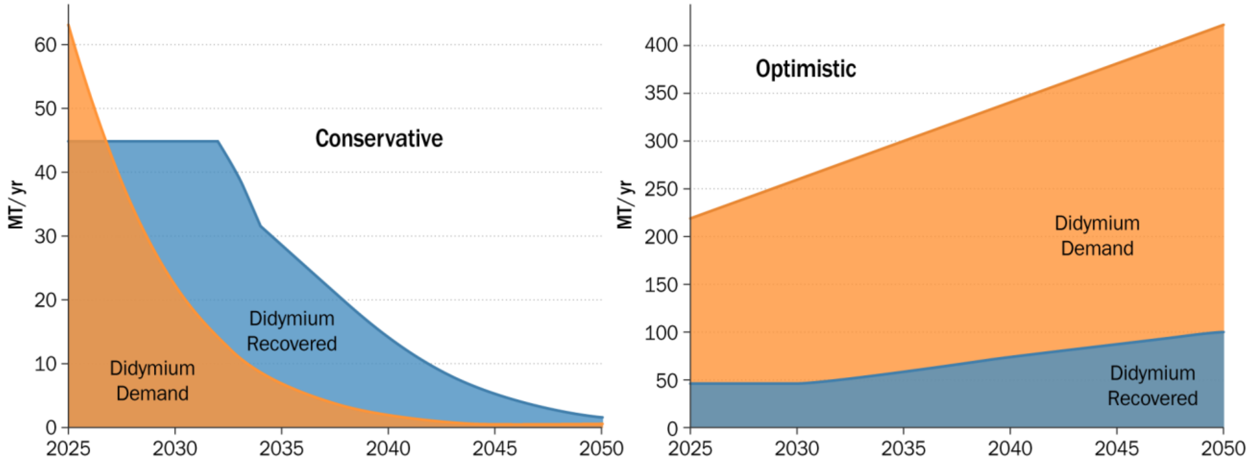


Figure 68. Didymium embedded in HEV NiMH batteries sold in the United States that could potentially be displaced by didymium recovered from the NiMH recycling for the Conservative and Optimistic Scenarios

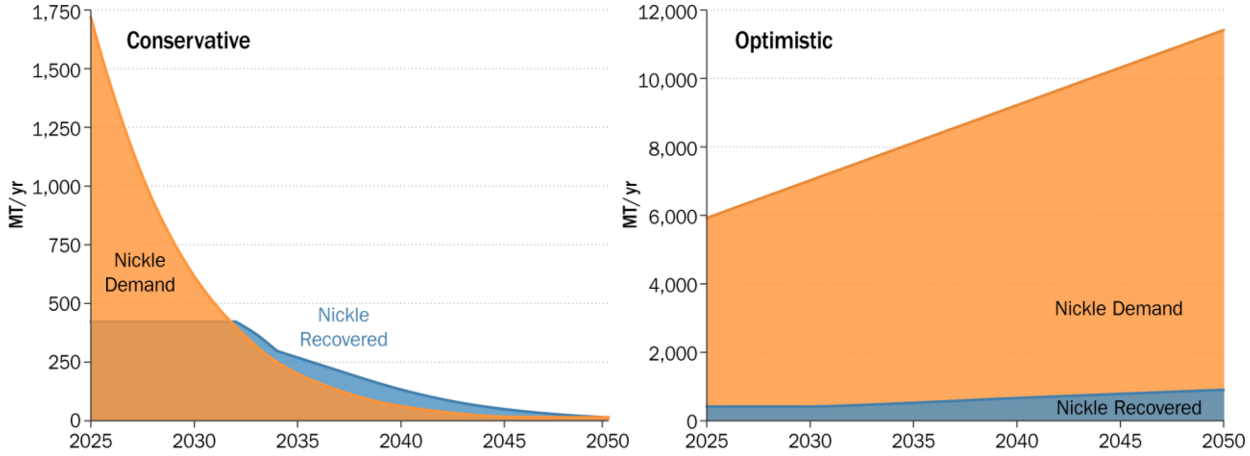


Figure 69. Nickel embedded in HEV NiMH batteries sold in the United States that could potentially be displaced by Nickel recovered from the NiMH recycling for the Conservative and Optimistic Scenarios

8.6.3 Economics of NiMH Battery Recycling

In Figure 70, we show the projected cost (per MT of battery processed) associated with the NiMH battery EOL reverse logistics network, averaged over the analysis window (2025–2050). Figure 70(a) highlights the major cost segments for the conservative scenario: capital (~17%), fixed operating (~23%), variable operating (~58%), and transportation (~2%). Figure 70(b) represents the same for the optimistic scrappage scenario, with variable costs comprising about 70% and transportation about 3%. Figure 70(c) shows the key contributors to the variable operating cost for the optimistic case. Core variable operating costs—primarily raw materials and labor—dominate variable costs (over 70%), with the rest coming from EOL battery acquisition, disposal, and storage costs.

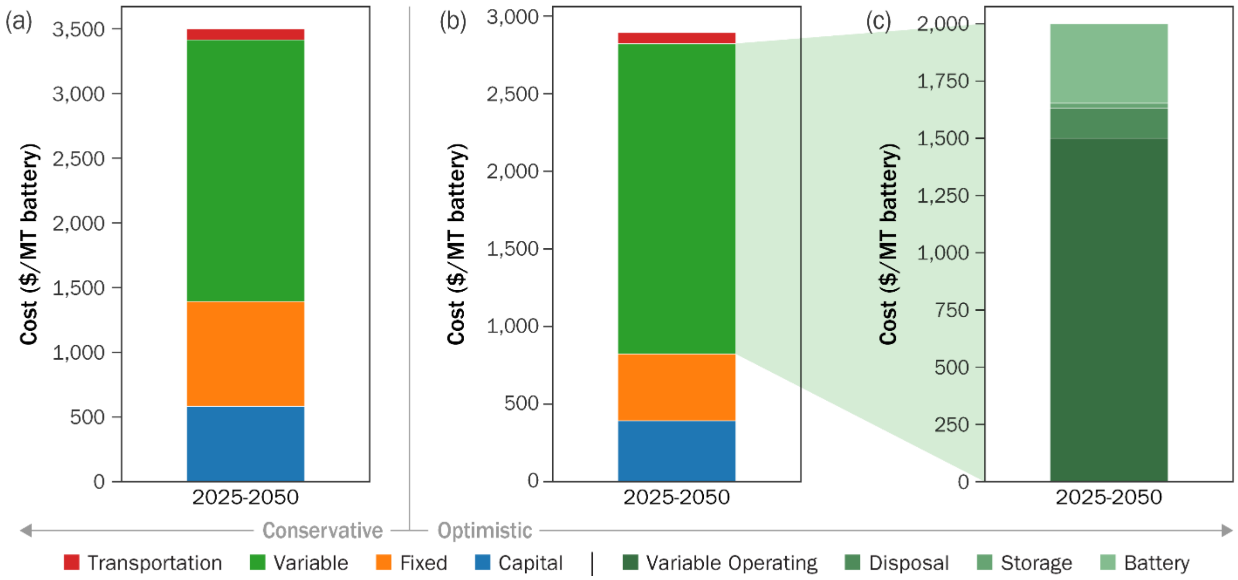


Figure 70. Averaged cost projection for the HEV NiMH reverse logistics supply chain per MT of battery.

Note: Including recycling facilities and transportation from 2025 to 2050. (a) Total cost showing major components, conservative scenario. (b) Total cost, optimistic scenario. (c) Variable operating cost breakdown (excluding transportation).

The total cost of recycling NiMH batteries to recover critical materials can be viewed several ways. One approach simply considers the average cost, determined by dividing the total costs over the analysis time frame by the number of batteries processed. This value comes to about $3,500 and $2,900 per MT of batteries processed for the conservative and optimistic scenarios, respectively, as shown in Figure 70(a) and Figure 70(b). The difference between the conservative and optimistic numbers captures an important contribution from the economy of scale; with larger volume of processing in the optimistic case, it can take better advantage of economies of scale to bring down specific recycling costs. Together, both scenarios provide reasonable bounds on the expected cost of battery recycling. Another perspective considers the marginal cost, which captures the cost of processing an additional MT of NiMH battery given the recycling infrastructure complex. This value depends on location and, given the peculiar features of NiMH battery availability in the United States, it also depends on the year in question. Figure 71(a) and Figure 71(b) show the selected locational marginal cost of recycling by state for both scenarios, showing variations of up to 8% across locations for any given year.

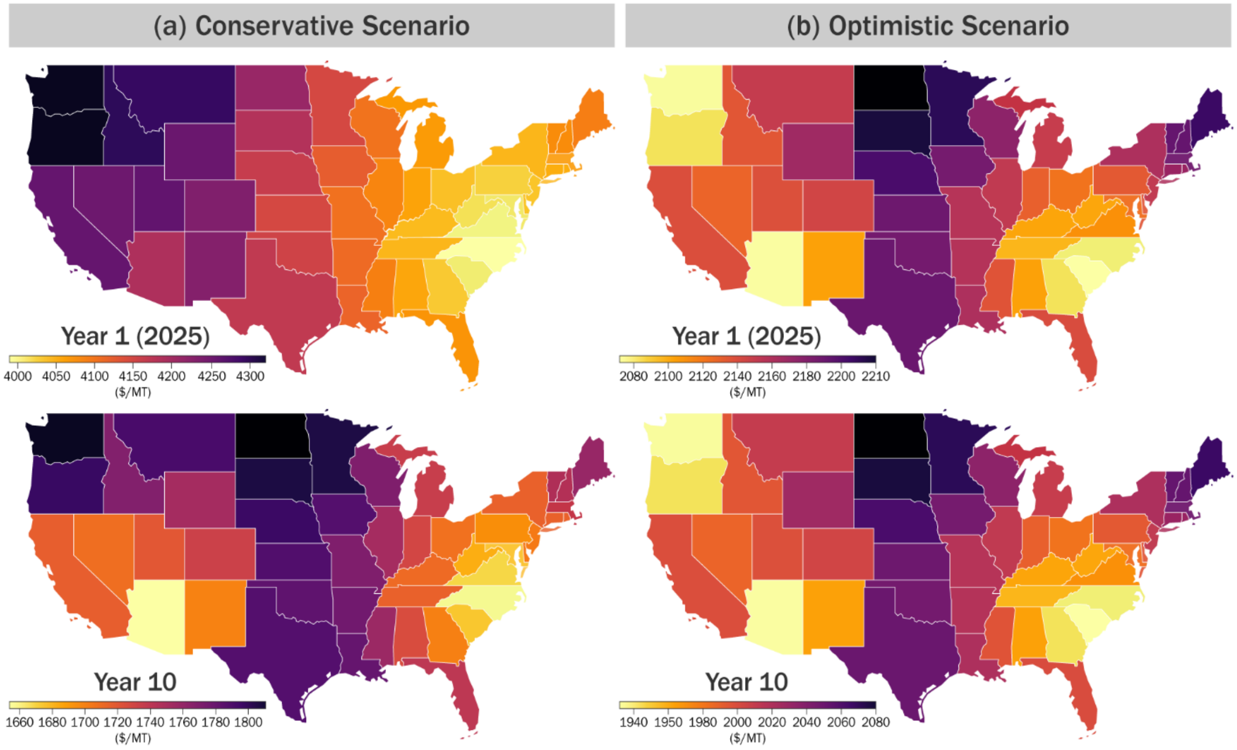


Figure 71. Location based marginal cost for selected years for Conservative Scenario (a) and Optimistic Scenario (b)

The cost predictions in Figure 70 and Figure 71 reflect the impact of the EOL battery availability profiles (shown in Figure 62, Page 100) and the annual processing constraint, which requires that the recycling facilities are sized to process all available batteries in any given year. We see from Figure 60 (Page 99) that for the conservative case, the quantity of EOL batteries is higher in the first years and decreases monotonically over subsequent years. So, the requirement that all available batteries in any given year be processed that same year results in an oversized system, with increasingly lower average utilization over time. While the variable costs drop with lower processing levels, the fixed costs, which depend on installed capacity, remain constant. Two factors limit the effect of this inadvertent oversize: first, the dominance of variable costs, compared to the fixed and capital costs over the 2025–2050 window (Figure 71); second, the practical decision to include storage, which allows the United States to open smaller plants and store excess material for subsequent processing. Thus, the additional cost of storage in the first 8 years largely explains why the marginal costs in Figure 71(a) are significantly higher in Year 1 compared to Year 10, where there is no storage.

8.6.4 Energy and Environmental Impacts

As shown in Figure 72(a), collecting and recycling 1 MT of spent NiMH batteries would lead to a global warming impact of 1,033 kg CO_2 eq, fossil resource use of 14,523 MJ_{ex} (MJ_{ex} defined in Table A.1), and metal resource use of 275 MJ_{ex}. The impacts are driven largely by the use of chemicals and reagents in the recycling process. Transportation of the spent batteries contributes to about 12% of the global warming impact. Environmental impacts of the recycling process are allocated to the end products (REEs, Ni, and scrap) using a revenue-based allocation (see Section 8.6.2 for details). The results, shown in Figure 72(b), indicate that compared to virgin production from Bayan Obo Mine in China based on Arshi, Vahidi, and

Zhao (2018), recycled neodymium, praseodymium, and cerium have a 67%, 84%, and 93% lower global warming impact, respectively. The corresponding results for the optimistic case are similar, as shown in Figure 72(c) and Figure 72(d), respectively.

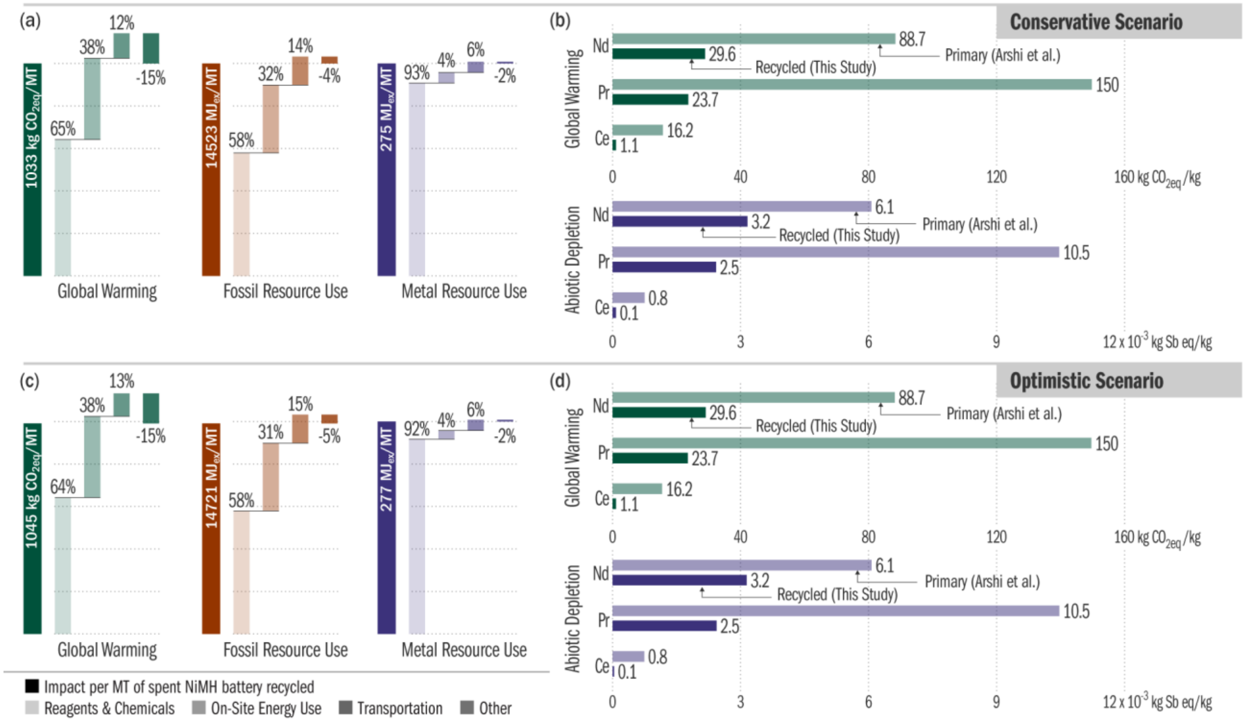


Figure 72. Environmental impacts for NiMH battery recycling

Note: (a) Life cycle environmental impacts of collecting and recycling 1 metric ton of spent NiMH batteries using hydrometallurgical process, conservative scenario; (b) environmental impacts per kilogram of recovered rare-earth metal and their comparison with impacts from virgin extraction based on Arshi et al. (2018), optimistic scenario; (c) life cycle environmental impacts of NiMH battery recycling, optimistic scenario; (d) comparative environmental impacts per kilogram of recovered rare-earth metal, optimistic scenario

Our results compare well with other battery recycling estimates in the literature. Excluding transportation, the ecoinvent process for hydrometallurgical recycling of Li-ion batteries, which assumes sulfuric acid as the leaching agent, has a global warming impact of 904 kg CO_2 per MT of spent battery (compared to 909 kg CO_2 eq). The GREET model, which assumes an organic acid and hydrogen peroxide as a reducing agent in its hydrometallurgical process, has a global warming impact of about 700–950 kg $CO_{2\text{ eq}}$. Note that the bench-scale hydrometallurgical process in GREET assumes a 90% acid recovery and reuse (Dunn, et al. 2012), which significantly reduces reagent use and therefore reduces the life-cycle GHG emissions.

8.7 Conclusion and Discussion

Hundreds of thousands of hybrid and mild-hybrid vehicles containing substantial stocks of NiMH batteries will be retired in the coming decades. However, the quantity of EOL NiMH batteries will diminish over time as Li-ion technologies replace vehicle batteries. We find that despite the declining EOL supply, recycling NiMH batteries in the United States could fill a substantial fraction of the country's demand for Di and supplement Ni supplies while reducing the cost and environmental impacts associated with their primary supplies.

This study assessed the feasibility and consequences of recovering critical rare earths and Ni from EOL NiMH batteries in the United States, accounting for stocks, time-resolved availability, and the logistics of procuring the EOL batteries. The analysis addresses questions concerning:

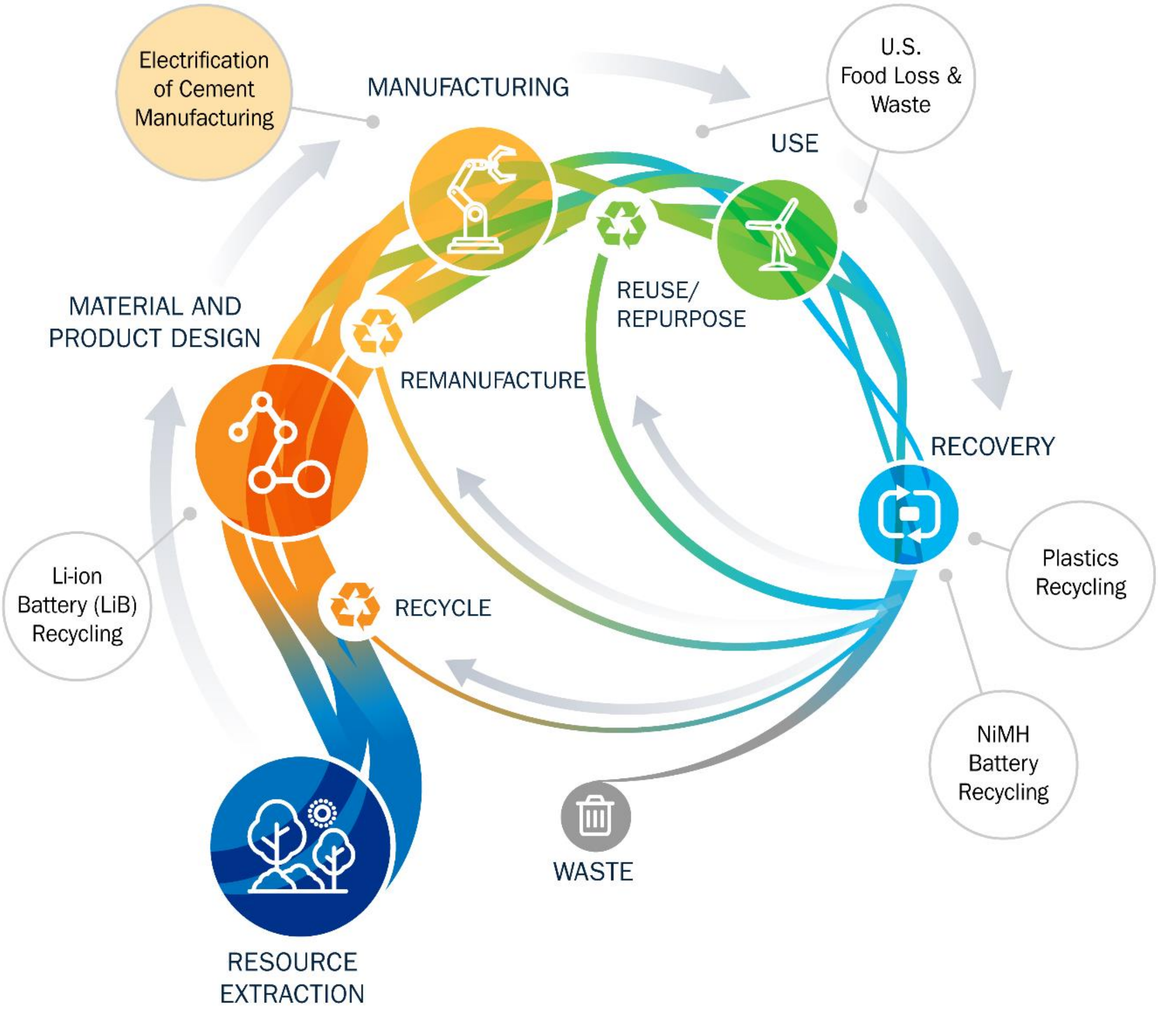
- The potential supply of critical materials from the NiMH reverse supply chain to offset primary imports
- The costs, energy use, and GHG emissions associated with this recycling complex
- How the environmental impacts compare with primary extraction
- Indirect impacts of NiMH recycling on the use of other strategic minerals and materials.

Spent NiMH batteries could act as a significant source of recycled REEs and Ni, producing 4.3–12.6 metric kilotons (KT) of REE product and 5.7–16.8 KT of Ni between the analysis time horizon of 2025–2050. The ranges reflect conservative and optimistic scenarios for the use of NiMH chemistry in HEVs in the future. Depending on the aforementioned scenarios for NiMH batteries in the future, two to three recycling plants with a recycling capacity of up to 5 KT/year of spent NiMH batteries would be sufficient to recycle all spent NiMH batteries (assuming 50% of spent batteries are recycled).

Given that the United States has no capacity for producing refined rare earths and refined Ni metals, domestic recycling can offset excessive dependence on imports for the supply of these materials, with the potential to reduce both the political and environmental burdens of primary supply. However, transporting and recycling 1,000 kg of NiMH batteries leads to a fossil depletion potential of 385 kg oil eq, global warming potential of 1,295 kg CO_2 eq, and an abiotic mineral resource depletion of 3.75 kg Cu eq. In all three categories of environmental impacts examined, chemical and reagent use in anode and cathode recycling and solvent extraction are the overwhelming contributors.

Important factors to consider in subsequent studies include uncertainty in EOL availability, regional regulatory constraints, and existing infrastructure. Further, the study should be extended to account for uncertain markets for recovered product, cash flow, and conditional value of risk analysis.

9 Case Study: Electrifying the Cement Clinker Production Process—Evaluating Impacts on Energy, Emissions, Materials, and of a Low-Carbon Electric Grid



9.1 Introduction

A circular economy envisions minimizing material waste and decoupling growth from the consumption of finite resources (e.g., materials, nonrenewable energy, fresh water). Electrifying thermal processes in industry by using low-carbon electricity, for instance, could help achieve this circularity with regard to the consumption of finite fossil fuels. Traditional thermal processes in industry typically derive heat from burning fossil fuel (e.g., coal, fuel oils, natural gas), resulting in the “leakage” of material from the economy in the form of unrecoverable fossil fuels. The fossil fuels used also must be mined, processed, and transported to the facility for use, adding to the energy and emissions footprint of using these fuel sources. Using low-carbon electricity directly for process heating tasks (e.g., melting, heating, drying) in the manufacturing sector is a potential pathway to decarbonize and achieve circularity in the use of coal and petroleum-based materials.

A few additional benefits are gained from electrifying thermal processes in industry. Electrotechnology systems could offer operational flexibility and regulate their system parameters to provide targeted heat to the product. This leads to heating the product more homogeneously, increasing the product quality significantly. Further, electromagnetic radiation interacts with different materials in unique ways. Electrotechnologies that make use of these radiations (e.g., microwave, radiofrequency, ultraviolet) may be used to develop entirely new or enhanced manufacturing processes, products, and materials. Given the unique characteristics of industrial thermal processes, the decision to replace them with electrotechnologies needs to be evaluated on a case-by-case basis by determining the holistic impact on process efficiency and productivity. The case study presented here aims to lay out the fundamental considerations for this comparison by evaluating the impact of electrifying the clinker production process in manufacturing Portland cement.

Various studies have evaluated the environmental impacts of Portland cement using cradle-to-gate LCA (Boesch, Koehler, and Hellweg 2009). An analysis by the Portland Cement Authority determined the impacts of producing cement using traditional fuel-fired kilns (PCA (Portland Cement Authority) 2016). A few studies have also explored the life cycle impacts of using alternative fuels (e.g., biological sludge) for the cement kilns and using alternative technologies (e.g., residue recycling) (Georgiopoulou and Lyberatos 2018), (Huntzinger and Eatmon 2009). A review of these LCA studies for estimating environmental impacts from cement production has also been done (Oluwafemi, et al. 2021). However, to the authors' knowledge, an analysis of electrifying major thermal processes in cement manufacturing using an LCA approach has not been conducted to date. Further, while the LCA literature on electricity generation technologies is enormous and has been synthesized in a meta-analysis by the National Renewable Energy Laboratory (NREL) (Burkhardt, Heath and Cohen 2012, Dolan and Heath 2012), a study integrating the impact of electrifying industrial thermal processes with renewable energy infrastructure needs has not been attempted. This case study will bridge this gap and put forth a comprehensive cradle-to-gate analysis model.

9.2 Overview of Cement and Concrete

With three metric tons per year used for every person in the world, concrete is the most heavily consumed material after water (Gagg 2014). The most important component and main binder for concrete is ordinary Portland cement. Cement acts as the binder between aggregates (fine and coarse rocks) in the formation of concrete.

The United States is the world's third-largest producer of cement. The U.S. industry produced 88.5 MMT of cement in 2019 from approximately 99 cement mills in 34 states, plus 2 plants in Puerto Rico. Approximately 10% of the cement used in the United States is imported (USGS 2020). Cement production is expected to grow further by an average of 1.2–1.6% annually through 2050 (EIA 2020a). However, factors such as economic growth, technological changes, population expansion, and labor productivity, among other macroeconomic variables, are often uncertain and can impact the potential production of cement in the future. To capture some of these uncertainties, this case study applies the average of two scenarios developed by U.S. Energy Information Administration (EIA) to estimate the projected demand. Figure 73 shows these two scenarios representing low economic growth and high economic growth.

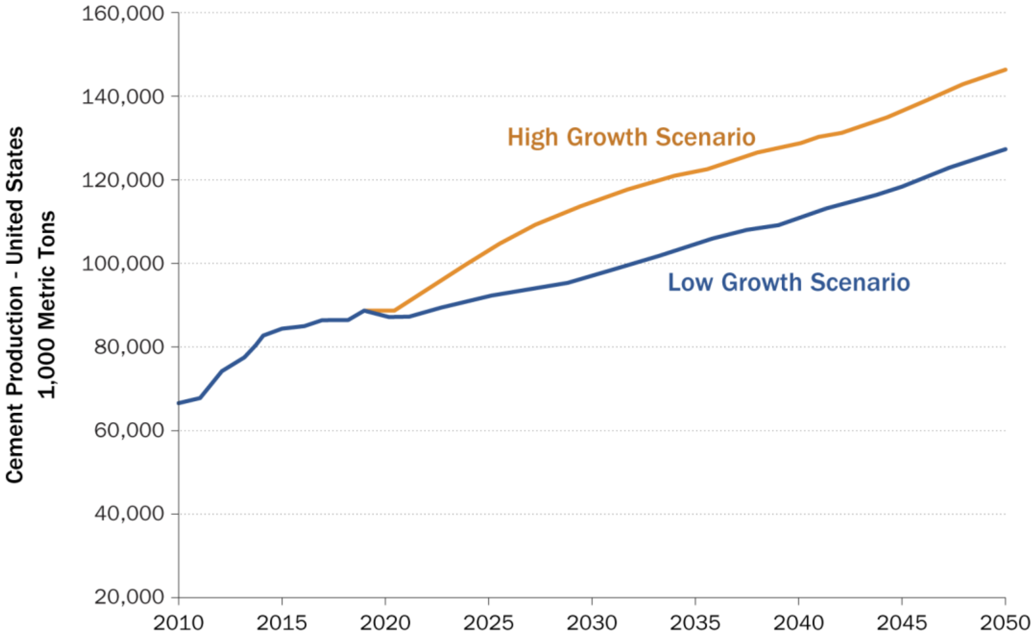


Figure 73. Projected U.S. cement production (thousand MT) for low- and high-growth scenarios

Note: Graph based on information in EIA (2020a).

Portland cement is a mixture of calcium silicates, calcium aluminate, and calcium aluminoferrite, usually with minor amounts of the oxides of sodium, potassium, and magnesium, as well as gypsum. The raw materials are typically sourced locally with the calcium usually coming from limestone; sometimes, shell or coral are used, depending on local availability.

In addition to being an extensively used resource, cement production is highly intensive in energy consumption and emissions. The U.S. cement industry used around 263 trillion British thermal units (TBtu) of heat from fuel combustion and 11,137 GWh of electricity in 2014 (U.S. EIA 2014). Figure 74 shows the share of different energy types used in the U.S. cement industry. Coal is the primary fuel in the U.S. cement industry; heat from fuel combustion accounts for 88% of total final energy consumption, while electricity use accounts for the remaining 12%.

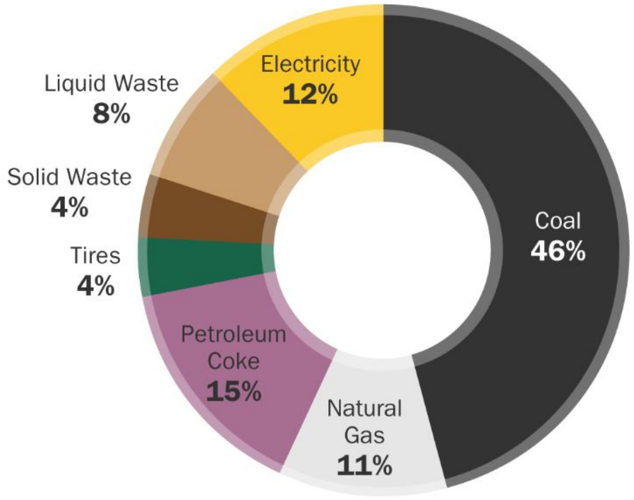


Figure 74. Breakdown of total energy consumed by the cement sector.

Note: Chart based on information in USGS (2016).

Because cement is an energy-intensive resource and national production is expected to increase, it is critical to examine the sector for material, energy, and emissions optimization. Various approaches may produce and use cement more sustainably. The potential pathways include the following:

- Energy efficiency/efficiency of energy inputs
- Use of carbon-free alternative fuels
- Electrification of thermal processes
- Carbon capture, utilization, and storage
- Alternative clinker materials
- EOL recycle and reuse.

Solutions involving the EOL reuse of cement/concrete and alternative material sources can help achieve greater material circularity (De Schepper, et al. 2014). On the other hand, innovative energy efficiency technologies, increased use of biomass fuel, and carbon capture techniques can help reduce emissions. Although these strategies have been discussed and analyzed in multiple studies and road maps (Czigler and Reiter 2020), electrification pathways that show promise to reduce emissions and energy consumption remain understudied. This case study reviews pathways for electrification and determines its holistic impacts on the finite resources used by the sector.

9.3 Case Study Goals and Scope

Manufacturing cement requires heating raw materials to high temperatures in a kiln in a fuel-intensive process known as pyroprocessing. The process results in clinker—small lumps of stony residue that are ground to a powder and combined with other ingredients to produce cement. The clinker production step which involves calcination (decomposition of limestone) and sintering (coalescence of cement clinker) accounts for 90% of the total energy consumed and 89% of the total CO_2 emissions (process- and energy-related emissions) associated with the cement manufacturing process (U.S. DOE 2017).

This case study aims to understand the environmental impacts of implementing innovative thermal heating solutions in the clinker production process. It explores two pathways for the electrification of the pyroprocess:

- **Pathway 1—Electric Calcining and Conventional Sintering**: A hybrid system that uses electric preheaters in combination with traditional fuel-fired rotary kilns to achieve clinkerization.
- **Pathway 2—Electric Calcining and Electric Sintering:** A completely electric solution for the thermal heating steps.

This case study seeks to understand the impact nationally from the widespread adoption of these two technologies. Specifically, exploring the trade-off between GHG emissions, material depletion, and energy consumption will help evaluate these pathways along with other alternatives. (Section 9.4.1 provides details about the proposed heating systems.)

Analysis of the environmental impact of electrification requires an unambiguous definition of its scope. This study considers a partial life cycle of cement from resource extraction (cradle), through processing, to the factory gate before the product is transported to the consumer. The “cradle-to-gate” scope thus defined considers all operations of the cement life cycle that would be impacted by changing the pyroprocessing step from a fuel-fired to an electricity-based system. Figure 75 lays out the life cycle of cement and indicates the operations excluded from the analysis with an asterisk.

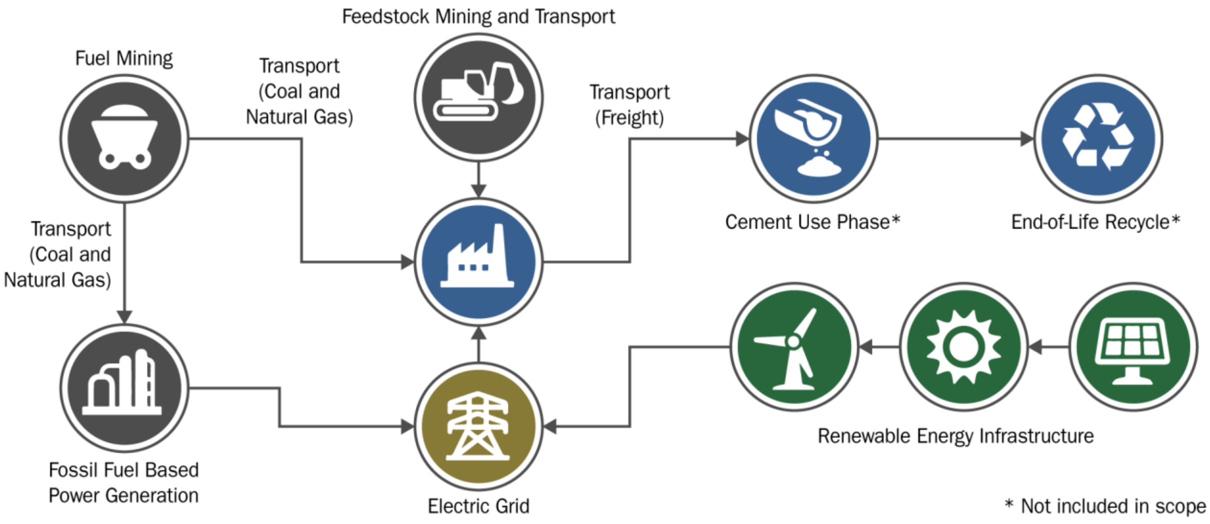


Figure 75. Flowchart of material and energy flow in the cement manufacturing process

Electrification has direct impacts within the boundary of the manufacturing facility in the form of reduced fuel consumption and increased electricity usage. These, in turn, indirectly impact multiple operations outside the facility boundary, including the mining and transport of fossil fuels and raw materials, electricity generation, and its transmission. The analysis considers these indirect impacts to provide a holistic comparison of the proposed electrification pathways with the current conventional scenario.

An analysis of the impact of electrification must account for a changing electric grid mix. The electrification of industries can help reduce reliance on fossil fuels only when low-carbon sources fuel the grid. The grid could be made “low carbon” by producing electricity using renewable sources, such as solar, wind, and hydropower. To generate renewable electricity, we need to produce photovoltaic cells, wind turbines, dams, and associated infrastructure. This study includes the materials needed to build this renewable energy infrastructure and compares them with the present-day grid materials.

Electrification does not typically affect the cement end product and has no negative impacts on the process. Product quality is expected to be similar if the mass flow rates and temperature profiles in the system are identical to, or kept very close to, the values in the fuel-fired system. Hence, the components involving the end product (cement), such as transport of the finished product and the product use phase, are not included in the scope of the study. Similarly, EOL recycling operations for cement/concrete are outside the scope of this work. Note that if electrifying the clinker manufacturing process impacted the quality of the cement product (e.g., increased lifespan, improved recyclability), the use phase of cement would need to be included within the LCA scope. This case study assumes the quality of the cement product is unchanged.

9.4 Cement Manufacturing—Baseline Material, Energy, and Emissions

This section presents the conventional cement manufacturing process and its energy use, material needs, and GHG emissions to form the baseline for analysis. Figure 76 shows the steps associated with cement manufacturing (Boldyryev 2018). The raw materials—predominantly limestone, sand, and clay from quarries (and other industrial waste streams)—move through a series of crushers and grinders to form the raw meal. In parallel, solid fuels such as coal and petroleum coke are prepared in a separate mill to be combined with the raw meal and fed into the kiln. A pyroprocessing setup consisting of cyclone preheaters and a rotary kiln achieves the calcination and sintering of the raw meal. In the preheaters, limestone starts to break down into its components (calcination), and the oxides of calcium and silicon begin to combine. The final heating occurs in the rotary kiln (sintering), where the material is slowly heated to 1,450°C as it moves through its long heating sections. The result is the end product of the pyroprocess: clinker.

Onsite energy: Energy consumption represents the largest part of the production cost for cement factories (Boldyryev 2018), and the clinker production step accounts for 90% of the total energy consumed in the cement manufacturing process (U.S. DOE 2017). As depicted in Figure 76, the traditional method of making cement has five subprocesses; Table 11 breaks down the energy consumption accordingly. The average U.S. primary energy conversion factor of 3 is used throughout this study to convert on-site electricity to its primary units.

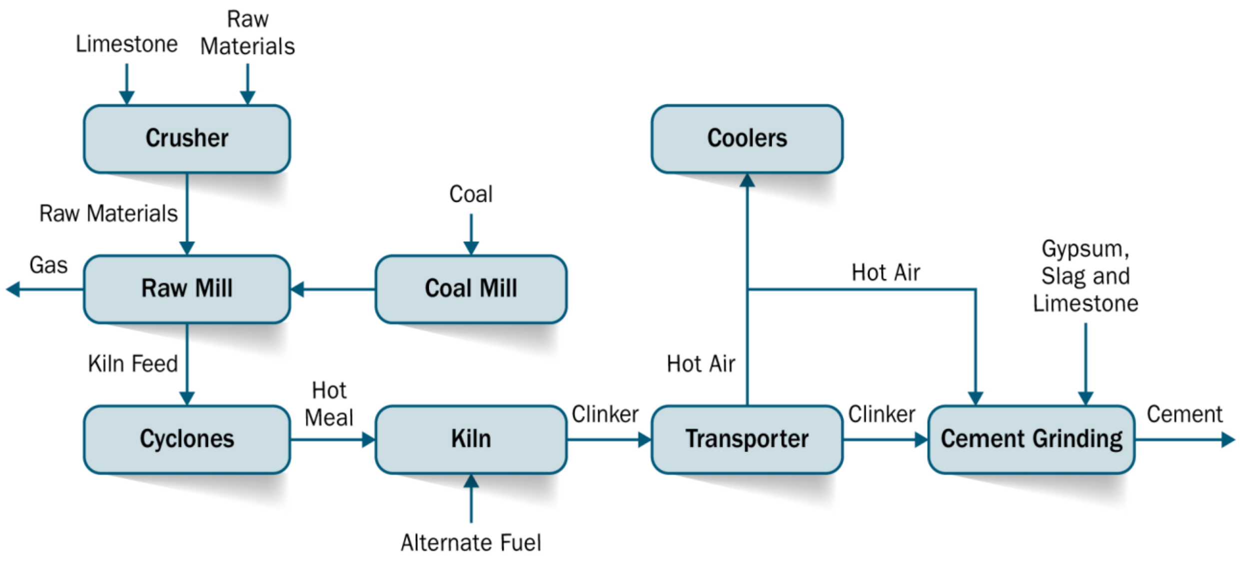


Figure 76. Flowchart of the cement manufacturing process

Note: Schematic based on information in Boldyryev (2018).

Table 11. Energy Consumption of Subprocesses in Conventional Dry Processing of Cement

Conventional Technology: Dry Processing					
Process	**Electric Use (GJ$_{electric}$ / MT of Clinker)**	**Fuel Use (Gj$_{thermal}$/MT of Clinker)**	**Total Primary Energy (Gj$_{thermal}$/MT of Clinker)**	**Process Description**	**Source**
Kiln feed preparation	0.220	–	0.660	Grinding and crushing raw materials using ball mills	(Worrell, Galitsky and Kermeli 2013)
Fuel preparation	0.011	–	0.035	Fuel preparation by ball and tube milling	(Worrell, Galitsky and Price 2013)
Pyro-processing	–	3.481	3.481	PH-PC operation[a]	(U.S. EPA 2007)
Clinker cooling	0.023	–	0.068	Kiln cooling[b]	(ECRA 2009)
Finish grinding	0.142	–	0.425	Finish milling system—lab tube mill	(Cembureau 1997)

[a] Preheater, precalciner, and rotary kiln.

[b] Reciprocating grate coolers.

The energy intensity numbers provided in Table 11 reflect the most commonly used technologies in the dry processing of cement, accounting for more than 90% of U.S. cement manufacturing. For example, ball milling is used for the crushing and grinding operations, as it is the most common form of milling in cement manufacturing. Similarly, the analysis uses the most common pyroprocessing system in modern dry processing: a rotary kiln with preheater and precalciner. In older cement kiln systems, all thermal processes occur within the kiln, leading to higher inefficiencies. These energy intensities help estimate the baseline energy, emissions, and material use in the cement sector.

On-site material use: It is helpful to understand the materials used along with the energy inputs and emission sources in manufacturing cement. Table 12 lists the minerals that make up the raw material of cement (Marceau, Nisbet and Vangeem 2006). Cement is manufactured through a closely controlled chemical combination of the oxides of calcium, silicon, aluminum, iron, and other ingredients. Common materials used to manufacture cement include limestone, shells, and chalk or marl (a rock mixture of clay and lime) combined with shale, clay, slate, blast furnace slag, silica sand, and iron ore.

Table 12. Raw Materials for Cement Manufacturing (kg raw materials/kg clinker)

Raw Material	Kg/kg Clinker
Limestone	1.130
Marl	0.249
Clay	0.054
Sand	0.041
Iron ore	0.014
Water	0.606
Shale	0.068
Bottom ash	0.009
Fly ash	0.012
Blast furnace slag	0.009
Other raw material	0.023

Source: Marceau, Nisbet, and Vangeem (2006)

On-site emissions: Figure 77 presents the two sources of emissions from a cement plant. Process-related emissions are generated directly from the cement manufacturing process, namely the decomposition of calcium carbonate ($CaCO_3$) to calcium oxide (CaO) and the release of CO_2. Non-process or energy-related emissions refer to the emissions associated with the generation and use of energy at the manufacturing site. Coal and petroleum coke, which have high carbon emissions when burned, account for over 60% of the energy use in the U.S. cement industry (see Figure 77). While decarbonization of cement manufacturing will require minimizing both energy- and process-related emissions, electrifying the heating process, the sole focus of this case study, would impact only energy-related emissions; this case study does not address the process-related emissions.

Off-site operations: Because cement is a bulky commodity, transportation can be a significant part of the life cycle energy use, material depletion, and emissions. To minimize these transportation impacts, cement plants are ideally located close to markets with access to efficient transportation, such as ships or rail. In addition, most cement plants are located close to limestone deposits. These upstream transportation operations, along with the mining of the fuel and raw materials, can be affected by the proposed process changes and are included in the LCA model. The analysis derives values for off-site operations from the ecoinvent database (Wernet, et al. 2016) with appropriate adjustments based on the boundary conditions defined.

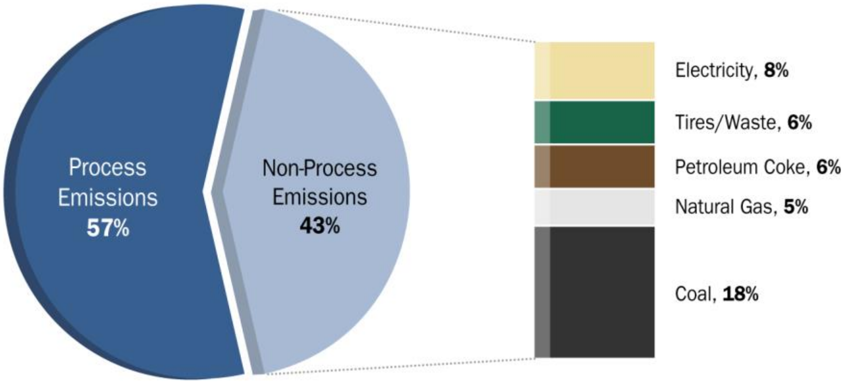


Figure 77. Sources of CO_2 emissions in the U.S. cement industry in 2015

Note: Based on information from USGS (2016).

9.4.1 Electrification of Cement Manufacturing

The proposed approach for reducing material use, energy consumption, and carbon emissions and helping to move the cement industry to a sustainable future is to electrify the clinkering process. This section highlights the relevant research and sets up the life cycle inventory for the proposed case study.

Research and development is ongoing for electrifying the clinker production process, and various options are discussed by Tokheim et al. (2019). The electric systems under development promise to increase energy productivity and, when combined with clean power, to potentially reduce GHG emissions. The use of established electric heating technologies, such as resistance heating, for indirectly heating calciners is well documented (Tokheim, et al. 2019). These hybrid systems use electric heaters in a first step followed by conventional fuel-fired sintering kilns to complete the pyroprocess. This hybrid approach is treated as **Pathway 1** in this analysis.

It is possible to expand the application of these established indirect electric heating techniques to rotary kilns (sintering step), allowing for complete electrification of the thermal processes in a cement plant. Indirect heating systems could switch the thermal energy source from a fuel to electricity without changing the basic purpose of the thermal processes involved. Other technologies under development are based on electromagnetic radiation and make use of the fundamental differences in the material energy interactions, changing the way a material is processed (Makul, Rattanadecho and Agrawal 2014). While both methods effectively lead to completely electrifying clinkerization, we consider electromagnetic radiation-based technologies under **Pathway 2**, given their unique capabilities in cement processing. The use of microwave-assisted heating for clinkerization is one such technique.

The major difference between microwave and conventional thermal processing is the heating method. Microwaves heat the material volumetrically, from the inside out, while conventional sintering heats from the outside, moving inward. Further, the unique energy-material interactions of microwave radiation provide additional processing benefits. In the case of cement processing, microwaves reduce the sintering temperature and processing times (Fang, Roy and Roy 1996). These translate to reduced energy consumption compared with conventional fuel-fired systems. Other electrical heating technologies (e.g., resistance heating), while replacing high-carbon fossil fuels, do not provide these additional energy efficiency and process benefits.

The earliest work on microwave processing of cement occurred in the 1980s (Quéméneur, et al. 1983). The study achieved microwave-based clinkering of cement raw meal using a grooved resonant applicator (0–3 kW, 2.45 GHz). X-ray diffraction and scanning electron microscopy analysis showed that the clinker produced using the microwave process was equivalent in composition to that produced in the laboratory. Compressive strengths of micro-specimens were also comparable.

Over the years, researchers in the laboratory environment have refined the technique and explored using various configurations of microwaves and associated plasmas. Microwaves enhanced the clinkering process of ordinary and colored Portland cement by reducing the clinkering temperature by 100°C and promoting the preferential growth of tricalcium silicate in the microwave field (Fang, Roy and Roy 1996). Further studies found that the presence of ferric oxide (Fe_2O_3) in the raw meal contributed to reducing the clinkering temperature, as Fe_2O_3 is a good absorber of microwave energy. Other studies explored preheating the raw meal before introducing microwave heating by an applicator and found preferential results (Liu, Jiang and Han 2011). Kaewwichit et al. (2017) proposed a complete electric pathway for clinkerization that uses a microwave-assisted technique to process cement; the study found that the process consumed significantly less energy than traditional methods. Figure 78 shows an example experimental setup of a microwave system used for cement clinkerization.

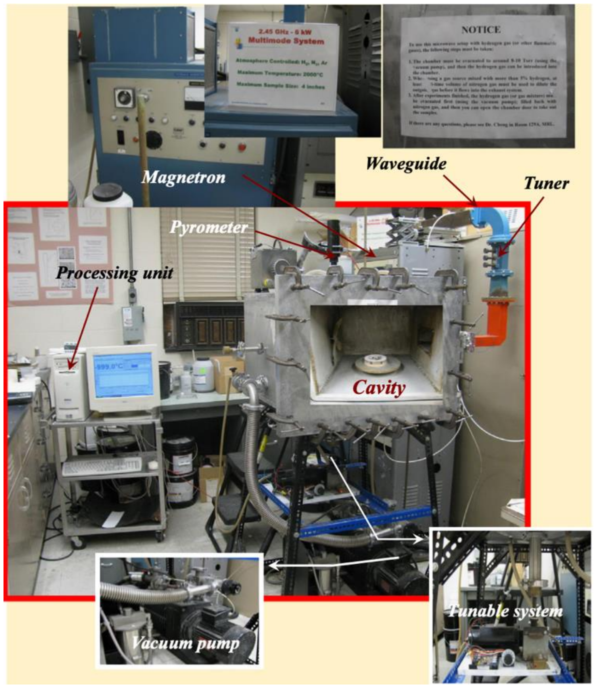


Figure 78. A high-temperature multimode microwave system presented as an example for the experimental setup used for cement clinkerization testing.

Note: Figure from Agrawal (2010). Used with permission from the author.

A pilot study undertaken by Ceralink, Inc. showed the potential for microwave-assisted technology (MAT) to process limestone. The clinkerization process could also apply this approach (Fall, et al. 2011). The most promising work in this area is a collaborative study undertaken by the Swedish Energy Agency, HeidelbergCement Group, and Vattenfall.[68] Their CemZero pilot study examined multiple emerging technologies to replace fuel-fired pyroprocessing in the cement manufacturing process with fossil-free electricity. It confirmed the potential for using microwave-based plasma technology (Vattenfall 2019). Based on these findings, the group is investigating the construction of a pilot plant that uses the technology. The results of this pilot will be critical to increase interest in, and possibly widen the adoption of, the technology.

These studies, while promising, are nascent, and the potential success of the technologies is uncertain. Given the lack of industry-scale results and an inadequate understanding of the economic impacts, electrifying the clinker production process commonly meets a high degree of skepticism from the cement industry. Also, as noted by Buttress, Jones, and Kingman (2015), the economics of a large-scale electric system would pose a significant barrier to its adoption, even if the technological barriers were overcome. These challenges are faced by any new technology that significantly changes the core process schemes. Therefore, a detailed LCA that goes beyond a simple system-level cost-effectiveness comparison, like the one undertaken in this case study, would be ideally suited to examine the holistic macroeconomic impacts in these scenarios.

Taking the above discussions into consideration, while recognizing the sizable barriers that exist for the adoption of the technology, this analysis is presented cautiously. It overcomes some uncertainties associated with the case study by performing a sensitivity analysis around them (e.g., the proposed improvements in energy intensity). The sensitivity metrics for the proposed system are determined by comparing the performance of MAT (previously discussed) in its research and scale-up phases. A 13% variance is established for the energy intensity metric of the electrotechnology based on this analysis (see details in Appendix B, Table B.2). Table 13 defines the energy intensity values used in the analysis.

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[68] The Swedish Energy Agency is a Swedish governmental agency that works for the use of renewable energy, improved technologies, a smarter end use of energy, and mitigation of climate change. HeidelbergCement, a German multinational company, is one of the largest building materials companies in the world. Vattenfall is a Swedish multinational power company owned by the Government of Sweden.

Table 13. Energy Intensity Values for Proposed Electrotechnology Steps – Dry Processing

Process	Electricity Use (GJ$_{electric}$ /MT of Clinker)	Fuel Use (Gj$_{thermal}$/MT of Clinker)	Total Primary Energy (Gj$_{thermal}$/MT of Clinker)	Process Description	Source
Kiln feed preparation	0.220	–	0.660	Grinding and crushing raw materials using ball mills	(Worrell, Galitsky and Kermeli 2013)
Fuel preparation	–	–	–	No fuel preparation	
Pyro-processing[b]	P1: 0.774; P2: 2.580 ± 0.33	P1: 1.315; P2: -	P1: 3.637 P2: 7.740 ± 1.01	Processing by hybrid and completely electric methods	(Tokheim, et al. 2019, Kaewwichit, et al. 2017)
Clinker cooling	0.023	–	0.068	Kiln cooling[a]	(ECRA 2009)
Finish grinding	0.142	–	0.425	Finish milling system—lab tube mill	(Cembureau 1997)

[a] Reciprocating grate coolers–

[b] P1 - Pathway 1: Electric Calcining and Conventional Sintering; P2 - Pathway 2 - Electric Calcining and Electric Sintering.

The case study follows the four stages typically associated with an LCA model as prescribed by ISO 14040 (ISO 2006b): goal definition and scoping, inventory analysis, impact assessment, and interpretation. The life cycle inventory (LCI) step involves identifying and quantifying all the resources used in the process flow. For the electrotechnology considered, the inventory should include not only the energy and material impacts within the facility but also changes to operations outside the facility indirectly impacted by electrification. These include the material and energy flow associated with generating and transporting energy to the point of use at the facility.

The LCI for conventional technology and electrotechnology is thus built through a rigorous assessment of the current state of research on the technology. Additional inventory components and associated sources used for impact assessment can be found in Appendix B, Table B.3.

9.4.2 LCA Model and Impact Assessment

In this case study, we use the ReCiPe-H impact assessment method (introduced in Table 4), and we focus on the global warming (kg CO_2 eq), fossil resource depletion (kg oil eq), and mineral resource depletion impacts (kg Cu eq). The LCA results are tabulated, and the impacts of the proposed electrotechnology on energy, materials, and emissions are presented using quad charts and wedge analysis. Figure 79 shows the LCA results. The figure's three charts present conventional technology (baseline case) and the two electrotechnology pathways (Pathways 1 and 2) for side-by-side comparison. Each data point on the chart represents a specific electricity source that fuels the operations. The size of each circle indicates its CO_2 equivalent emissions (GWP).

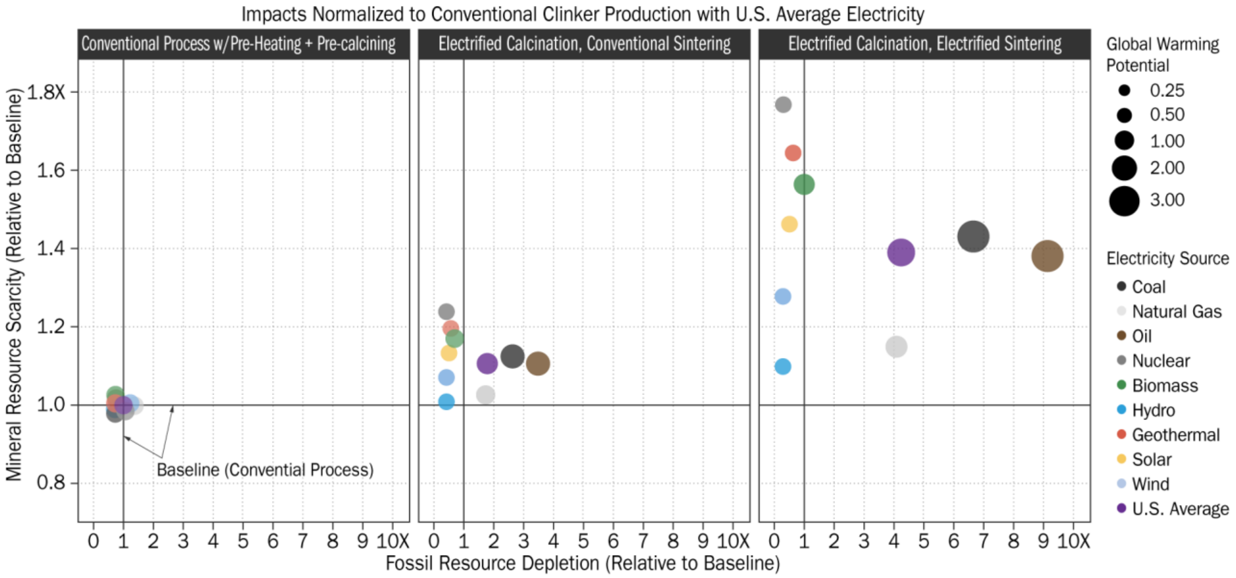

Figure 79. Environmental impacts per metric ton of clinker from conventional production (left panel) and the two electrification pathways (center and right panel) shown as a quad chart.

The electricity source significantly impacts the sustainability parameters analyzed. Comparatively studying the impact of individual sources provides valuable insights, although a grid driven by a single fuel source does not correspond to a real-world scenario. One observation is that lowering fossil use and global warming impacts requires both a more renewable grid (e.g., wind, solar, hydro) and appropriately electrifying the manufacturing process. This finding is clearly established by analyzing the impact of the two electrification scenarios on the environmental impact metrics when using the U.S. average electricity sources compared with using renewable electricity sources. The U.S. average electricity source represents the current national grid. With the current grid mixture, the implementation of either of the electrification pathways increases environmental impacts for the three impact metrics examined.

Under the current grid scenario, implementing electrotechnologies increases the fossil resource depletion because of the high fuel consumption upstream for electricity generation. This also leads to increased emissions at the generation site. The trade-off in these cases is that decreasing on-site emissions leads to relatively higher upstream emissions, resulting in an overall larger global warming impact. Figure 80 shows this quantitatively by plotting these two emission components separately.

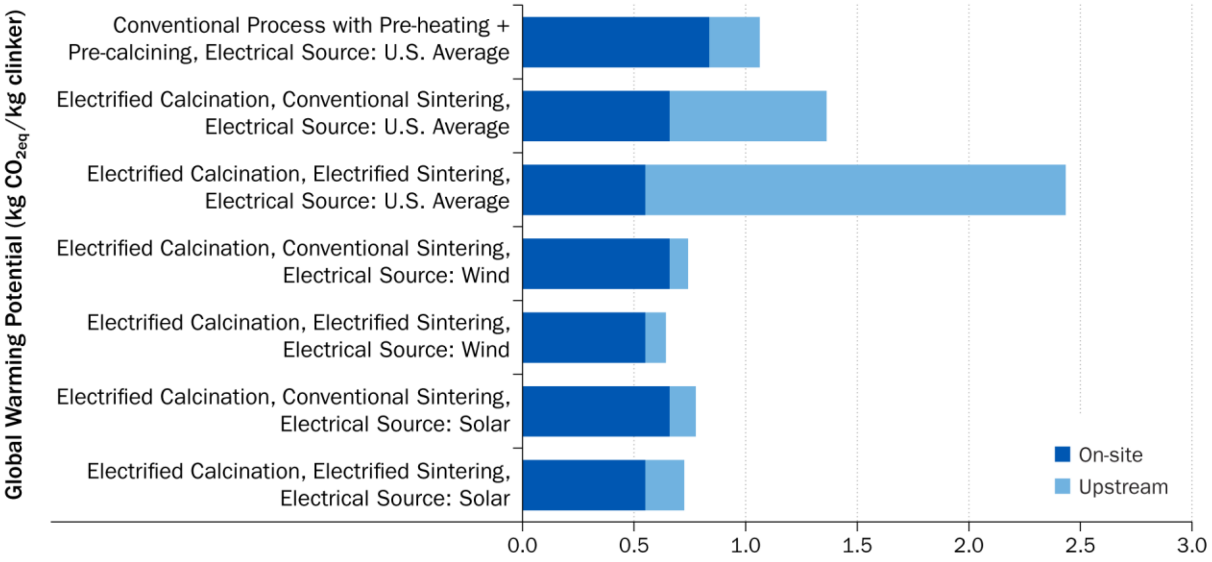

Figure 80. LCA results for GWP divided into its components for the three scenarios analyzed under different grid configurations.

Thus, this analysis quantitatively establishes the need for more renewable sources of electricity. However, increasing the amount of renewable energy should be approached with a holistic understanding of the sustainability impacts, as there will be material trade-offs to increasing renewable capacity. Further, each renewable source will vastly differ in its impact on the sustainability parameters considered. It is critical to understand these impacts to achieve the desired sustainability benefits.

Figure 79 provides more insight into these impacts; the figure shows that while all renewable sources fall to the left (lower) of the U.S. average in terms of fossil resource depletion, they have a much higher material depletion impact. Also, although the individual renewable sources (wind, solar, hydro) have a relatively similar impact on fossil fuel reduction, they are vastly different in terms of their abiotic material footprints. This difference is more pronounced for Pathway 2 (electric calcining and electric sintering), as this scenario has the highest electricity demand; correspondingly, its renewable energy infrastructure needs are higher.

The capacity factor (i.e., ratio of the actual to the maximum possible electrical energy output) of the electricity sources plays a significant role in quantifying these impacts. The LCA model applies a national average capacity factor of 35% for wind and 25% for solar (EIA 2020a). With technological advancements, optimized design, and careful siting, the capacity factor can be increased, leading to a smaller material and emission footprint for these renewable sources.

Figure 81 explores the impact of electrification on the overall primary energy consumed under different grid configurations. These results were calculated using the Cumulative Energy Demand impact assessment method. Primary energy is the energy contained in the raw fuel, as opposed to secondary energy (e.g., electricity), which is derived from primary sources. Primary energy can be nonrenewable or renewable. As Figure 81 shows, total primary energy increases with electrification, irrespective of grid configuration. However, when wind or solar power is the electricity source, the nonrenewable fossil fuel is greatly reduced for the proposed pathways. From a material circularity perspective, this fossil fuel depletion is most relevant, as it represents a material leak from the economy. Hence, subsequent analysis steps focus on the fossil fuel depletion when considering energy use.

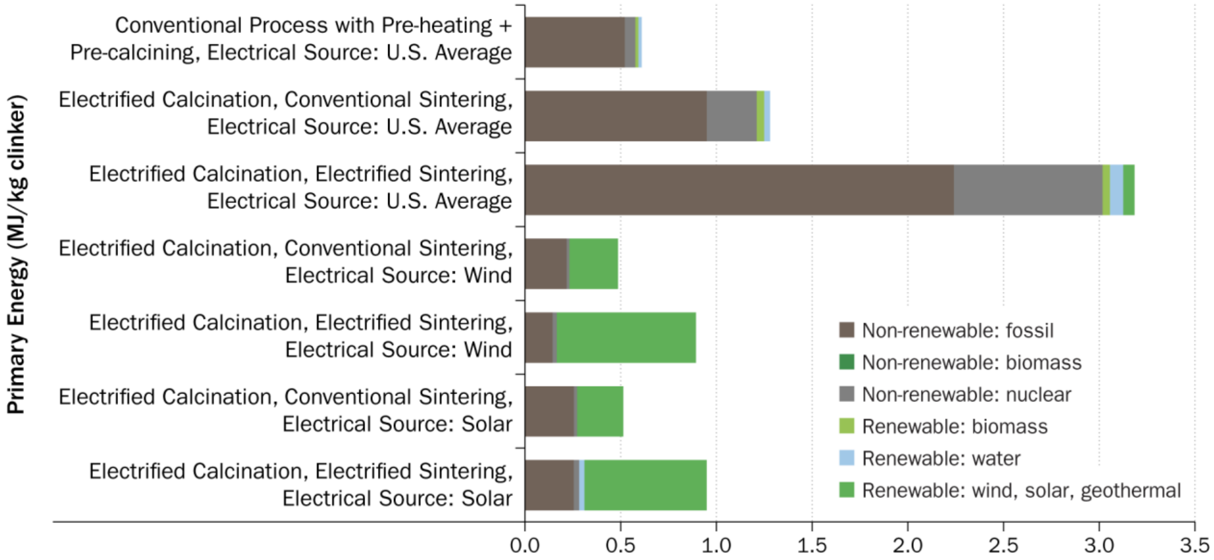

Figure 81. LCA results for primary energy divided into its components for the three scenarios analyzed under different grid configurations.

Note: Renewable and Non-renewable Biomass: Biomass consumption is considered renewable when the feedstocks (example: energy crops, agricultural crop residues, forestry and wood processing residues) are sustainably sourced and considered non-renewable when it contributes to deforestation of primary forest or forest/grasslands degradation

9.4.3 Forecast Model and Impact Assessment

To analyze the impact of deploying the technology over time, the study integrated results from the LCA analysis into a forecasting model to develop wedge charts. The forecasting model estimates the impact of electrotechnology from 2020 to 2050 and accounts for a nonlinear technology adoption trajectory, different grid projection scenarios, and changing cement demand. (Appendix B, Table B.1 summarizes the LCA model results.)

Figure 82 summarizes the steps and data sources of the entire analysis.

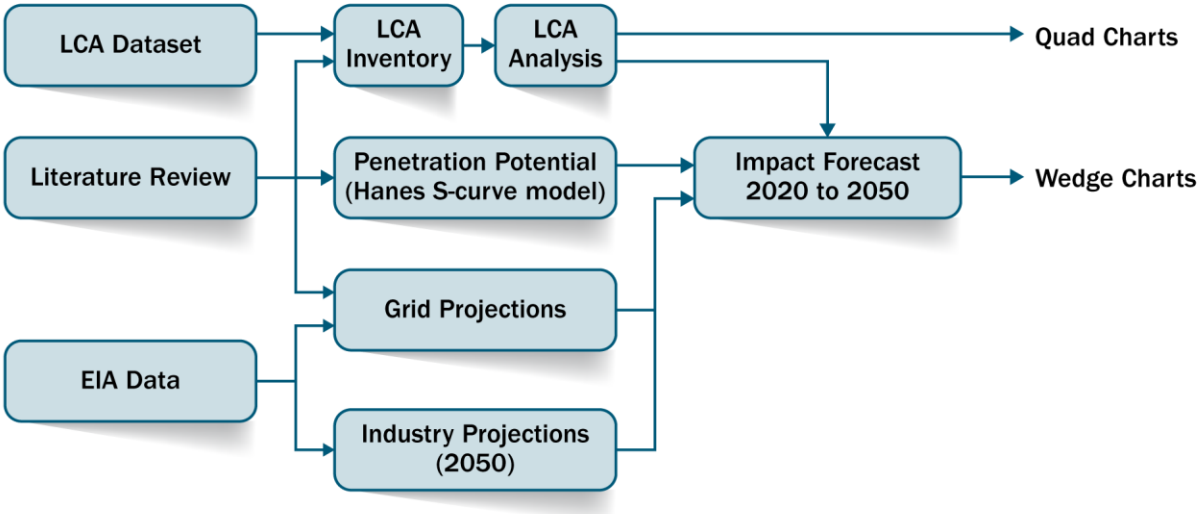

Figure 82. Summary of steps and input data sources for the analysis

The study derived the technology adoption trajectory from the methodology laid out by Hanes et al. (2019) for quantifying the adoption rates over time for advanced energy-efficient manufacturing technologies. The adoption was modeled as an S-curve. It considered eleven technology characteristics (e.g., internal rate of return and scope of impact) and weighted those characteristics to reflect their impact on the overall technology adoption. (Appendix B, Figure B.1 provides details of the modeling.)

Integrating the forecast model with the different grid scenarios highlights the importance of the grid mix to the potential impact of electrification. The BAU case adopted the reference grid scenario projected by the EIA (2020a). An 80% renewable energy grid projected by NREL (Mai, et al. 2012) provided the alternative high–renewable energy scenario. Figures B.2 and B.3 in Appendix B present a comparison of these two grid scenarios.

The growth in cement demand considered in the analysis is derived from EIA's forecast for the value of shipment estimated for the sector. The value of shipment is considered a better indicator for clinker production than an energy or CO_2 metric, as the latter projections have energy and process performance improvements built into the forecasts. The EIA estimates the cement industry's shipment values under various scenarios; the analysis used an average of its high-growth and low-growth scenarios. The effect of inflation on the value of shipment was discounted to ensure that the year-to-year increase in values was not driven by it.

The results from the forecasting model are presented in the form of wedge charts for energy consumption, material depletion, and GHG emissions. Figure 83 and Figure 84 show the charts for energy and material depletion.

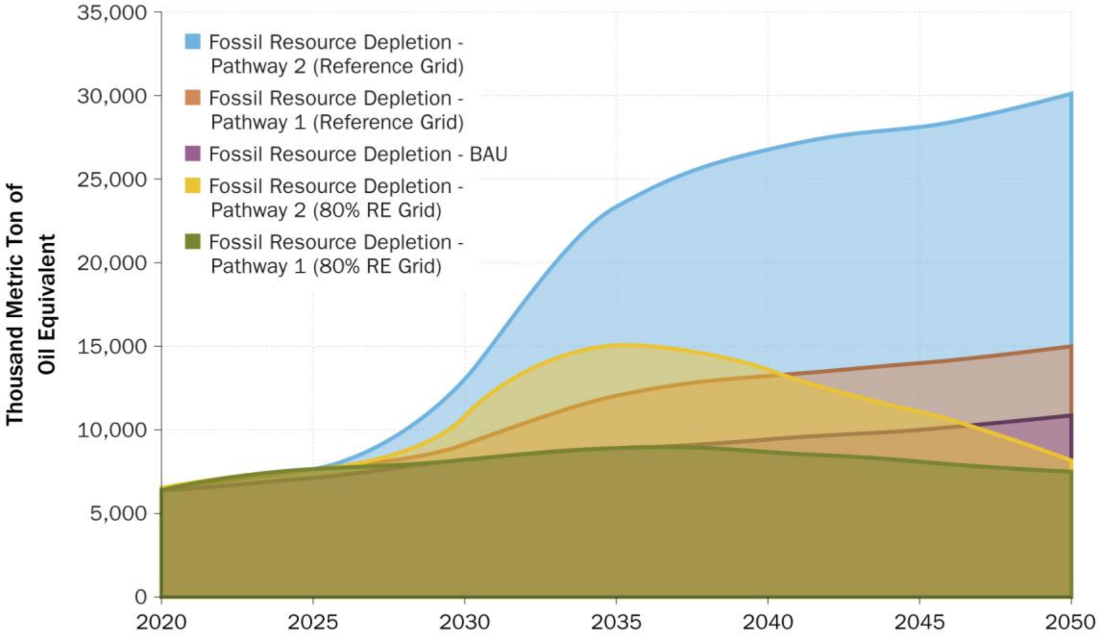

Figure 83. Forecasted fossil fuel depletion from 2020 to 2050 with the gradual adoption of electrification under a Reference Grid and a High-Renewable Grid

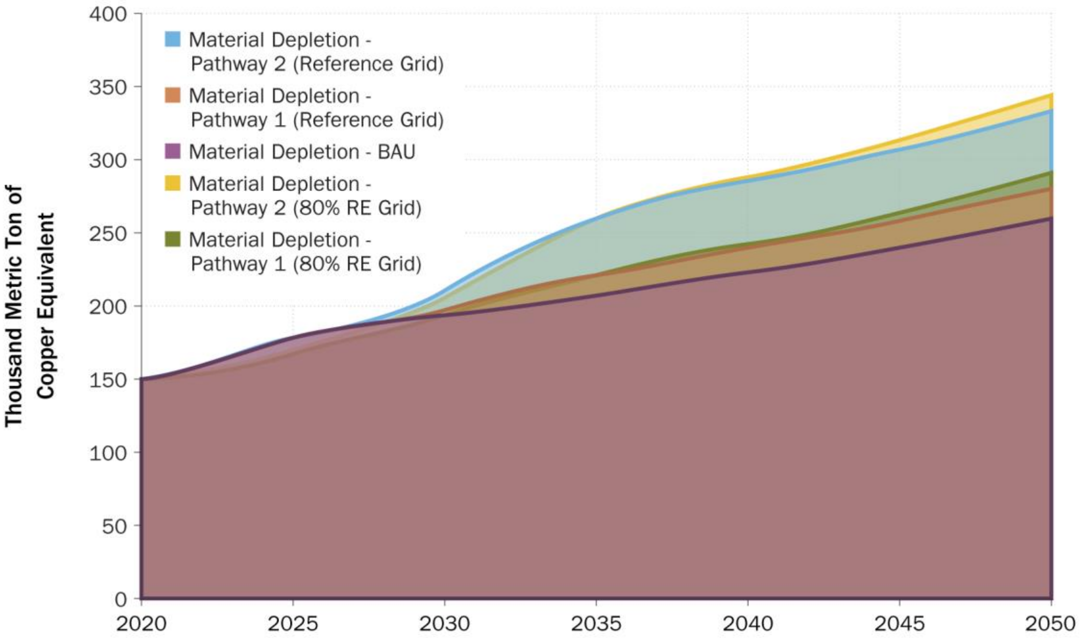

Figure 84. Forecasted material depletion from 2020 to 2050 with the gradual adoption of electrification under a Reference Grid and a High-Renewable Grid

Figure 83 and Figure 84 represent the cradle-to-gate (all processes upstream of the manufacturing facility) impact of electrification on total fossil energy and material depletion under different scenarios. Applying electrotechnology under the Reference Grid scenario increases both fossil fuel use and overall material depletion. Fossil fuel depletion increases because of losses associated with generating and transporting electric power to the facility, in contrast to using fossil fuels at the site in the clinkerization process. Under a High-Renewable Grid, fossil use decreases in the long run, but material depletion increases, especially as the grid comprises more solar or wind energy. This is because the new renewable energy infrastructure used to satisfy the increased demand is more material-intensive than conventional energy generation operations.

Figure 85 presents the wedge chart for GHG emissions. The chart shows that applying the new technology reduces emissions, but only when the corresponding grid is predominantly composed of renewable energy sources. In fact, total emissions increase with the deployment of electrotechnology in a high-fossil fuel grid (Reference Grid), making the adoption of electrotechnology counterproductive to decarbonizing cement production. Because the Reference Grid relies heavily on coal and natural gas, deploying the electrotechnology without changing the grid merely displaces emissions to the electricity generation site.

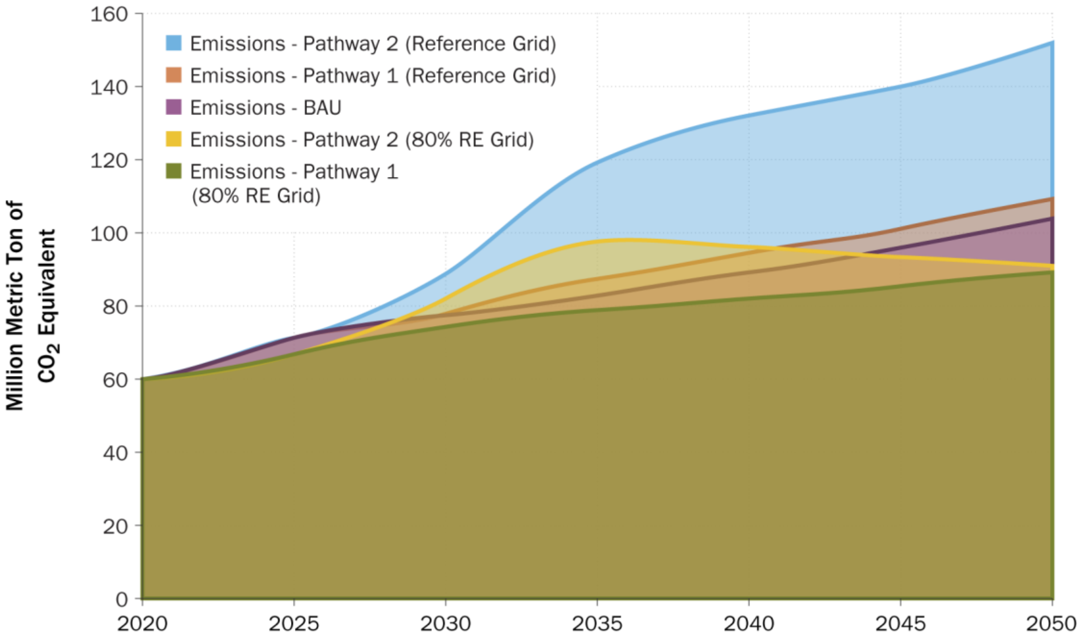

Figure 85. Forecasted emissions from 2020 to 2050 with the gradual adoption of electrification under a Reference Grid and a High-Renewable Grid

Furthermore, under a High-Renewable Grid (80% RE Grid) scenario, the results for 2050 reveal lower emissions with partial electrification (Pathway 1) compared to complete electrification (Pathway 2). This difference is a product of the various parameters (e.g., technology penetration, grid mix, capacity factor) and their interactions considered as part of the model. While complete electrification could lead to higher emissions initially, when the grid is not sufficiently clean (2030–2040), the emissions rapidly reduce with the integration of more renewable sources (2040–2050). Analyzing the trajectory of GHG emission volumes over these periods suggests that complete electrification will eventually have a smaller emission footprint than partial electrification.

With the implementation of electrification under a highly renewable grid, it is projected that a 25–32% reduction in fossil resources and a 12–13% reduction in GHG emissions can be achieved for the two pathways by 2050. These reductions come with a 12–32% increase in the use of recyclable metals and minerals driven by the higher material intensity of renewable electricity plants.

9.5 Interpretation and Discussion

The analysis found that deploying the two electrification pathways with existing renewable grid technologies reduces fossil energy consumption and emissions; however, the more material-intensive electric generation infrastructure leads to additional abiotic depletion of metals and other critical materials. This result is not ideal from a circular economy perspective.

A more material-efficient renewable energy infrastructure with greater material reuse and/or increased generation efficiency would substantially reduce material consumption in the life cycle of cement. Thus, a crucial step in qualifying electrification as truly sustainable is to develop suitable methods to recycle and reuse renewable energy equipment at EOL. The importance of recycling for clean energy technologies is well understood and documented (St. John 2022). Developing more efficient electrotechnology can also lead to changes in the dynamics of the impacts estimated.

Material circularity: While a review of methods for recycling renewable energy infrastructure is outside the scope of this effort, it is worth noting the types of materials for which consumption increases, as well as understanding, at a high level, their potential recyclability. Figure 86 shows a side-by-side comparison of the material consumption in terms of mineral resource depletion for making a metric ton of clinker under conventional and electrified scenarios and categorized as either raw meal or other.[69]

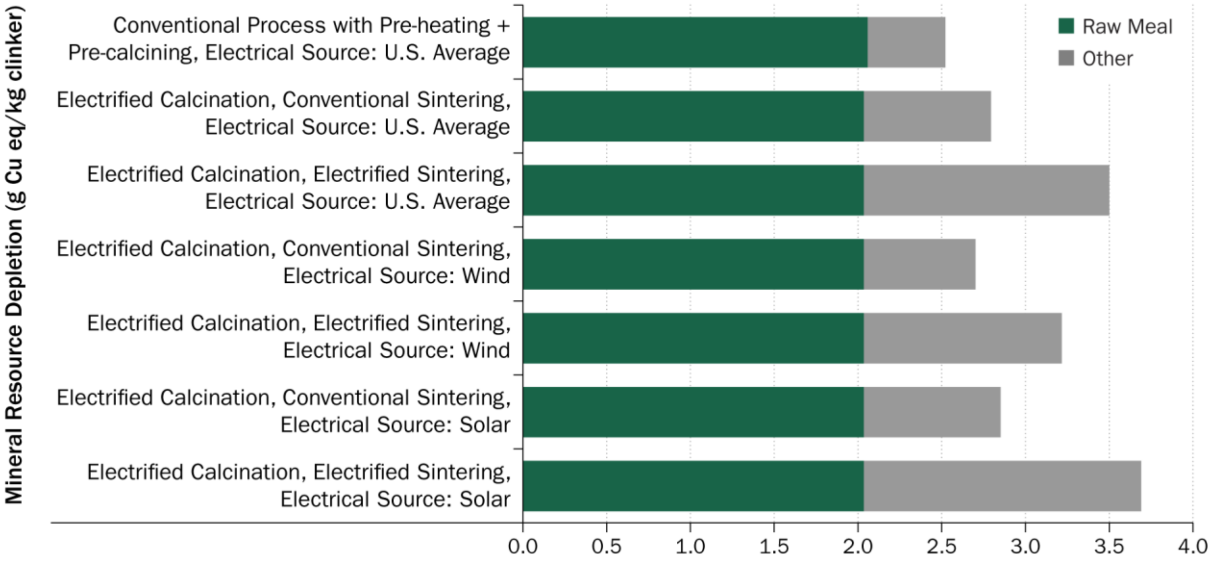

Figure 86. Material breakdown for mineral resource depletion in cement production operation

The increase in material consumption is driven predominately by steel, iron, glass fiber, copper, and aluminum. These materials are highly recyclable. Improving the rate at which they are recycled from renewable energy infrastructure will help achieve circularity of material flow. Even if higher volumes of material are used under an electrified scenario, effective recycling of these materials can result in more sustainable operation overall.

The key reason for increased material use would be the construction of wind and solar capacity. Wind turbine blades are 85% recoverable on average, and efforts are under way to solve the technical challenges of turbine blade recycling. Research in advanced recycling technologies, such as recyclable blade resins, is being conducted by groups like the Institute for Advanced Composites Manufacturing Innovation (IACMI), Institute for Reducing Embodied-energy And Decreasing Emissions (REMADE), the American Composites Manufacturers Association (ACMA), and the Electric Power Research Institute (EPRI). DOE is also active in this space, exploring advanced manufacturing techniques for better recyclability of future blades, performing R&D to enable cost-effective recycling of existing fiberglass blades, and analyzing to quantify blade waste (U.S. DOE WETO 2020). Industry efforts to increase wind

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[69] "Raw meal" refers to the volume of raw materials (e.g., clay, limestone) needed to make a unit of product (clinker), converted to copper equivalence. "Others" is the sum of all the materials consumed in upstream processes defined within the scope of the study (e.g., materials needed for energy infrastructure). The amount of raw material needed does not change with any of the proposed technology pathways; hence, the "raw meal" remains constant under all scenarios.

turbine blade recycling have already begun (Frangoul 2021). Eventually, blade components may also be recycled and used to manufacture new blades, creating a circular economy of use.

Entrepreneurs are also exploring new business opportunities that may arise from recycling wind turbine blades. Several startups are developing processes to break down the blades and refabricate them into useful products for vehicles, other renewable energy system components, agricultural products, and performance sports equipment. Vestas, one of the world’s oldest and largest wind turbine manufacturers, has committed to producing only zero-waste wind turbines by 2040, with intermediate targets along the way.

Solar technologies have a more challenging path to recyclability. Solar panels manufactured using crystalline silicon (C-Si) technology hold the largest share in the photovoltaic panel market. Recycling Si-based photovoltaic panels entails separating the layers, which necessitates removing the encapsulant from the panel and the silicon cells to recover the metals.

Electrification and carbon capture: Fully decarbonizing the cement sector requires addressing clinker-making process emissions. Regardless of the heat source used (fossil fuel or clean [low-carbon] electricity), the solution must include carbon capture—either carbon capture of both the combustion and process emissions or capture of concentrated process emissions combined with electrification (Somers 2020).

The operation of a cement kiln system is affected not only by the chemical composition of the raw materials used but also by the fuel used. The type of fuel can introduce some material components (through combustion) that might interfere with the chemistry of the cement materials as well as affect the system’s operation (Chinyama 2011). Electrical technologies are ideally suited for retrofitting projects, as electrification would not affect the combustion chemistry and would not require a complete overhaul of all systems. Given that pyroprocessing does not involve a flue gas, the complications associated with carbon capture are also dramatically reduced.

Electrification could thus aid the more effective implementation of carbon capture technologies. An electrified system will produce a cleaner and more concentrated stream of CO_2 compared with a fuel-fired system. The carbon can be captured directly without expensive processes like amine scrubbing (a technology typically used to separate CO_2 from natural gas and hydrogen).

Conclusion: This case study explores electrification of the cement manufacturing process and, through an LCA-based approach, quantifies its holistic impacts on energy consumption, emissions, and materials depletion under various scenarios. It examines the interplay among these components in detail and devises opportunities to comprehensively address sustainability. Finally, it quantitatively establishes the importance of a low-carbon electric grid for electrification applications.

10 Discussion and Conclusion

Adopting circular and material efficiency strategies could play a significant role in decarbonizing the manufacturing industry. The case studies reported here examine the challenges and opportunities for adopting particular circular economy strategies and evaluate material consumption, waste generation, and associated environmental trade-offs. Each case study assumes (unknowable) future conditions for estimating the volume of EOL materials and products, the direction next-generation products take, and how quickly technologies are adopted. These are challenges any analysis projecting future scenarios will encounter. Therefore, it is vital not only to recognize these uncertainties and try to minimize them but also to interpret and use the information from the analyses with these uncertainties in mind.

We highlight the critical challenges revealed by the case studies, as follows:

- The *Food Loss and Waste Case Study* revealed that most food waste occurs at the final consumption stage of the value chain. Improved packaging technologies and labeling standards, along with consumer education, are needed to help reduce consumer food waste generation (and its associated impacts). While reducing waste generation is always the best means of reducing waste impacts, if the original consumer will not use the food and it is still good when the consumer will not eat it, food donation is the next best pathway for reducing waste. If the waste that cannot be donated (e.g., food scraps or partially consumed), the best options are to utilize it for animal feed, anaerobic digestion, or dispose of it via wastewater treatment.
- The *Plastics Case Study* found that while R&D is exploring and developing several technologies to convert plastic waste into functional materials for manufacturing, careful trade-off analysis will be needed to assess these novel technologies. For instance, while triboelectrostatic separation results in higher plastics recovery but higher energy and GHG emissions, other forms of waste diversion (PET upcycling and solvent separation) could result in decreased energy use and reduced GHG emissions. In addition, the need remains for solutions to divert plastic waste from landfills, for products that are better designed for recycling and for separation systems that enable conversion to secondary materials used in manufacturing.
- The *LiB Case Study* found that innovations in EV battery design, the reverse supply chain, and recycling processes dramatically decreased demand for virgin raw materials for EV batteries. Overall, direct recycling combined with a shift to low-cobalt chemistries demonstrates a promising environmental solution. However, there is some risk that direct recycling would recover material that is no longer in high demand, which could negatively impact the profitability of recycling.
- The *NiMH Battery Case Study* shows that recycling has a lower-carbon footprint than primary production of critical rare-earth materials, recycling incurs significant costs and environmental impacts primarily driven by reagents and chemicals used in the hydrometallurgical processes.
- The *Cement Case Study* evaluated the material impacts of applying electrification to cement/clinker manufacturing processes. It found that renewable power was essential to decarbonize the sector but that the demand for renewable energy systems would significantly increase abiotic resource consumption.

Therefore, new technologies with lower chemical intensities are needed to sustainably offset primary material demand.

The case studies also revealed that adopting circular economy strategies could realize material efficiency benefits:

- The *Food Loss and Waste Case Study* found that improved packaging and labeling not only helps mitigate waste at the final consumption stage but also results in decreases in both energy and GHG emissions and may help increase food security.
- The *LiB Case Study* and the *NiMH Battery* Case *Study* found that certain circular economy strategies could reduce the demand for critical materials in the supply chains, where the bulk of the mining can negatively impact the social and environmental sustainability of the countries sourcing those materials.
- The *Cement Case Study* found that electrification substantially reduces fossil fuel consumption and carbon emissions when combined with a sufficiently renewable grid. From a material circularity perspective, fossil fuel depletion represents a nonrecoverable material leak from the economy. While renewable energy drives higher material use intensity, the adoption of proper recycling methods can contain the increased use of these metals and minerals. This improves the overall recyclability of the materials used in the cement life cycle even if the total volume of materials increases.

10.1Strategic Conflicts

The analysis reported here also helps underscore the challenges in evaluating the impacts of adopting circular economy approaches and technologies. For example, in many cases, with the current state of technology, implementing particular circular economy strategies will be more energy-intensive than the business-as-usual practice of extracting raw materials and then disposing of them in landfills at EOL. An exception to this would be to increase use intensity or life extension.

Though the circularity benefit to making products last longer is clear, this strategy can sometimes conflict with realized energy savings, particularly for industries experiencing rapid technological change, such as consumer electronics. As an example, the energy use of televisions, as measured by average on-mode power consumed per unit display area, which comprises the majority of power consumption (Urban et al., 2021), has declined more than 75% between 2003 and 2015 (Urban and Roth 2017), while the devices themselves typically last between 4 to 10 years (Quantum 2021). For consumer electronics as a whole, the energy use per device in the U.S. declined ~50% between 2006 and 2017, but increased in 2020 due to pandemic-related usage (Urban and Roth 2017); given lifespans of 1.5 to 13 years depending on the device (Quantum 2021), device efficiency often improves more rapidly than product turnover.

10.2 Time Horizon

Adopting circular economy strategies presents significant challenges in forecasting future markets and infrastructure. Three case studies especially highlighted the importance of the time horizon when designing solutions:

- The *LiB Case Study* found that direct recycling might recover materials no longer in high demand in future markets.

- The *NiMH Battery Case Study* showed the decreasing use of NiMH technology will limit how much material can be recovered and used in other applications when looking past 2030.
- The *Cement Case Study* illustrated that a slowly changing U.S. electric grid can result in increases in GHG emissions in the near term, but much lower emissions by 2050.

10.3 Dependencies and Interdependencies

Because of the nature of the circular economy, impact analysis needs to capture the complexity and economic, environmental, and logistic challenges of EOL and material efficiency approaches. The approaches and strategies are interconnected with other industries and nonindustrial activities in the United States and the global economy (as most industries rely on global supply chains). An example for battery recycling would be that the materials used in batteries have a demand signal in different types of batteries as well as different types of products, and the ability to recover these materials might be managed by facilities owned by local or regional organizations or governments or driven by policies enacted at some level of government. Manufacturers need to be familiar with the multiple EOL pathways, stakeholders, and the multiple opportunities and markets that can create demand or supply for the EOL materials on both the front and back ends of the value chain. The opportunity and solution spaces will likely evolve as industries shift their activities and as technologies to manage and recover materials at EOL improve and ultimately achieve cost parity for secondary materials and products and become economically sustainable and profitable. Beyond the challenge of economics, however, there will be the challenge of incorporating into the design of these next-generation products and technologies the ability to identify, assess, and minimize the potential social and environmental impacts. Perhaps, as McDonough and Braungart (2002) envision, “we stop trying to be less bad and we start figuring out how to be good.”

This work has also revealed path dependencies and interdependencies, where material and energy savings depend on the evolution of circular economy adoption by multiple individual manufacturers and associated services. For example:

- The *Food Loss and Waste Case Study* identified supply chain phases and energy sources with the highest use and GHG emissions, allowing for the identification of areas where innovation could have significant impacts, for example, in the manufacturing energy of pesticides and fertilizers or low-carbon electricity infrastructure.
- The *LiB Case Study* demonstrated that refurbishment and reuse could delay the availability of some materials for recycling. Additionally, it found that the evolutions of battery chemistries could affect economic decision-making regarding recycling technologies.
- The *NiMH Battery Case Study* explored the structure and features of a large-scale infrastructure network for battery recycling. It found that future domestic demand for EVs will affect the feasibility and build-out of domestic recycling plants for EV batteries.
- The *Cement Case Study* demonstrated that the success of electrification of the cement sector relies heavily on the availability of renewable energy sources for the sector and that the level of demand from the cement sector will impact the mineral demands for those manufacturing those technologies.

10.4 Focus for Future Work

Possibilities for future work fall into four categories: design solutions, landfill impacts, policies and standards, recycling infrastructure needs and social dimensions.

Design Solutions: Managing and recovering products and materials at obsolescence will improve with products innovatively designed for reuse, for remanufacturing, for recycling, and other sustainable design approaches. This effort includes designing renewable power systems for Re-X and designing single-use packaging to allow for food preservation, recovery, and recycling.

Landfill Impacts: A more comprehensive understanding of the impacts of landfilling (beyond energy and economics) would provide a more holistic view of the full environmental and social impacts. We need to fully explore the complexity of the impacts and their connections to societal activities to improve our ability to incorporate the EOL perspective and thinking into future analyses. The value of the material that is lost to landfills has been quantified for plastics at $7.2 billion market value for 2019 (Milbrandt, et al. 2022) and at $62.5 billion for electronics in 2017 (WEF and PACE 2019).

Policies and Standards: The case studies in this report do not cover the impacts of policies or standards. However, policies can have a prominent role, as we have observed from policies implemented in other countries or regions. In our work, the *Food Loss and Waste Case Study* showed that standardizing the meaning of food expiration labels could significantly impact how much food is wasted. Researchers could expand this finding to interrogate other food-related policies and standards. Future policy analysis might also consider how implementing extended producer responsibilities would impact EOL management of materials, what barriers manufacturers will face, and the options for overcoming them. Researchers should explore the incentives and disincentives that could increase industrial material efficiency and decarbonization. For example, which kind of financial support would help overcome the challenge of U.S. dependency on foreign sources of critical materials? Which policies and incentives might help advance the reuse, refurbishing, and recycling of products, packaging, and other decarbonization technologies?

Social Dimensions. Additional study is needed to explore the social dimensions of circular economy approaches, including employment; health and safety of employees, customers, and communities; customer satisfaction; product responsibility; and social and environmental justice. For example, new domestic employment opportunities would likely arise to support the collection, transport, and manufacturing associated with EOL and decarbonization processes and technologies. On the other hand, jobs could be lost in competing industries, possibly further disadvantaging supply chain workers in developing countries. Future analysis of circular economy technologies and approaches should consider social dimensions across their supply chains, including how they might affect the affordability of essential products; employee, customer, and community health and safety; environmental injustices associated with local air and water pollution; and job opportunities and accessibility for disadvantaged persons.

Appendix A. Life Cycle Impact Assessment Metrics

Table A-1 presents metrics and definitions of abiotic depletion (*AD*) as expressed in commonly used life cycle impact assessment metrics. It is compiled from sources listed in the table and review papers by Klinglmair, Sala, and Brandão (2014) and Sonderegger et al. (2020). *P, R, g, and CRE,* respectively, represent the annual extraction rate, resource availability, ore grade, and cumulative resource extracted of an abiotic mineral, and *X* represents the extraction of the resource *i*, for which the resulting abiotic depletion is calculated. Methods used in this report are marked in **bold**.

Table A.1. Metrics And Definitions of Abiotic Depletion (*AD*) as Expressed in Commonly Used Life Cycle Impact Assessment Metrics

Impact Assessment Method	Source(s)	Abiotic Depletion Metric and Characterization Factor	Abiotic Depletion Units	Notes
CML 2001	(Guinée and Heijungs 1995) (J. Guinée 2002)	$AD_i = X_i \times ADP_{i,ref} = X_i \times \left(\frac{P_i / R_i^2}{P_{ref} / R_{ref}^2} \right)$ ref = Antimony (Sb) R = Ultimate reserves	kg Sb eq.[70]	- Considers metallic and nonmetallic elements, metallic and nonmetallic mineral ores, and fossil resources. - ADP = Abiotic depletion potential.
ILCD 2011	(Chomkhamsri, Wolf and Pant 2011) (EC-IES-JRC 2012)	$AD_i = X_i \times ADP_{i,ref} = X_i \times \left(\frac{P_i / R_i^2}{P_{ref} / R_{ref}^2} \right)$ ref = Antimony (Sb) R = Ultimate reserves	kg Sb eq.	- Considers metallic and nonmetallic elements and fossil resources. - ADP = Abiotic depletion potential. - Based on CML 2001.
Eco-Indicator 99	(Goedkoop and Spriensma 2001) (Müller-Wenk 1998)	$AD_i = E_i\left(P_i, R_i\right)$ R = Economic reserves	$\mathrm{MJ}_{\mathrm{surplus}}$	- Considers metallic elements, metallic mineral ores, and fossil resources. - *E* = Marginal energy expenditure function. - *E* assumes present day extraction of a resource would reduce the ore grade and thus increase the energy required to extract the same amount of material from its ore in future. - *E* is estimated based on current technology (in 1990), and is calculated as the difference in energy needed for extracting a unit amount of material from its ore in 1990 and the energy needed to extract the same amount of material in future when humankind has extracted 5 times the amount of material extracted in 1990.

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[70] As explained by Van Oers, Guinée, and Heijungs (2020), antimony was chosen due to the relative stability in estimates of *P* and *R* compared to other minerals at the time the metric was developed. Mathematically, the choice of reference material has no bearing on the relative abiotic depletion (*AD)* of a given material to that of another.

Impact Assessment Method	Source(s)	Abiotic Depletion Metric and Characterization Factor	Abiotic Depletion Units	Notes
ReCiPe-2008	(Goedkoop, Heijungs, et al. 2013)	$AD_i = X_i \times ADP_{i,ref}$ $X_i \times \left(\frac{E_i\left(P_i,R_i\right)}{E_{ref}\left(P_{ref},R_{ref}\right)} \right)$ $ref =$ Iron (Fe) $R =$ Economic reserves	kg Fe eq.	• Considers metallic elements, metallic mineral ores, and fossil resources. • ADP = Abiotic depletion potential. • Definition of *E* same as in Eco-Indicator 99 method above.
ReCiPe-2016	(Huijbregts, et al. 2016) (Vieira, et al. 2017)	$AD_i = X_i \cdot ADP_{i,ref}$ $X_i \cdot \left(\frac{SOP_i\left(R_i,g_i,CRE_i\right)}{SOP_{ref}\left(R_{ref},g_{ref},CRE_{ref}\right)} \right)$ $ref =$ Copper (Cu) $R =$ Ultimately extractable reserve (ReCiPe-E) OR Economic reserve (ReCiPe-H)	kg Cu eq.	• Considers metallic and nonmetallic elements, bulk resources, and fossil resources. • SOP = Surplus ore potential function. • SOP is calculated with the basis that each unit of extraction of a mineral leads to a reduction in the ore grade, *g*.
IMPACT 2002+	(Jolliet et al. 2003)	$AD_i = X_i \times ADP_{i,ref}$ $X_i \times \left(\frac{E_i\left(P_i,R_i\right)}{E_{ref}\left(P_{ref},R_{ref}\right)} \right)$ $ref =$ Iron (Fe) $R =$ Economic reserves	kg Fe eq.	• Considers metallic elements and metallic mineral ores. • Definition of *E* same as in Eco-Indicator 99 method above.
EDIP 97, EDIP 2003	(Hauschild and Wenzel 1998), (Hauschild and Potting 2005) (Potting and Hauschild 2005), (Stranddorf, Hoffmann and Schmidt n.d.)R	$AD_{i,r} = PR_{i,r}$ $= X_i \cdot \left(\frac{Pop_{world}}{\sum_{j \in M} R_j} \cdot \frac{Pop_r}{Pop_{world}} \right)$ $R =$ Economic reserves	Person Reserves (PR)	• Considers metallic elements, metallic mineral ores, and fossil resources. • *PR* = Per-capita proportion for a reference region *r* relative to known reserves of all resources aggregated per person consumed in 2004. • *Pop* = Population.
EDIP 2003 (Resources Only)	(Hauschild and Wenzel 1998)	$AD_i = X_i$	kg	• Considers metallic elements and fossil resources. • Characterization factor of 1 is used to allow representation of abiotic mineral depletion in terms of amount of resource consumed without any consideration for availability, historical extraction, or energy needed for extraction.

Impact Assessment Method	Source(s)	Abiotic Depletion Metric and Characterization Factor	Abiotic Depletion Units	Notes
EPS 2000, EPS 2015d, EPS 2015dx	(B. Steen 1999a), (B. Steen, A Systematic Approach to Environmental Priority Strategies in Product Development (EPS). Version 2000 - Models and Data of the Default Method 1999b), (B. Steen, The EPS 2015d Impact Assessment Method - an Overview 2015), (Steen and Palander, A Selection of Safeguard Subjects and State Indicators for Sustainability Assessment 2016)	$AD_i = WTP_i\left(P_i, R_i\right)$	ELU (€ or $)	• Considers metallic elements and fossil resources. • ELU = Environmental Load Unit, expressed in terms of willingness to pay (*WTP*) in € or $ as cost of substituting a given resource by a sustainable alternative in future due to use of the resource in the present. • *WTP* for each resource is calculated using economic models in which future generations, assumed to be sustainable (much more efficient than today) in their resource consumption, bid on present stocks of a resource.
Cumulative Exergy Demand (CExD)	(Bösch, et al. 2006)	$CExD = \sum_i X_i \cdot Ex_{(ch),i} + \sum_j n_j \cdot r_{ex-e(k,p,n,r,t),j}$ Where $Ex_{(ch),i}$ = exergy per kg of substance *i* n_j = amount of energy from energy carrier *i* $r_{ex-e(k,p,n,r,t),j}$ = exergy to energy ratio of energy carrier *j*	MJ_{ex}	• Considers metallic and nonmetallic elements, metallic and nonmetallic mineral ores, bulk resources, fossil, nuclear and renewable resources. • *CExD* is expressed as the cumulative exergy stored in a resource in the form of chemical, thermal, kinetic, potential, nuclear and radiative energy.

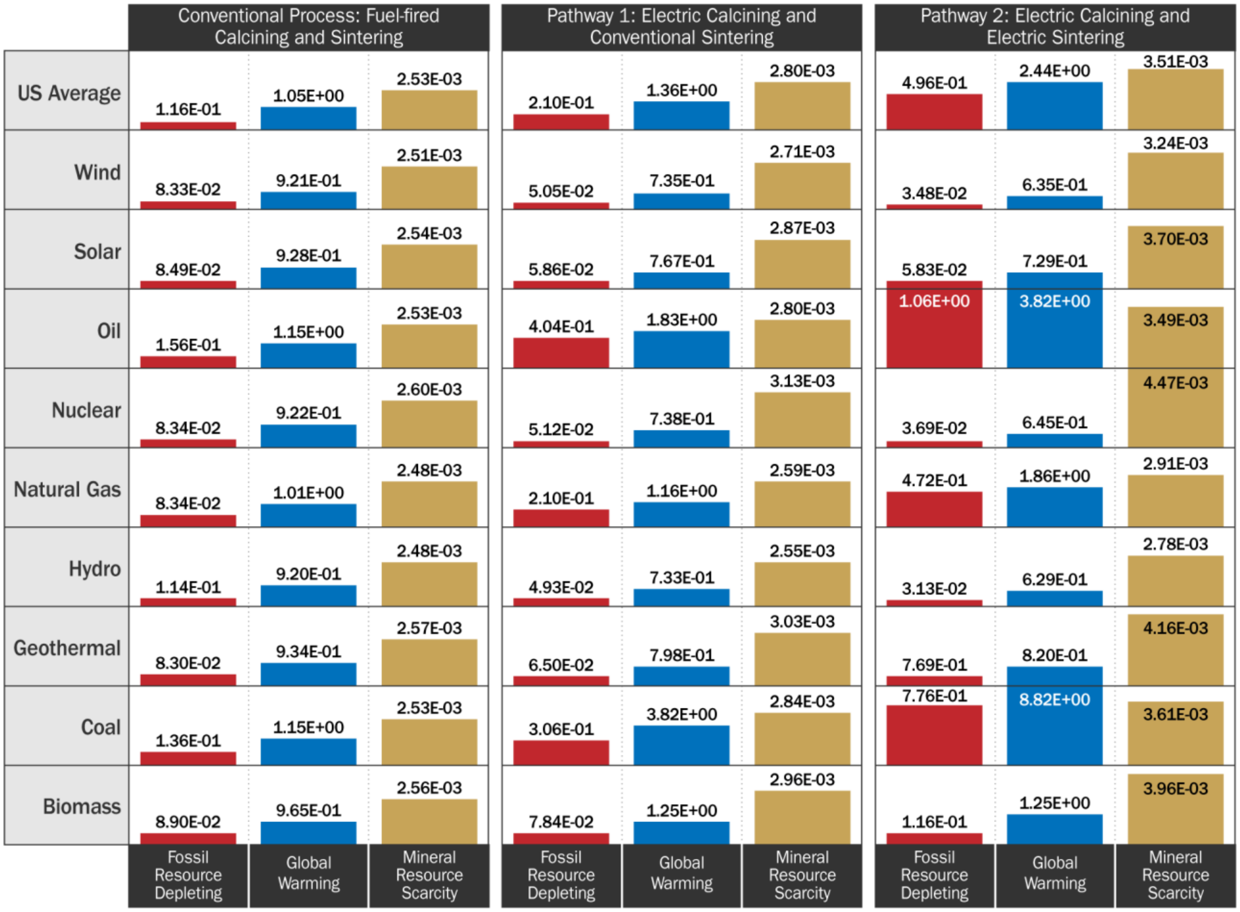
Impact Assessment Method	Source(s)	Abiotic Depletion Metric and Characterization Factor	Abiotic Depletion Units	Notes
Cumulative Exergy Extraction from Natural Environment (CEENE)	(Dewulf, et al. 2007)	$CEENE_i = \sum_{n=1}^{N} x_n \cdot a_{ni}$ Where, x_n = $CEENE$ of the nth resource used to produce a unit of material X_i a_{ni} = Amount of nth resource used to produce a unit of material X_i	MJ_{ex}	• Considers metallic and nonmetallic elements, metallic and nonmetallic mineral ores, bulk resources, fossil, nuclear, land, renewable, and atmospheric resources. • *CEENE* is expressed as the cumulative exergy extracted from the environment to produce a material resource. • x_n includes fossil fuels, metal ores, nuclear energy, biomass, land, renewable energy flows, atmospheric resources, and water resources.

Appendix B. Supplements to Case Study: Electrifying the Cement Clinker Production Process—Evaluating Impacts on Energy, Emissions, Materials, and of a Low-Carbon Electric Grid

LCA Model Results

Table B.1 provides the complete LCA model results.

Table B.1. LCA Model Results



	Conventional Process: Fuel-fired Calcining and Sintering			Pathway 1: Electric Calcining and Conventional Sintering			Pathway 2: Electric Calcining and Electric Sintering		
	Fossil Resource Depleting	Global Warming	Mineral Resource Scarcity	Fossil Resource Depleting	Global Warming	Mineral Resource Scarcity	Fossil Resource Depleting	Global Warming	Mineral Resource Scarcity
US Average	1.16E-01	1.05E+00	2.53E-03	2.10E-01	1.36E+00	2.80E-03	4.96E-01	2.44E+00	3.51E-03
Wind	8.33E-02	9.21E-01	2.51E-03	5.05E-02	7.35E-01	2.71E-03	3.48E-02	6.35E-01	3.24E-03
Solar	8.49E-02	9.28E-01	2.54E-03	5.86E-02	7.67E-01	2.87E-03	5.83E-02	7.29E-01	3.70E-03
Oil	1.56E-01	1.15E+00	2.53E-03	4.04E-01	1.83E+00	2.80E-03	1.06E+00	3.82E+00	3.49E-03
Nuclear	8.34E-02	9.22E-01	2.60E-03	5.12E-02	7.38E-01	3.13E-03	3.69E-02	6.45E-01	4.47E-03
Natural Gas	8.34E-02	1.01E+00	2.48E-03	2.10E-01	1.16E+00	2.59E-03	4.72E-01	1.86E+00	2.91E-03
Hydro	1.14E-01	9.20E-01	2.48E-03	4.93E-02	7.33E-01	2.55E-03	3.13E-02	6.29E-01	2.78E-03
Geothermal	8.30E-02	9.34E-01	2.57E-03	6.50E-02	7.98E-01	3.03E-03	7.69E-01	8.20E-01	4.16E-03
Coal	1.36E-01	1.15E+00	2.53E-03	3.06E-01	3.82E+00	2.84E-03	7.76E-01	8.82E+00	3.61E-03
Biomass	8.90E-02	9.65E-01	2.56E-03	7.84E-02	1.25E+00	2.96E-03	1.16E-01	1.25E+00	3.96E-03

Supporting Calculations for Energy Intensity Values

All the figures for conventional technology have been corroborated and found to be consistent with data provided in (U.S. DOE 2017). Table B.2 presents the calculations for the energy intensity values used in the analysis.

Table B.2. Calculations for Energy Intensity Values

Process	Test Total Primary Energy (GJ Therm/MT of clinker)	Calculation Description Test
Kiln feed preparation	0.660	Worrell, Galistsky, and Kermeli (2013) list the energy for kiln feed preparation to be 25–35 kWh/MT of raw material. From (U.S. DOE 2017), for a clinker production of 120,371 million lb/year, the kiln feed production is 210,175 million lb/year in the United States for dry processing. Converting to energy per MT of clinker production to primary energy, $35\frac{kwh}{ton} \times 0.0036\frac{GJ}{kwh} \times \frac{210{,}175}{120{,}371} \times 3 = 0.660\frac{GJ}{ton}$
Fuel preparation	0.035	A tube mill is considered with EI of 26 kWh/MT based on the numbers provided by Worrell (2008). From DOE (2017), for a clinker production of 120,371 million lb/year, the fuel production is 14,176 million lb/year in the United States for dry processing. Converting to energy per MT of clinker production to primary energy, $27\frac{kwh}{ton} \times 0.0036\frac{GJ}{kwh} \times \frac{14{,}176}{120{,}371} \times 3 = 0.035\frac{GJ}{ton}$
Pyro processing—conventional	3.481	EPA (2007) lists the Heat Input for a PH/PC kiln type in Table 3.3 as 3.3 MMBtu/ MT of clinker. Converting the units, $3.3\frac{MMBTU}{ton} \times 1.05536\frac{GJ}{MMBTU}$
Pyro processing—electrotechnology (Pathway 1)	3.637	From Tokheim et al. (2019), the on-site thermal and electric intensity for the hybrid process is determined separately. Thermal Intensity – 1.315 $\frac{GJ}{ton}$; Electric Intensity – 0.774 $\frac{GJ}{ton}$ Converting to primary energy, $1.315\frac{GJ}{ton} + (0.774 \times 3)\frac{GJ}{ton}$
Pyro processing—electrotechnology (Pathway 2)	7.74	From Table 4 of Kaewwichit et al. (2017), total energy of the new microwave assisted method is 2.58 x 10^6 kJ per MT. It is concluded that the energy calculated is for a MT of clinker given it is compared with the theoretical energy for conventional clinker production process in the paper. Converting to primary energy using a grid factor of 3, $2.58 \times 10^6\frac{kJ}{ton} \times \frac{1}{10^6} \times 3 = 7.74\frac{GJ}{ton}$
Clinker cooling	0.068	From Table 1 of Atmaca and Yumrutas (2015), it is calculated that grate coolers consume 6.3 kWh/MT. This is corroborated with the ECRA (2009) study, which states that grate coolers consume 3–6 kWh/MT of clinker. The heat recovered from the clinker cooler exhaust has been neglected. Converting to primary energy using a grid factor of 3, $6.3\frac{kwh}{ton} \times 0.0036\frac{GJ}{kwh} \times 3 = 0.068\frac{GJ}{ton}$

Process	Test Total Primary Energy (GJ Therm/MT of clinker)	Calculation Description Test
Finish Grinding	0.425	From estimates provided by Cembureau (1997), modern ball mills may use 32–37 kWh/t. From DOE (2017), for a clinker production of 120,371 million lb/year, the cement production is 131,513 million lb/year in the United States for dry processing. Converting to energy per MT of clinker production and to primary energy, $36 \frac{kwh}{ton} \times 0.0036 \frac{GJ}{kwh} \times \frac{131{,}513}{120{,}371} \times 3 = 0.425 \frac{GJ}{ton}$

Table B.3 compares the savings achieved by microwave-assisted technology for limestone sintering over conventional methods during lab-scale testing and when scaled up for deployment.

Table B.3. Comparison of Savings Achieved by Microwave-Assisted Technology

Process	Savings Compared With Conventional Methods	Sources
Lab Test—Limestone	18–23%	(Fall, et al. 2011), Appendix A
Scale-up Test—Limestone	10%	(Fall, et al. 2011), Table 4

Estimating Adoption Potential for Electric Cement Processing Technology

Figure B.1 presents a projected technology adoption curve, derived from the methodology laid out by Hanes et al. (2019). The inputs for the model are based on the adoption profile followed by electric arc furnace (EAF). The EAF used by the iron and steel sector was determined to be a suitable technology to model the adoption of an electric kiln, given the similarity in technical and scaling challenges. Further, a maximum adoption rate of the electric kiln technology is assumed to be 75%, with adoption starting in 2030.

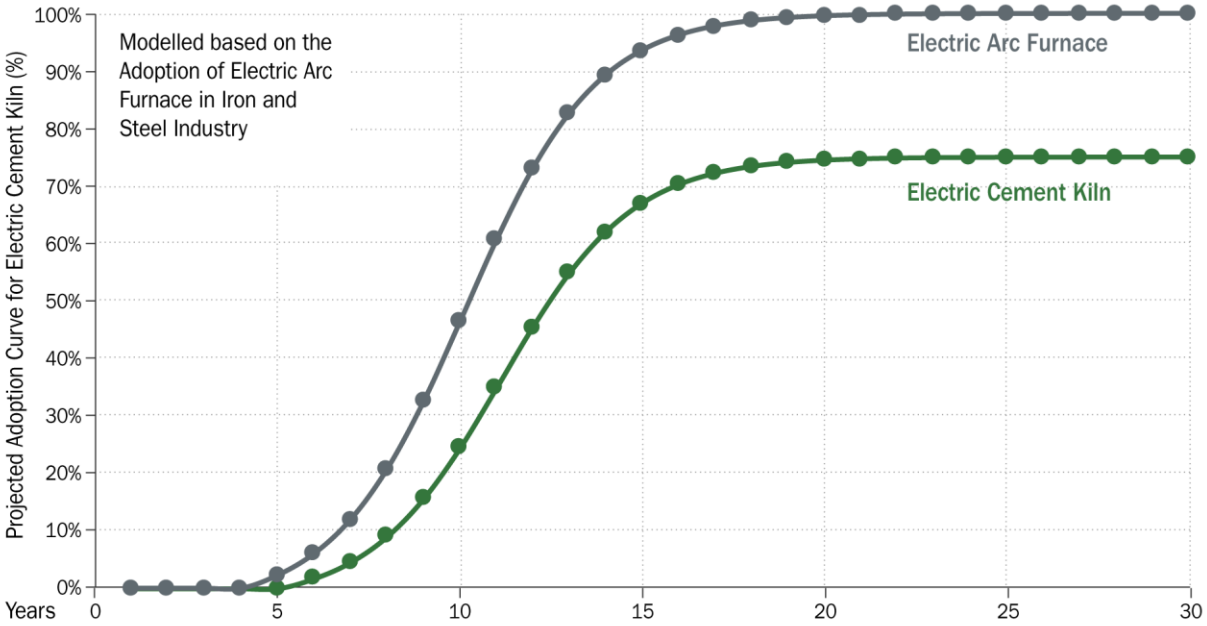


Figure B.1. Projected technology adoption curve for electric cement kiln

Note: Figure derived from the methodology laid out by Hanes et al. (2019)

Reference and High-Renewable Grid

Figures B.2 and B.3 provide a comparison of the two Reference Grid and High-Renewable Grid scenarios.

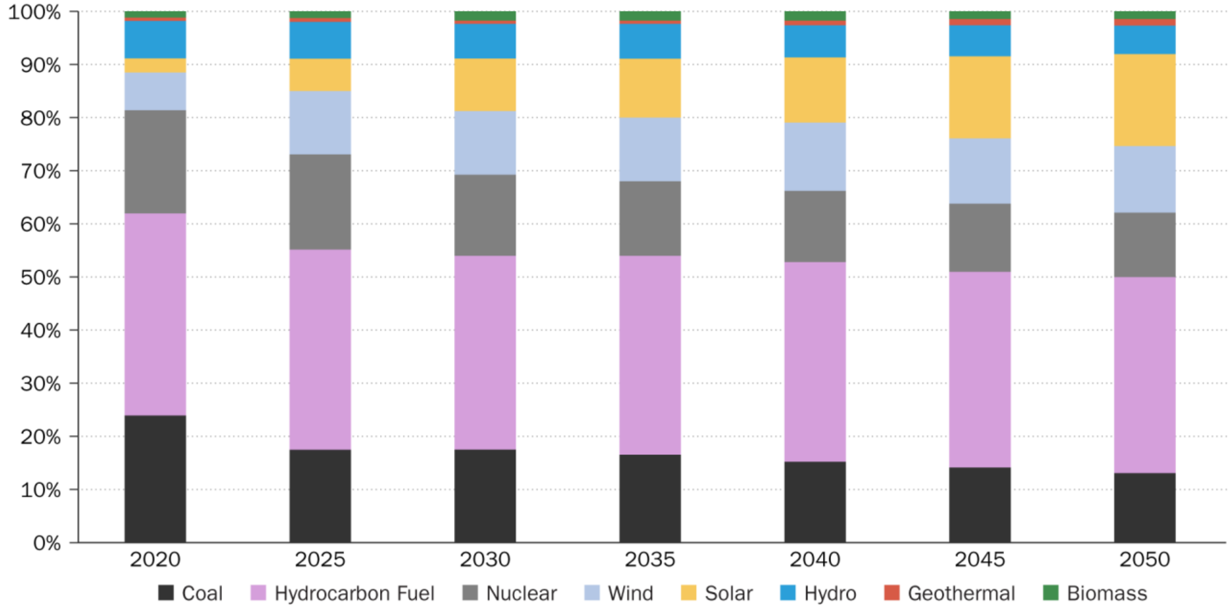


Figure B.2. Reference Grid Projection

Note: Based on data in U.S. EIA (2020a).

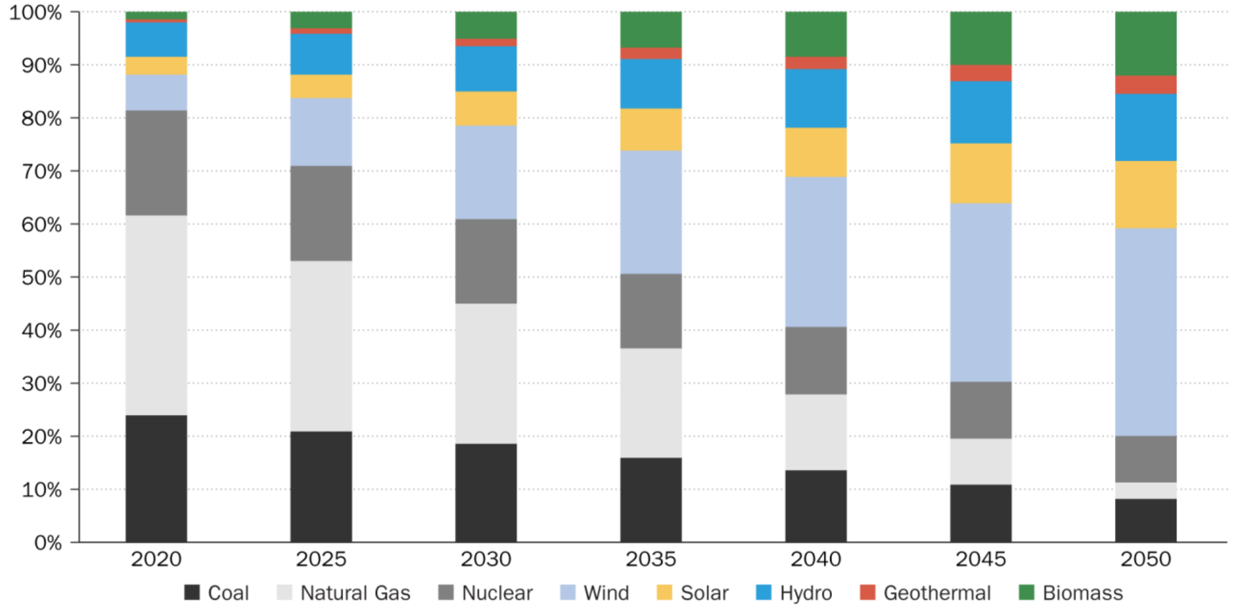


Figure B.3. High-Renewable Grid Projection

Note: Based on data in Mai et al. (2012).

Appendix C. Supplements to Case Study: Recycling Plastics—Projecting Costs, Energy Use, and Emissions

Embodied Energy for Polymer Separation and Reuse: The energy savings from the use of recycled polymers varies based on its application. Energy savings are measured by the embodied energy of all materials in the final product plus any processing required in the final manufacturing of the polymer. As end-of-life materials, energy and subsequent emissions are assigned to the recycled polymer using the cut-off approach, where all inputs from the original life of a material are assigned only to that first use. Using an approach such as the expanded system boundary, where energy and emissions are shared across all lifetimes of a material, would confer a large burden on the recycled material due to the relatively larger energy values of a typical polymer (Tuladhar 2019). For this work, we measured only energy, as opposed to including potential GHG emission reductions from the use of a waste product. The methods examined are:

- PVC solvent recovery and reuse
- Tribo-electrostatic charging
- PET upcycling in fiber-reinforced polymer.

PVC Solvent Recovery and Reuse: The energy savings for PVC in solvent recovery is listed on the website of VinyLoop Ferrara S.p.A., the company that had first commercialized the process (VinyLoop Ferrara S.p.A. 2012). The reclaimed PVC product has a total of 75% recycled content to achieve the listed energy savings.

Tribo-Electrostatic Charging: Tribo-electrostatic charging is the only direct separation method examined here. (We include other separation methods being tested for plastic separation, such as solvent dissolution or optical separation based on material color, only as a reference for potential applications. Tribo-electrostatic separation involves the grinding of a polymer to a manageable particle size, washing to remove impurities, drying, charging the particles, and separation (Li, Wu and Xu 2016). This method is applied to mixed streams of plastic to obtain a pure polymer for reuse. The energy inputs for each step are described as follows.

Grinding: The energy for grinding has been examined in several papers, but a median value of those available was chosen for this input (Macko 2012). Use of the alternate energy requirement values would not cause a significant change in the final energy for tribo-electrostatic separation (Mello, Coelho and Calmon 2016) (Zulcão, Coelho and Calmon 2016) .

Charging: The energy for charging is provided by a vibrating box containing the polymer particles, although a fluidized bed may also be used (Park, Jeon and Cho, et al. 2007).The charge applied is on the order of a nano-Coulomb / gram. The energy for rotating the tube (or box vibration) is estimated to be negligible compared to the other inputs required for the separation.

Separation: The voltage applied for separation is listed as 30 kV (Park, Jeon and Yu, et al. 2008). Assuming 20 amperes of electric current and 40 grams/second of material (20 g/s each of two polymers) are passed through the separation zone, estimated electricity consumption is calculated as:

$$\boldsymbol{Energy \approx 30kV * 20\,A * \frac{1\,second}{40\,g} = 15\,MJ\,/\,kg}$$

Separation energy can range from 5 to 150 MJ/kg based on varying the material flow rate from 5 to 200 g/s, with 40 g/s chosen as an average value. This method has not yet been optimized for energy use, as the goal of experimentation was to find the ideal method and parameters for efficient and effective separation of several polymer combinations.

Washing: The washing removes outside contamination and requires little energy, so in this step most of the energy use comes from particle drying. The heat required to evaporate a mass of water equivalent to the polymer weight is used as a proxy for drying energy.

PET Upcycling in Fiber-reinforced Polymer: When PET is upcycled into fiber reinforced polymer (FRP), energy savings vary based on the allocation method for the recycled product and the type of plastic being recycled (Rorrer, et al. 2019). For this example, the allocation method is either a cut-off approach or a waste valuation of the PET being recycled. The PET can be green or clear, and the acrylic for the FRP is either petroleum or bio-based. The energy savings in Table C.1 reflect a clear petroleum-based acrylic PET with the cut-off allocation method. All methods provide some degree of energy savings compared to using virgin PET.

Table C.1. Energy Savings from the Use of Plastic in New Materials

Plastic Waste Handling Technology	Original Energy Use (MJ/kg)	Technology Energy Use (MJ/kg)	Energy Difference Savings (MJ/kg)	Percent Energy Savings (%)
PVC Solvent Separation	3.36	2.60	-0.76	23%
Upcycled PET Fiber Reinforced Polymer	88	45	-43.00	49%
Triboelectric Charge Separation	-	15	15	-

Appendix D. Supplements to Case Study: Recycling EV Lithium-Ion Batteries—Evaluating the Role of Innovation in Battery Economics and Circularity

Table D.1. Static Baseline Used for Comparing Scenarios

Case	BAU Static Baseline Case
Description	Basic recycling assuming current (2020) batteries (with some increase in size but not composition)
Vehicle sales	BNEF passenger xEV vehicle sales to 2040 (McKerracher, et al. 2020), linear increase 2040 to 2050, EIA historical data 2010 to 2017 (EIA 2020a) (https://www.eia.gov/outlooks/aeo/data/browser/#/?id=48-AEO2020&sourcekey=0)
Battery chemistry	Static at BNEF 2020 values
Innovation (relative to static baseline)	None
Technology adoption	Increasing battery size, see sales and retirements for time period. Battery capacities (kWh), weight, and range (miles) are derived from the Argonne National Laboratory - BatPac4.0 19FEB2020 Model (https://www.anl.gov/tcp/batpac-battery-manufacturing-cost-estimation)
Battery Sizes	Based on all-electric miles from EIA AEO2019 projections
Battery life	Gradual increase from 8 to 10 years; no second life. Ten-year life starting in 2020. Source: Richa et al. (2014), Chevy Bolt EV 2020, 8 years/100,000 miles (https://www.chevrolet.com/electric/bolt-ev); Hyundai Kona EV 2020, 10 years/100,000 miles (https://www.hyundaiusa.com/us/en/vehicles/kona-electric/compare-specs); Tesla 2020 models, 8 years/100,000–150,000 miles (varies by model) and 70% battery capacity retention (https://www.tesla.com/support/vehicle-warranty)
Disassembly	To cell assembly level (module casing and wiring removed - materials recoverable at the module level and pack level)
Collection rate	N/A
Electrolyte recycling	No
Refurbishment	No
Battery recycling	N/A

Table D.2. Battery Chemistries Included in this Study

Chemistry	Cathode $/kg	Include for Direct Recycling—Aggressive Adoption Case	Include for Direct Recycling—Low-Co Adoption Case[3]	Co Content (~%)	Ni Content (~%)
LFP (lithium, iron, phosphate)	14	No	No	0	0
LMO (lithium, manganese oxide)	10	No	No	0	0
NCA (nickel, Co, aluminum)	21.5	Yes	Yes	14	73
NMC 333 (nickel$_{33}$, manganese$_{33}$, cobalt$_{33}$)	21	Yes	Yes	31	30
NMC 532	20	Yes	Yes	18	45
NMC 622	20.6	Yes	Yes	18	54
NMC 811	22	Yes	Yes	9	72
NCAplus[1,4]	21[2]	Yes	No	5	82
NMC 955 (nickel$_{90}$, manganese$_{5}$, cobalt$_{5}$)[1]	21[2]	Yes	No	5	81
NMCA[1] (nickel, manganese, cobalt, aluminum)	21[2]	Yes	No	6	75

[1] Not included in BatPac4.0

[2] Price assumed to be roughly the same as BatPac4.0 cost for NMC 622 at full adoption

[3] Direct recycling of batteries with Co percentage of 9% or greater

[4] Designation for Tesla/Panasonic very-low Co NCA battery (e.g., https://techcrunch.com/2020/07/30/panasonic-boosts-energy-density-trims-cobalt-in-new-2170-battery-cell-for-tesla)

Sources are Argonne National Laboratory BatPac4.0 19FEB2020 Model (https://www.anl.gov/tcp/batpac-battery-manufacturing-cost-estimation) and NREL analysis. Co and Ni content based on McKerracher et al. (2020) and BatPac4.0 19FEB2020.

Table D.3. Recycling Process Material Recovery Fractions

Material	Pyrometallurgy	Hydrometallurgy	Direct Physical	Notes
Copper	0.9	0.9	0.9	
Steel	0.9	0.9	0.9	
Aluminum		0.9	0.9	
Graphite		0.9	0.9	
Plastics		0.5	0.5	
Lithium		0.9		Li+ in product
LCO			0.9	
NMC333			0.9	
NMC622			0.9	
NMC811			0.9	
NCA			0.9	
LMO			0.9	
LFP			0.9	
Cobalt	0.98	0.98		Co2+ in product

Material	Pyrometallurgy	Hydrometallurgy	Direct Physical	Notes
Nickel	0.98	0.98		Ni2+ in product
Manganese		0.98		Mn2+ in product
PF6		0.5	0.5	Electrolyte Organics

Source: EverBatt 2019 Model, Table 1.11 Recovery efficiency (https://www.anl.gov/egs/everbatt) (5/23/2019)

Table D.4. Prices and Recycling Costs for Materials Recoverable from Li-ion Batteries

GREET (baseline prices)			LME (sensitivity prices)		
Material	**Unit prices ($/kg)**	**Metals Price ($/MT) used in analysis**	**Material**	**Unit prices ($/kg)**	**Metals Price ($/MT) used in analysis**
CoSO4	19.51	51,300	Co (2/1/21)	41.240	41,200
NiSO4	4.29	11,300	Ni (2/1/21)	17.807	17,800
Li2CO3	7.9	42,100	LiOH20 (1/28/21)	9	53,100
Refurbished Batteries					
Current Price (50% of Price of a New Battery)			**Future Price (50% of Price of a New Battery)**		
Refurbished batteries	New $137/kWh	$68.50/kWh	New $100/kWh	$50/kWh	
Recycling Process Costs					
Process	**Source**		**Cost**	**Unit of Measure**	
Pyrometallurgy	EverBatt 2019 Model. https://www.anl.gov/egs/everbatt (5/23/2019), 5,000 MT/year input cells		3.69	$/kg cells	
Hydrometallurgy			3.09	$/kg cells	
Direct Recycling			3.57	$/kg cells	
Battery refurbishment	(Neubauer, et al. 2015) Assumptions; average of 3.4 kWh/module; advanced diagnostics; lowest-cost scenario (short handling and short testing); average of low and high cell fault rate cases		20	$/kWh	

Sources: GREET (https://greet.es.anl.gov/index.php), London Metals Exchange (LME) (https://www.lme.com/Metals), EverBatt 2019 Model (https://www.anl.gov/egs/everbatt) (5/23/2019), Bloomberg predicts <$100/kWh by 2023 (https://about.bnef.com/blog/battery-pack-prices-cited-below-100-kwh-for-the-first-time-in-2020-while-market-average-sits-at-137-kwh)

Table D.5. Scenario 1: Design for Disassembly

Case	Design for Disassembly (20%)
Description	Battery design facilitates rapid state-of-health screening and sorting. Batteries that meet refurbishment criteria (assumed to be 20% of EOL EV batteries) are disassembled and modules are reassembled into new vehicle batteries, which have a lifespan of one-half the original lifespan of the batteries.
Vehicle sales	Same as static baseline.
Battery chemistry and capacity	Same as static baseline.
Battery life	New batteries are assumed to have the same lifespan characteristics as the static baseline. Refurbished batteries have one-half the expected life of new batteries.
Disassembly	Battery design facilitates rapid state-of-health screening and sorting. Batteries that meet refurbishment criteria (based on state-of-health monitoring during their lifetime) are routed to rigorous testing and refurbishment. Battery design features, such as bolted rather than glued assembly, would allow batteries to be rapidly disassembled to the cell level for repackaging into new modules. However, this technology is not assumed for this case study.
Collection and reverse supply chain	95% collection, labeling and state-of-health monitoring for all EOL batteries. This translates to a refurbishment rate of ~21% for all collected batteries.
Electrolyte recycling	Yes
Refurbishment	Yes, assume 20% of all EV EOL batteries starting in 2030. Modules that meet the criteria are repackaged into new batteries.
Battery recycling	Pyrometallurgy with 98% recovery of Ni and Co for nonrefurbished and EOL refurbished batteries. Batteries are not refurbished a second time. Li recovery from electrolyte of refurbished batteries.

Table D.6. Scenario 2: Design for Recycling

Case	Design for Recycling
Description	Basic recycling assuming current batteries (with some increase in size but not composition).
Vehicle sales	BNEF passenger xEV vehicle sales to 2040, linear increase from ~9M/year to 2050, Argonne National Laboratory historical data 2010 to 2017 (Gohike and Zhou 2020).
Battery chemistry	Static at BNEF 2020 values.
Battery sizes	Based on all-electric miles from EIA AEO2019 projections and BatPac4.0 19FEB2020 Model (https://www.anl.gov/tcp/batpac-battery-manufacturing-cost-estimation).
Battery life	Gradual increase from 8 to 10 years. Batteries made from recovered cathode material are assumed to have the same lifespan as new material.
Disassembly	To cell level (module casing and wiring removed; materials recoverable at the module level and pack level), cells are separated, crushed, and recycled to recover intact cathode.
Collection and reverse supply chain	95% collection rate for recycling, Only batteries with high-value cathode chemistry are slated for direct recycling.
Electrolyte recycling	Yes
Refurbishment	No
Battery recycling	Pyrometallurgy, 98% recovery of Ni and Co for batteries not sent to direct recycling, direct recycling with full recovery of Li for design-for-recycling batteries. Rate determined by tech adoption.
Tables and figures	Nominal rate of tech adoption = 100% adoption of design for recycle cells by 2030 for “high value” cathode materials = >= $20/kg in 2020; See Table 5. BatPac4.0 19FEB2020 Model (https://www.anl.gov/tcp/batpac-battery-manufacturing-cost-estimation).

Table D.7. Scenario 3: Material Substitution

Case	Low-Co Batteries
Description	Recovery of additional materials to compensate for loss of high-value Co in the EOL stream (EverBatt 2019 assumptions).
Vehicle sales	BNEF passenger xEV vehicle sales to 2040, linear increase to ~9.5M/yr to 2050, Argonne historical 2010 to 2017 (this is the same assumption as for other cases).
Battery Chemistry	Smoothed BNEF estimates (McKerracher, et al. 2020). BNEF to 2030 and smoothed to a steady value by ~2035.
Innovation (relative to static baseline)	Low-Co Battery adoption with hydrometallurgy recovery of most cell materials for low-Co chemistries. Basic pyrometallurgy dominant Umicore process 2020–2030 with recovery of Ni, Co, no Li, (5,000 MT/year/facility). Plots for active+ materials; Active+ = Li in cathode, Co, Ni, Mn (BatPac4.0).
Technology adoption	Hydrometallurgy for low-Co batteries (see below) and Pyrometallurgy for high-Co batteries. TOXCO/Retriev hydrometallurgy dominant process with recovery of Li, 4,500 MT/year capacity 2030–2040, 10,000 MT/yr starting in 2040.
Battery Sizes	Based on all-electric miles from EIA AEO2020 projections (EIA 2020a, Table 38) (https://www.eia.gov/outlooks/aeo/tables_ref.php).
Battery life	Gradual increase from 8 to 10 years; no second life. Ten--year life starting in 2030 for lifespan (https://www.energy.gov/sites/prod/files/2016/06/f32/es000_howell_2016_o_web.pdf; https://www.myev.com/research/buyers-sellers-advice/evaluating-electric-vehicle-warranties).
Disassembly	To cell assembly level (module casing and wiring removed - materials recoverable at the module level and pack level). Crushing of cells.
Collection rate	95% collection rate for recycling.
Electrolyte recycling	PF6 and Li are recovered in Hydrometallurgy. For low-Co batteries, which have lower metals value (i.e., low-Co <18%), batteries will be sent to hydrometallurgy. Low-Co batteries are: LFP, LMO, NMC811, NCA, NCA+, NMC955, NMCA.
Refurbishment	No
Battery recycling	Pyrometallurgy for higher-Co batteries (Co % of cathode >= 18%, 98% recovery of Ni and Co, no Li recovery; hydrometallurgy with the following recovery for low-Co batteries - Al (90%), Li (90%), Co (98%), Mn (98%), Ni (98%), electrolyte organics (50%).

Table D.8. Value of Cathode by Chemistry

Chemistry	Battery 1	Battery 2	Battery 3	Battery 4	Battery 5	Battery 6	Battery 7
NCA	$1,404	$1,735	$ 2,060	$2,387	$650	$2,276	$3,239
NCAplus	$1,371	$1,695	$2,013	$2,332	$635	$2,223	$3,164
NMC 622	$1,616	$1,975	$2,334	$2,693	$718	$2,424	$3,770
NMC 333	$1,954	$2,388	$2,822	$3,256	$868	$2,931	$4,559
LFP	$1,506	$1,840	$2,175	$2,509	$669	$2,258	$3,513
LMO	$1,240	$1,515	$1,791	$2,066	$551	$1,860	$2,893
NMC 811	$1,465	$1,790	$2,116	$2,441	$651	$2,197	$3,418
NMC 532	$1,797	$2,196	$2,595	$2,995	$799	$2,695	$4,193
NMC 955	$1,398	$1,709	$2,019	$2,330	$621	$2,097	$3,262
NMCA	$1,371	$1,695	$2,013	$2,332	$635	$2,223	$3,164

Source: BatPac4.0 19FEB2020 Model (https://www.anl.gov/tcp/batpac-battery-manufacturing-cost-estimation)

Environmental Impact

Figures D.1 and D.2 highlight the relative contribution of the major processes involved in the life cycles of virgin NMC111 and NMC811 production, respectively, to the overall environmental impact. The impacts associated with battery cathode production play a significant role for most of the impact categories (e.g., global warming, terrestrial acidification, mineral resource scarcity, fossil resource scarcity, water consumption). In addition, copper and graphite anode are major sources of the environmental impact associated with virgin battery production for both the cathode chemistries, indicating the need to recycle these materials and consequently improve recycling rates and efficiencies for the BEV sector.

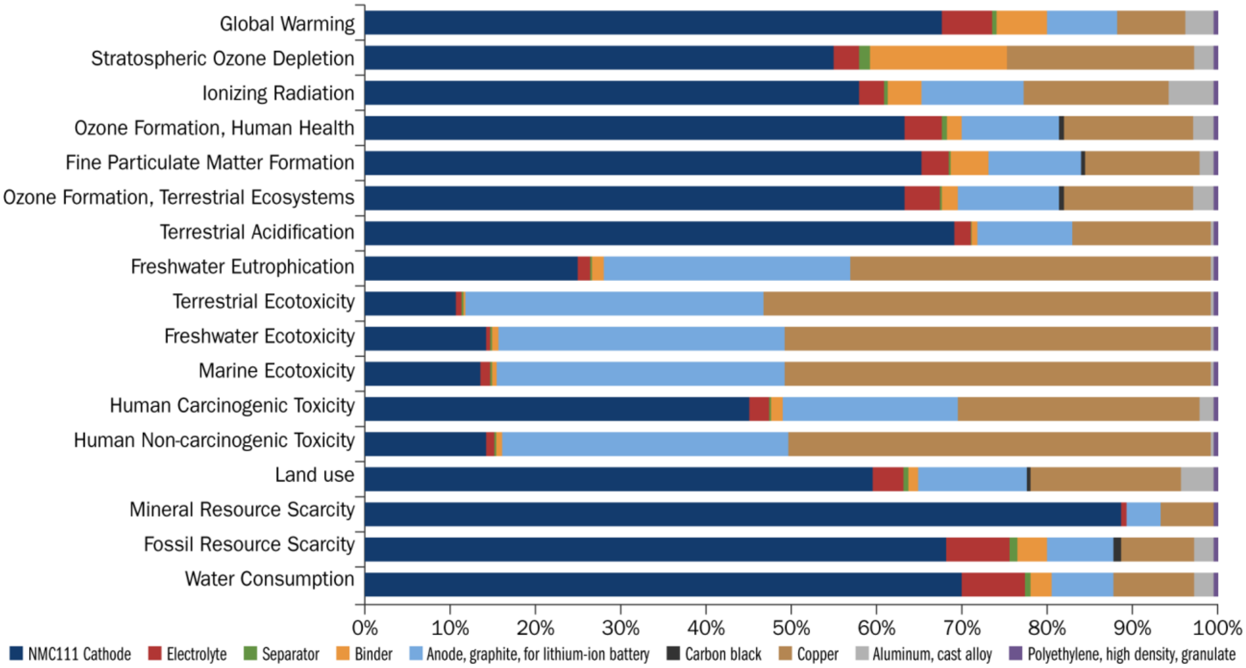


Figure D.1. Hotspot analysis of environmental impacts from virgin NMC111 cell production

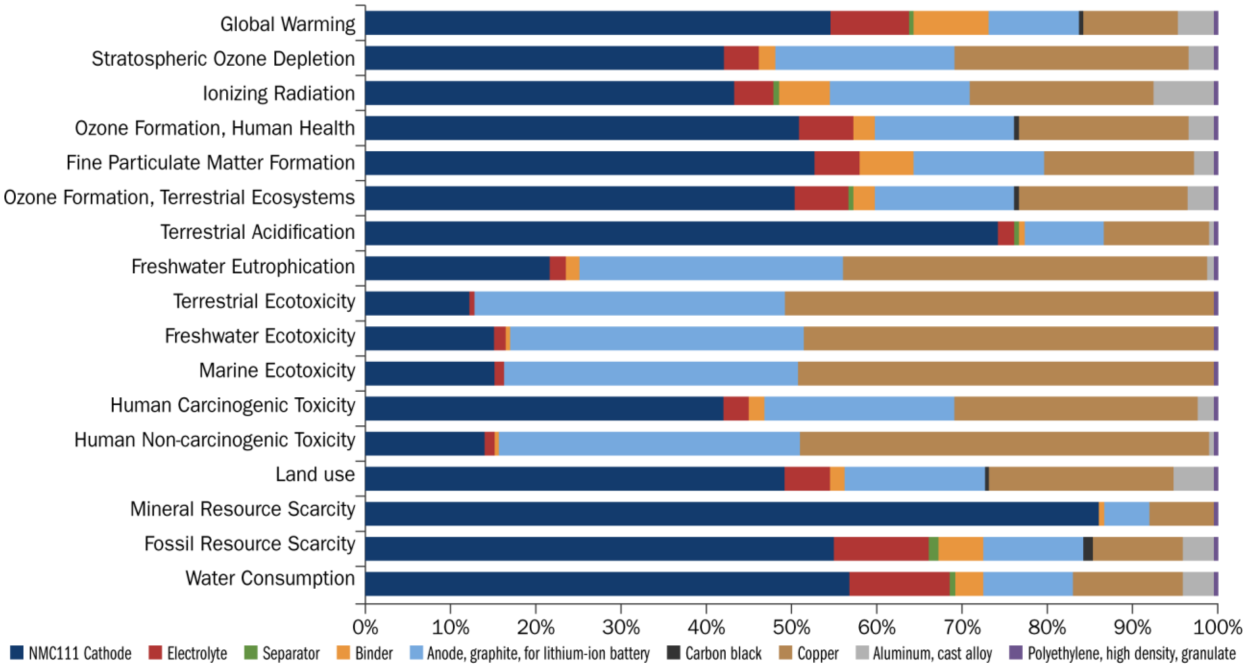


Figure D.2. Hotspot analysis of environmental impacts from virgin NMC811cell production

Table D.9. Life Cycle Impact Analysis for LiB NMC811 and LiB With and Without Recycling (see Figure 54)

Impact Category	Unit	LiB with NMC811 Chemistry	LiB with NMC811 Chemistry and Recycling	LiB with NMC111 Chemistry	LiB with NMC111 Chemistry and Recycling
Human noncarcinogenic toxicity (kg 1,4-DBC eq)	kg 1,4-DBC e	115.53	52.78	122.96	56.30
Global warming (kg CO_2 eq)	kg CO_2 eq	63.47	19.74	102.06	55.59
Fossil resource scarcity (kg oil eq)	kg oil eq	15.64	5.94	27.18	16.87
Freshwater ecotoxicity (kg 1,4-DCB eq)	kg 1,4-DCB e	22.94	10.354	23.87	10.49
Mineral resource scarcity (kg Cu eq)	kg Cu eq	11.49	-0.54	15.18	2.39
Ionizing radiation (kBq Co-60 eq)	kBq Co-60 eq	7.48	3.38	10.99	6.64
Marine ecotoxicity (kg 1,4-DBC eq)	kg 1,4-DBC e	7.78	3.52	8.10	3.56
Land use (m2a crop eq)	m2a crop eq	2.87	0.96	4.14	2.11

Impact Category	Unit	LiB with NMC811 Chemistry	LiB with NMC811 Chemistry and Recycling	LiB with NMC111 Chemistry	LiB with NMC111 Chemistry and Recycling
Terrestrial acidification (kg SO_2 eq)	kg SO2 eq	1.97	0.78	1.75	0.48
Water consumption (m^3)	m3	0.89	0.35	1.59	1.02
Terrestrial ecotoxicity (kg 1,4-DCB eq)	kg 1,4-DCB e	0.30	0.14	0.31	0.14
Ozone formation, Terrestrial ecosystems (kg NOx eq)	kg NOx eq	0.20	0.04	0.30	0.14
Ozone formation, Human health (kg NOx eq)	kg NOx eq	0.19	0.04	0.30	0.14
Freshwater eutrophication (kg P eq)	kg P eq	0.14	0.06	0.16	0.07
Fine particulate matter formation (kg PM2.5 eq)	kg PM2.5 eq	0.09	0.03	0.13	0.07
Human carcinogenic toxicity (kg 1,4-DBC eq)	kg 1,4-DBC e	0.05	0.02	0.06	0.03
Stratospheric ozone depletion (kg CFC11 eq)	kg CFC11 eq	4.032E-05	1.40E-05	5.81E-05	3.00E-05

Appendix E. Supplements to Case Study: Recovering Critical Materials from NiMH Batteries—Assessing Costs and Environmental Impacts

List of Critical Materials

Table E.1. U.S. Department of the Interior's List of Critical Minerals

U.S. Critical Materials				
Aluminum (bauxite)	Antimony	Arsenic	Barite	Beryllium
Bismuth	Cesium	Chromium	Cobalt	Fluorspar
Galium	Germanium	Graphite (natural)	Hafnium	Helium
Indium	Lithium	Magnesium	Manganese	Niobium
Platinum group metals	Potash	Rare earth elements (REE) group	Rhenium	Rubidium
Scandium	Strontium	Tantalum	Tellurium	Tin
Titanium	Tungsten	Uranium	Vanadium	Zirconium

Source of data: Federal Register (Petty 2018)

Price Reference Data

Table E.2. Product Prices

	Ce/La	Nd/Pr	Misch (battery)	Ni	Mixed Iron	Ni-Scrap	Mixed Hydroxides
Units	($/kg)	($/kg)	($/kg)	($/kg)	($/kg)	($/kg)	($/kg)
Value	4.384	65.318	21.92	18.045	3.284832	4.409171	1.84

Sources: Rare-earth metal prices: https://www.metal.com/Rare-Earth-Metals; Scrap metal prices: https://www.scrapmetalpricer.com; https://www.metalary.com/scrap-metal-prices; Mixed hydroxides: https://www.zauba.com/import-mixed-hydroxide/hs-code-29420090-hs-code.html

Economic Model Factors

Table E.3. Direct Factors for Capital Equipment Cost Estimates

	Delivery	Installation	Instruments and Controls	Piping	Electrical Systems	Buildings and Services	Yard Improvements	Service Facilities
Disassembly	0.10	0.45	0.26	0.18	0.1	0.56	0.2	0.55
Anode Material Recycling	0.10	0.39	0.26	0.31	0.1	0.18	0.10	0.55
Cathode Material Recycling	0.10	0.39	0.26	0.31	0.1	0.18	0.10	0.55
Rare-Earth Separations	0.10	0.39	0.26	0.31	0.1	0.18	0.10	0.55

Source of data: Peters, Timmerhaus, and West (2002)

Table E.4. Indirect Factors for Capital Equipment Cost Estimates

	Engineering and Supervision	Construction Expenses	Legal Expenses	Contractors' Fee	Contingency	Working Capital
Disassembly	0.32	0.34	0.04	0.19	0.37	0.75
Anode Material Recycling	0.32	0.34	0.04	0.19	0.37	0.75
Cathode Material Recycling	0.32	0.34	0.04	0.19	0.37	0.75
Rare Earths Separations	0.32	0.34	0.04	0.19	0.37	0.75

Source of data: Peters, Timmerhaus, and West (2022)

Table E.5 Miscellaneous Cost Factors for Fixed Operating Cost Estimates

	Taxes	Insurance	Maintenance	Supplies
Value*	0.02	0.01	0.06	0.15
Basis	Fixed Capital ($)	Fixed Capital ($)	Fixed Capital ($)	Maintenance

*Default data from Peters, Timmerhaus, and West (2022)

Table E.6. Utilities Cost

Utilities	
Purchased electricity ($/kWh)	0.07
Natural gas ($/GJ)	3
Steam ($/Kg)	0.008
Hazardous Waste ($/kg)	0.203
Nonhazardous Waste ($/kg)	0.0504
Cooling Water ($/m3)	0.112
Process Water ($/m3)	0.74

Source of data: Peters, Timmerhaus, and West (2022)

Table E.7. Raw Material Costs

Material	Units	Value
PC88A	($/kg)	6[1]
Kerosene	($/L)	0.966975[2]
HCl (31%)	($/kg)	0.09[3]
NaOH (50%)	($/kg)	0.33[4]
NaOCl (9%)	($/kg)	0.46[4]

[1] https://www.made-in-china.com/showroom/lyzhongda1109/product-detailsMbmBxyHZTrS/China-PC88A-P507-.html; [2]https://www.nyserda.ny.gov/Researchers-and-Policymakers/Energy-Prices/Kerosene/Average-Kerosene-Prices; [3]www.sunsirs.com/uk/prodetail-355.htmlwww.sunsirs.com/uk/prodetail-355.html; [4] (Sabatini, et al. 1994)

NiMH Battery Composition

Table E.8. Battery Elemental Composition by Weight

Material	Sources						
	(Holmberg 2017)*	(Rodrigues and Mansur 2010)*	(Rodrigues and Mansur 2010) ***	(Reutschi, Meli and Desilvestro 1995)	(Reutschi, Meli and Desilvestro 1995)	(Sabatini, et al. 1994)	This Study
Ni	0.735	0.528	0.25–0.46	0.36–0.42	0.38–0.4	0.292	31%
Fe		0.009	0.20–0.25	0.22–0.35	0.06–0.09	0.435**	7%
La	0.104	0.115	0.01–0.07	0.08–0.1	0.07–0.08	0.032	5%
Ce	0.028	0.034	0.004–0.05	0.08–0.1	0.07–0.08	0.002	1%
Pr	0.006	0.032	0.003–0.013	0.08–0.1	0.07–0.08	0.014	0%
Nd	0.018	0.109	0.01–0.041	0.08–0.1	0.07–0.08	0.002	1%
Co	0.061	0.051	0.025–0.04	0.03–0.04	0.02–0.03	0.017	2%
Mn	0.024	0.014	0.008–0.03	0.02–0.03	0.03–0.04	0.008	2%
Al	0.010	0.011	0.005–0.20	0.02–0.03	0.03–0.04	0.003	1%
O				0.15–0.17	0.16–0.18	0.043	3%
H						0.003	10%
K		0.109	0.024–0.031	0.01–0.02	0.02–0.03		1%
Ti		0.001					5%
Zn	0.013	0.003	0.001–0.016				12%

* Casing not included; ** Mostly from casing; *** Multiple references

Candidate Facility Locations

Candidate facility locations provide a set of destination options, for batteries from collection centers or materials from intermediate processing facilities, from which the optimization model selects the final facility locations. Figure E.1 shows an example set of candidate locations, created by randomly sampling in proximity of U.S. highways to ensure easy access for transportation, and filtered to remove redundancy by limiting one candidate location per county. Each location can accept multiple plants as long as they are not the same type.

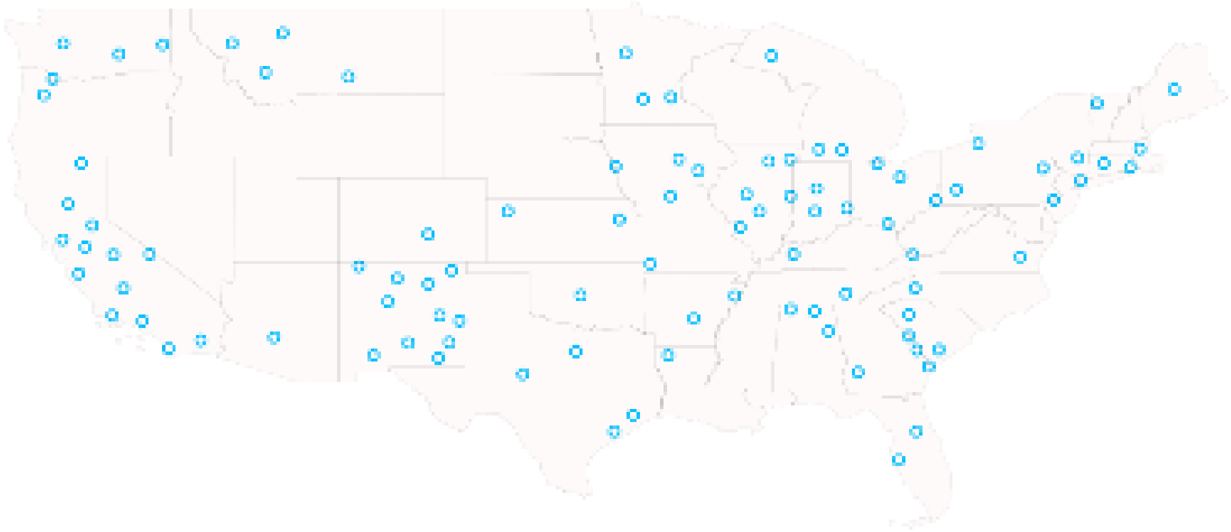


Figure E.1. GIS plot illustrating sample candidate locations for each of the recycling facility types.

Note: These locations are randomly generated and filtered to exclude multiple facilities within a single county, and ensure locations are adjacent to the highway network.

Life Cycle Analysis Data

Table E.9. Recycling Facility Material, Energy, and Emission Intensities

Plant Category	Relative Feed Mass Ratio *	Energy GJ/MT Feed
Disassembly facility	1	1.14*
Anode material recycling facility	0.485	5.6*
Cathode material recycling facility	0.3	9.1*
Rare earths separations facility	0.08	25.7**

* Derived from (Sabatini, et al. 1994); ** (Talens and Mendez 2013)

Tables E.10 to E.13 provide details of life cycle analysis results.

Table E.10. Conservative EOL Battery Supply Case, ILCD 2011 Impact Assessment Method

Impact Category	Reference Unit	Result
Acidification	molc H+ eq	5.735E+00
Climate change	kg CO2 eq	1.033E+03
Freshwater ecotoxicity	CTUe t	1.155E+04
Freshwater eutrophication	kg P eq	7.980E-01
Human toxicity, cancer effects	CTUh	6.725E-05
Human toxicity, non-cancer effects	CTUh	4.427E-04
Ionizing radiation E (interim)	CTUe	6.389E-04
Ionizing radiation HH	kBq U235 eq	1.894E+02
Land use	kg C deficit	1.363E+03
Marine eutrophication	kg N eq	1.102E+00
Mineral, fossil & ren resource depletion	kg Sb eq	1.103E-01
Ozone depletion	kg CFC-11 eq	3.882E-04
Particulate matter	kg PM2.5 eq	1.134E+00
Photochemical ozone formation	kg NMVOC eq	2.835E+00
Terrestrial eutrophication	molc N eq	1.051E+01
Water resource depletion	m3 water eq	1.719E+01

Table E.11. Conservative EOL Battery Supply Case, CExD Impact Assessment Method

Impact Category	Reference Unit	Result
Nonrenewable, fossil	MJ	1.452E+04
Nonrenewable, metals	MJ	2.751E+02
Nonrenewable, minerals	MJ	5.670E+01
Nonrenewable, nuclear	MJ	2.782E+03
Nonrenewable, primary	MJ	4.947E-01
Renewable, biomass	MJ	2.352E+02
Renewable, kinetic	MJ	2.374E+02
Renewable, potential	MJ	9.514E+02
Renewable, solar	MJ	1.816E-01
Renewable, water	MJ	5.305E+03

Table E.12. Optimistic EOL Battery Supply Case, ILCD 2011 Impact Assessment Method

Impact Category	Reference Unit	Result
Acidification	molc H+ eq	5.735E+00
Climate change	kg CO2 eq	1.045E+03
Freshwater ecotoxicity	CTUe t	1.162E+04
Freshwater eutrophication	kg P eq	7.990E-01
Human toxicity, cancer effects	CTUh	6.725E-05
Human toxicity, non-cancer effects	CTUh	4.456E-04
Ionizing radiation E (interim)	CTUe	6.446E-04
Ionizing radiation HH	kBq U235 eq	1.903E+02
Land use	kg C deficit	1.408E+03
Marine eutrophication	kg N eq	1.109E+00
Mineral, fossil & ren resource depletion	kg Sb eq	1.111E-01
Ozone depletion	kg CFC-11 eq	3.905E-04
Particulate matter	kg PM2.5 eq	1.139E+00
Photochemical ozone formation	kg NMVOC eq	2.863E+00
Terrestrial eutrophication	molc N eq	1.059E+01
Water resource depletion	m3 water eq	1.721E+01

Table E.13. Optimistic EOL battery supply case, CExD impact assessment method

Impact Category	Reference Unit	Result
Nonrenewable, fossil	MJ	1.472E+04
Nonrenewable, metals	MJ	2.766E+02
Nonrenewable, minerals	MJ	5.759E+01
Nonrenewable, nuclear	MJ	2.785E+03
Nonrenewable, primary	MJ	5.022E-01
Renewable, biomass	MJ	2.359E+02
Renewable, kinetic	MJ	2.376E+02
Renewable, potential	MJ	9.525E+02
Renewable, solar	MJ	1.878E-01
Renewable, water	MJ	5.311E+03

The transportation cost was obtained by combining diesel truck cost per mile data (Murray and Glidewell 2019), single unit long haul truck mileage data (Wang, Elgowainy and Burnham 2020) weight capacity data (TBGFS 2020).

Table E.14. Transportation Cost

Parameter	Value
Cost ($/km/MT)	0.097
Energy (J/km/MT)	2.72E6

Glossary

Al	Aluminum
CaO	Calcium Oxide
CaO3	Calcium Carbonate
Ce-La	Cerium-Lanthanum
Circular economy	An industrial system that is restorative or regenerative by intention and design, replacing the end-of-life (EOL) concept with restoration, shifting to renewable energy, and eliminating toxic chemicals, which impair reuse. It aims to eliminate waste through the superior design of materials, products, systems, and related business models. (Kirchherr, Reike, and Hekkert (2017))
Clean energy	Energy sources that include wind, solar, geothermal, water, bioenergy, nuclear, and hydrogen and fuel cells.[71]
Co	Cobalt
Construction demolition and debris	“Construction and demolition (CD&D) debris is a type of waste that is not included in municipal solid waste (MSW). Materials included in the C&D debris generation estimates are steel, wood products, drywall and plaster, brick and clay tile, asphalt shingles, concrete, and asphalt concrete. These materials are used in buildings, roads and bridges, and other sectors.”[72]
cradle-to-cradle	“Cradle to cradle’ goes beyond ‘cradle to grave’ and conforms more to the model of the circular economy. In a cradle to cradle model products would be designed in a way so that at the end of their initial life they can be readily reused, or recycled, and therefore avoid landfill altogether.”[73]

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[71] “Clean Energy,” DOE, https://www.energy.gov/clean-energy.

[72] “Construction and Demolition Debris: Material-Specific Data,” EPA, https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/construction-and-demolition-debris-material.

[73] “Cradle to Cradle,” European Commission, https://knowledge4policy.ec.europa.eu/glossary-item/cradle-cradle_en.

cradle-to-gate	"Cradle to gate is a boundary condition associated with embodied carbon, carbon footprint and LCA studies. A study to these boundaries considers all activities starting with the extraction of materials from the earth (the cradle), their transportation, refining, processing and fabrication activities until the material or product is ready to leave the factory gate."[74]
cradle-to-grave	"Cradle to site is a boundary condition associated with embodied carbon, carbon footprint and LCA studies. A study to these boundaries includes the cradle to site results but also includes the GHG emissions associated with the in use of the material or product (maintenance) and the end of life (disposal, reuse, recycling)."[75]
Critical material	"Materials with a high demand in industry, but a global supply that is unstable or uncertain due to one or more risk factors. Certain substances provide essential capabilities, such as light emission or magnetism, and when the supply of one of these substances is at risk, it becomes a "critical" material. The DOE has identified five rare earth elements—neodymium, europium, terbium, dysprosium, and yttrium—as critical materials essential for America's transition to clean energy technologies. The DOE has identified two additional elements, lithium, and tellurium, as "near-critical" materials. These non-rare-earth materials play an indispensable role in emerging energy storage and battery technologies, such as hybrid and electric vehicles, wind turbines, and photovoltaic thin films."[76]
Cu	Copper
Cumulative energy demand	"The Cumulative Energy Demand (CED) of a product represents the direct and indirect energy use throughout the life cycle, including the energy consumed during the extraction, manufacturing, and disposal of the raw and auxiliary materials" (M. A. Huijbregts 2006)
Di	Didymium

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[74] "Environmental Glossary of Terms and Definitions," Circular Ecology, https://circularecology.com/glossary-of-terms-and-definitions.html.

[75] "Environmental Glossary of Terms and Definitions," Circular Ecology, https://circularecology.com/glossary-of-terms-and-definitions.html.

[76] https://cmi.ameslab.gov/materials/factsheet

Embodied energy	"Embodied energy is the energy associated with the manufacturing of a product or ser-vices. This includes energy used for extracting and processing of raw materials, manufacturing of construction materials, transportation and distribution, and assembly and construction. Cradle-to-grave approach in calculating embodied energy also includes energy required for refurbishing and maintaining infrastructures during their service life and for demolition and waste management at end of life" (Tuladhar 2019)
End of life (EOL)	Stage of a product's life cycle when it is no longer used for its intended application and is recycled or disposed of
Energy efficiency	"Energy efficiency is using technology that requires less energy to perform the same function."[77]
energy-related emissions	"Emissions from combustion of waste materials derived from petroleum and emissions from some types of geothermal power plants. … Electricity-related CO2 emissions are based on electric power sector electricity sales to the other sectors and the emissions associated with the generation of that electricity."[78]
Environmental justice	Environmental justice (EJ) is the "fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation and enforcement of environmental laws, regulations and policies."[79]
Externality	"A cost or benefit that affects a party that did not choose to incur that cost." (U.S. DOE 2015) This cost can be economic, environmental, or social in nature.
Fe2O3	Ferric Oxide
Food loss waste (FLW)	Food intended for human consumption that has left the food supply chain
Food supply chain (FSC)	On-Farm Production, Manufacturing Food, Distribution, Wholesale and Retail (W&R), and Consumption

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[77] "Use of Energy Explained: Energy Efficiency and Conservation," EIA, https://www.eia.gov/energyexplained/use-of-energy/efficiency-and-conservation.php.

[78] "Frequently Asked Questions (FAQS): What are U.S. Energy-Related Carbon Dioxide Emissions by Source and Sector?" EIA, https://www.eia.gov/tools/faqs/faq.php?id=75.

[79] "Learn About Environmental Justice," EPA, https://www.epa.gov/environmentaljustice/learn-about-environmental-justice.

Fossil resource depletion	Abiotic resource depletion indicator specific to fossil resources and measure in kg of oil equivalents (kg oil eq).
Global warming potential	“The Global Warming Potential (GWP) was developed to allow comparisons of the global warming impacts of different gases. Specifically, it is a measure of how much energy the emissions of 1 ton of a gas will absorb over a given period of time, relative to the emissions of 1 ton of carbon dioxide (CO_2).”[80]
Global warming	“Global warming is the long-term heating of Earth’s climate system observed since the preindustrial period (1850–1900) due to human activities, primarily fossil fuel burning, which increases heat-trapping greenhouse gas levels in Earth’s atmosphere. The term is frequently used interchangeably with the term climate change, though the latter refers to both human- and naturally produced warming and the effects it has on our planet.”[81]
Li	Lithium
Life cycle	The “[c]onsecutive and interlinked stages of a product (or service) system, from raw material acquisition or generation from natural resources to final disposal” (ISO 2016)
Life cycle assessment	The “compilation and evaluation of the inputs, outputs, and the potential environmental impacts of a product system throughout its life cycle” (ISO 2006a)
Material criticality	The determination of which materials that flow through an industry or economy are most important to the production process. Material criticality evaluation criteria consist of three dimensions: supply risk, vulnerability to supply restriction, and environmental implications (T. R. Graedel 2012)
Material efficiency	“The use of material resources per unit output for a product system” (U.S. DOE 2015)

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[80] “Understanding Global Warming Potentials,” EPA, https://www.epa.gov/ghgemissions/understanding-global-warming-potentials.

[81] “Overview: Weather, Global Warming and Climate Change,” NASA, https://climate.nasa.gov/resources/global-warming-vs-climate-change/.

Material footprint	"Material footprint is a consumption-based indicator of resource use. It is defined as a global allocation of used raw material extraction to the final demand of an economy" according to Weidmann et al. (2013) (LidijaČuček 2015).
Material resources	Biotic or abiotic resources. While both types of resources have finite total stocks,[82] biotic resources are intrinsically renewable due to the continual biological activity that regenerates them. In contrast, abiotic resources are not generally renewable without deliberate human input.[83] Biotic material resources in the economy include those from agriculture, forestry, fisheries, and hunting. Abiotic material resources in the economy include metals; nonmetals and their mineral ores; bulk minerals, such as those used in industry and construction; and fossil energy carriers (Giljum, et al. 2011).[84]
Mineral resource depletion	An abiotic resource depletion indicator that is specifically accounting for loss of mineral resources (kg Cu eq).
Mn	Manganese
Municipal solid waste (MSW)	Municipal Solid Waste (MSW)—more commonly known as trash or garbage—consists of everyday items we use and then throw away, such as product packaging, grass clippings, furniture, clothing, bottles, food scraps, newspapers, appliances, paint, and batteries. This comes from our homes, schools, hospitals, and businesses. https://archive.epa.gov/epawaste/nonhaz/municipal/web/html/
NdFeB	Neodymium Iron Boron
Ni	Nickel

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[82] Stocks of a resource in this report encompass the amount of the resource currently in anthropogenic and/or natural use. For instance, the number of vehicles already on the road in a time period constitutes the "stock" of vehicles in that period.

[83] Some abiotic resources are renewed through naturally occurring chemical, physical, or biological mechanisms (i.e., weathering, mineralization, and accumulation). However, the timescales at which these natural phenomena occur are too long to be meaningful in the context of sustainable human consumption.

[84] Water (specifically, fresh water) represents a "middle ground" that is not the (main) product of biological systems; it is produced at important levels by these systems, and the timescales at which abiotic phenomena (e.g., evaporation, transport, precipitation, and filtering through mineral systems) replenish it is very meaningful for human consumption, unlike other abiotic resources. Moreover, the LCA community does not yet use an elementary flow nomenclature for tracking water that is standard for other abiotic resources. Therefore, water is typically handled differently from other resources, though efforts are underway to address this. See for example, Berger and Finkbeinger (2010).

Non-process or energy-related emissions	Emissions from fuel combustion
Oil equivalent	"A barrel of oil equivalent (BOE) is a term used to summarize the amount of energy that is equivalent to the amount of energy found in a barrel of crude oil."[85]
Onsite energy	Energy consumed at a plant, building or facility site.
Post-consumer waste	Post-consumer is the waste generated at the end of life of products. (ISO 2016)
Pre-consumer waste	Pre-consumer waste is waste that is produced during the manufacturing process, sometimes referred to as manufacturing waste or in-plant waste. (ISO 2016)
Primary energy	"Energy in the form that it is first accounted for in a statistical energy balance before any transformation to secondary or tertiary forms of energy. For example, coal can be converted to synthetic gas, which can be converted to electricity; in this example, coal is primary energy, synthetic gas is secondary energy, and electricity is tertiary energy."[86]
Primary material	Material produced: from a mine or a souce extracted from the earth, such as that extracted from mined ore, extracted from ores, natural brines, or ocean water. Also called virgin material.[87]
Process-related emissions	"Process emissions generally include emissions from chemical transformation of raw materials and fugitive emissions" (U.S. EPA n.d.).
ReCiPe	ReCiPe is life cycle impact assessment (LCIA) methodology that considers metallic and nonmetallic elements, bulk resources, and fossil resources.[88]
Reverse supply chain	The series of activities required to retrieve a used product from a customer and either dispose of it or reuse it.
Re-X	Reuse, repair, remanufacturing, refurbishment, recycling, and other circular processes.

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[85] "Barrel of Oil Equivalent (BOE)," Investopedia, Adam Hayes, updated May 20, 2021, https://www.investopedia.com/terms/b/barrelofoilequivalent.asp.

[86] EIA Glossary. https://www.eia.gov/tools/glossary/index.php?id=Primary%20energy.

[87] Based off of definition for primary metals from https://www.mindat.org/glossary/primary_metal.

[88] For more information about the ReCiPe model, see "LCIA: the ReCiPe model," National Institute for Public Health and the Environment (The Netherlands), https://www.rivm.nl/en/life-cycle-assessment-lca/recipe.

Secondary Material	Material recovered from scrap and products recovered from end of life.
Sustainable development	Sustainable development is defined globally as meeting the needs of the present without compromising the well-being of future generations (United Nations General Assembly 1987). For the United States, sustainable development means a commitment "to create and maintain conditions under which humans and nature can exist in productive harmony, that permit fulfilling the social, economic and other requirements of present and future generations" (U.S. NEPA 1969).
Sustainable manufacturing	Sustainable manufacturing is the "creation of manufactured products through economically sound processes that minimize negative environmental impacts while conserving energy and natural resources" (U.S. EPA 2020). The Department of Commerce extends that definition to require safety for employees, communities, and consumers (Howard n.d.)(slide 6).
Trade-off	Occurs when the reduction of impacts in one stage of the life cycle, geographic location, or impact category result in increased impacts in another. When not explicitly accounted for can be considered burden-shifting.
UN Sustainable Development Goals	The Sustainable Development Goals or Global Goals are a collection of 17 interlinked global goals designed to be a "blueprint to achieve a better and more sustainable future for all." The SDGs were set up in 2015 by the United Nations General Assembly and are intended to be achieved by the year 2030.
Upcycling	The process of transforming by-products, waste materials, useless, or unwanted products into new materials or products perceived to be of greater quality or value.

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