

LIFE CYCLE IMPACTS OF ALKALINE BATTERIES WITH A FOCUS ON END-OF-LIFE

A STUDY CONDUCTED FOR THE NATIONAL ELECTRICAL MANUFACTURERS ASSOCIATION

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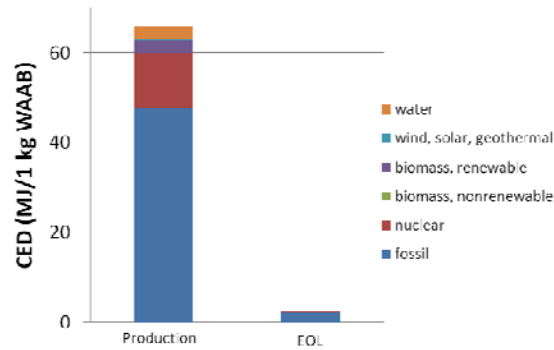
EXECUTIVE SUMMARY

Approximately 80% of portable batteries manufactured in the US are so-called alkaline dry cells with a global annual production exceeding 10 billion units. Today, the majority of these batteries go to landfills at end-of-life. An increased focus on environmental issues related to battery disposal, along with recently implemented battery directives in Europe and Canada and waste classification legislation in California, has intensified discussions about end-of-life battery regulations globally. The logistics of battery collection are intensive given the large quantity retired annually, their broad dispersion, and the small size of each battery. Careful evaluation of the environmental impacts of battery recycling is critical to determining the conditions under which recycling should occur. This work compares a baseline scenario involving landfilling of alkaline batteries as municipal solid waste with several collection schemes for battery recycling through pyrometallurgical material recovery. Network models and life cycle assessment methods enable the evaluation of various end-of-life collection and treatment scenarios for alkaline batteries.

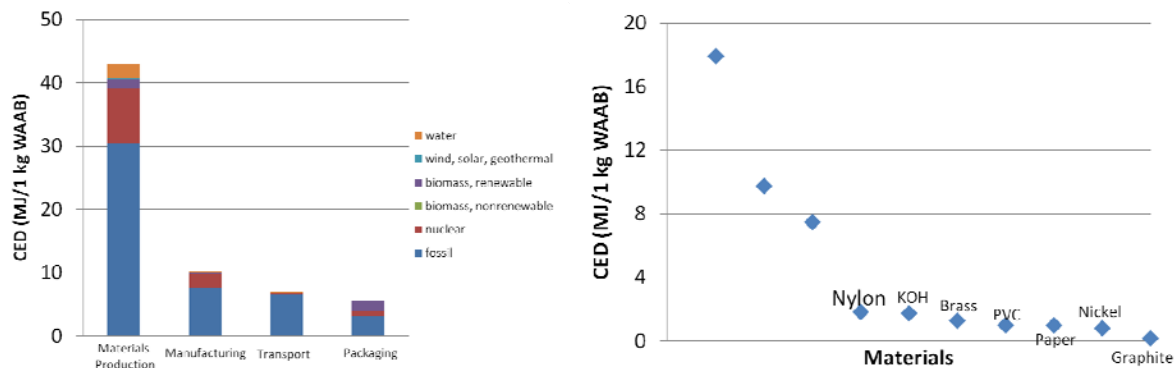
The study employs life-cycle assessment techniques in accordance with the ISO 14040 standard. This approach is applied to each end-of-life scenario to provide a comprehensive means of accounting for environmental impacts. The scope of the analyses includes raw material extraction and refining, battery manufacturing, end-of-life disposition, and transportation. Secondary life cycle inventory datasets from the ecoinvent 2.2 data set were employed when primary data were not available. Metrics evaluated include Cumulative Energy Demand (CED), Global Warming Potential (GWP), and Ecosystem Quality, Human Health, and Resources indicators (the latter three are damage categories from the Ecoindicator 99 methodology). Because of the country-specific grid mixes used in the recycling scenarios, the amount of radioactive waste is also presented as a metric of interest in for the recycling scenarios.

To summarize the full life cycle implications of alkaline batteries, the production of raw materials dominates the life cycle with the transport of those raw materials to manufacturing having a minimal environmental impact as shown in the figures below using the proxy environmental impact metric, CED. A few materials dominate this materials production impact, with manganese dioxide, zinc, and steel having the highest impacts.

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios



LIFE CYCLE IMPACT USING CED FOR 1 KG SALES-WEIGHTED AVERAGE ALKALINE BATTERIES INCLUDING PACKAGING (~30 BATTERIES)



PRODUCTION IMPACT (LEFT GRAPH) AND MATERIALS IMPACT (RIGHT GRAPH) USING CED FOR 1 KG SALES-WEIGHTED AVERAGE ALKALINE BATTERIES INCLUDING PACKAGING (~30 BATTERIES)

There are complex and uncertain potential impacts associated with placing primary alkaline batteries in landfills at end-of-life and recycling may reduce those impacts, but may cause additional burdens that outweigh benefits. The primary factors that drive the environmental impact of alkaline battery recycling, compared to the baseline landfill scenario, include the recycling technology used, the amount of materials recovered, and the state of the recovered materials. Study findings indicate recovering more than zinc for metal value (replacing virgin material) is important for reducing environmental impact and technologies involving high temperature are energy intensive. The principal drivers of end-of-life environmental performance of batteries vary depending on the metrics of impact assessment. Findings indicate metrics around energy and carbon are strongly dependent on recovery technologies, metrics for ecosystem quality depend on landfill scenario assumptions as well as the materials benefits associated with recycling. For the latter, if one assumes little to no landfill leachate resulting from batteries (in other words, batteries remain intact in the landfill or leachate is collected and not of concern over the time horizon considered), the main benefit from recycling stems from the recovery of zinc, manganese and steel. The same is true for metrics around human health. Several conclusions related to the transportation of the batteries to their end-of-life disposition step are also of interest. For the recycling scenario where batteries are dropped off by individual consumers at a retail or municipal

facility, the assumed allocation of the trip (dedicated versus non-dedicated) drives the burden. In addition, the fact that there are just three facilities modeled for North America that are able to take alkaline batteries for recycling drives the large transportation burden associated with taking batteries from an intermediary consolidation facility to the recycling facility. This study modeled several scenarios for collection and recycling of batteries that resulted in an overall net environmental burden as compared to landfilling as well as several that resulted in an overall environmental benefit. When considering metrics related to energy or global warming potential, the recycling scenarios appear more environmentally burdensome whereas for metrics of human health and ecosystem quality, the recycling scenarios appear more environmentally beneficial. This study does not intend to explicitly compare the technologies; rather this work investigates the specific sites and contexts under which the technologies operate.

For the collection burden associated with battery recycling, the greatest burden was associated with the scenario wherein individual consumers dropped batteries off at municipal locations, such as transfer stations. The crucial assumptions were around the degree of dedication for this leg of the journey, and literature indicated a higher likelihood of a dedicated trip for municipal drop-off along with greater distances traveled than the retail drop-off. Municipal drop-off was on average 3-4 MJ and 0.2 kg CO₂ eq/kg of batteries disposed greater than retail drop-off (1.3×10^{-7} DALY, 0.015 pdf*m²yr, and 0.25 MJ surplus /kg surplus for the other metrics). Curbside pickup (both MSW and Recycle co-collection) for the recycling scenarios was determined to be lower in impact than both of the drop-off scenarios.

For the materials recovery burden at the end-of-life, the overall conclusions indicated that the burden or benefit of recycling depends on the scenario assumed in recycling. The materials recovery credit associated with zinc often drives the environmental benefit of all the recycling scenarios, so investigating how this material acts within the metals market would offer further insight to its benefit. Five overall scenarios were examined with their specific transportation, fuel mix and materials recovery contexts. Scenario A, involving a metal fuming furnace in the Northwest, employed coal and a hydropower electrical grid to recover zinc for metal value with steel and manganese dioxides reporting to slag which is sold to cement market. It exhibits an environmental benefit as compared to the baseline MSW landfilling scenario for metrics of ecosystem quality for both municipal and retail drop-off. This scenario results in a more significant environmental burden than landfilling as measured by CED, GWP, human health, and resources using any modeled collection method.

Scenario B, located in the Midwest used natural gas and coal, to recover zinc and steel to metal value with some metal value from manganese dioxides and the remainder reporting to slag which is sold to cement market. This scenario exhibits an environmental benefit or neutrality as compared to the baseline MSW landfilling scenario for metrics of ecosystem quality and human health for both municipal and retail drop-off. However, this scenario results in a more significant environmental burden than landfilling as measured by CED, GWP, and resources using any modeled collection method.

Scenario C recovered steel to metal value and zinc and manganese dioxides sold for micronutrient replacement using a lower temperature process powered by natural gas and a Canadian electrical grid. It exhibits an environmental neutrality or slight benefit as compared to the baseline MSW landfilling scenario for metrics of human health and ecosystem quality. For CED, ecosystem quality, and resources

the value of this scenario is generally environmentally burdensome. Therefore, for the scenarios currently used in the US, the impact oscillates between environmentally beneficial and environmentally burdensome depending on the indicator metric investigated and the transportation scenario used.

There were two scenarios investigated that are not currently used in the US. The first of these scenarios is modeled after using the EAF infrastructure in the US where zinc and steel (with volume of manganese) are recovered for metal value and exhibit an environmental benefit compared to a baseline landfilling scenario for all metrics used in this study except for an apparent neutrality in the case of CED and in municipal drop-off for ecosystem quality. Because of the copper poisoning to steel production, this scenario is limited by capacity (although based on the number of batteries recovered, this value would not be exceeded). EAF facilities would require permitting for this scenario to be possible for EOL battery processing in California. This scenario is sensitive to zinc recovery as shown in the scenario analysis. The sensitivity analyses indicate that for the metrics of CED and GWP the burden for recycling with less than 32-40% zinc recovery exceeds the impact of landfilling.

The final scenario, not currently used in the US, recovers zinc, steel and manganese for metal value based on European recycling facilities incorporating low carbon intensity electrical grids from France and Switzerland. The transportation scenario assumes transport by road and ship to the EU. The results indicate for the majority of cases, there is an environmental benefit to this scenario, except for CED and GWP where it is environmentally burdensome, and for resources where the transportation scenario (retail versus municipal drop-off) dictates whether it is a burden or benefit.

When all the examined recycling scenarios are assumed to use the US electrical fuel mix as an electricity source and an average, equivalent transportation burden, Scenario D may demonstrate environmental benefit of recycling compared to landfilling while Scenario A, C and E are more environmentally burdensome and the environmental benefit of Scenario B is metric dependent. For ecosystem quality all the scenarios are environmentally beneficial except for C.

Incineration, as part of MSW management, performs similarly to the hypothetical EAF scenario, except that it is burdensome due to reduced materials recovery and increased transportation burden. Incineration may be preferable to landfilling because of the potential for material recovery.

CONTENTS

Executive Summary.....	2
Chapter 1: Introduction	7
Chapter 2: Introduction to life cycle assessment.....	9
Chapter 3: Overall Goal and scope definition	13
Section 1: Whole Life Cycle Assessment.....	15
Chapter 4: Alkaline battery life cycle assessment	15
Methodology.....	15
Scope.....	15
Data sources and assumptions	16
ecoinvent data gaps	20
Data Quality/Source Matrix	20
Results.....	21
Section 2: Alkaline Battery end of life focus	29
Chapter 5: End-of-Life investigations.....	29
Goal, Scope and Methodology.....	29
Spent Battery Chemistry	30
Logistics assumptions.....	31
Baseline and Curbside Pickup scenarios	31
Consumer Drop-off scenarios (municipal/retail)	33
Previous work - logistics.....	35
Landfilling and Incineration Toxicity issues.....	36
Incineration	37
Landfilling	40
Recycling technologies.....	42
Scenario A	43
Scenario B	44
Scenario C.....	44
Scenario D modeled after recycling with steel in an EAF	44
Scenario E modeled after aggregating European recyclers	45

Chapter 6: End of life scenario analysis results.....	48
Incineration scenario	63
Chapter 7: Parameter Analysis.....	64
Conclusions	72
Recommendations for actions to reduce the environmental impact	76
Life cycle impact.....	76
End of life impact	76
Future Work	76
References	78
Appendix A: NEMA survey questionnaire	81
Appendix B: Detail around LCI	84
Appendix C: Numeric results of Chapter 6.....	93
Appendix D: Figures for 100% Primary offset for zinc and steel.	95
Appendix E: External Reviews.....	102
Reviewer 1	102
Reviewer 2	104
Reviewer 3	108

CHAPTER 1: INTRODUCTION

End-of-life issues for consumer non-durables has become increasingly subject to the critical eye of individuals, local government and producers. Most products follow a linear lifecycle, beginning as raw materials in the earth, passing through refining, manufacturing, and use, and finally returning to the earth in a landfill. While this linear lifecycle has been the norm for many products in the US, increasing focus on environmental issues has drawn attention to the apparent wastefulness of a linear lifecycle. To address this, adding loops to the linear lifecycle, often in the form of reuse, remanufacturing, and recycling, has been proposed. However, these loops, and in particular the recycling loop, are not without debate, as the economic and environmental impacts of such loops are often uncertain.

In the case of alkaline batteries, the economic and environmental impacts of recycling are particularly interesting. In the US, the large quantity of alkaline batteries that are retired each year, the broad dispersion of those batteries, and the small size of each individual battery, makes the logistics of battery collection particularly challenging. The material composition of alkaline batteries adds another layer of complexity to the recycling dilemma, as there are disparate views about whether materials found in alkaline batteries are harmful when landfilled. Although alkaline batteries pass all U.S. EPA hazardous

waste criteria and are therefore not deemed to be hazardous in the U.S., the state of California has deemed them harmful, and some consumers are under the impression that alkaline batteries contain harmful materials¹. Given these and other issues, careful evaluation of both the economic and environmental impacts of alkaline battery collection and recycling is critical prior to deciding whether or not alkaline battery recycling should take place and, if so, under what conditions. In the US, clearly understanding such impacts is particularly relevant, as an increased focus on environmental issues, along with a recently implemented end-of-life battery directive in Europe and regulatory interpretation in California impacting alkalines, has intensified the discussions about end-of-life battery regulations in the US.

As background to the legislation that impacts batteries in the US, the situation in California presents one perspective. Most batteries are considered hazardous waste in California when they are discarded including batteries of all sizes, both rechargeable and single use. Therefore alkaline batteries, as of February 8, 2006, must be recycled, or taken to a household hazardous waste disposal facility, a universal waste handler, or an authorized recycling facility. Large and small quantity handlers are required to ship their universal waste to another handler, a universal waste transfer station, a recycling facility, or a disposal facility. Several other states in the US have legislated a restriction on disposal in landfills of particular rechargeable chemistries such as nickel cadmium, but these do not cover alkaline single use batteries and as of this report writing no other state has legislation banning alkaline batteries from landfills. Canada and the EU mandate collection of alkaline batteries.

This study evaluates the environmental impacts of different end-of-life strategies, such as disposal and recycling for alkaline batteries in the United States. The analysis is divided into two sections. The first section of the analysis encompasses the entire life cycle of the battery, accounting for impacts from production in a manufacturing facility to use and eventual end-of-life treatment. The focus of the second section of the study is more detail on the end-of-life treatment. The scope and approach are outlined in more detail below and in the two sections of the document. The geographic scope of the study includes batteries manufactured and disposed of in the United States. The results of the analysis were generated in accordance with the ISO 14040 standard for life cycle assessments (LCAs).

For the second section of the study, defining multiple scenarios enabled investigation of the implications of several end-of-life treatments for battery recycling. These consist first of a baseline scenario including municipal solid waste (MSW) pickup of batteries with regular household waste accompanied by disposal in a “typical” landfill. This baseline was contrasted with a series of recycling scenarios that included multiple collection schemes and recycling technologies. The collection schemes included curbside with MSW, curbside with municipal recycling, and drop off to both municipal and retail locations. Incineration of batteries with regular household waste will also be commented on throughout this document; however it is not a major focus because of the dominance of landfilling in the US, as shown in Figure 1.

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¹ There is no mercury added to US OEM-produced alkaline batteries, but may be present in trace quantities from other sources such as batteries produced before mercury was not added, imported or counterfeit batteries.

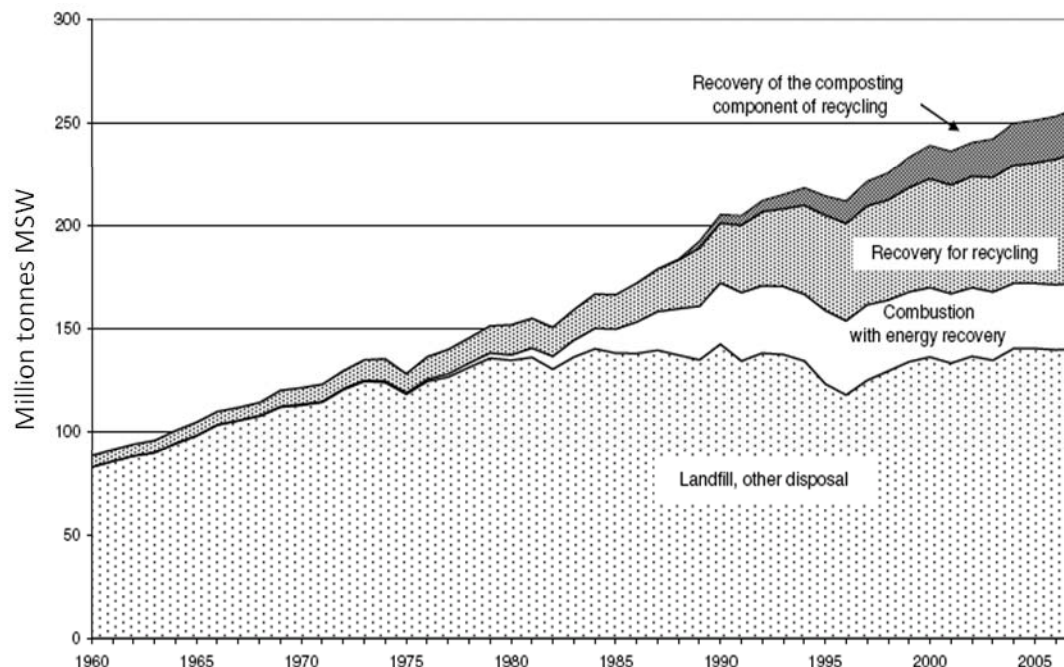


FIGURE 1. MUNICIPAL SOLID WASTE MANAGEMENT FROM 1960 TO 2007. REPRODUCED FROM [1]

The analysis aimed to quantify the total vehicle miles traveled (VMT) by the batteries as a function of collection scheme through a series of network models and to measure the burdens and benefits of treatment at end of life through life cycle assessment techniques. The results describe the impact and the sensitivity of the analysis to several components including the effect of collection scheme, the effect of regional variation in VMT, and the energy needed to treat at end-of-life. Through these scenario analyses, this work explores some of the condition (or conditions) under which recycling demonstrates environmental benefit compared to landfilling.

This document begins with a brief introduction to the LCA methodology, followed by the goal and scope definition of the overall study. The first section describes the scope, methodologies and results of the full LCA for alkaline batteries, and then an investigation into the end-of-life alternatives is shown in the second section along with a discussion of the sensitivity of the results to several key assumptions. The document concludes with a summary of the study results and recommendations for actions that may be taken to reduce the environmental impact associated with the products.

CHAPTER 2: INTRODUCTION TO LIFE CYCLE ASSESSMENT

Life cycle assessment is an approach to analyzing the environmental impact of a product or industrial system throughout its entire life cycle, from cradle to grave. The life cycle under consideration generally encompasses all stages of a product's life, including raw material production, product manufacture, use, and end-of-life disposal or recovery, as depicted in Figure 2. The arrow in Figure 2 demonstrates the transport that takes place between each phase in the life cycle. The comprehensiveness of LCA is one of its strengths; it includes many details that are not part of more focused environmental impact analyses.

However, the complexity and level of detail necessitate a strict adherence to a consistent methodology. A brief overview of the LCA methodology is presented here; more thorough references are available for additional details [2, 3].

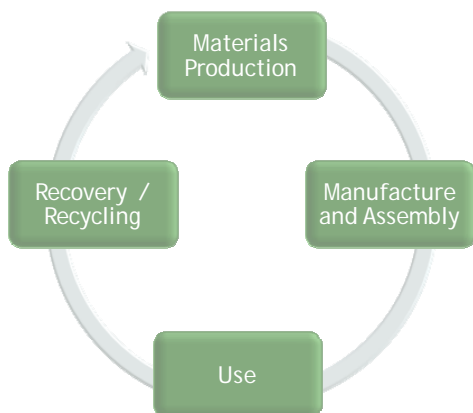


FIGURE 2. PHASES IN A PRODUCT LIFE CYCLE.

The International Organization for Standardization has developed a standard methodology for life cycle assessment as part of its ISO 14000 environmental management series. The LCA standard, ISO 14040 [4], outlines four main steps in an LCA: goal and scope definition, inventory analysis, impact assessment, and interpretation of results. These steps are shown in Figure 3, and explained below.

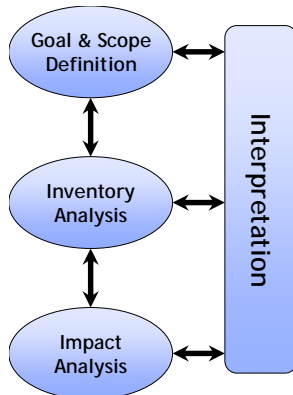


FIGURE 3. LIFE CYCLE ASSESSMENT METHODOLOGY (ADAPTED FROM THE ISO 14040 STANDARD).

- *Goal and scope definition* articulates the objectives, functional unit under consideration, and regional and temporal boundaries of the assessment.
- *Inventory analysis* entails the quantification of energy, water, and material resource requirements and emissions to air, land, and water for all unit processes within the life cycle, as depicted in Figure 4.
- *Impact assessment* evaluates the human and ecological effects of the resource consumption and emissions to the environment associated with the life cycle.

- *Interpretation* includes an evaluation of the impact assessment results within the context of the limitations, uncertainty, and assumptions in the inventory data and the scope.

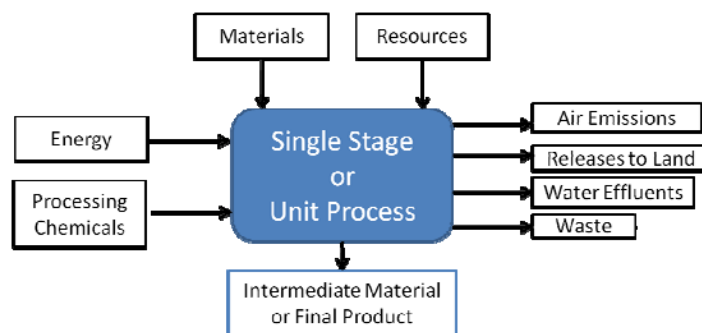


FIGURE 4. INVENTORY ANALYSIS: INFLOWS AND OUTFLOWS OF A UNIT PROCESS.

The accumulation of the life cycle inventory step transforms a detailed list of all the inputs and outputs of the process to and from the technosphere into inputs from and outputs to nature, therefore containing only resources and emissions.. The impact assessment step is particularly challenging because of the difficulty associated with aggregating and valuing numerous types of resource consumption and emissions to the environment. There is uncertainty in the modeling used to produce midpoint and endpoint indicators. There are metrics that include a single attribute such as energy or global warming potential, but there are also metrics that try to capture multiple impacts. The impact assessment methods (and proxy metrics) that were used in this study are described below.²

Single Issue metrics:

- Cumulative Energy Demand (CED) includes all direct and indirect energy consumption associated with a defined set of unit processes. It does not directly account for the impact of non-energetic raw material consumption or emissions to the environment. Values for CED are measured in terms of energy (e.g., joules). Note: CED is a proxy metric and not a formal impact assessment method. This method considers energy from multiple sources, including renewable and non-renewable. For the CED results, all energy sources are presented as both renewable and non-renewable are used. This number is broken down by source where it is of interest. [5]
- Global Warming Potential (GWP) incorporates the impact of gaseous emissions according to their potential to contribute to global warming based on values published in 2007 by the Intergovernmental Panel on Climate Change. The impacts for all gaseous emissions are evaluated relative to carbon dioxide. Impact assessment values for GWP are measured in terms of an equivalent mass of carbon dioxide (e.g., kg CO₂ equivalent) [5].

Multi impact metric:

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² For the recycling technologies, the amount of nuclear waste generated is also presented as a result because of the grid mixes used for some of the technologies.

- Ecoindicator 99 (EI) is a damage-oriented method that calculates environmental impact in three categories: damage to human health, ecosystem quality, and damage to resources. The characterization of damage by inventory items is based on scientific methods (e.g., effects of toxic materials in drinking water on human health) and these damage categories serve as endpoints in ISO 14040. Damage to human health is in units of disability adjusted life years (DALY), implying that different disability caused by diseases are weighted. Ecosystem quality is reported in units of potentially disappeared fraction of plant species (PDF*m²yr). The final category is Resources, which includes assessment of minerals and fossil fuels in units of MJ surplus, or the additional energy requirement to compensate for future ore grade. Damage is then normalized by average European impacts. The egalitarian perspective was used for this study. The valuation of the relative importance of various environmental impact categories is determined by survey responses from an expert panel; these responses determine the relative weighting of all of the impacts. These steps enable all of the impacts to be aggregated into a single value which has the units of “points”, where 1000 points represents the average environmental impact of a European in one year. This European weighting may present a limitation given the US geographic scope of this analysis; however, this impact assessment method is used primarily for its damage categories rather than the single combined metric for this study. Also, because this study examines various options for end of life and is therefore classified as comparative assertion, weighting results are not used for communication. A few different approaches exist, however for the purposes of this study, **the damage category impacts are reported including human health, ecosystem quality and resources [6].**

These methods were used for this work because they provide separate lenses with which to evaluate environmental burden. Global warming potential is of interest to the audience because of the focus on greenhouse gases in some pending legislation. Because of the potential ecosystem quality and human health concerns associated with batteries in landfills, the EI 99 method was also chosen. It can be challenging to have a “feel” for reasonable values calculated by life cycle impact assessment methodologies as such values often represent abstract concepts or non-physical quantities. Indeed, such values are most useful when presented in the context of a comparison so that relative quantities may be evaluated. Several products and processes have been evaluated using the methodologies for LCAs detailed above, and the results appear in Table 1. These values provide some basis of comparison for the results presented in this study.

TABLE 1. LIFE CYCLE IMPACT ASSESSMENT VALUES FROM FOUR LCAS. VALUES CALCULATED USING THE ECOINVENT 2.2 DATABASE.

Product or Process	CED (MJ)	GWP (kg CO ₂ eq)	Human Health (DALY)	Ecosystem quality (PDF*m ² yr)	Resources (MJ surplus)
Production of 25 g PET beverage bottle (20 fl oz/590 ml)	2	0.07	6.8 x 10 ⁻⁸	0.005	0.15
Production of 14 g aluminum beverage can (12 fl oz/350 ml)	3	0.2	2.6 x 10 ⁻⁷	0.007	0.17
100 km fuel consumption in a European passenger car	305	18	2 x 10 ⁻⁵	1	21
Coffee pot: 5 years	5400	220	2 x 10 ⁻⁴	30	190

Now that the concept of life cycle assessment has been briefly outlined, the next section will define the goal and scope of the work undertaken in this study.

CHAPTER 3: OVERALL GOAL AND SCOPE DEFINITION

This life cycle assessment is divided into two sections. The goal of the first section, described in Chapter 4, was to determine the life cycle impact of an industry average alkaline battery, based on input from four battery manufacturers. The whole life cycle of the battery is established as a first goal of the study to provide a context for these end-of-life impacts. The second section, described in the remainder of the report (Chapter 5 & 6) focuses just on the end of life treatment for alkaline batteries. The goal of this second part of the study is to compare different disposal and recycling scenarios for alkaline batteries to weigh the environmental burdens and benefits of each specific situation for battery disposition at end of life. More detail around scope for the second section will be established at the beginning of Chapter 5. The intended audience of the study is NEMA, local and state government agencies including waste management entities, as well as the general environmental community through journal publication. The geographic region of interest for product sales and use is the United States and portions of Canada. System boundaries are defined in accordance with the ecoinvent life cycle inventory database³ unless otherwise specified to include all life-cycle steps from material extraction to end-of-life (this database presents some limitations because of its EU focus, given the US geographic scope of this study). For the first section of the study the cut-off for EOL materials to recycling is applied.

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³ <http://www.ecoinvent.ch/>

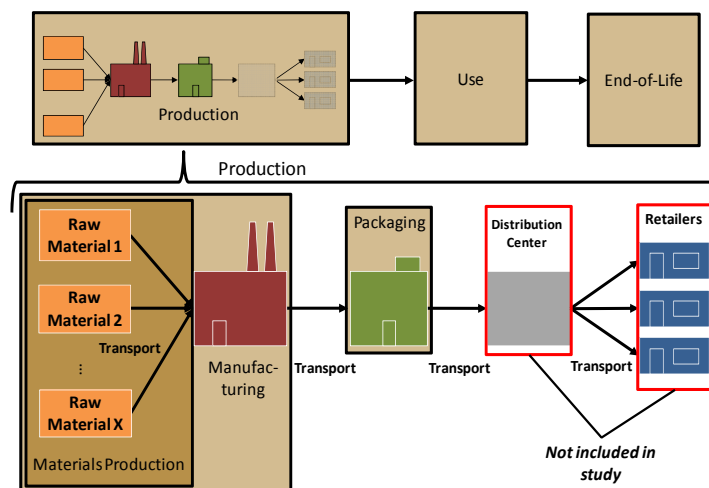


FIGURE 5. PHASES WITHIN THE PRODUCT LIFE CYCLE.

The terminology for the phases within the product life cycle is defined in Figure 5. At the highest level, the life cycle is broken into three phases: production, use, and end-of-life. The production phase is further broken into manufacturing (i.e., battery manufacturing), packaging, and distribution. The environmental impacts of distribution centers and retailers are ignored in this analysis, but the transportation to them is included. Finally, the manufacturing phase is further broken down into the production of raw materials (i.e. acquisition and refining of raw materials) and the actual manufacturing of the battery. Transportation of raw materials from suppliers to the manufacturing facility is included in materials production. For the purposes of this study the use phase contributes no environmental impact because the investigation focuses only on primary alkaline batteries. For primary batteries the beneficial work they do for the consumer in use derives from the chemical potential of the materials contained within the device. As stated previously, the end-of-life (EoL) phase provides the primary area of interest for this particular investigation in the second section of the study. Regardless of the scenario under investigation at EoL the batteries go through some intermediate transportation and consolidation phase and are then taken to a particular disposal facility. The EoL boundaries and scenarios will be fully outlined after the full life cycle results are presented. The functional unit used in this analysis is 1 kg weighted average alkaline batteries, equivalent to 30 batteries; this quantity was chosen as most relevant to a consumer. This weighted average is the sales weighted average of the batteries sizes sold in the US (as described in detail in Chapter 4). For the full life cycle assessment described in chapter 4, impacts are shown of 1 kg of these weighted average batteries including packaging and for the end-of-life analysis the functional unit is the treatment of 1 kg of batteries.

The data for the study will be gathered through the battery manufacturers participating, the potential battery recyclers, literature, modelling efforts and interviews with collection system providers. Therefore the data are of varying quality and are required to be transparent to the research team for interpretation. Specifics on the temporal, geographic and representativeness of these data are provided within the chapters below. The critical review process for this study will be done by three external reviewers sequentially from three different institutions in academia and state government. Those reviews can be found in the appendices. The interpretation of this study involves identifying the

significant issues found from the inventory and analysis steps, evaluating the completeness of the work and providing a description of the gaps and recommendations. As mentioned above, the specific impact assessment methods used in this study were: cumulative energy demand, global warming potential and the three damage categories of Ecoindicator 99. Value choices are made by selecting just these five metrics to present in the results. Future work should look at other characterization approaches.

SECTION 1: WHOLE LIFE CYCLE ASSESSMENT

CHAPTER 4: ALKALINE BATTERY LIFE CYCLE ASSESSMENT

This section describes the full life cycle analysis for the primary alkaline battery to provide the relevant scale of end-of-life compared to the rest of the life cycle. This full alkaline battery LCA is not directly applicable to the second study goal but provides important context. There are other reasons to focus on end-of-life besides its impact on the whole battery life cycle. The set of parameters outlined in the upcoming tables are used for all analyses in this section.

METHODOLOGY

For the alkaline battery life cycle assessment, each phase of the life cycle is identified. Following this, materials and energy are quantified and environmental impacts are calculated for each phase. This section describes the methodology in detail by identifying the scope of the analysis for the alkaline battery and then describing the sources of data – including necessary assumptions. The data used in this analysis was gathered through a survey of the firms participating in the study and then averaged by a statistician within NEMA. The survey used is provided in Appendix A.

SCOPE

As described in Figure 5, the life cycle consists of production, use, and a standard end-of-life treatment. The production phase for the alkaline battery consists of producing the raw materials, transporting the raw materials to the manufacturing facility, manufacturing the battery, transporting the battery to the packaging facility, packaging the battery, transporting the packaged battery to distribution facilities throughout the United States, and finally transporting the packaged batteries to retail facilities. It is assumed that no environmental impact is associated with the use phase of the alkaline battery because it is single use and any emissions to air, land or soil in this use of a battery in a product would be attributed to that product. The end of life treatment for this first section is just taken as standard landfill and incineration without any materials recovery. More detail around end of life is investigated in the second section of the study. For the allocation of recycling materials at end of life (for example in the manufacturing process), the cut off approach is used, so no credit or burden is assigned to the portion of material recycled in that process at end of life. While a scenario analysis on this could be performed, the overall impact of the recycled scrap after manufacturing provides a small amount of the overall impact. The same is true for the recycled packaging at end of life.

A single alkaline battery is actually represented as the **weighted average** of each size of battery (AA, AAA, C, D and 9V) based on percentage sales in 2007 as shown in Table 2. This weighted average is used

to determine the weight of the battery, the bill of materials, the amount of packaging and the weighted distance traveled in each transport step. Finally, the baseline end-of-life scenario assumed for the alkaline battery consists of landfilling and incineration (87% landfill and 13% incineration) [7].

TABLE 2. ALKALINE BATTERY SALES BROKEN DOWN BY SIZE FOR 2007

	2007 Sales	Weight of battery (g)
AA	60%	23
AAA	24%	11
C	4%	71
D	8%	147
9V	4%	45

DATA SOURCES AND ASSUMPTIONS

This section describes the data gathered and processed for implementation. It should be noted that the data shown here do not represent a company-specific battery bill of materials or manufacturing process but instead are an aggregation of data from four separate OEMs. First, the bill of materials is established for a single weighted-average alkaline battery. The mass of a single weighted average alkaline battery (WAAB) is 33 grams. The six top components by mass and each component's mass within the battery calculated from the weighted mass percentages of battery constituents and these 33 grams are shown in Table 3. The chemistry of each of the sizes of alkaline battery (AA, AAA, C, D and 9V) are assumed to be the same (although the weight percentage of materials for the 9V is slightly different). Line 2 contains 35wt% aqueous potassium hydroxide for the electrolyte.

Table 3 also shows the supplier locations for each material (by country) and the one-way distance traveled from that supplier to the manufacturing facility by truck and boat (backhaul distances were ignored). A few supplier locations for each material were provided and the distances reported in the table are the average of each country listed. For overseas shipments, specific ports-of-call were assumed when they were not specified. There was no information provided about manufacturing scrap, so the partial bill of materials shown below is the actual amount of material in the weighted battery. However, a few exceptions to this include 1) it was difficult to determine the amount of water present in the electrolyte as received (described below) and 2) the remaining mass was divided by the "other materials" (brass, plastics [incl. PVC], paper, and galvanized steel) [8].

TABLE 3. TOP TEN COMPONENTS BY MASS WITHIN THE BILL OF MATERIALS FOR A SINGLE WAAB.

No.	Material	Mass (g)	Supplier Locations	Nominal Distance from Supplier	
				Truck (km)	Boat (km)
1	Electrolytic Manganese Dioxide	13	Japan, South Africa, US	1600	11000
2	Potassium hydroxide (35wt% aqueous KOH)	3.7	US	1000	-
3	Graphite	1.2	Brazil, Canada, Switzerland	1130	5900
4	Nickel-Plated Steel	6.0	Japan, Netherlands, US	1600	7700
5	Zinc	5.8	Canada, Japan, US	1450	9000
6	Brass	1.0	-	-	-
7	Galvanized steel	0.52		-	-
8	Nylon	0.51		Not specified	
9	Paper	0.51			
10	PVC	0.51			
	Total weighted battery	33			

Also included as an input in the analysis but not shown in the bill of materials are the excess materials needed in production that ends up as scrap (as reported consisting primarily of steel). Information on the additional material, water and energy inputs and waste outputs (including the scrap materials) from the manufacturing facility were also provided, through the survey, in units of input or output per million weighted average batteries (based on production within the facility). The values for these inputs and outputs from manufacturing are shown in Table 4, allocated by unit to a single WAAB. The inputs to the facility were electricity, natural gas and light fuel oil as well as water. Outputs from the facility were also provided, including the waste for recycling steel, waste to landfill, water treatment, and air emissions in the form of volatile organic compounds.

It is common industry practice that the water used in production is to dilute the as-received 50% potassium hydroxide to the concentration used in the final electrolyte (35%). This water is included in the bill of materials. The incoming water used in production that is not accounted for in the bill of materials in Table 3 is accounted for as an input to the facility and output of wastewater leaving the site.

TABLE 4. INPUTS AND OUTPUTS FROM THE BATTERY MANUFACTURING FACILITY ALLOCATED TO A SINGLE WAAB.

Inputs	Amount per Battery	Units
Water	32	g
Electricity	0.02	kWh
Natural Gas, burned in industrial furnace	25	kJ
Fuel Oil, burned in industrial furnace	9.3	kJ
Outputs	Amount per Battery	Units
VOC	0.02	g
Waste (for recycling)	1.1	g
Waste (for disposal)	0.52	g
Waste Water	32	g

*The number for VOC's in this inventory is observed to be high, but this was as reported from the company survey described above. This turns out to be a minimal contribution to the results.

The next phase of the life cycle is packaging of the battery. Packaging occurs at a facility 460 km from manufacturing (weighted distance by sales). The materials used to package the batteries in “blister packs” of 2 or 4 (depending on the size), as shown in Table 5, are polyvinyl chloride, paperboard and corrugated cardboard for shipping. Approximately 6% of the final mass of the packaged battery is attributed to these packaging materials.

TABLE 5. MATERIALS USED IN THE PACKAGING OF A SINGLE WAAB.

No.	Material	Amount per Battery	Units
1	Polyvinyl Chloride	0.4	g
2	Corrugated board	1	g
3	Paper board	0.8	g

The inputs and outputs from the overall operation of the packaging facility are provided in Table 6 allocated to a single WAAB.

TABLE 6. INPUTS AND OUTPUTS FROM THE PACKAGING FACILITY ALLOCATED FOR A SINGLE WAAB.

Inputs	Amount per Battery	Units
Water	1.7	g
Electricity	5.5	Wh
Natural Gas, burned in industrial furnace	6.1	kJ
Outputs	Amount per Battery	Units
Waste Water	1.7	g

After the batteries are packaged they are shipped to the distribution centers. The nominal distance for transport from packaging to the distribution centers (based again on the weighted average of sales) is 630 km. In addition, 1100 km is used as a weighted average distance from the distribution centers to the retailer. Table 7 provides a summary of each segment of transport from manufacturing to distribution. The mass of the battery up until the packaging facility is 33 g; after packaging, the mass of the battery and packaging is approximately 35 g in total.

TABLE 7. TRANSPORTATION FOR ALKALINE BATTERY FROM MANUFACTURING TO PACKAGING AND FINALLY DISTRIBUTION.

From	To	Nominal Distance (km)	Method
Manufacturing	Packaging	460	Truck
Packaging	Distribution Center	630	Truck
Distribution Center	Retailer	1100	Truck

As mentioned previously, there is no additional environmental impact from the use phase of the alkaline battery since no energy is added beyond the production of the cell detailed above. The final phase in the life cycle is the end-of-life treatment of the battery. This analysis assumed that 13% of alkaline batteries are incinerated at end-of-life as that is the percent of total generation of MSW that is combusted in the US [7]. The remaining percentage of alkaline batteries is landfilled; both scenarios assumed to involve 100 km of MSW vehicle transport. A generic landfilling and incineration scenario is used for this baseline case, just to understand the order of magnitude comparison between production and EoL.

Table 8 summarizes the baseline end-of-life scenario described above for the battery.

TABLE 8. END OF LIFE DESCRIPTION FOR THE ALKALINE BATTERY.

Transport	Distance (km)
Disposal Truck	100
Waste Scenario	Percent
Landfill	87%
Incineration	13%

Table 9 summarizes the end-of-life scenario for the battery packaging, which involves recycling of 30% of the cardboard packaging and the disposal of the remaining packaging through landfill and incineration. Because of the assumption of cut off allocation at end of life no burden or benefit is associated with the quantity of recycled packaging material; a reduced end of life burden is assumed.

TABLE 9. END OF LIFE DESCRIPTION FOR THE ALKALINE BATTERY PACKAGING.

Transport	Distance (km)
Recycle Truck	400
Disposal Truck	100
Material Recycled	Percent
Cardboard	30%
Waste Scenario	Percent
Landfill	87%
Incineration	13%

ecoinvent DATA GAPS

The source of data for implementation of this portion of the analysis is the ecoinvent 2.2 database. A few major assumptions were necessary for implementation and they are mentioned here, especially in the case when an inventory is not available for a particular item in the bill of materials. The first major limitation of ecoinvent is its European focus, while the geographic scope of this study was the US.

One gap in the ecoinvent database was an inventory for manganese dioxide, the major component in the alkaline bill of materials (~40 wt% or 13 g in a WAAB). Previous studies, for example the Defra-commissioned report mentioned earlier, have substituted the manganese inventory for manganese dioxide, altering the mass stoichiometrically [9]. Substituting an inventory for titanium dioxide for manganese dioxide is another possible approach, as both materials can be produced using similar processes. Manganese dioxide is produced through a series of steps, including roasting of manganese ore, dissolution of the roasted manganese in acid, filtration of impurities, and the electrowinning of the final product. The inventory for manganese(III) oxide developed for lithium ion batteries (Mn_2O_3) was used as a proxy. The steel inventory used includes 40% steel from an electric arc furnace and the remainder converted pig iron from a blast furnace. The zinc inventory includes 30% from combined zinc production. The full outline of the inventories used is provided in Appendix B.

A 16-32-tonne truck is assumed to perform all land transport, while all boat transport is performed by transoceanic freight ship, both from the ecoinvent database.

DATA QUALITY/SOURCE MATRIX

Where primary data was not available secondary sources were used, including published reports, specifications, and the ecoinvent LCI database. The quality of the data has been assessed on the following criteria:

- Source--primary or secondary
- Temporal --when was the data collected and over what amount of time was it aggregated

- Representativeness—how closely the data collected represents the supply chain of the system, including geographic and operational considerations

Stage	Data Source	Temporal	Representativeness
Alkaline Battery Materials	Primary data regarding types of material and quantities. Secondary data (ecoinvent) for upstream extraction and processing	Primary data from 2009, based on ecoinvent processes (global mix used when available), Zinc and steel inventories are EU focused	Limits on data based on global focus of ecoinvent inventories
Battery Primary Packaging	Primary data regarding types of material, and quantities. Secondary data (ecoinvent) for upstream extraction and processing	Primary data from 2009, other based on ecoinvent processes	Limits on data based on European focus of ecoinvent data
Battery Manufacturing Facility	Primary data on energy consumption quantities and waste produced. Secondary data (ecoinvent) for inventory, electricity grid mix for US used.	Primary data from 2009, other based on ecoinvent processes	Limits on data based on EU focus of some inventories, however electricity mix US-based
Packaging Facility	Primary data regarding energy consumption quantities and waste produced. Secondary data (ecoinvent) for inventory	Primary data from 2009, other based on ecoinvent processes	Limits on data based on EU focus of some inventories, however electricity mix US-based
Transportation	Primary data regarding transportation distances Secondary data (ecoinvent) for inventory	Primary data from 2009, other based on ecoinvent processes	Limits on data based on European focus of ecoinvent data 50% load factor truck likely overestimate of burden as trucks are likely more full.
Use	No environmental burden associated with use as the chemical energy stored in the battery through use of manganese dioxide and zinc		
Disposal (for this section)	Secondary data (ecoinvent) for disposal scenarios.	Data does not well represent the geographic scope	

RESULTS

The following section describes the results from the full life cycle analysis for 1 kg WAAB including their packaging where the end of life fate is 87% landfill, 13% incineration w/o steel recovery as this is investigated in more detail in the second section of this work. Results describe the impact assessments from the life cycle inventory using Cumulative Energy Demand (CED) in MJ, Global Warming Potential (GWP) in grams of CO₂ equivalent, and Eco-Indicator (EI) midpoints for Human health (in DALY),

Ecosystem quality (in PDF*m²yr) and Resources (MJ Surplus). The methods differ in focus, as CED emphasizes total energy consumption, GWP stresses global warming contributing gases, and EI may highlight perceived human health risks (Human health indicator) or ecosystem toxicity (Ecosystem quality). Furthermore, the final EI damage category, Resources, comments on perceived resource scarcity of a particular material or the upstream impacts associated with that material. In some cases only the results of one impact assessment method are presented. The beginning of this analysis explores the “hot spots” for impact within the life cycle of an alkaline battery, resolving where the biggest impacts are. The full Eco-Indicator 99 breakdown is provided at the end of this section.

Figure 6 shows the relative contribution of each phase on the full life cycle impact and plots the values in for CED and

Figure 7 shows the relative contribution of each phase for GWP.

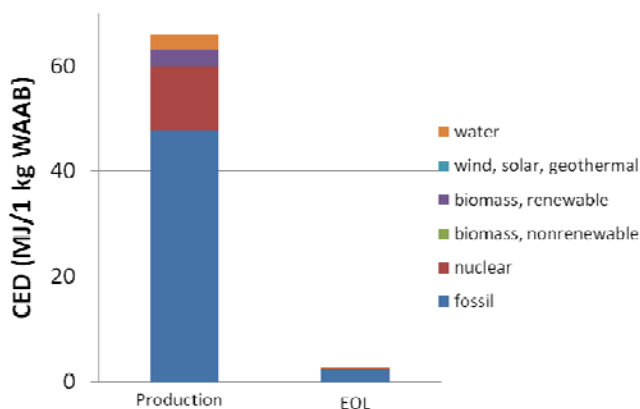


FIGURE 6. LIFE CYCLE IMPACT USING CED FOR 1 KG WAAB INCLUDING PACKAGING(~30 BATTERIES)

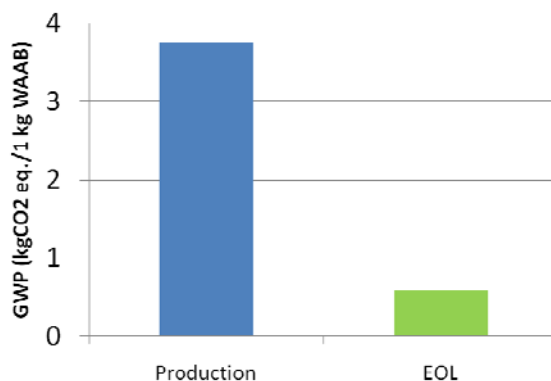


FIGURE 7. LIFE CYCLE IMPACT USING GWP FOR 1 KG WAAB INCLUDING PACKAGING (~30 BATTERIES)

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Table 10 shows the life cycle impact of 1 kg WAAB using all three LCIA methodologies, CED, GWP and the three damage categories of EI 99. Table 10, Figure 6, and

Figure 7 show that the production phase dominates the life cycle impact.

TABLE 10. LIFE CYCLE IMPACT OF 1 KG WAAB, INCLUDING PACKAGING USING THREE LCIA ASSESSMENT METHODS (THE DAMAGE CATEGORIES OF EI 99 ARE SHOWN, HUMAN HEALTH, ECOSYSTEM QUALITY AND RESOURCES).

Life Cycle Phase	CED (MJ/1 kg WAAB)	GWP (kg CO ₂ eq./1 kg WAAB)	Human Health (DALY/1 kg WAAB)	Ecosystem Quality (PDF*m ² yr/1 kg WAAB)	Resources (MJ surplus/1 kg WAAB)
Production	66	3.8	1.1x10 ⁻⁵	1.5	4.7
End-of-Life	2.5	0.6	9.7x10 ⁻⁷	0.62	0.18
TOTAL	68	4.3	1.2x10⁻⁵	2.1	4.9
% EoL contribution	4%	13%	8%	29%	4%

Table 1 is also repeated below in Table 11 but with the impact of one WAAB and 1 kg WAABs included to provide some perspective for the impacts shown in these results (both include packaging).

TABLE 11. LIFE CYCLE IMPACT ASSESSMENT VALUES AS SHOWN IN TABLE 1 WITH THE ADDITION OF ONE WEIGHTED-AVERAGE ALKALINE AND 1 KG WEIGHTED AVERAGE ALKALINE BATTERIES.

Product or Process	CED (MJ)	GWP (kg CO ₂ eq)	Human Health (DALY)	Ecosystem Quality (PDF*m ² yr)	Resources (MJ surplus)
Production of 25 g PET beverage bottle (20 fl oz/590 ml)	2	0.07	6.8 x 10 ⁻⁸	0.005	0.15
1 weighted-average battery (33 g)	2	0.14	4 x 10 ⁻⁷	0.07	0.16
Production of 14 g aluminum beverage can (12 fl oz/350 ml)	3	0.2	2.6 x 10 ⁻⁷	0.007	0.17
1 kg weighted-average batteries	68	4.3	1.2 x 10 ⁻⁵	2.1	4.9
100 km fuel consumption in a European passenger car	305	18	2 x 10 ⁻⁵	1	21
Coffee pot: 5 years	5400	220	2 x 10 ⁻⁴	30	190

GWP, Human Health and Ecosystem Quality have a higher relative contribution from the end-of-life scenario than CED and Resources resulting from the generic landfilling and incineration processes. These will be examined in much more detail in the subsequent analysis.

Because the production phase dominates the life cycle, drilling further down into the production phase reveals the drivers of impact within that phase. Table 12 shows the values of this breakdown for 1 kg of WAAB. Figure 8 shows the absolute values using CED plotted side by side.

TABLE 12. BREAKDOWN OF PRODUCTION IMPACTS FOR 1 KG OF WA ALKALINE BATTERIES USING FIVE INDICATORS (DOES NOT INCLUDE END-OF-LIFE, WHICH WAS PRESENTED IN TABLE 11)

Life Cycle Phase	CED (MJ/1 kg WAAB)	GWP (kg CO ₂ eq./1 kg WAAB)	Human Health (DALY/1 kg WAAB)	Ecosystem Quality (PDF*m ² yr/1 kg WAAB)	Resources (MJ surplus/1 kg WAAB)
Materials Production	43	2.5	9.3x10 ⁻⁵	1.4	3.4
Manufacturing	10	0.6	5.6x10 ⁻⁷	0.022	0.55
Transport	6.9	0.42	5.1x10 ⁻⁷	0.037	0.51
Packaging materials	5.6	0.22	2.2x10 ⁻⁷	0.04	0.24
TOTAL	66	3.7	9.4x10⁻⁵	1.5	4.7

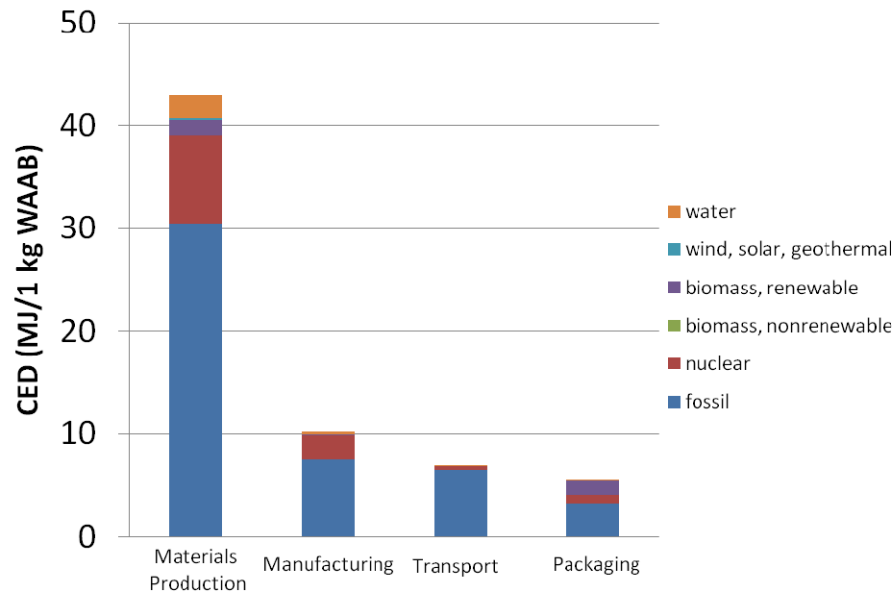
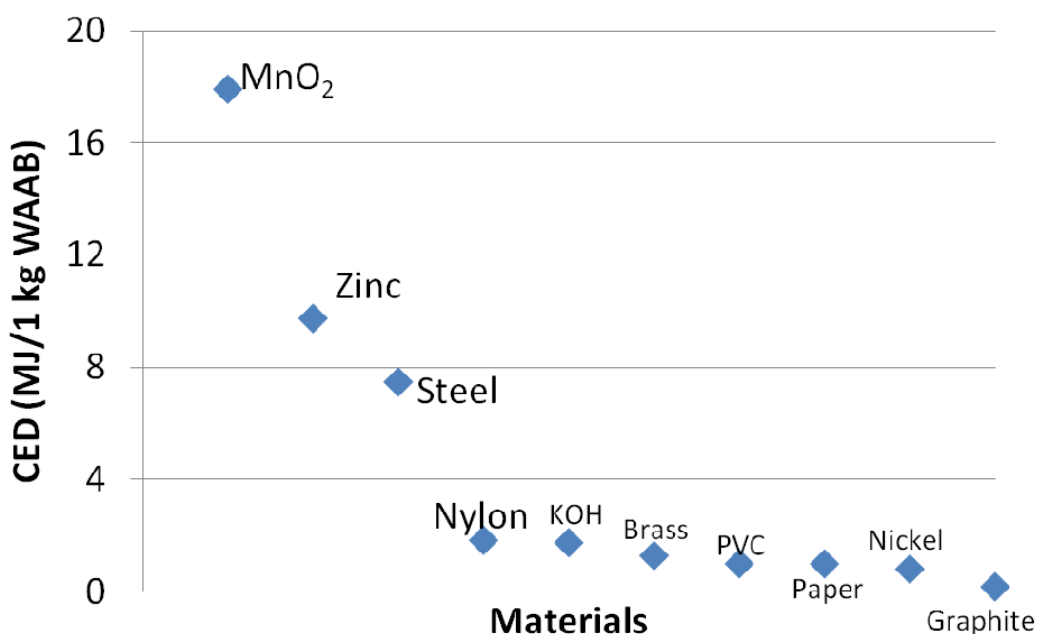


FIGURE 8. BREAKDOWN OF PRODUCTION IMPACTS FOR 1 KG WAAB USING CED.

There are a few differences between the indicators used, which will be discussed later in this section. However, all three indicators show that the **impacts of materials production dominate production impact**; therefore the next analysis focuses in on the raw materials going into the alkaline battery.

The results so far have combined the transport of raw materials from the suppliers to the manufacturing facility with the production of the raw materials themselves in the category 'materials production'. The manufacturing of the raw materials dominates this impact, as transportation is a very small percentage of the burden.

Figure 9 shows the ranking of materials within materials production for CED. This ranking changes depending on which indicator is used as shown in Table 13 (while the uncertainty in the results makes it difficult to differentiate between the materials in the last several places in the table, the first few materials are much different in impact even with the attendant uncertainty). In CED and GWP, and Resources manganese dioxide has the largest impact; in EI, the zinc has the largest impact. This reflects the perceived toxicity of zinc by this metric. Figure 9 demonstrates how quickly the impact falls off after the first three materials. The bulk of the burden is thus focused on three materials: manganese dioxide, zinc ingot, and steel.



**FIGURE 9. MATERIALS PRODUCTION IMPACTS FOR 1 KG WAAB USING CED
(THESE VALUES INCLUDE THE TRANSPORT OF RAW MATERIALS TO THE MANUFACTURING FACILITY).**

TABLE 13. RANKING OF ENVIRONMENTAL BURDEN FOR MATERIALS FOR FIVE IMPACT METHODS

CED (MJ/1kg WAAB)	GWP (kg CO ₂ eq/1kg WAAB)	Human Health (DALY/1kg WAAB)	Ecosystem Quality (PDF*m ² yr/1kg WAAB)	Resources (MJ surplus/1kg WAAB)	Mass in 1kg WAAB (g)
MnO ₂ : 17	MnO ₂ : 1.1	Brass: 3.4x10 ⁻⁶	Zinc: 0.9	MnO ₂ : 1.1	MnO ₂ : 390
Zinc: 9.8	Zinc: 0.52	Zinc: 2.7x10 ⁻⁶	Brass: 0.25	Steel: 0.76	Steel*: 190
Steel: 7.5	Steel: 0.46	MnO ₂ : 1.2x10 ⁻⁶	Steel: 0.12	Brass: 0.6	Zinc**: 170
Nylon: 1.8	Nylon: 0.11	Steel: 1.0x10 ⁻⁷	MnO ₂ : 0.071	Zinc: 0.5	KOH***: 110
KOH: 1.8	KOH: 0.094	Nickel: 8.3x10 ⁻⁷	Nickel: 0.036	Nickel: 0.17	Graphite: 36
Brass: 1.3	Brass: 0.04	KOH: 1.0x10 ⁻⁷	Paper: 0.02	Nylon: 0.13	Brass: 31
PVC: 1	Nickel: 0.047	Nylon: 7.6x10 ⁻⁸	KOH: 0.006	KOH: 0.1	Paper: 15
Paper: 1	PVC: 0.033	Paper: 2.5x10 ⁻⁸	Nylon: 0.0018	PVC: 0.062	Nylon: 15
Nickel: 0.8	Paper: 0.014	PVC: 2.0x10 ⁻⁸	Graphite: 0.001	Paper: 0.014	PVC: 15
Graphite: 0.2	Graphite: 0.011	Graphite: 1.7x10 ⁻⁸	PVC: 0.0008	Graphite: 0.01	Nickel: 3.6

*Includes steel in can and galvanized steel;**Just zinc in electrode and galvanized steel, not brass;***KOH including water

This analysis shows that for CED, GWP, and resources, the greatest environmental impact of alkaline batteries comes from the materials production of manganese dioxide. For all three of these metrics, approximately 1/3 of the total environmental impact from production comes from a single material. This general trend mirrors the highest materials by mass within the battery. However, for the case of the human health and ecosystem quality indicator, zinc has the highest environmental burden, reflecting the relative toxicity of zinc in human health and the ecosystem as described by this method. Brass also comes to the forefront for similar reasons in human health and ecosystem quality indicator. Neither of these components is highest by mass, but because of their relative perceived toxicity, they come to the top of the list for ecosystem quality or toxicity. In the ecoinvent inventory process zinc emissions to air in the primary production process (from mining and processing) dominates the environmental impact associated with both human health and ecosystem quality, accounting for ~90% of the burden of zinc.

Finally, it is useful to examine the burden of the manufacturing facility in a bit more detail. Table 14 and Figure 10 demonstrate the relative impact of electricity, natural gas, diesel, water and waste in the battery manufacturing facility.

TABLE 14. RELATIVE CONTRIBUTIONS OF IMPACTS OF THE MANUFACTURING FACILITY FOR 1 KG WAAB.

Manufacturing Facility	CED (MJ/1 kg WAAB)	GWP (kg CO ₂ eq./1 kg WAAB)	Human Health (DALY/1 kg WAAB)	Ecosystem Quality (PDF*m ² yr/1 kg WAAB)	Resources (MJ surplus/1 kg WAAB)
Electricity	8.9	0.53	5.3x10 ⁻⁷	2.1x10 ⁻²	0.45
Natural Gas	0.9	0.051	1.6x10 ⁻⁸	4.1x10 ⁻⁴	0.071
Diesel	0.36	0.024	1.1x10 ⁻⁸	1.4x10 ⁻³	0.028
Water	0.018	7.7x10 ⁻⁴	1.0x10 ⁻⁹	6.6x10 ⁻⁵	7x10 ⁻⁴
Waste	0.0086	4.8x10 ⁻⁴	1.1x10 ⁻⁹	6.3x10 ⁻⁵	5.5x10 ⁻⁴
TOTAL	10	0.6	5.5x10⁻⁷	2.3x10⁻²	0.5

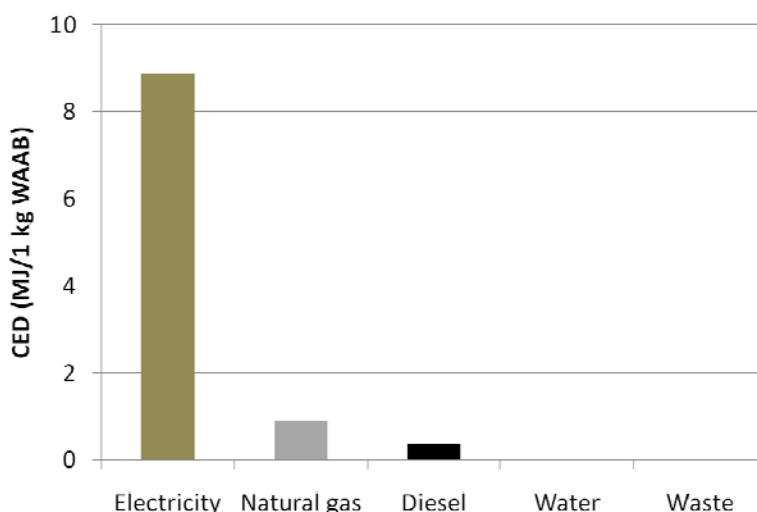


FIGURE 10. BREAKDOWN OF THE IMPACTS OF MANUFACTURING FOR 1 KG WAAB USING CED.

From Figure 10 it is clear that the electricity use within has the greatest effect on the environmental burden of the manufacturing facility. Another element of note in Figure 10 is the negative value (or credit) associated with the waste and recycling burden (indicated by the black bar below the y-axis). This credit is associated with the steel recycling within the manufacturing facility.

As described above there is no environmental burden from the use phase of an alkaline battery. The impact of the end-of-life scenario consisting of 87% landfilling and 13% incineration was a small portion of the overall impact as described in Table 10. The EOL scenario will be further investigated in the remainder of this report.

As a final set of summarizing analyses for this full life cycle assessment, two more details around the Eco-Indicator 99 metric are presented. Table 15 outlines the specific results of each characterization for a 1 kg WAAB weighted alkaline battery.

TABLE 15. A) PRODUCTION VERSUS END-OF-LIFE IMPACTS AND B) WITHIN RAW MATERIALS, PRODUCTION, TRANSPORTATION AND PACKAGING FOR EACH CHARACTERIZATION WITHIN ECO-INVENT FOR 1 KG WAAB.

Impact	Units	Production	End-of-Life
Carcinogens	DALY/1 kg WAAB	5.9E-06	6.9E-07
Respiratory organics		3.4E-09	8.4E-10
Respiratory inorganics		3.9E-06	1.7E-07
Climate change		7.9E-07	1.1E-07
Radiation		2.2E-08	3.0E-10
Ozone layer		2.6E-10	2.5E-11
Ecotoxicity	PDF*m2yr/ 1 kg WAAB	1.3	0.61
Acidification/Eutrophication		0.1	0.006
Land use		0.1	0.003
Minerals	MJ surplus/ 1 kg WAAB	1.1	0.001
Fossil fuel use		3.5	0.180

Impact	Units	Materials Prod.	Manufacturing	Transportation	Packaging
Carcinogens	DALY/1 kg WAAB	5.7E-06	1.3E-07	3.5E-08	5.5E-08
Respiratory organics		2.1E-09	5.2E-10	5.7E-10	2.4E-10
Respiratory inorganics		3.1E-06	2.9E-07	3.8E-07	1.3E-07
Climate change		5.2E-07	1.3E-07	8.8E-08	5.7E-08
Radiation		1.7E-08	3.3E-09	7.8E-10	1.2E-09
Ozone layer		1.6E-10	2.5E-11	6.9E-11	1.2E-11
Ecotoxicity	PDF*m2yr/ 1 kg WAAB	1.3	8.6E-03	1.3E-02	0.005
Acidification/Eutrophication		0.1	0.0094	1.8E-02	0.0042
Land use		0.1	0.0047	6.6E-03	0.033
Minerals	MJ surplus/ 1 kg WAAB	1.1	0.0033	6.1E-03	0.0028
Fossil fuel use		2.3	0.55	5.0E-01	0.25

To summarize the full life cycle implications of alkaline batteries, the production of raw materials dominates the life cycle with the transport of those raw materials to manufacturing having a minimal environmental impact. A few materials dominate this materials production impact, with manganese dioxide, zinc, and steel having the highest relative impacts. To return to the data quality assessment for this phase of the analysis, despite the ubiquity of European data based on ecoinvent, the relative impact of materials and elements of manufacturing are assumed to be reliable. Furthermore, within manufacturing the electricity burden dominated which was modeled after a US grid, therefore also more relevant geographically. Therefore the data gaps, in particular around geography, are not expected to have an effect on the dominant elements of the alkaline battery life cycle.

SECTION 2: ALKALINE BATTERY END OF LIFE FOCUS

CHAPTER 5: END-OF-LIFE INVESTIGATIONS

The focus of the second section of this work was to investigate the scenarios for battery disposal at end of life. This chapter describes in detail the scenarios and the relevant assumptions. A literature review of end-of-life issues in alkaline batteries is provided throughout as well. As illustrated in Figure 11 below, the batteries go through a consolidation step to an intermediate facility by car or truck. They are then transported by truck to the final disposition landfill, incineration or recycling (materials recovery). There is a collection burden associated with truck or car transport due to fuel consumption. There is a processing burden associated with landfill or recycling process (primarily energy use in the case of the recycling) and the materials benefit associated with avoided resource extraction and contamination.

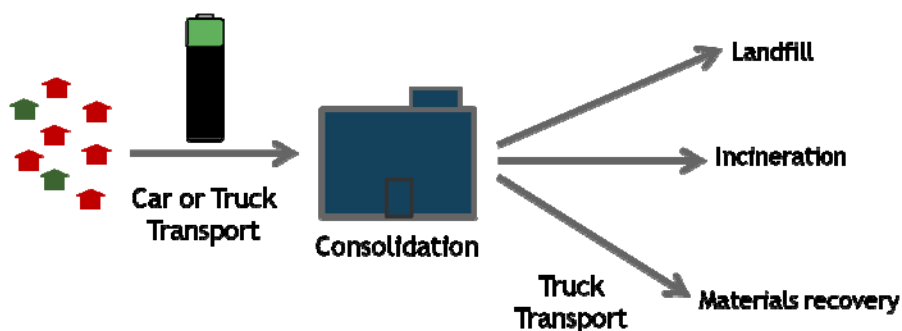


FIGURE 11. END OF LIFE SCHEMATIC FOR ALKALINE BATTERY DISPOSAL

GOAL, SCOPE AND METHODOLOGY

The goal of this section of the study is to assess various specific methods of battery collection and disposition in the US based on their environmental burdens and benefits. The allocation method assumed here is different than in the section one analysis. This section uses a substitution approach quantifying the burdens associated with the particular disposition process as well as the credits associated with materials recovery. This analysis will look at the burden from materials disposition of 1 kg of batteries focused on the US with the initial assumptions around the situation in California. This is a prospective study so in addition to looking at existing alkaline recycling scenarios, we also explore hypothetical scenarios as will be detailed below. The choice of California was motivated by the existing battery regulation impacting that state. However, other regions of the US may have differing vehicle distances and frequency of consolidation sites, therefore scenarios are run on the results as a function of population density, etc. The overall scope is outlined here and further detail provided below. The disposal scenarios will include collection with MSW trucks based on assumptions found for CA MSW vehicles with landfill treatment (and incineration). This will be compared with collection for recycling according to a few different scenarios. The retail/municipal drop-off scenario makes use of the existing California sites established by the Rechargeable Battery Recycling Corporation (RBRC). For this scenario the collection burden includes consumer drop-off at a nearby RBRC site (municipal or retail) and transport from the RBRC site to a battery recycler (average between North American recyclers).

The recycling treatments quantified are all pyrometallurgical and therefore the burden of processing includes that from three existing North American recyclers, an electric arc furnace to reflect a hypothetical minimill/steel recycling scenario, and two EU recyclers. It should be noted that the European recycling technologies used in the model do not currently exist in the U.S. Likewise, there are no steel minimill EAFs⁴ in the U.S. that currently accept EOL batteries for processing. The battery industry in Europe has spent several years qualifying EAFs for use in processing EOL batteries, but have found there to be technical and regulatory impediments that still prevent their widespread use. According to the EPBA, of the 180 EAFs in Europe of which about 60 are suitable for batteries i.e. produce construction steel (rebar), of which only 3 currently take waste batteries. The analysis captures variation in the recycling processes based on the different amounts of materials recovered, and energy used as well as the variation credits assumed for the materials benefits in recycling.

A few components remain outside the scope of this analysis, due to their expected smaller contribution to the total impact than the included elements. These include the impact of the intermediate consolidation facility such as a sorting plant or transfer station. Data were not available to differentiate the intermediate facility between scenarios. There could be a case where a particular scenario requires many more intermediate facilities and this is commented on throughout the analysis. Another component left out of this particular analysis due to its small expected contribution to the total burden is the collection vessel, such as the box associated with the RBRC program or other container for consolidation. Therefore, the report does not quantify the impacts arising from fabrication and distribution of collection containers, their replacement and disposal, as these were considered small compared to the burdens that were quantified.

There are a few scenarios around volumes of collection. The collection value is based on both the European Union Battery Recycling Directive, which mandates a 25% battery recycling rate by 2012, up to 45% by 2016 and on the current recycling rates in a handful of EU countries with longstanding battery collection systems, including Belgium, Austria, and Germany, which exceed the 25% recycling rate mandated by the directive.

Several studies have looked at the recycling of household batteries of several chemistries but these have focused on geographies other than the US and have therefore included different travel distances and different recycling technologies [10-14]. One such study by Rydh and Karlstrom examined the recycling of portable nickel cadmium batteries in Sweden. This study found that the transportation of batteries in collection had no significant effect on the energy consumption and emissions [15]. A major study by ERM in 2006 covered the recycling situation in the UK looking at several battery chemistries concluded that, although there was an inherent cost, battery recycling was beneficial [9]. The details of how this study differs from this work of Fischer et al. will be detailed throughout this document. It is difficult to fully compare the results because of the multiple chemistries examined in the Fischer work.

SPENT BATTERY CHEMISTRY

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⁴ Other than those associated with a particular recycling scenario.

The components of a weighted-average alkaline battery were identified above in Table 3. When the battery is used, several of the species undergo a chemical reaction according to the equations below [8]. Table 16 below outlines the constituents within 1 kg WAAB in the form they would be assigned value, such as Mn rather than MnO_2 . In reality the chemical composition of a spent battery will include several species of zinc and manganese oxide due to incomplete chemical reaction and the completeness of reaction varies with the cell [16]. The second reaction below is the most probable but the products may change depending on discharge conditions. Table 16 connects the masses found in Table 3 so that the impact of battery disposal can be determined. The masses are scaled to the 1 kg WAAB and the electrolyte, brass, nickel –plated steel, and galvanized are divided into their constituents (nickel, copper and steel).

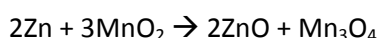


TABLE 16. THE COMPOSITION OF 1 KG SPENT WAAB.

Material	Mass (g) resulting from 1 kg batteries
Mn	250
Zn	190*
Steel	190
K	26
Graphite	36
Copper	20
Nickel	4
PVC	15
Nylon	15
Paper	15
Moisture content ~6 wt%	

* This number combines the zinc from the electrode as well as the brass and galvanized steel.

LOGISTICS ASSUMPTIONS

Several scenarios for the logistics portion of the end-of-life disposition were modeled and the assumptions for distances traveled and allocations are described in this section. Modeled distances are calculated using actual road distances where possible or when simplifications are necessary, great circle distances multiplied by a road factor of 1.3 as discussed in [17].

BASELINE AND CURBSIDE PICKUP SCENARIOS

First of all, a baseline assumption was made for the MSW disposal of alkaline batteries collected for landfill or incineration (Scenario: MSW). Two other similar scenarios are modeled for the recycling with curbside pickup, the first where a separate box is placed with curbside pickup of traditional MSW (Scenario: MSW co-collection) and the second where the batteries are placed to be collected with curbside recycling (Scenario: Recycle co-collection). The general pattern for residential waste collection proceeds as follows: Collection vehicles leave from a vehicle garage at the beginning of each workday to

begin collecting residential refuse along predetermined routes. The length of a collection route is determined by the number of households that a vehicle can service before it is filled to capacity. This varies as a function of population density as the truck fills faster in more dense areas. The fully loaded vehicles drive to a treatment or disposal facility, unload, and drive back to the starting point of another collection route. At the end of the workday the vehicles travel from the treatment or disposal facility back to the vehicle garage and this leg of the journey is allocated over the entire day. Efficiencies differ between the MSW vehicle and the curbside recycling vehicle [18].

The vehicle miles traveled (VMT) by municipal solid waste vehicles (21-tonne MSW vehicle) were determined from the literature and from discussions with the waste collection industry. Models of MSW transport have been developed by organizations such as the US EPA, Research Triangle International and other universities. One source for the average distance traveled by an MSW truck from the curbside pickup to an intermediary consolidation facility was assumed to range from 20-40 miles (~30-60km) [19] and the EPA WARM model [20]. More recent modeling and survey work by collaborators at RTI, NC State and others developed models for MSW collection. This model assumes a baseline distance of 35 miles for the MSW vehicle traveling from the garage to the collection route, to the facility and then back to the garage with additional length added as a function of population density. Several studies in the literature pointed to an upper limit of 100 miles [21] or 125 miles (NYC 2007). This is assumed to be a milk run distance, where the starting point and end point are in close proximity, therefore no backhaul is part of this journey. For the baseline scenario, this distance is then allocated to the study functional unit of 1 kg of batteries leading to a ton-km burden for this leg of transport, assuming a truck that is 80% capacity [22]. After consolidation and preliminary sorting, the distance traveled to a landfill or incineration facility was modeled based on the location of landfills and recovery facilities in the United States. Similar assumptions were used for incineration facilities [23].

For the scenario involving battery pick up with MSW trash trucks to be then recycled a similar mass allocation was made except that it was assumed the truck does not travel as full because of different rates of filling for the various materials put in the truck. For the recycling scenario, it is assumed that the average miles traveled are longer based on less recycling mass per household and varies as a function of population density [24]. The scenarios for the baseline pickup scenarios are outlined in Table 17. The truck used for modeling the burden for the baseline and pickup scenarios was a 21-tonne MSW truck. After collection by MSW vehicles the batteries for recycling are transported to a recycling consolidation facility and recycling facility. These legs of transportation are modeled based on leg 2 and leg 3 defined below for the drop-off scenarios. (see “drop-off” scenario, next section for further detail).

TABLE 17. END-OF-LIFE CURBSIDE PICK UP SCENARIOS FOR ALKALINE BATTERY DISPOSAL

Scenario	Truck capacity	Distance traveled (km)	Disposition
Baseline: MSW 1	80%	32-250	Landfill
Baseline: MSW 2	80%	50-240	Incineration
MSW co-collection	60%	32 +	Recycling
Recycle co-collection	80%	60 +	Recycling

Another thing to consider in the pickup scenario is the reality of whether infrastructure is currently in place to enable US-wide pickup. In 2007, nearly 60 percent of the U.S. population had access to curbside recyclables collection programs (based on data from states representing over 80 percent of the U.S. urban population). The Northeast region had the largest population served – 43 million persons. In the Northeast about 84% of the population had access to curbside recyclables collection, while in the West 76% of the population had access to curbside recycling [7].

CONSUMER DROP-OFF SCENARIOS (MUNICIPAL/RETAIL)

All of the drop-off scenarios consist of three transport legs as described below and shown in Figure 12.

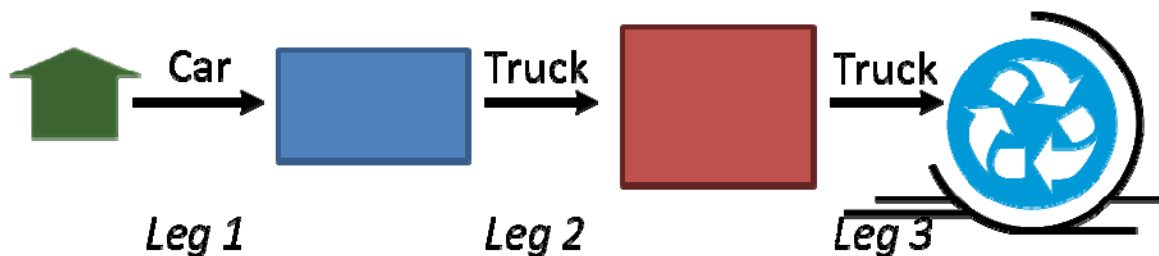


FIGURE 12. DROP OFF SCENARIO SCHEMATIC DEPICTING CAR TRANSPORT TO DROP-OFF LOCATION, TRUCK TRANSPORT TO CONSOLIDATION FACILITY AND TRUCK TRANSPORT TO RECYCLING LOCATION

For the first leg or *leg one* the consumer travels to a retail or municipal drop-off center with batteries for recycling. The average shopping trip as determined by the Department of Transportation is 14 miles [25]. The range found in the literature was about 2 - 20 miles. In addition, distances were modeled using assumptions around population distributions in and around city centers to 600 of the existing RBRC sites, predicting travel distances from 5-20 miles [26]. These values vary as a function of population density with the shorter distances corresponding to more dense groupings of individuals. Studies in Europe have indicated ranges for hazardous waste recovery centers on the order of a 5-mile collection radius, but these also vary between tightly clustered urban conurbations to sparsely populated rural areas. These facilities are frequently located on industrial estates with typical visitor travel times between 2 and 5 min [27].

The degree to which trips made to drop off batteries are part of dedicated trips to a retail or municipal site dictates the impact this leg contributes to the entire burden for recycling. A great deal of difficulty exists in predicting this factor. Several values in the literature provide some basis for this number, which will vary by population density, nature of trip, season and amount of waste. One study indicated that between 60 and 80% of municipal drop-off site users make special dedicated journeys [27]. Speirs and Tucker, based on a study of Scotland, indicated that 77% of participants combined the journey with another activity [28]. In addition, this study found that those who make dedicated trips generally traveled shorter distances compared with those combining trips (between 1.97 and 3.02 mi) and brought heavier loads (412 km/ton, 28 kg/trip) compared to (1304 km/ton, 17 kg/trip). For the trip made by the consumer a baseline allocation was assumed to describe how many other reasons the trip was made (5% for the retail and 10% for municipal drop-off) of the passenger car VMT. The allocations for the retail drop off are lower than for the municipal drop off because often the retail destination

lends itself to more potential other reasons for the trip, although the municipal drop-off trip is likely to include other similar recycling measures [29].

For *leg two* of the journey the batteries are consolidated using a delivery truck (9 tonnes; 20K lbs) at a shipping hub by a shipping company such as FedEx or UPS. For the municipalities this trip is assumed to be similar to the MSW trips. For the retail facilities this trip is made primarily on delivery truck backhaul so two scenarios are assumed, the first where no miles are allocated to this portion of the journey as backhaul to the shipping hub is primarily done empty for the case when the batteries are at a municipal pickup and a second where the truck is assumed to be mostly full on backhaul therefore the batteries are displacing some other cargo so the burden of transport for 1 kg of batteries will be signed to this transport leg for that scenario. The miles traveled from the facility to the hub was estimated from discussions with industry ranged from 128 km – 1000 km and also modeled [30] Therefore the two scenarios modeled here are 1) no allocation and 2) 0.128-1 tkm/kg WAAB allocated to the batteries.

Finally in *leg three*, the batteries travel by larger truck from the shipping hub to the recycler in one of 4 locations (see recycling descriptions in the Recycling technologies section below). For this scenario the truck is assumed to hold 80% round trip capacity. For the various recycling facilities, Scenarios A-E as described below, the average distances traveled respectively are, 1990 km, 4610 km, 4509 km, and 200 km. In addition, for a final drop-off recycling scenario an average of these distances were assumed. For the burdens associated with EU recycling scenarios, the leg three distances involved shipping by water ~ 5500km and road trucking on either end from the battery collection point to the recycler (for a total of 2500km road transport, the majority of this transport is within the US to take the batteries to the port to the EU).

Finally, estimates were made using a model of k-means clustering [31] to investigate the changes with population density within the US. The k-means clustering algorithm is a systematic way of grouping data into a desired number of clusters such that the mean value of each cluster is minimized. This algorithm can be used to determine the optimal locations for multiple sites and weighted by factors such as population. These estimates of distance were not connected to any particular existing collection sites but were just modeled based on the clustering algorithm to understand how total mileage will change as a function of population density [26]. These resulted in ranges for the values found in the logistics assumptions.

Table 18 summarizes the drop-off scenario legs and indicates the baseline assumptions for the results to be shown in Chapter 6. For the uncertainty shown in results below triangular distributions were assumed between the ranges shown for mileage and allocation. Lognormal distributions were assumed using the data quality index approach for theecoinvent inventory related to the passenger vehicle, the delivery truck and the leg three truck.

TABLE 18. END-OF-LIFE CURBSIDE BASELINE DROP-OFF SCENARIOS (MUNICIPAL AND RETAIL) AND PICK-UP FOR ALKALINE BATTERY DISPOSAL. THE TRUCK DISTANCE RANGE IN LEG 3 COVERS THE VARIOUS SCENARIOS FOR RECYCLING.

Scenario	Leg 1 – Car		Leg 2 – Delivery Truck		Leg 3 - Truck	
	Car allocation	Distance range (km)	Round trip truck capacity	Distance range (km)	Round trip truck capacity	Distance (km)
Municipal Drop off	Most likely: 10%, Range 0-10%	17 (5 – 25)	50%	120 - 400	50%	Scenario dependent, either: 1900, 4509, 4610, 200 or 8000km (truck and boat); For range assume 14% above and below
Retail Drop off	Most likely 5%, Range 0-20% Parameter analysis on 0%	8 (2 - 16)	Two scenarios: 1) Assume trip happens on backhaul, no burden 2) 50%	120 - 400		
	Leg 1 - Truck		Leg 2 – Truck		Leg 3 - Truck	
Curbside pickup: MSW and Recycle co-collection	As described in previous section and Table 17		Round trip truck capacity	Distance range (km)	Round trip truck capacity	Distance
			50%	120 – 400	80%	See above

PREVIOUS WORK - LOGISTICS

Beyond the studies mentioned throughout the section above, it is useful to summarize the assumptions that have gone into other studies of battery collection. The most comprehensive of which was found in the study by Fisher [9]. One significant difference in Fisher et al.'s study and the one performed here was that personal travel was excluded in Fisher's study, which was found to be a significant driver of environmental impact in the drop-off scenario in the current study. The scenarios modeled in the Fisher study assumed 50% capacity traveling out empty and returning full. For the scenario equivalent to the pickup scenarios modeled here Fisher assumed a typical collection round will visit between 800 and 1800 households. Furthermore, batteries were assumed to be collected from centralized locations and undergo sorting via a centrally located facility. The total collection round is approximately 250 miles with all vehicles collecting to capacity. For the drop-off scenario, a van (equivalent to the FedEx/UPS delivery truck) travels 161 km that makes visits to sites gathering smaller quantities of batteries and delivering to a consolidation point. Then larger trucks make deliveries to centralized sorting facilities as in the previous scenario. A typical transit collection route is approximately 100 miles, and satellite sites are planned to be an average distance of approximately 250 miles from centrally-located sorting plants.

Rydh estimates consumer car transport to recycling site and local truck transports of batteries from a study of glass collection. Recovery rates increase with increasing recycling densities and shorter distances per kg battery recovery are driven at higher sites densities, range of 30-250 km with a 100 km average [15]. At higher recovery rates, the fuel consumption increases rapidly due to longer distances to cover all sites and the decreasing amount of material available per site. This study found that recycling rates greater than 90% local transport for emptying collection boxes and delivery to sorting plants

increases rapidly. Mohareb *et al.* model MSW management strategies in Canada and assumed 21 km one-way distances for MSW vehicles and assumed these vehicles completed 2 loads per day [32].

LANDFILLING AND INCINERATION TOXICITY ISSUES

The issue of the environmental impact of landfilling and incineration of alkaline batteries in general is fraught with uncertainty and intense discussion. This report does not intend to offer a definitive position one way or another about the exposure and effects of the disposed battery within a landfill, but instead presents a series of scenarios based on literature values. This section will describe a few of the basic elements to consider in the landfilling or incineration of batteries. The next section outlines the first key question around landfilling of alkaline batteries: how much and what type of material leaches to the soil or water or is emitted through the air. This is a question of fate and exposure of the elements within the battery. The section after that outlines the next question, which assumes some amount of exposure and then asks what the impacts are of that exposure on human health and the ecosystem.

First, Figure 13 shows the number of landfills in the United States in 2007 as a function of region within the country. This figure is presented to provide a sense of scale of landfills in the US where the total number of landfills is around 1800.

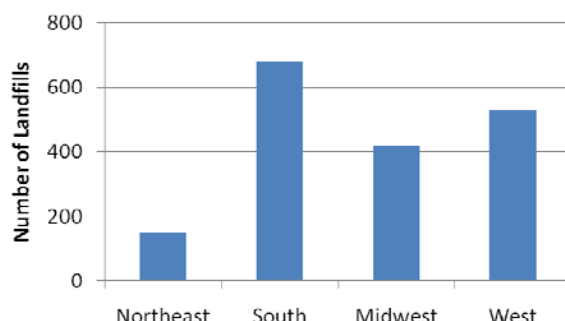


FIGURE 13. NUMBER OF LANDFILLS IN THE US IN 2007, REPRODUCED FROM [1].

One major source of environmental burden within a landfill overall is the release of methane due to anaerobic decomposition of organic matter [33]. In order to determine the relevant greenhouse gas emissions from landfilling batteries, one must first look at the amount of methane recovery within the US. Most landfill methane recovery in the US, both for flaring and electricity, is occurring in response to a 1996 EPA rule that requires a well-designed and well-operated landfill gas collection system at landfills that have designed capacity of at least 2.5 million metric tons and 2.5 million cubic meters (EPA 2000). For the year 2003 an estimated 59% of landfill CH_4 was generated at landfills with gas recovery systems. For the case of the battery, organic carbon in the form of paper within the battery may be released as methane in a landfill if the battery is in an anaerobic environment and potentially broken or crushed. The total amount of graphitic carbon in the battery was outlined in the section Spent Battery Chemistry (Table 15), however, this carbon is unlikely to be released as methane [3]. Therefore a trivial methane release burden is associated with landfilling of batteries in terms of the release of methane from decomposing organic, accessible carbon. The primary burdens due to landfilling from an energy and

greenhouse gas emissions perspective are more a function of the landfill operation and infrastructure (as well as the collection burden).

Another source of environmental impact from a landfill is through the soil or water in the form of leachate. Leachate is produced as water percolates through landfills. Factors affecting leachate formation include the quantity of water entering the landfill, waste composition, and the degree of decomposition. Because it may contain materials capable of contaminating groundwater, leachate (and the carbon it contains) is typically collected and treated before being released to the environment. However, leachate is increasingly being recycled into the landfill as a means of inexpensive disposal and to promote decomposition while the containment system is operating at peak efficiency. Another consideration is the long term status of landfills after they have been filled and are no longer actively managed. At this point the leachate will no longer be collected and treated and therefore will enter the environment. In addition, the characteristics of the soil matrix into which any leachate moves will impact of the environmental burden of these substances, for example metals may become immobilized in the soil matrix [34]. This potential impact is discussed in more detail in a subsequent section.

INCINERATION

Combustion of waste results in the potential for environmental impact from a few different sources as well, both from the carbon dioxide and other emissions released to the air as a result of the burning process and subsequent landfilling of the incineration ash residue. Most of the municipal solid waste combustion currently practiced in this country incorporates recovery of an energy product (generally steam or electricity), approximately 60% [33]. For the particular case of metals, MSW combusted with energy recovery in the US may be combusted in Waste-to-Energy (WtE) plants that recover ferrous metals such as the steel cans of batteries. This is an integral part of the operations of many combustors and then this material is recycled in steel minimills after incurring incineration burden. This section will provide a bit more detail about the capacity potential from an incineration perspective and the details around incineration of batteries are described below. Among operating US WtE plants, 77% have onsite ferrous metal recovery programs. These facilities recover more than 702 thousand tons of ferrous annually. Most of these metals are recovered at mass-burn WtE plants, from the bottom ash after combustion. In addition, 43% of the operating facilities recover other materials on-site for recycling (e.g., non-ferrous metals, plastics, glass, white goods and WtE ash that is used for road construction outside landfills); over 780 thousand tons of these recyclables are recovered annually [35, 36]. In mass-burn plants, the MSW is fed as collected into large furnaces while in refuse-derived fuel facilities plants, the MSW is first shredded into small pieces and most of the metals are recovered before combustion. Figure 14 shows the regional distribution of municipal waste to energy capacity, while Table 19 and Table 20 provide some quantification of the capacity for WtE [33].

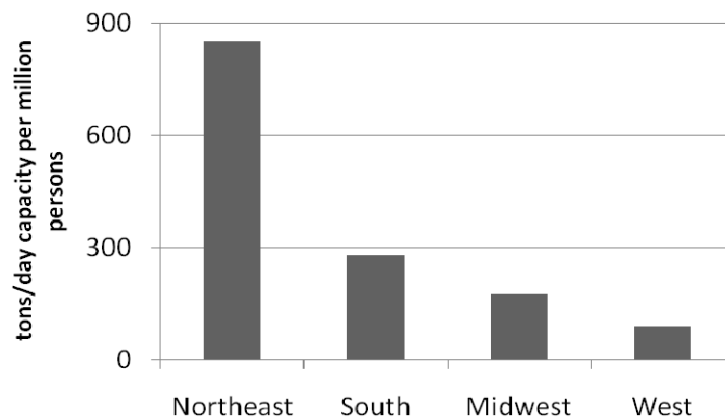


FIGURE 14. MUNICIPAL WASTE-TO-ENERGY CAPACITY, 2007, PLOTTED IN TERMS OF CAPACITY IN TONS PER MILLION PERSONS REPRODUCED FROM [7]

TABLE 19. MAJOR USERS OF WASTE-TO-ENERGY IN THE US. REPRODUCED FROM [37].

State	Number of WTE plants	Capacity (tons/day)
Connecticut	6	5897
New York	10	10070
New Jersey	5	5625
Pennsylvania	6	7620
Virginia	6	7530
Florida	13	17509
Total	53	63140

TABLE 20. OPERATING US WASTE-TO-ENERGY PLANTS [37]

Technology	Number of plants	Capacity (tons/day)	Capacity (mtons/yr)
Mass burn	65	64730	20.05
Refuse derived fuel	15	18162	5.71

For the case of incineration of batteries if combusted rather than recovered, the temperatures are typically lower than that of an electric arc furnace, around 1200- 1300 K [38]. For a study performed on the incineration of spent alkaline batteries, the emissions of zinc were the most emitted metal, ~6.5% of Zinc in the batteries while the emissions of manganese were negligible (the other significant source of emissions were from PVC found within the battery) [38]. MSW incineration facilities are equipped with air and water equipment to capture pollutants however small amounts may escape through filters, metals are stored in both bottom ash and flue ash. Organic sulfides are added to the ash for complexation of metals and this stabilized sludge is landfilled at municipal sanitary landfills.

Fisher et al. describe the assumptions in incineration as follows particular to metals and carbon emissions, “we have assumed that 0.5% of the metals in batteries are emitted to air from the

[incinerator]...and the remaining are removed through flue gas treatment and bottom ash.” The residues from incineration are then disposed to landfill with another 2.5% of the metal content leached to water (Fisher et al. makes an assumption that the residues are more inert than raw MSW despite the fact that it primarily consists of steel, zinc and manganese oxide). Furthermore, no energy recovery benefit is assigned because they are considered of low calorific value [9].

Details: incineration of MSW containing spent alkaline batteries [38]

Some of the zinc metal present in the anode vaporizes from spent batteries. Most of the zinc will be found in the bottom ash, except for particles carried by injected gases. In large part, manganese and its oxides collect within the bottom ash as an oxide and may substitute for road bed material (however due to the contained zinc, there may be complication with this substitution). All the steel from the can may be recovered with other iron scrap in the bottom ash, however, much of the iron will be strongly oxidized and may result in a less valuable iron scrap and recycling this material incurs further burden associated with EAF minimill activity. The majority of plastics and paper (a small weight percent of the battery) are used as energy, similar to the other recycling processes at high temperature, described below. KOH vaporizes at most common incineration temperatures and may act to neutralize some of the agents within the incineration process and potentially decrease the emissions of CO₂. Graphitic carbon will burn and may react to form CO and CO₂. The energy and GWP burden associated with battery MSW incineration process itself may be minimal due to the large majority of incinerators that burn refuse or mass that would otherwise be landfilled as the batteries do not contain much carbon, however emissions from the incineration process are included from the incineration of this waste regardless of the source of fuel. These elements are combined to form the incineration scenario: 1) Burden associated with incineration (increased metal content in MSW incineration will lead to more fuel consumption than typical MSW), 2) Burden associated with EAF recovery of steel, 3) Credit associated with zinc, steel and slag recycling. For #1 the 2006 EPA report on GHG emissions from solid waste management provides guidance on the differential energy content values on metals versus “typical” MSW [39]. For steel in a mass burn facility the “avoided utility CO₂ per ton combusted” is equivalent to -0.01 metric tonne carbon equivalents (MCTE)/tonne because of the -0.7 million Btu/tonne of steel provided in the energy content of this material. This is in contrast to the equivalent avoided utility CO₂ metric for paper, plastic and mixed MSW of ~0.2, ~0.4, and ~0.14 MCTE/tonne respectively. In general, incineration of alkaline batteries at MSW plants may result in steel that is not recycled and the gaseous emissions must be cleaned by off gas systems [35, 36], therefore this incineration analysis assumes an optimistic case of battery incineration. The experience of those in the industry and in waste management is that batteries remain largely intact in the incineration process and would therefore report to the bottom ash as whole products Based on data availability for the incineration scenario only GWP is presented in the results. These data were derived from the 2006 EPA report and include (for the battery relevant materials): 1) combustion CO₂ and N₂O emissions, and 2) *avoided* utility emissions at mass burn facilities. These are combined with the credit associated with steel recovery and the burden associated with transportation as described above. If we were to consider assessment methods other than GWP, Zn emissions and transfer of materials to the slag would also have to be considered.

The next section describes various values found in the literature and other similar studies on the question of “how much metal leaches” and the following section outlines the many metrics that have been suggested to quantify the toxicity for the elements of concern in the battery.

LANDFILLING

How much metal leaches or methane is emitted

The amount of metals leaching from and within a landfill is dependent on a great many factors such as the age of the landfill, the conditions of the landfill liners and the processes in place to control and minimize landfill effluent. There is a great deal of research in this space that cannot be fully covered in this work, however, a few key studies are highlighted regarding systems of interest.

For metallic materials, one prevalent release mechanism is the process of corrosion. The release rate to leachate (water) equals the corrosion rate [40]. While the landfill is managed the leachate is treated. There is a significant change in the concentration of the leachate as the conditions in the landfill change (i.e. become more acidic, anaerobic or methanogenic). The leachate concentration appears to be lowest during the methanogenic stage. The report by Fisher *et al.* mentioned previously assumes that in landfilling batteries “5% of these metals in batteries are leached to the environment, the remainder remaining locked in landfills as non-compromised batteries or as mineralized compounds resistant to leaching.” The timeframe of this leaching was not clarified in this study although the time period of the study was 25 years. The source for this 5% number was not clear so the value reported by Fisher was not used directly in this work. As mentioned above the residues from incineration are disposed to landfill with 2.5% of the metal content leached to water[9].

A few times in the literature results are presented over several time periods, for example, Rydh and Karlstrom and other works assume the worst case scenario of an infinite time perspective when all the metals have been completely released from the landfill [15, 41]. Several studies assume that landfilled metals are corroded but immobilized as solid compounds and therefore not biologically available. However these products may be continuously leached out by percolation. Slack et al. modeled landfill leachate migration and determined that for copper, nickel and zinc the ground-water off site compliance point concentrations were, respectively, 0.61, 0.45, and 0.34 µg/L [42]. There are several tests used to determine the limits to chemicals in the environment and several studies comment on these levels in relation to electronics, such as cell phones. Two such tests, the Waste Extraction Test and the Total Threshold Limit Concentration investigate limits related to the following as metals of concern Zn, Ni and Cu [43].

The particular scenarios of interest for this work are as follows:

1. The Cal Recovery report results indicated no significant mobilization of zinc into the leachate. The concentration of organic and nitrogen compounds, salts, Ca, Mg, Na, K and Mg in the reactor leachate showed no significant differences during the investigation, except for higher but inconsequential concentrations of manganese and potassium (indicating potential 1.5 factor higher for the battery containing lysimeter) over the control. Therefore, for this scenario 0% was assumed released into the leachate [44].

2. Research by Slack and coworkers quantified 0.02% of metals in landfills released to the environment [34, 42]. For this battery analysis, since this is not a battery-specific value, the percentage was used as a percent of metals in batteries..
3. A recent article described a landfill lysimeter test of several different battery types and identified the following regarding controlled-compared values for Ni (0.09 mg/kg dry waste) and Zn (4 mg/kg dry waste) [45].
4. The authors in [41] present the following values for MSW landfill leachate release over a 30 year time frame. For this battery analysis, these values were modified to reflect the small amount of these substances resulting from batteries present in MSW.
 - 0.00007 kg Cu emitted/kg Cu landfilled
 - 0.00010 kg Fe emitted/kg Fe landfilled
 - 0.00020 kg Zn emitted/kg Zn landfilled
 - 0.005 kg Ni emitted/kg Ni landfilled

For this study, as described below, we run a scenario around landfilling that assumes an average of number 2, 3 and 4 above. The average of the quantities above for Zn, Cu and Ni were combined with the toxicity values described below for ecotoxicity to provide a total ecosystem quality impact due to metals present in the leachate. Over a longer term scenario, when the landfill is no longer managed or the landfill leachate collected and tested, these quantities could be different. Therefore the long term emissions to ground water values modeled and published in the ecoinvent data set were also considered. Those values for copper, manganese, nickel, potassium, and zinc are 0.00109 kg, 0.000223 kg, 0.000104 kg, 0.0017 kg, and 0.000344 kg / kg MSW, respectively.

Toxicity measures

For the purposes of this discussion the components of interest from a potential toxicity perspective present within an alkaline battery are zinc, manganese, brass and nickel. The various oxide forms of these components are also of interest, however, very little data exist on the toxicity of the oxide forms. There are several metrics within the impact assessment tools used in LCA that comment on the toxicity to the ecosystem or impact to human health of particular elements. Therefore the LCIA midpoints or damage assessment values from the following the human health and ecosystem quality categories were used for the impact of the baseline landfilling scenario. For example, for the substances of note within Ecoindicator 99 there are toxicity measures corresponding to Zn, Cu and Ni [46]. In order to determine the impact of the metals in the leachate scenarios referenced in the literature and described above, the impacts to water associated with ion leaching of Zn, Cu and Ni were used as shown in Table 21 for Ecoindicator 99. Each of the factors described below were applied to the scenarios of leaching from the batteries in a landfill as described in the previous section (also as a function of how much is present within the battery). Note that, as described above, 0 was also a scenario considered as reflected in the CalRecovery study. The scenario incorporating the metals from the batteries into the leachate is termed the MSW-leach scenario.

TABLE 21. ECOTOXICITY IMPACTS FROM NICKEL COPPER AND ZINC FROM THE ECOINDICATOR 99 LCIA METHODOLOGY.⁵

Ion form	Ecotoxicity in water (PDF*m ² yr/kg)
Nickel	1.4E+02
Copper	1.5E+02
Zinc	1.6E+01

RECYCLING TECHNOLOGIES

The last areas to consider in more detail are the recycling technologies that are modeled in this analysis. This analysis focuses on pyrometallurgical techniques for recycling that use high temperature to transform metals. There are several different approaches to pyrometallurgical recycling and these vary in terms of the pretreatment and post-treatment, other feedstocks required, energy consumed (and fuel used to supply that energy) as well as the materials recovered [47]. The review article by [47] elaborates on details surrounding hydrometallurgical approaches to battery recycling. These will likely have differential impacts than the ones discussed here and should be further explored from a life cycle perspective as they become more industrially available.

There are three specific facilities that were modeled, Metal fuming furnace in the Northwest; Nickel reclamation furnace in the Midwest and low temperature process in the Great Lakes region. Historically, there has also been interest by the battery industry in recycling alkaline batteries along with steel in electric arc furnaces. This scenario is also considered, however, it is currently only a hypothetical scenario, as there are currently no EAF's in the U.S. that accept EOL batteries. There are a number of technical issues that limit the loading of batteries potentially acceptable to EAF's. There are also regulatory and permitting hurdles, especially for EOL batteries generated in California, where they are considered universal wastes. The data from two European facilities are also included from France and Switzerland. As mentioned above, there are other technologies that may be applicable in the hydrometallurgical domain, but these were considered outside of scope at the time of the commissioning of this study. They could be looked at in future work. It is important to note that this study does not intend to explicitly compare the technologies; rather this work investigates the specific sites and contexts under which the technologies operate. In other words, we investigate the scenarios in detail rather than comparing the technologies head to head. An analysis is presented at the end of the document that uses the same electrical grid and transportation distance for all the scenarios.

Determining the credit allocated to the materials recovered in recycling is a topic of much academic research in the field of life cycle assessment [48-51]. For the purposes of the baseline it is assumed that the burdens and benefits of recycling are directly applicable to the life cycle of these materials that are directly related to alkaline batteries. The assigning of these "credits" should also be understood in the context of the market the particular materials operate in. In other words, does the recycled material

⁵ Goedkoop, M and R Spriensma, "The Eco-indicator 99, A Damage Oriented Method for Life Cycle Impact Assessment - Methodology Report", PRe Consultants, 2001.

from battery recovery replace primary or virgin material or does it instead replace recycled material from some other system, thereby not offsetting virgin material extraction. This implication can be explored in further analysis on this topic but is not directly covered in this report. The recovering of zinc, steel and manganese oxides of various forms are the most significant elements of this discussion. Overall metals recovery in recycling has been described as favorable given the energy savings potential, the ability to recover mostly to the characteristic of primary, and the generally effective economics [41]. This study excludes detailed modeling around the transport leg of the material recovered from batteries at the recycling facility to the “metal market” where the material will be sold as there was not sufficient information available to distinguish this transport leg between the recycling scenarios considered. Therefore the following assumptions were made around this distance for the three categories of materials recovered: slag for cement, 150 km; micronutrient, 350 km, and metal value (steel, ferromanganese and zinc), 350 km.

SCENARIO A

The metal fuming furnace in the Northwest operates by batch operation, which includes a charge stage whereby the cold, blended feed is metered into the furnace (along with the required amount of coal) and heated to approximately 1200°C. Each batch cycle with a cycle time of four hours consists of a charge stage, fuming stage and tap stage. The alkaline batteries (2-5 tons per 50 tons of feed) are charged to the furnace as whole batteries and not separated into components. Following the charge stage, the processing conditions are made reducing to fume Zn vapor, which is then re-oxidized in the gas space and condensed to solids in the boiler. The ZnO solids are moved to the baghouse for filtration with the solids collected and conveyed to downstream plants for further processing and Zn recovery. ZnO is first converted to ZnSO₄ in downstream leaching processes. At the end of fuming, the furnace bath is deficient in metal value and therefore the Mn oxide and steel with high yield remain with the tail slag, which is tapped with the granulated slag then sold for cement manufacture. The furnace is fueled by coal, which is injected through tuyeres along with blast air and oxygen. The coal not only provides energy to maintain 1200°C bath temperature but also supports bath metallurgy in converting metals to their zero oxidation state, which then fume according to their high vapor pressures. The subsequent downstream processing of fume material yields 90% recovery of Zn-in-fume to metal for market. The assumed electricity burden for subsequent product is modeled off of the Zn process wherein the Zn is then recovered from Zn electrolyte (H₂SO₄) at the Electrolytic and Melting plant. Hydroelectricity is used for the Zn recovery operation [52]. The yields described by the company for this process are as follows: ~80% yield from the original amount of Zn and ~95% from the original amount of Mn and steel. - Scenario A in the results below is derived from this process, where Zn is credited for the market mix between co-mined and primary mined Zn (using the value of 30% co-minedecoinvent inventory to correspond to secondary production). This market mix credit could be considered more appropriate for a short term perspective, while for a longer term perspective in a market where demand is rising, one could also argue that a 100% primary metal credit is more appropriate. The appendices present the same results but using 100% primary zinc offset to illustrate this longer term perspective.

SCENARIO B

For the nickel reclamation furnace process the batteries may be mixed with other non-battery scrap materials and reducing agents such as coke or coal. The resulting composition of reclaimed metal is controlled by this feed composition. During the process, the feed material is put into a rotary hearth furnace and heated to 1250°C. This heating stage causes a partial reduction reaction to occur where the some of the exposed metal oxides (iron) release oxygen to the carbon, resulting in gas release.

The hot pellets are then transferred to an electric arc furnace where it is heated to 1700°C, smelted, and tapped out. Reduction occurs due to the presence of carbon so oxides are converted to their metallic forms and zinc is vaporized and drawn from the furnace. This zinc is collected in a bag house and recovered at a separate facility. A molten metal alloy is formed with the steel, some of the manganese, and other metals (nickel, chromium, etc.) and cast into pigs to be used as secondary steel feedstock. The impacts of this facility and transport leg were included in the analysis. There is also some remaining material which is used in asphalt. The main product is iron, nickel, chromium, molybdenum for use in steel alloy plants. Battery recycling at this facility was originally set up around nickel-containing battery chemistries. The yields described by the company for this process are as follows: ~90% yield from the original amount of zinc and ~90% from the original amount of Mn (with some material reporting to steel and other to the material used in asphalt) and ~95% yield from the original amount of steel. Scenario B in the results below is derived from this process, where steel is credited for the market mix of 40% recycled and 60% converter steel. The appendices present the same results but using 100% converter steel offset.

SCENARIO C

This recycling process takes place in the Great Lakes region. After collection and sorting the batteries are managed through batch processing on a daily basis. The process involves electricity inputs to a mechanical shredding and pulverizing step that results in a fine particulate material. Steel is pulled off through magnetic separation prior to complete pulverization and sent to steel recycling. There are also a set of rinsing steps and natural gas inputs to a heating step (well below the melting temperature of the materials to drive off moisture). Zinc and manganese oxides are recovered to a micronutrient application for the agricultural industry. The recovered paper and plastic go to a waste to energy process. Scenario C in the results below is derived from this process.

SCENARIO D MODELED AFTER RECYCLING WITH STEEL IN AN EAF

An electric arc furnace (EAF) is a furnace that heats charged material by means of an electric arc. Arc furnaces differ from induction furnaces in that the charge material is directly exposed to the electric arc, and the current in the furnace terminals passes through the charged material. This process was modeled using the ecoinvent data set for steel production from scrap. The furnace reaches temperatures of approximately 1700°C. These furnaces or minimills are typically used to make reinforcing bar or chromium steel. The capacity of the EAF is 120 tons and charges are approximately one hour in duration. There are a few concerns with recycling batteries with steel, one is the copper present in the battery that is a poison in steel processing, and therefore the battery capacity is limited to 5% of the charge. Another concern is the possible presence of mercury in the feedstock which is a risk in all public

collections; mercury is not added to OEM-produced alkaline batteries, but may be present in trace quantities from other sources such as older batteries made before mercury was not added, imported, or counterfeit types⁶. Zinc is recovered from a baghouse and the manganese within the battery is assumed to provide alloying content to the steel. The increase in zinc in the baghouse from the batteries is a benefit to the minimills. Legislation regarding waste designation around alkaline batteries presents a challenge in permitting EAF facilities to take the waste cells. Therefore, as of writing this report, there were no EAF-type facilities taking waste batteries in North America, so this is a hypothetical, generic scenario. Leg 3 of the transportation burden associated with the EAF scenario, Scenario D, was modeled using the locations of US minimills (assuming a hypothetical situation where all minimills would accept EOL batteries), resulting in a lower overall average leg 3 transport burden than the other scenarios, approximately 200 km. Therefore, this transport scenario is in contrast to the other scenarios (A-C and E) as it assumes that the batteries travel to facilities that are distributed throughout the country rather than all traveling to one recycler. At the end of the document, a parameter analysis investigates the impact of this assumption. The yields assumed based on conversations with workers at a steel minimill who had been involved in a battery trial in this process are as follows: ~75% yield from the original amount of zinc and ~95% from the original amount of Mn and steel. There is a sensitivity analysis around the recovery of zinc found after the baseline results of the study.

SCENARIO E MODELED AFTER AGGREGATING EUROPEAN RECYCLERS

The process in Switzerland begins with manual sorting and pyrolysis at 700°C. The metallic components resulting from this step are passed into an induction furnace where they are reduced through smelting at a temperature of 1500°C. Iron and manganese remain in the melt to form ferro-manganese and zinc is recovered after vaporization. The process in France begins with a grinding and mechanical pre-treatment before being fed into an arc furnace. Ferromanganese is also obtained along with a zinc oxide dust. As mentioned above, the leg 3 collection burden assumed for this scenario quantifies the transport of 1 kg WAAB to the EU facilities by boat and road. This transportation burden is broken out in the results below (c.f. Figure 18 and the subsequent description). If the waste batteries were to be processed using these processes, they would need to be transported from the US to Europe. The yields assumed in this process from documentation found in the literature are as follows: ~90% yield from the original amount of zinc and ~95% from the original amount of Mn and Steel.

Several scenarios of recycling are outlined according to the descriptions provided in this section. These scenarios along with a description of the materials recovered are shown in Table 22. Metal value in Table 22 indicates that the material replaces virgin metal as indicated (steel, zinc, or manganese).

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⁶ The mercury level of the entire waste stream is estimated through battery sorting studies to be less than 20 ppm.

TABLE 22. RECYCLING SCENARIOS FOR BATTERY RECOVERY. METAL VALUE INDICATES REPLACING VIRGIN MATERIAL

Scenario	Materials recovered
A	Zinc (metal value) Steel and Manganese (cement/road construction)
B	Steel (metal value) Zinc (metal value) Manganese (part metal value part cement/road construction)
C	Steel (metal value) Zinc/Manganese (micronutrient)
D	Steel, Zinc and Manganese (metal value)
E	Steel, Zinc and Manganese (metal value)

There are three categories of information tracked for the modeling of the recycling processes, each with inherent uncertainty. First the inventory contains the energy used for the process in terms of electricity, coal, or natural gas. That information, for scenarios A, C was gathered from the recyclers themselves. For scenarios B and partially E that information was discerned from the literature and for scenario D ecoinvent inventories were used. Then emissions from the combustion of natural gas and coal are identified based on data from the Energy Information Agency. The specific substances quantified in that combustion were carbon dioxide, carbon monoxide, nitrogen oxides, sulfur dioxide, particulates and mercury (for coal and oil). Finally emissions from the particular recycling furnace processes should be present in these inventories. The substances (air, water emissions) of interest from the recycling process were determined based on those whose data could be found from the literature and also contributed to the impact (determined by running the relevant ecoinvent processes, such as the EAF scenario or the EU based scenarios). Where no quantitative data were available for those substances listed above, the same values were assumed between the scenarios when they contained similar forms of heating (furnace processes). These data were not readily obtainable therefore are a data gap in this work. However, these direct emissions were approximated from Fisher et al., literature, and the ecoinvent electric arc furnace process (more details provided in the inventories in the appendices). The inventories focused on including those substances that would have demonstrate an effect within the impact assessment method used. For the case of scenario C, the non-fuel derived emissions were based on the shredding process within ecoinvent (Shredding, electrical and electronic scrap/GLO U). Because various sources are being used here, the scope of what is included in the emissions is of varying quality and some are more complete than others. The particulate and metal emissions may be particularly influential in the human health and ecosystem indicators so this gap in data should be further investigated.

The table below summarizes the data used and the quality of this data evaluated on the same criteria as above: primary versus secondary, temporal and representativeness (geographic scope).

As should be evident from the description throughout this chapter, there is a great deal of uncertainty associated with these end of life scenarios. Therefore, several types of distributions will be presented

around the numbers obtained in this analysis. That uncertainty and variation captured in this analysis is a function of the mileage travelled, the energy used in the processing (and associated direct emissions from fuel use), the amount of material recovered, and the nature of the credit assigned to those materials. For mileage traveled, and degree of dedication for consumer travel a triangular distribution was used; a normal distribution was assumed around the quantity of energy used in recovery (using a 10% coefficient of variation on the energy and emissions data); a uniform distribution around the amount of materials recovered; and lognormal distributions using the data quality index approach around the magnitude of credit assigned (i.e. the inventories used from ecoinvent). The results presented below are based on propagating the uncertainty within the amount of recovered material, energy use, credit assigned, and the variation within the mileage traveled, etc. through a series of statistical Monte Carlo simulations. Further detail on the ranges assumed in the uncertainty analysis is provided in the appendices.

Stage		Data Sources	Temporal	Representativeness
Spent battery chemistry		Literature data on quantities and materials. ecoinvent data on inventories for materials and recovery credit	Literature data from 2002 and 2009, other based on ecoinvent processes	Limits on data based on European focus of ecoinvent data
Collection				
	Curbside pickup (not explicitly presented in the results)	Literature and industry data on mileage and fuel consumption. Secondary data (ecoinvent) for modeling of upstream processing of MSW truck.	Literature data from 2000-2008, industry data from 2008	Data representative of US MSW mileage and fuel consumption, burden of truck infrastructure limited by EU focus
	Municipal and Retail drop-off	Primary data on existing RBRC collection sites. Literature data on consumer travel, mileage and trip dedication. Secondary data for life cycle inventories for car and truck transport (ecoinvent).	RBRC site data from 2007/2008 Literature data from 2003, 2007 and 2008	Limits on data based on European focus of ecoinvent data
Disposal				
	Landfilling	Landfill infrastructure impact based on ecoinvent Leachate and emissions data based on literature Impact of leachate based on EI 99	Literature data from 2000-2006 Toxicity data from 2008	Limits on data based on European focus of ecoinvent data. Leachate data based on secondary sources, includes Calrecovery lysimeter test
Recycling				
	Scenario A:	Primary data on energy consumption quantities, Secondary data for life cycle inventories Emissions based on literature	Energy consumption quantities data from 2008	Limits on data based on European focus of ecoinvent data

		data		
	Scenario B:	Primary data on energy consumption quantities, Secondary data for life cycle inventories Emissions based on literature data	Energy consumption quantities Inventory data from 2007	Limits on data based on European focus of ecoinvent data
	Scenario C:	Primary data on energy consumption quantities, Secondary data for life cycle inventories Data gap around emissions	Energy consumption quantities Inventory data from 2009	Limits on data based on European focus of ecoinvent data
	Scenario D: Hypothetical EAF	Secondary data for process modeling (ecoinvent).	Based on ecoinvent processes ⁷	Limits on data based on European focus of ecoinvent data
	Scenario E:Hypothetical EU	Literature data for energy consumption quantities and ecoinvent processes	Based on ecoinvent processes	Data may well represent as EU focused

CHAPTER 6: END OF LIFE SCENARIO ANALYSIS RESULTS

This section will describe the results of the analysis using the impact assessment indicators Cumulative Energy Demand, Global Warming Potential, Ecosystem Toxicity (including ecotoxicity and acidification/eutrophication), Human Health, and Resources. The impact assessment results are relative expressions and do not predict impacts on category endpoints. The summary of the information to be assembled for each scenario is shown in Figure 15. The vehicle miles traveled in the collection step are quantified according to the scenarios outlined in the previous sections. Then the burden of battery disposition is quantified and then any credits due to materials recovery are quantified. It is critical to recall that each of the existing recycling scenarios is associated with its respective leg 3 transport. Therefore in the comparisons below the specific locations of the facilities are bundled into the analysis and the analysis does NOT directly compare each recycling **technology**, rather the recycling **scenarios** are compared.

From a vehicle miles traveled perspective, the greater burden for transport in the recycling scenarios was found to be associated with the drop-off scenario logistics as opposed to the scenarios around curbside pickup. **Therefore, to reduce the complexity of the results presented, the collection scenarios presented in these results are those based on both municipal and retail drop-off because they demonstrate the higher transport burden for leg one.** The MSW pickup scenario will be discussed below.

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⁷ Chapter 12 of EPA's AP 42 emissions factors for the metallurgic industry indicates that US EAF emissions factors are similar to that calculated using ecoinvent 2.2, therefore ecoinvent was assumed to be realistic and sufficient for the purposes of this study.

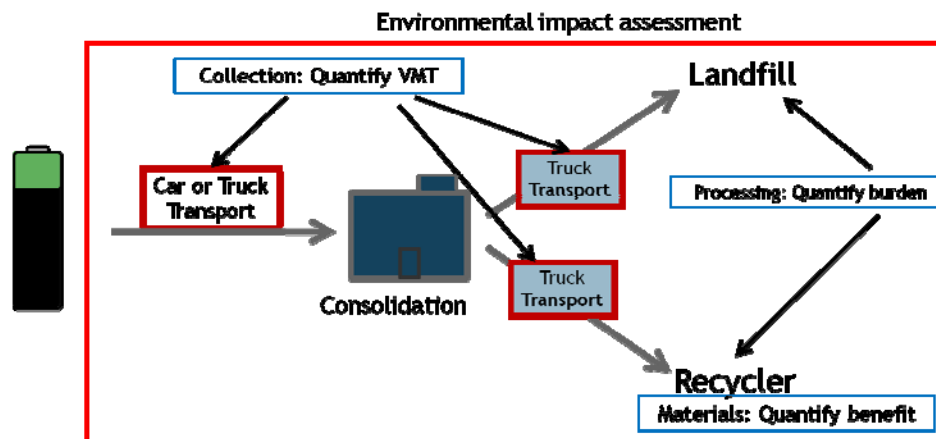


FIGURE 15. SCHEMATIC OF END-OF-LIFE SCENARIOS TO BE ASSEMBLED IN THE RESULTS

Figure 16 outlines the way results will be presented throughout this section. A value plotted above the x-axis is considered an environmental burden and below the x-axis the benefits offered through materials recovery outweigh the burdens, so an environmental benefit is demonstrated. Then moving along the x-axis, the landfill scenario is shown to the left of the dotted red line. The first light blue bar indicates the burden associated with driving batteries around with the MSW truck and the dark blue bar indicates the processing burden of the landfill. The purple bar to the right of that is the superposition of the light and dark blue bars indicating the net impact of landfilling batteries. To the right of the red dotted line two materials recovery scenarios are depicted. Each has a different collection burden noted by the light blue bar and a different processing burden noted by the dark blue bar above the x-axis. The blue bar below the x-axis shows the credit associated with each material recovered in recycling, such as zinc, steel or manganese dioxide. For the two scenarios shown here the two materials are recovered with a particular credit associated with what they would offset in primary extraction. The final light and dark green bars in each materials recovery scenario are the superposition of all three previous blue bars depicting the net impact of recycling alkaline batteries. The net impact for scenario 1 aggregates to provide an environmental benefit while the net impact for scenario 2 shows an environmental burden.

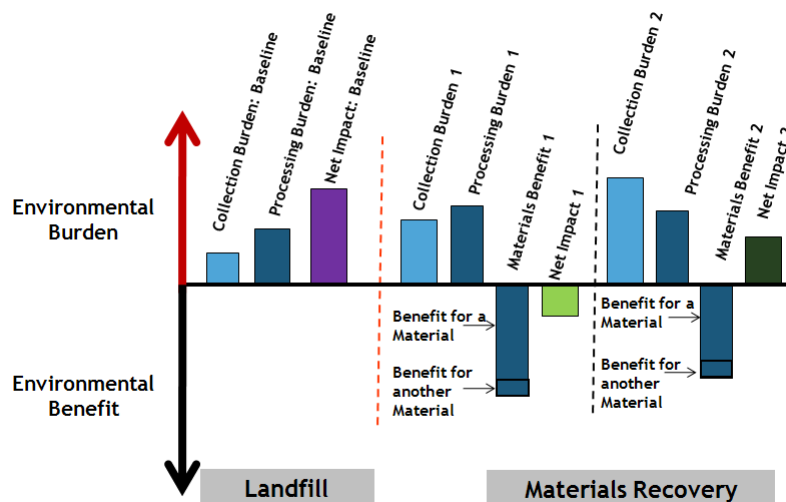


FIGURE 16. SCHEMATIC OUTLINING THE FORMAT OF RESULTS PRESENTATION

Before presenting the full results, it is useful to build up the plots based on the description provided above. Two plots will be used to do this, the landfilling scenario and one of the recycling scenarios. Figure 17 below shows the two impacts and their combination for the MSW 1 scenario, baseline landfilling of batteries, in terms of cumulative energy demand, shown in MJ. This plot is shown with a wide range on the y-axis so it can be compared to the recycling scenarios. The light blue bar in the figure indicates the burden associated with municipal solid waste collection of one kg of weighted average alkaline batteries and the associated uncertainty based on the distance traveled and the fuel economy of the vehicle. This includes a transportation range of 32 – 250 km. The dark blue bar represents the burden associated with landfilling alkaline batteries, which is primary due to the operation of the landfill. The final purple bar shows the combination of the blue bars and thereby the net impact of MSW 1. The net impact uncertainty was calculated by the square root of the sum of the squares on the uncertainty within each portion of the analysis. The functional unit for all the graphs presented in this section is 1 kg WAAB; because the batteries are the focus of the EOL analysis, battery packaging has been excluded in all the results presented below.

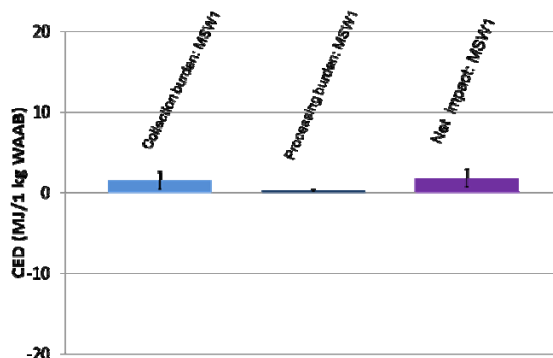


FIGURE 17. MSW1: COLLECTION AND DISPOSITION OF LANDFILLED BATTERIES FOR 1 KG WAAB.

As mentioned previously an analysis of incineration is provided at the end of this report.

The presentation of the results of the recycling scenarios is similar to that of MSW 1 except for the addition of materials recovery (materials benefit) into the equation. Figure 18 shows the collection burden of Drop-off retail, the processing and materials benefit for Scenario E, and the aggregated European battery recycling scenario in terms of the proxy indicator, cumulative energy demand. The light blue bar shows the collection burden and the associated uncertainty based on the ranges described in Table 18. *As described above, this scenario assumes the batteries are transported by truck within the US to a port, shipped to the EU and driven the last portion by truck to the recycling facility, the total burden for leg 3 from the port to the EU is ~4.7 MJ/1 kg WAAB.* The collection burden is comprised of ~20% Leg one (drop off by consumers) and ~80% Leg three (recycling center trip) and is shown in the first bar of the graph shown in Figure 18. The dark blue bar above the x-axis indicates the burden associated with the processing in this scenario driven by the energy to drive the high temperature steps. The next bar to the right depicts the materials benefit associated with this recycling scenario based on the amount of material in the battery that is recovered and at what value. For Scenario E it contains two sections: the darker blue associated with the benefit of Zn recycling and the lighter blue associated with Fe and Mn metal value for steel. For the case of Valdi and Baltrec ferromanganese is recovered. The final bar shows the net impact of drop off municipal collection with Scenario E disposition. The average net impact is greater than zero indicating that it is likely a net environmental burden.

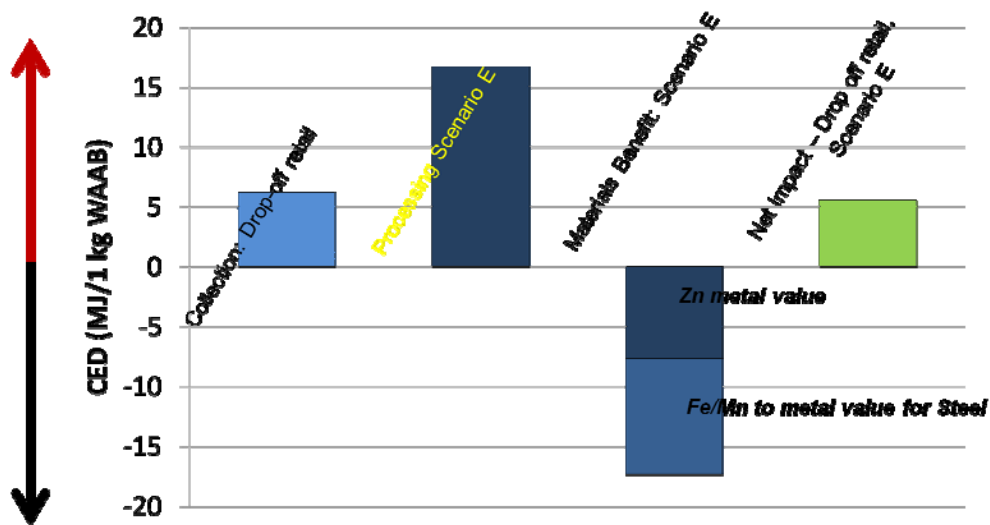


FIGURE 18. COLLECTION ACCORDING TO DROP-OFF RETAIL, DISPOSITION AND MATERIALS BENEFIT OF RECYCLED BATTERIES IN SCENARIO E FOR 1 KG WAAB.

If instead this same recycling scenario were in conjunction with the drop off municipal scenario, which has slightly different range of values as show in Table 18, the burden of collection becomes higher and

therefore the net impact shifted more towards an environmental burden as shown in Figure 19. Here the burden assumed in collection involves 3 changes from the retail scenario: the allocation of the personal vehicle trip is greater, the average miles traveled is greater, and a portion of the consolidation journey is allocated to the batteries.

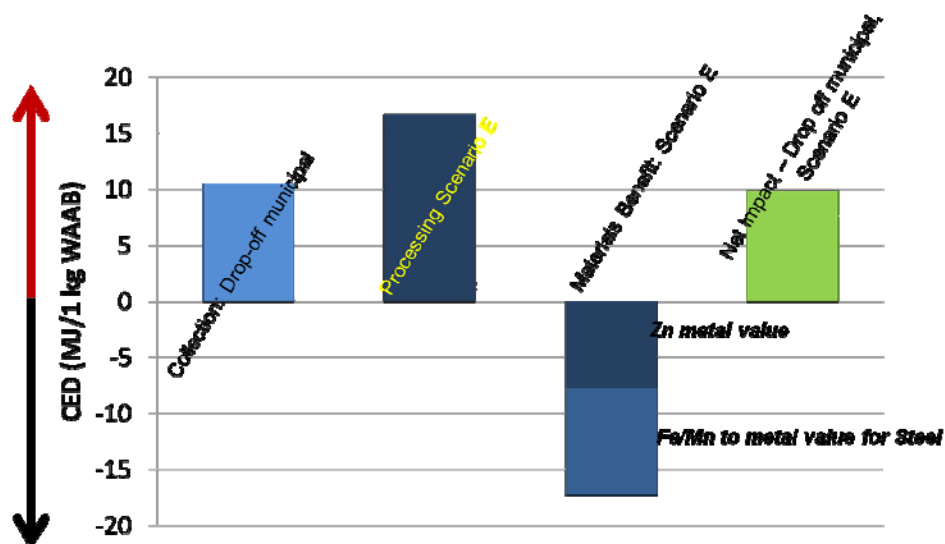


FIGURE 19. COLLECTION ACCORDING TO DROP-OFF MUNICIPAL, DISPOSITION AND MATERIALS BENEFIT OF RECYCLED BATTERIES IN SCENARIO E FOR 1 KG WAAB.

The previous set of figures has demonstrated the convention for presenting results in this report and the next set of figures show only the net impact of several scenarios considered to compare the various burdens and benefits due to alkaline battery disposition including the uncertainty associated with the scenario. Figure 20 compares the cumulative energy demand impact of MSW1 with Recycling scenarios A – E including retail drop off.

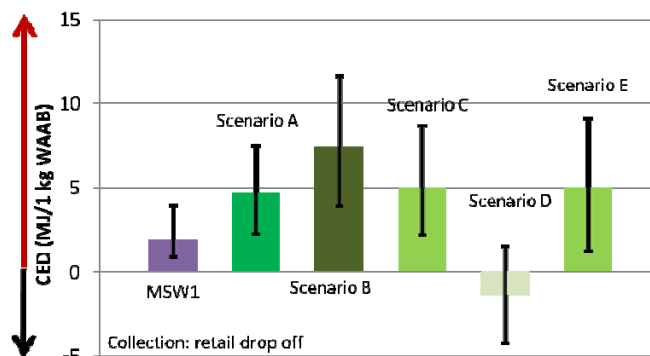


FIGURE 20. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH RETAIL DROP OFF USING CED FOR 1 KG WAAB.

The first and last bars on Figure 20 are the same bars from Figure 17 and Figure 18 for MSW1 and Scenario E with retail drop-off respectively. The middle bars show the net impact of recycling Scenarios

A – D with retail drop-off. We see from this figure that the recycling scenarios cover a wide range of impacts, from more than four times the impact for Scenario B, to at about neutral impact to slightly beneficial for Scenario D (potentially environmentally burdensome or beneficial when considering the uncertainty). These scenarios are a strong function of the materials recovered and the energy required in the recycling process. Because of the way these results are aggregated they do not allow for direct comparison of the recycling **technologies** because each specific facility location and energy source is bundled into the results. The results are rather a comparison of the recycling **scenarios**.

- For Scenarios A, C, and E the energy required in the recycling process about equals the benefits of materials recovery so that the transportation burden serves to bring the net impact above 0. For Scenario B, the processing burden is even less offset by the materials recovery.
- For Scenario D the materials benefit just offsets the burden in recycling such that the transportation burden is balanced or negated through the materials benefit credit. The transportation burden for Scenario D is quite low compared to the others (over a factor of ten lower, because of the distribution of EAF facilities compared to the few sparsely located battery recycling facilities), so that drives its environmental performance; however, this distribution of EAF's is only hypothetical at this point, as they are not currently used, or permitted for use, in recycling EOL batteries.

There is also a great deal of overlap between the scenarios when variation is taken into account, indicating the uncertainty in this result. For example Scenario A's lower bound is within the uncertainty of MSW1. The error bars in the plot indicate the 5% and 95% percentiles of the combined uncertainty from the Monte Carlo simulation accounting for the variation among the transportation distances assumed, the uncertainty in the energy for the recycling process, and the credit assigned to the materials recovered. Figure 21 shows the same recycling scenarios as Figure 20 but now the collection is via municipal drop-off rather than retail drop-off. Because the modeled burden of municipal drop-off is higher than that for retail (both due to allocation and distance assumptions) the overall impact for all scenarios increases. In this case, only Scenario D seems to have the slight possibility of being beneficial as compared with MSW1 when the scenario uncertainty is factored in. The environmental impact between the retail and municipal drop off scenarios increases by about 3-4 MJ/kg of WAAB.

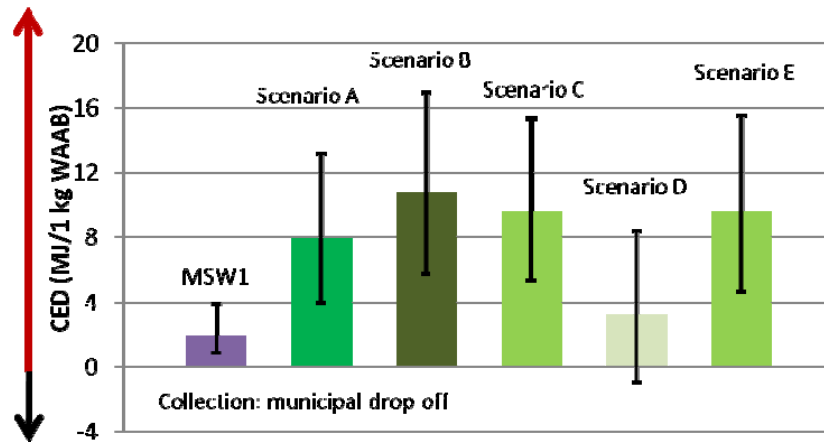


FIGURE 21. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH MUNICIPAL DROP OFF USING CED FOR 1 KG WAAB.

The results graphs so far have demonstrated the environmental impact of various recycling scenarios according to the proxy metric, cumulative energy demand. The MSW and Recycle co-collection schemes are not depicted graphically, but will be commented on here. Both co-collection schemes modify the first leg of the recycling scenario (replacing the individual consumer drop off travel with the MSW vehicle journey). The modeling of the MSW co-collection results in values similar to the municipal drop-off scenario when the US population density range is accounted for. This average population density is also the value that is used in the MSW1 scenario. For a low population density this number rises and for a high population density the total vehicle miles traveled is lower than the average burden, but this also lowers the transportation impact in MSW1 so the overall relative impacts do not change. For the Recycle co-collection the burden is higher than the Municipal drop-off but lower than the Retail drop-off so this result falls between Figure 20 and Figure 21.

Based on the uncertainty analysis the results for each scenario which factors have the most impact on the results using CED are outlined below:

- Scenario A - Retail: Impact of recycling, credit for Zn & Leg one allocation; Municipal; Leg one burden and allocation
- Scenario B - Retail: Impact of recycling, credit for Zn & Leg three burden; Municipal; Leg one allocation, Impact of recycling & Leg three burden
- Scenario C - Retail: Leg three burden, credit for Zn & Mn (micronutrient); Municipal; Leg one burden and allocation, Leg three burden
- Scenario D - Retail: Impact of recycling, credit for Zn & Leg one allocation; Municipal; Leg one burden and allocation, Impact of recycling

- Scenario E – Retail: Impact of recycling & credit for Zn; Municipal: Impact of recycling & Leg one burden

Figure 22 below show the same result as above in Figure 20 but with a breakdown of the CED impact for the recycling scenarios which draw fuel for energy from a variety of different sources.

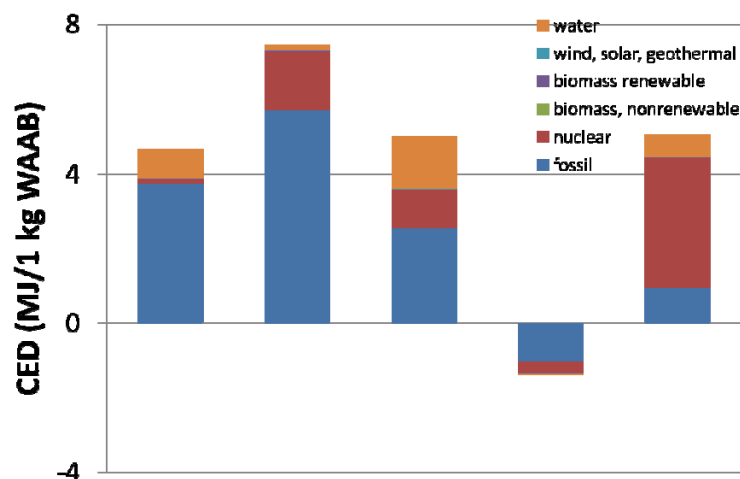


FIGURE 22. NET IMPACT OF SCENARIOS A – E WITH RETAIL DROP OFF USING CED SHOWING BREAKDOWN BY ENERGY SOURCE FOR 1 KG WAAB.

Because of the dominance of nuclear energy in the French and Swiss electrical grid mix, the metric of total amount of radioactive waste was also extracted for the recycling scenarios A-E. Those totals are presented below in Table 23.

TABLE 23. AMOUNT OF RADIOACTIVE WASTE FOR THE PROCESSING BURDEN ASSOCIATED WITH RECYCLING SCENARIOS A-E.

Scenario	Amount of radioactive waste (mm ³ /kg WAAB)
A	1
B	11
C	3
D	10
E	80

Leg 2 is not a large part of the transportation impact (accounting for less than 5% of the burden). However, there are scenarios when leg two is on an empty backhaul for a delivery truck so Figure 23 below shows the impact with no burden allocated to leg 2.

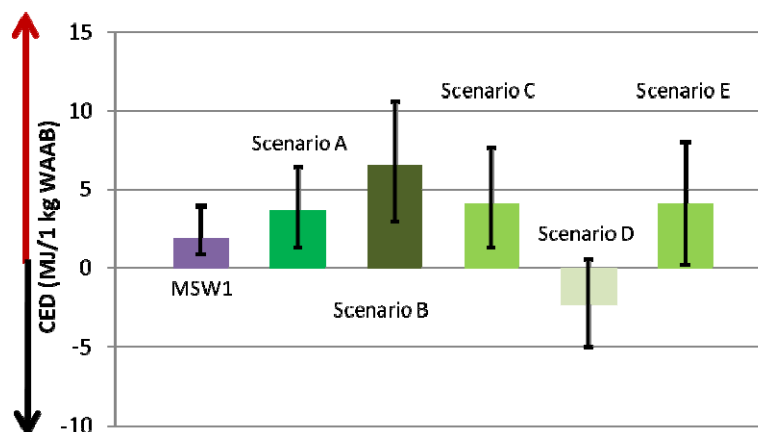


FIGURE 23. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH RETAIL DROP OFF USING CED FOR 1 KG WAAB, NO BURDEN ASSOCIATED WITH LEG 2.

The next series of plots show similar scenarios but now using global warming potential, a method that tracks the quantity, atmospheric longevity, and warming intensity of green house gases. The trends are similar as for cumulative energy demand but there are differences. The impact assessment method GWP, or global warming potential, presents results in units of kg CO₂ equivalents, rather than MJ (per 1 kg of WAAB), therefore the values (and therefore the axes) between the graphs to follow and those above are not directly comparable. Figure 24 shows the net impact of MSW1 and each of the recycling scenarios with retail drop-off collection using GWP. The main difference between the results from the perspective of global warming gases stems from the fuel or energy carriers used in the recycling scenario. The energy for Scenario A is primarily derived from coal, which has the highest global warming potential of the fuels incorporated in this study. There is also some hydropower used in Scenario A, but the total amount of electricity in that scenario is small and it is this coal-derived CO₂ burden that increases this recycling technology over Scenario B as compared to CED. Scenarios B and D derive their energy for heating from natural gas, coal, and the average US electricity grid is also used (composed of coal, natural gas and nuclear power). Scenario C incorporates some natural gas and an average Canadian electricity grid, which is dominated by hydropower, lowering the overall global warming potential for this scenario as compared to the energy burden. The final scenario, which is hypothetical for the US, includes the Swiss and French electricity grid mixes dominated by hydro and nuclear power. The GWP of Scenario E would be much higher if US-average grids were used (see the energy result in Figure 20 and Figure 21). This figure indicates that Scenarios D and E could be environmentally beneficial or slightly less than landfilling.

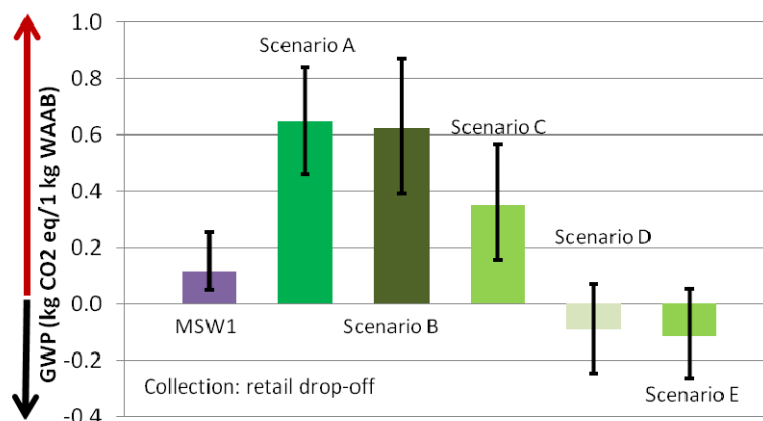


FIGURE 24. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH RETAIL DROP OFF USING GWP FOR 1 KG WAAB.

Figure 25 is analogous to Figure 21 but for GWP rather than CED. Again, because the modeled emissions from municipal drop-off are greater than that of retail drop off (both due to allocation and distance assumptions) the burden is increased for municipal drop off is greater. In this case, Scenario C and E with the uncertainty included are on the order of MSW1, and only hypothetical Scenario D seems beneficial as compared with MSW1. The environmental impact between the retail and municipal drop off scenarios increases by about 0.2 - 0.25 kg CO₂/ 1 kg of WAAB.

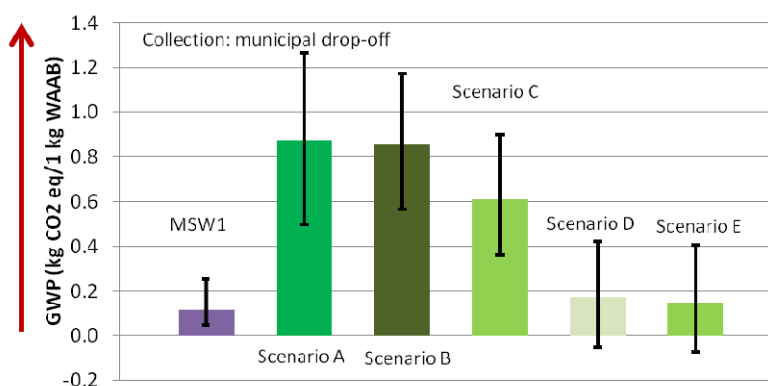


FIGURE 25. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH MUNICIPAL DROP OFF USING GWP FOR 1 KG WAAB.

The previous plots have demonstrated the environmental impact of various recycling scenarios according to GWP. For the case of GWP, Scenarios A, B and C are generally more environmentally burdensome than MSW1 and Scenarios D & E are generally environmentally similar in magnitude to MSW1, but may be environmentally beneficial considering uncertainty.

Based on the uncertainty analysis the results for each scenario which factors have the most impact on the results using GWP are outlined below:

- Scenario A - Retail: Impact of recycling, leg 3 burden, credit for Zn & Leg one allocation; Municipal; Impact of recycling, Leg one burden and allocation
- Scenario B - Retail: Impact of recycling, Leg three burden; Municipal; Leg one allocation, Impact of recycling & Leg three burden
- Scenario C - Retail: Leg three burden, credit for Zn & Mn (micronutrient); Municipal; Leg one burden and allocation, Leg three burden, credit for Zn & Mn (micronutrient)
- Scenario D - Retail: Impact of recycling, credit for Mn & Leg one allocation; Municipal; Leg one burden and allocation, Impact of recycling
- Scenario E – Retail: Leg one and Leg three burden & credit for Zn; Municipal: Leg one burden

The next results focus on the damage category indicators of the impact assessment method, Ecoindicator 99: Ecosystem Quality, Human Health and Resources.

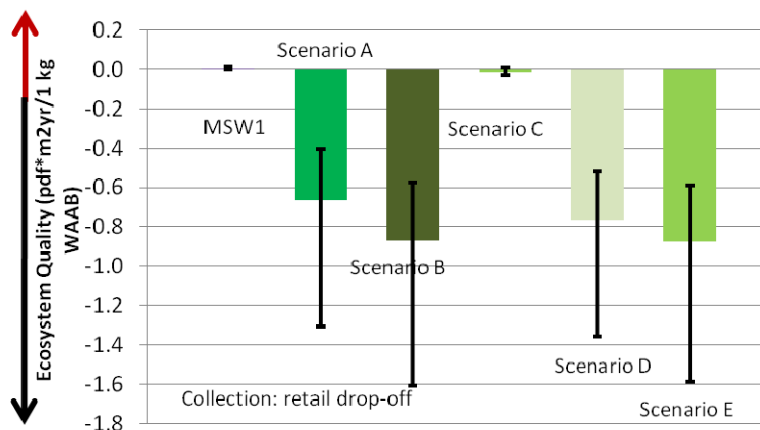


FIGURE 26. NET IMPACT OF MSW1 AND RECYCLING SCENARIOS A – E WITH RETAIL DROP OFF USING THE ECOSYSTEM QUALITY CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

Figure 26 and Figure 27 show the net impact of MSW1 and Scenarios A-E with retail and municipal drop-off, respectively, using ecosystem quality as a measure of environmental burden. This damage assessment category includes ecotoxicity as well as acidification and eutrophication in units of pdf*m²/yr/1 kg WAAB, where pdf represents the potentially disappeared fraction of plant or animal species and therefore these units represent the amount of those species that disappear in a given area over a year. The values for this environmental impact indicator (and therefore the axes) are not directly comparable with those above for GWP and CED. This indicator is driven primarily by the recovery of zinc for metal value from each of the recycling scenarios A, B, D and E, which are more environmentally beneficial than Scenario C, which does not receive metal value credit for the Zn and Mn recovered (rather credited as a replacement for micronutrient manufacture). This burden, according to the ecoinvent database inventory used in this analysis, is driven by the associated toxicity of air emissions of zinc, copper, and cadmium released to the environment in the process of extraction and beneficiating as

well as the blasting and mining processes. These perceived avoided toxicity impact associated with zinc production dominate the materials benefit portion of the recycling scenarios. Therefore, a larger credit is associated with these scenarios. However, there is a large variation in the metrics of this indicator as measures of toxicity and impact on ecosystems have a great degree of uncertainty associated with them. Looking at the uncertainty delineated on these figures, they may be on the order of the MSW1 scenario. A sensitivity to zinc recovery for Scenario D is discussed in the scenario section below.

It can be challenging to have a feel for this indicator as compared to MJ or CO₂/1 kg WAAB, therefore comparing y-axis of Figure 26 - Figure 28 with the values found in the full lifecycle of the battery may be useful. The Ecosystem Quality impact for the whole life cycle of the battery was ~1.8 pdf*m²yr/1 kg WAAB, well above the values seen here.

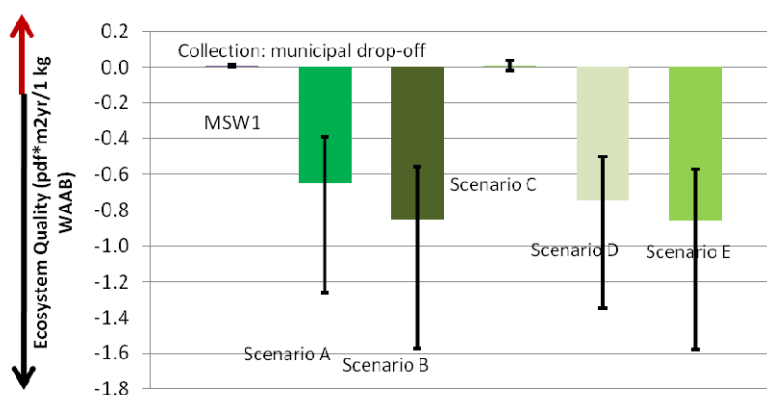


FIGURE 27. NET IMPACT OF MSW1 AND RECYCLING SCENARIOS A – E WITH MUNICIPAL DROP OFF USING THE ECOSYSTEM QUALITY CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

An important scenario of interest for the case of ecosystem quality relates to the potential for substances from the spent alkaline battery to enter the landfill leachate as outlined in the assumptions section above on pages 40 and 41. As was indicated above, the baseline assumption as supported by the CalRecovery report [44] was that none (0) of the metals within the battery are present in the leachate. The other scenarios found in the literature that were used to demonstrate the potential impact of this risk were outlined above and are from [34, 40, 45]. The average of the toxicity measures for Zn, Cu and Ni were combined with the toxicity indicators for those values from Table 21 to provide a total ecosystem quality impact due to metals present in the leachate. This result is presented in Figure 28 under the MSW-leach scenario. Figure 28 demonstrates that with the values of metals in leachate found in the literature andecoinvent above the CalRecovery scenario, the burden of landfiling increases from 0.005 to 0.17 pdf*m²yr/1 kg WAAB or a factor of around 30. Obviously, this value is highly uncertain and because of the zinc credit the majority of recycling scenarios investigated are environmentally beneficial.

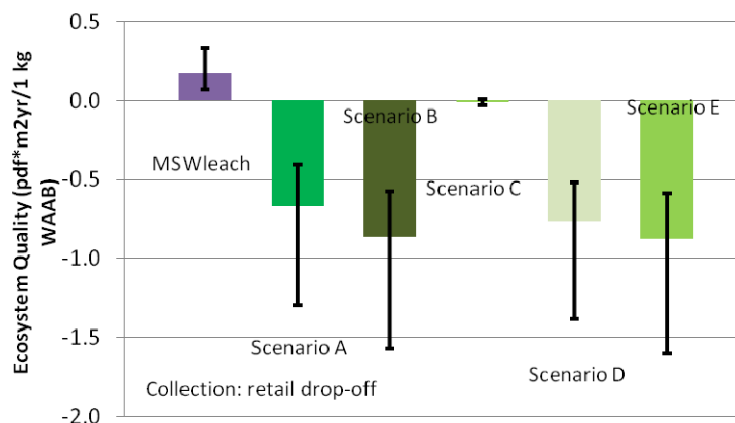


FIGURE 28. NET IMPACT OF MSW-LEACH AND SCENARIOS A – E WITH RETAIL DROP OFF USING THE ECOSYSTEM QUALITY CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

If the metals within the landfilled batteries are mobile species within the landfill and then leach outside of the landfill once it is not managed any more or the leachate is not collected, the far left purple bar in Figure 28 could be larger. The ecoinvent landfilling inventories models landfill leachate simulating tens of thousands of years after the initial disposition of material. These values could be considered as a further scenario in this study. However, in order to provide a meaningful comparison the recycling scenarios and their byproducts should also be subject to the same tens of thousands of year timescale. The data to consider this scenario are extremely speculative and difficult to obtain.

Based on the uncertainty analysis the results for each scenario which factors have the most impact on the results using the damage category ecosystem quality are outlined below:

- Scenario A, B, D & E for both retail and municipal drop off the credit from zinc is the most significant parameter
- Scenario C - Retail: Leg three burden, credit for Zn & Mn (micronutrient); Municipal; Leg one burden and allocation, Leg three burden, credit for Zn & Mn (micronutrient)

The next category demonstrated here is for the Human Health damage assessment category found within Ecoindicator 99 and is in units of DALYs/1 kg WAAB or disability adjusted life years, describing the impact of various releases on human health in terms of diseases caused. Therefore, as before, the values are not comparable between this indicator and the others used in this study. The category includes carcinogens, respiratory inorganics and organics, radiation, climate change and ozone depletion. Figure 29 and Figure 30 show the net impact of MSW1 compared to Scenarios A – E with retail and municipal drop off collection, respectively. Overall, the variation in this metric is not as great as with Ecosystem Quality. With uncertainty considered, scenario A for recycling is environmentally burdensome and greater than MSW1. The other recycling scenarios are beneficial from the perspective of human health with Scenario B potentially being environmentally burdensome.

The benefit associated with Scenarios B and C according to human health is driven fairly evenly by the balance of the transport, recycling contribution and offset by the materials benefit from the metal value of Zn or steel in the case of C (although less significantly than the Ecosystem Quality benefit perceived by avoided extraction of these metals). Scenario D is dominated by the value perceived in avoided steel extraction (the manganese and steel report to the metal value substituting basic oxygen furnace-steel or even ferromanganese in some cases where Mn would have been added⁸). Scenario E is dominated by the value perceived in avoided zinc and ferromanganese extraction and production.

Another factor of interest for this scenario is that the difference between the collection scenarios (municipal versus retail drop-off) in Figure 29 and Figure 30 is not as significant as previous indicators. This will be further illustrated in the scenario analysis below.

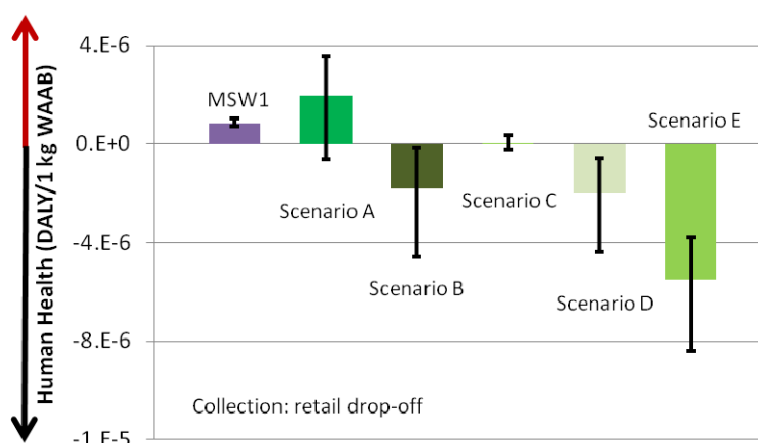


FIGURE 29. NET IMPACT OF MSW1 AND SCENARIOS A – D WITH RETAIL DROP-OFF USING THE HUMAN HEALTH CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

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⁸ Conversations with steel minimill facility workers indicated that Mn would not always be needed as an additive in production.

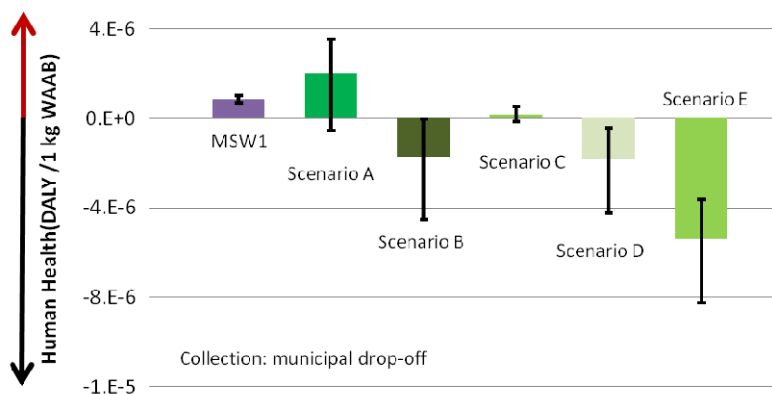


FIGURE 30. NET IMPACT OF MSW1 AND SCENARIOS A – D WITH MUNICIPAL DROP-OFF USING THE HUMAN HEALTH CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

Based on the uncertainty analysis the results for each scenario which factors have the most impact on the results using Human Health are outlined below:

- Scenario A - Retail: Impact of recycling, credit for Zn; Municipal: credit for Zn
- Scenario B - Retail: credit for Zn; Municipal: Leg one allocation, credit for Zn
- Scenario C - Retail: Leg three burden, credit for Zn & Mn (micronutrient); Municipal: Leg one burden and allocation, Leg three burden, credit for Zn & Mn (micronutrient)
- Scenario D – Retail & Municipal: credit for Mn & Zn
- Scenario E – Retail & Municipal: credit for Zn, Mn and steel.

Finally, the Resources damage category is considered for the same scenarios of retail and municipal drop off in units of MJ surplus as shown in Figure 31 and Figure 32. This metric looks similar to GWP with Scenario D looking like it would be more beneficial than in GWP (for the case of retail drop off).

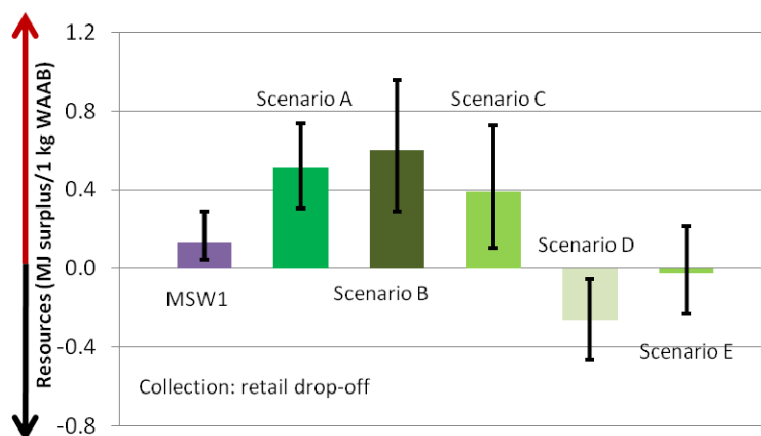


FIGURE 31. NET IMPACT OF MSW1 AND SCENARIOS A – D WITH RETAIL DROP-OFF USING THE RESOURCES CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

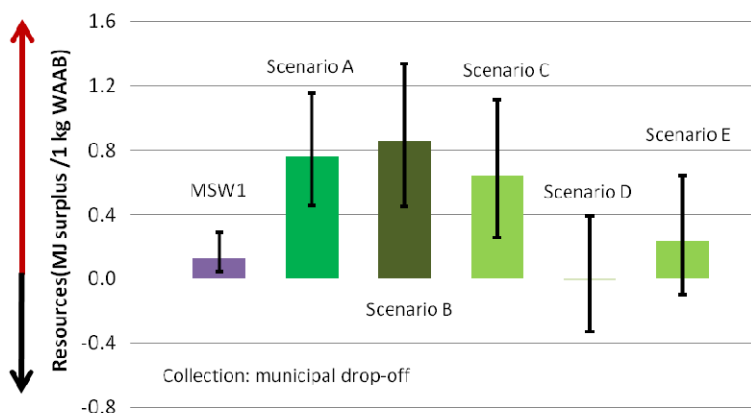


FIGURE 32. NET IMPACT OF MSW1 AND SCENARIOS A – D WITH MUNICIPAL DROP-OFF USING THE RESOURCES CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

INCINERATION SCENARIO

As described in detail in Chapter 5, incineration provides another option for end of life alkaline batteries, although not common in the US. The assumptions in this analysis are that batteries are transported through the MSW scenarios to incineration facilities that require further distances traveled than for the landfill because there are fewer incinerators throughout the country. There is a burden associated with incineration despite refuse-derived fuel ubiquity because less energy recovery would be present in an incineration with higher metal content than typical. That steel is recovered for recycling in a steel minimill (transport added). There is a burden associated with the mass of steel sent to EAF and credit awarded to this recovery based on more oxidized iron content than the recycling scenarios. Zinc is recovered for metallic value in small percentage related to the amount of Zn vaporization and the remainder of the slag used for road construction. This scenario for incineration is based on a situation

where ferrous recovery was occurring at the incineration facility regardless of whether the batteries remain intact. The environmental impact of this scenario is presented below for GWP in Figure 33.

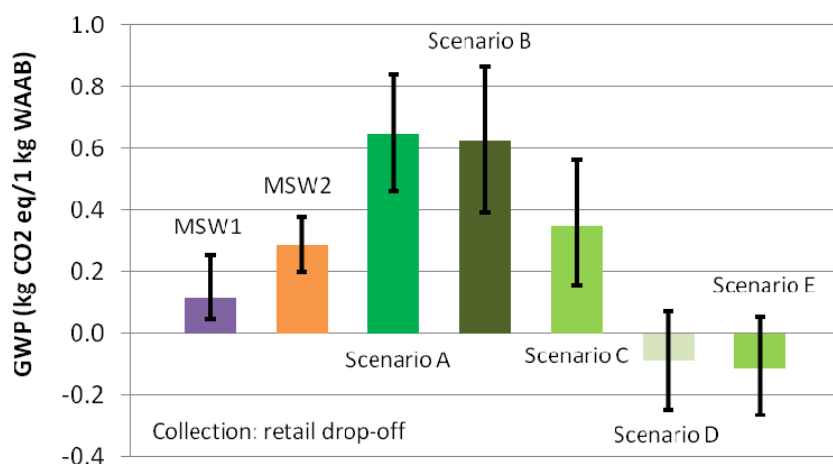


FIGURE 33. NET IMPACT OF MSW1: LANDFILL, INCINERATION AND SCENARIOS A – E WITH RETAIL DROP OFF USING GWP FOR 1 KG WAAB.

The burden associated with the materials recovery in incineration is driven in part by the EAF used to recover steel after it reports to bottom ash and is separated magnetically. Some minimal zinc recovery is possible within incineration from the fly ash (although given the lower temperatures than typical EAF, this amount may be lower as all zinc may not completely volatilize at these lower temperatures, despite being above the melting point of Zn). Therefore, incineration provides some potential materials recovery in the case of steel and some zinc recovery for metallic value. MSW2 and Scenario D present similarities in that the primary burden derives from EAF energy inputs, however in the case of Scenario D, more zinc recovery may be indicated and potential more mass recovery of the manganese oxides.

The next section presents several scenario analyses on several key parameters within the results section.

CHAPTER 7: PARAMETER ANALYSIS

A few parameter analyses or sensitivities are presented in this section to explore the influence of parameters other than those covered in previous sections. First, the impact of collection percentage was examined based on the collection targets set forth by the European Commission. This analysis demonstrates what would occur were these collection targets adopted in the US as well. These targets are “collections-to-sales” performance goals of 25% by 2012 and 45% by 2016. These are the EU values rather than those in the US, but they provide a reasonable number for what might be adopted in the US. These sensitivities captured in Figure 34 and Figure 35 project these percentages onto a US scenario such that the remainder is sent to landfill. Therefore, Figure 34 shows the burden of taking the 1 kg battery functional unit and allocating 25% of it to recycling and 75% of it to landfill. Figure 35 does a similar division of the burden except that 45% are sent to recycling in each of the scenarios and 55% are sent to landfill.

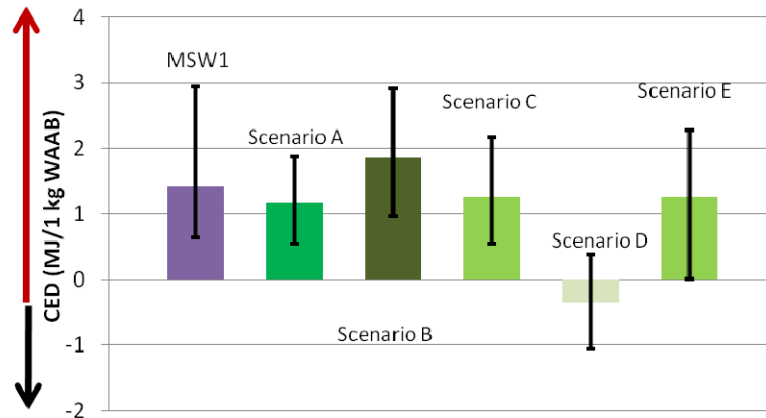


FIGURE 34. NET IMPACT OF MSW1 AND RECYCLING SCEN. A – E WITH RETAIL DROP OFF USING CED WITH 75% LANDFILL AND 25% RECYCLING (FOR EACH SCENARIO INDEPENDENTLY) FOR 1 KG WAAB.

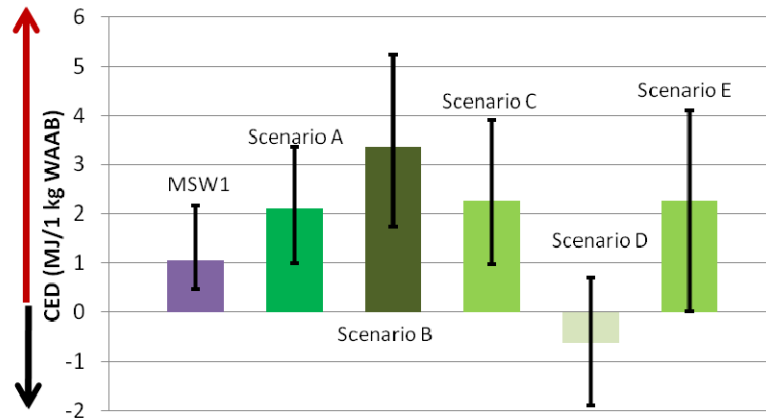


FIGURE 35. NET IMPACT OF MSW1 AND RECYCLING SCEN. A – E WITH RETAIL DROP OFF USING CED WITH 55% LANDFILL AND 45% RECYCLING (FOR EACH SCENARIO INDEPENDENTLY) FOR 1 KG WAAB.

The next analysis explores the impact of the individual consumer transport leg of the drop off collection scheme to the two parameters: allocation and mileage traveled. The allocation percentage contains a high degree of uncertainty. These two elements are the focus of this analysis because *leg one* provides a significant portion of the environmental burden using CED, GWP, and Resources because of this sensitivity to allocation of customer transport. As explained in the assumptions section, determining the allocation for the individual consumer travel is fraught with uncertainty as each individual consumer will likely make a different journey with a different destination or multitude of destinations. This may be one reason why this leg of the journey was left off in the work done by [9]. However, this is not an insignificant burden as shown by the results. Table 24 demonstrates the crossover point between the MSW1 scenario and each recycling scenario based on changes in the allocation or distance traveled parameters for each environmental impact assessment indicator used in this study. This analysis shows, for example, that if the allocation of a personal consumer trip is either 14% or the consumer travels for 22 km, then Scenario D will be equivalent in impact to MSW1. One interpretation of this allocation

percentage could be that there are slightly greater than 6-7 other reasons (or that one other reason is ~6 times more important to the consumer) that equally share the burden of the consumer's personal travel. The 22 km drop-off distance is more reflective of lower population densities requiring consumers to travel greater distances.

TABLE 24. SENSITIVITY ANALYSIS ON LEG ONE WITHIN DROP-OFF SCENARIOS FOR THE INDICATORS USED IN THIS REPORT (CED, GWP, ECOSYSTEM QUALITY, HUMAN HEALTH, AND RESOURCES) DEMONSTRATING THE CROSSOVER POINT WITH MSW1 WITH EITHER THE ALLOCATION VARIABLE –OR– THE DISTANCE TRAVELED.

	Cross over with MSW1 for:	
CED		
<i>Scenarios of interest</i>	Allocation for fixed distance	Distance (km) for fixed allocation
Scenario D	14%	22
GWP		
<i>Scenarios of interest</i>		
Scenario D	11%	18
Scenario E	9%	15
Ecosystem Quality		
<i>Scenarios of interest</i>		
Scenario B	62%	100
Scenario C	22%	35
Scenario D	42%	100
Scenario E	68%	>100
Human Health		
<i>Scenarios of interest</i>		
Scenario B	55%	125
Scenario C	90%	> 200
Scenario D	N/A	375
Resources		
<i>Scenarios of interest</i>		
Scenario D	22%	35

It is of interest to examine a retail drop-off scenario where no allocation is made to the first leg of travel for the retail drop-off scenario. This assumes an entirely non-dedicated trip to deliver spent batteries to the drop off location, the retail store. This parameter analysis is presented below for CED, GWP and Resources.

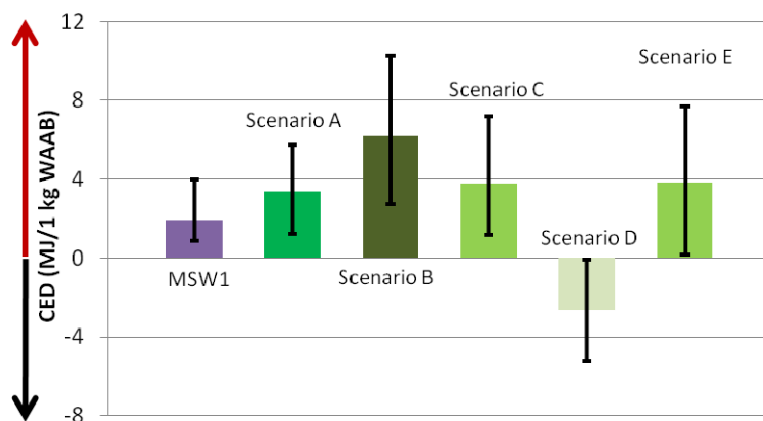


FIGURE 36. NET IMPACT OF MSW1 AND RECYCLING SCENARIOS A – E USING CED WITH NO ALLOCATION TO LEG 1 OF TRANSPORT FOR 1 KG WAAB

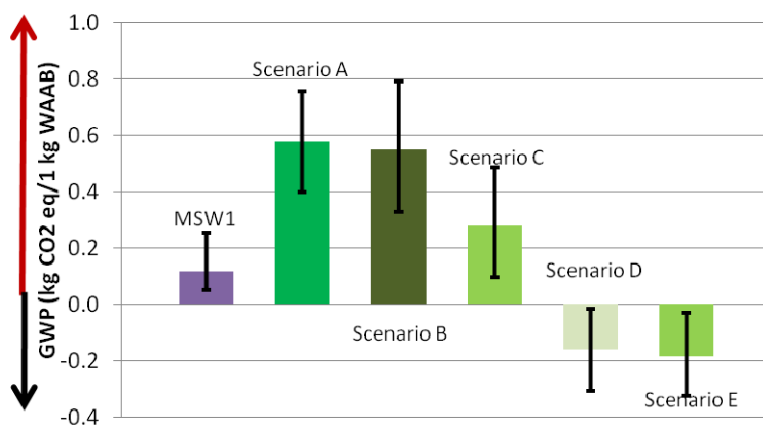


FIGURE 37 NET IMPACT OF MSW1 AND RECYCLING SCENARIOS A – E USING GWP WITH NO ALLOCATION TO LEG 1 OF TRANSPORT FOR 1 KG WAAB

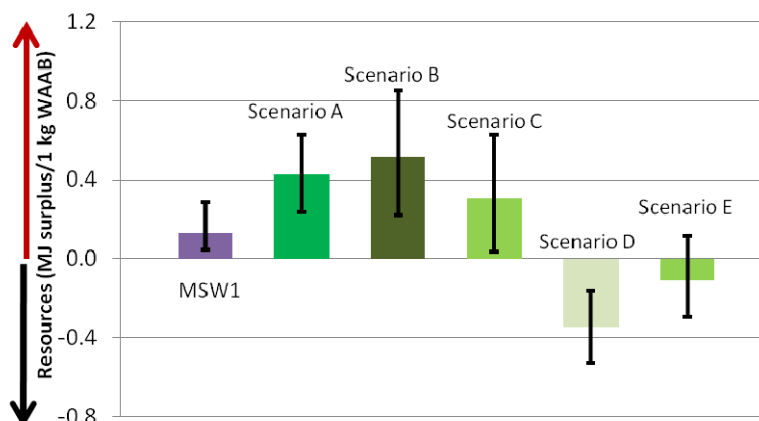


FIGURE 38. NET IMPACT OF MSW1 AND RECYCLING SCENARIOS A – E WITH RETAIL DROP OFF USING RESOURCES WITH NO ALLOCATION TO LEG 1 OF TRANSPORT FOR 1 KG WAAB

Table 25 illustrates the percentage of transport for the retail drop-off scenario attributed to leg 1 for each of the scenarios for the retail drop off collection method using the impact assessment methods used in this study.

TABLE 25. PERCENTAGE OF THE TRANSPORT IMPACT ASSOCIATED WITH LEG 1

	Scenario A	Scenario B	Scenario C	Scenario D	Scenario E
CED	25%	15%	14%	50%	18%
GWP	20%	12%	11%	45%	15%
HH	12%	6%	6%	31%	7%
EQ	21%	12%	12%	41%	14%
R	19%	11%	11%	42%	14%

Next, zinc recovery within the EAF scenario, Scenario D, is considered. The reason for this particular sensitivity around materials recovery is that the benefit of this scenario is dependent on the zinc recovery that is achieved. The baseline assumption for Scenario D was a ~75% recovery on zinc. Figure 39 presents a comparison between the MSW 1 burden and Scenario D as a function of percent zinc recovery for four indicators of environmental performance, CED, GWP, Ecosystem quality (EQ), Human Health (HH), and Resources (R). This analysis demonstrates that as zinc recovery decreases from 75% to 10% the benefit of Scenario D decreases and approaches the burden of MSW1, in the case of CED and GWP, it crosses the purple MSW1 line at around 40% and 32% zinc recovery, respectively, and becomes a net burden in comparison. For resources, it has crossed the x-axis, indicating generally a net environmental burden but perhaps not greater than the MSW1 scenario. It should be noted that there should be an error bar on each of the points shown below, indicating the uncertainty in the calculation.

The methods are shown side by side in normalized space in Figure 40. This figure shows all five indicators CED, GWP, EQ, HH and R as a function of zinc recovery normalized to the impact of the MSW1

scenario. Therefore, the metric on the y-axis shows the burden of Scenario D over that of MSW1, a positive number indicates the burden associated with Scenario D is greater than MSW1.

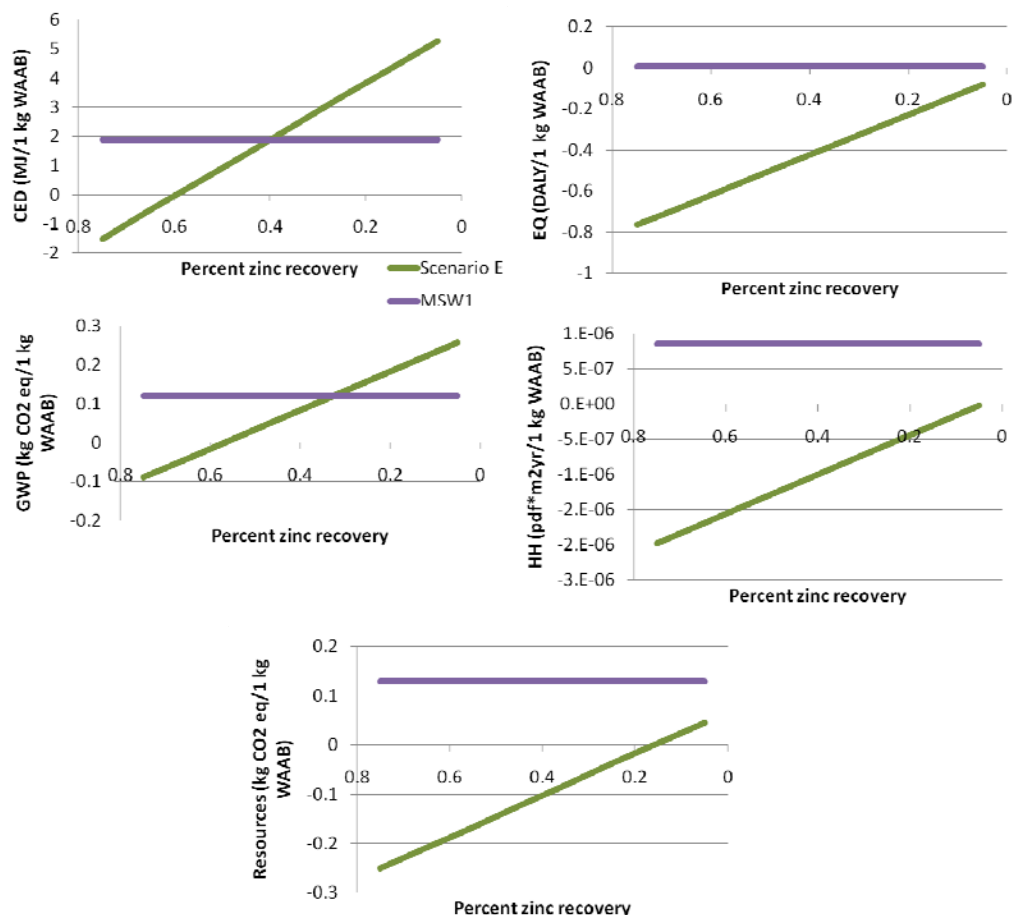


FIGURE 39. SENSITIVITY AROUND ZINC RECOVERY FOR CED, GWP, ECOSYSTEM QUALITY (EQ) AND HUMAN HEALTH (HH) FOR SCENARIO D FOR 1 KG WAAB.

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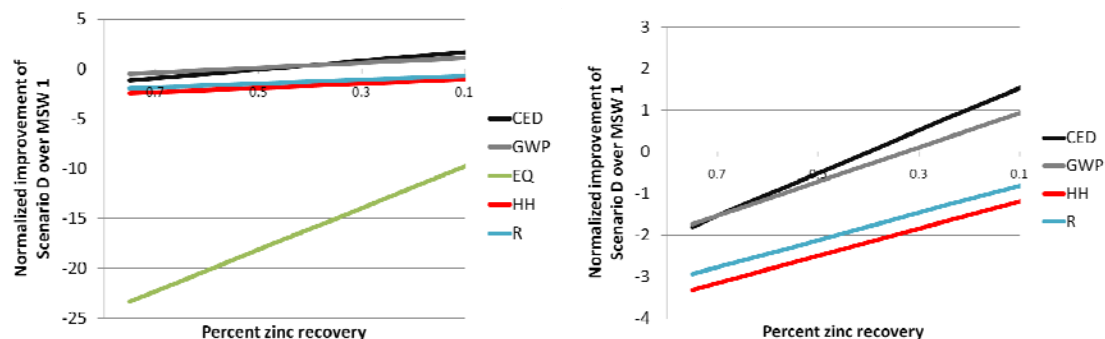


FIGURE 40. NORMALIZED IMPROVEMENT OF SCENARIO D OVER MSW 1 AS A FUNCTION OF PERCENT ZINC RECOVERY FOR FOUR INDICATORS, CED, GWP, EQ, HH AND R.

The final parameter analysis of interest attempts to place each of the recycling scenarios on a level comparison using the same electrical grid for any electricity required in processing and an average transportation distance to the recycler based on all the scenarios used in the study. The results of this parameter analysis are shown in Figure 41 and Figure 42. For both figures the electrical grid used is the US electrical grid (the other energy sources remain the same) and for Figure 42 an average GWP impact for leg 3 transport (to the recycler) is used for all of the recycling scenarios. It is important to note that using an average fuel mix for the US electrical grid locally has uncertainty due to significant regional variation in this mix (and therefore uncertainty in GWP).

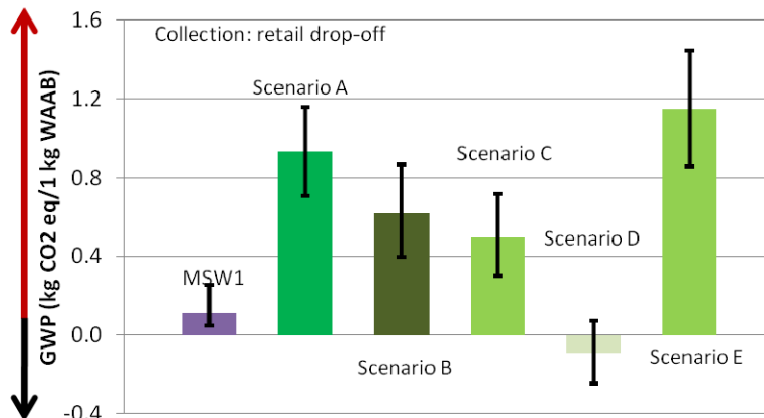


FIGURE 41. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH US ELECTRICAL GRID WITH RETAIL DROP OFF USING GWP FOR 1 KG WAAB.

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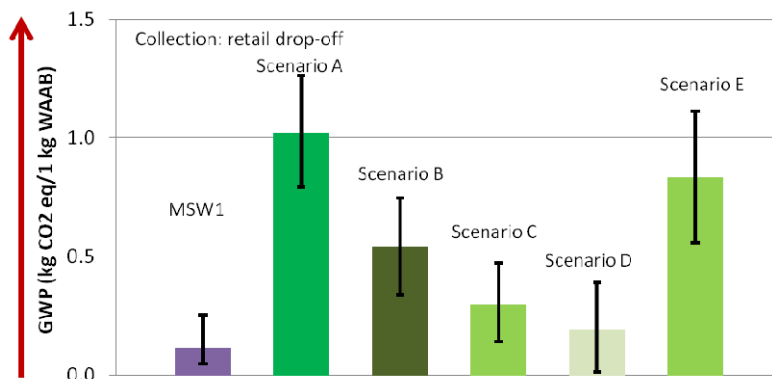


FIGURE 42. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH US ELECTRICAL GRID AND AVERAGE TRANSPORTATION FOR LEG 3 WITH RETAIL DROP OFF USING GWP FOR 1 KG WAAB.

The GWP impact with a US electrical grid is reminiscent of Figure 20 for CED. The impact from Scenario A shows an increase from the baseline (c.f. Figure 24) because of the less carbon intensive hydropower electricity generation for the baseline case. Scenario B remains unchanged and Scenario D becomes less environmentally beneficial. The biggest difference between the previous GWP results is found in Scenario E. The grids assumed in the baseline analysis were both of low carbon intensity: France and Switzerland, both dominated by nuclear and hydropower. Scenario E is now more environmentally burdensome and the results are similar to Scenario A (the increased materials recovery is offset by the increased energy intensity process). Figure 42 shows that with an average transportation burden Scenario C is less burdensome and Scenario D could be similar in environmental performance according to GWP to landfilling.

A similar analysis around common electrical grid is shown below for ecosystem quality, human health and resources in Figure 43, Figure 44, and Figure 45, respectively.

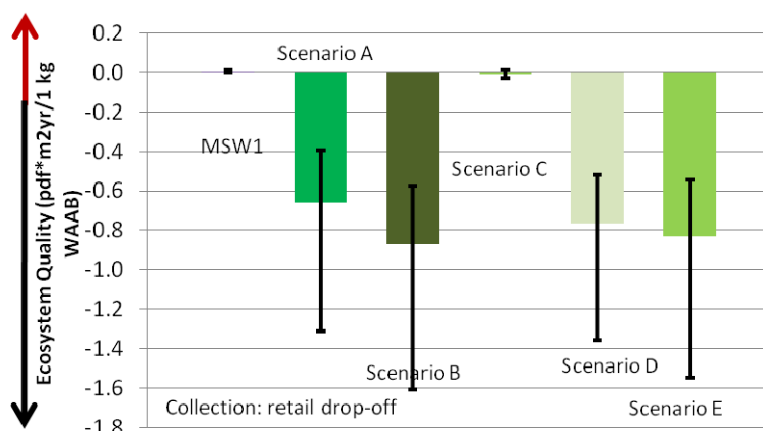


FIGURE 43. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH US ELECTRICAL GRID WITH RETAIL DROP OFF USING THE ECOSYSTEM QUALITY CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

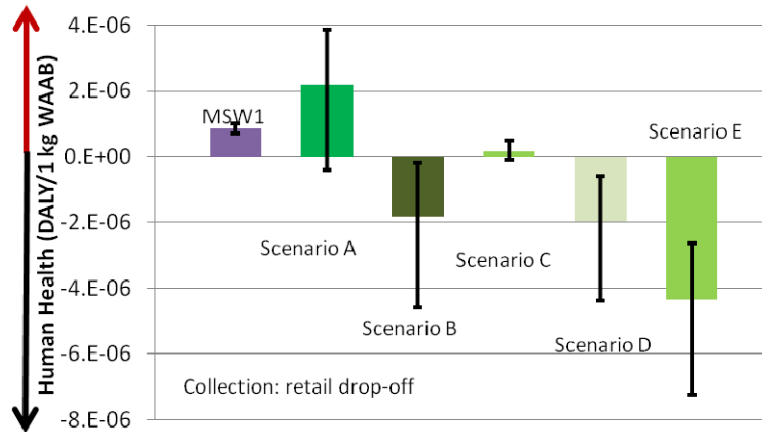


FIGURE 44. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH US ELECTRICAL GRID WITH RETAIL DROP OFF USING THE HUMAN HEALTH CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB.

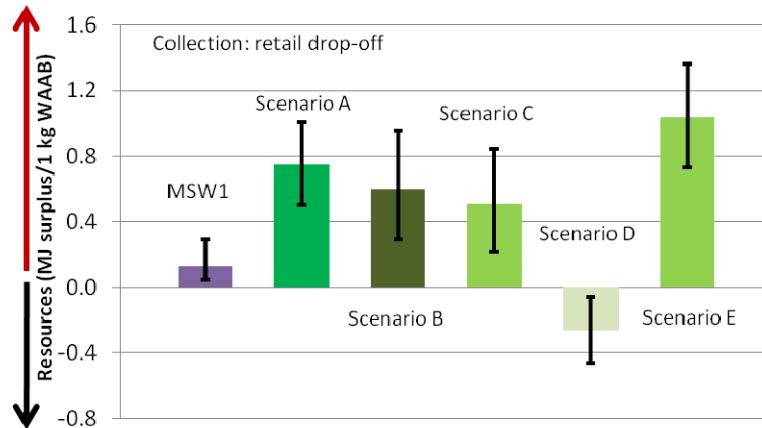


FIGURE 45. NET IMPACT OF MSW1 AND SCENARIOS A – E WITH US ELECTRICAL GRID WITH RETAIL DROP OFF USING THE RESOURCES CATEGORY WITHIN ECOINDICATOR 99 FOR 1 KG WAAB

CONCLUSIONS

Because of the importance of data within life cycle assessment, we begin the conclusions with an assessment of the data quality throughout the study. For section 1, the bill of materials was of reasonable quality and it is assumed that the conclusions derived from that bill of materials are acceptable for the level of analysis that was required here. It was obtained through survey collection with the industry participants and was aggregated by the consortium. This made it difficult to track

down potential errors and there were concerns about a few of the manufacturing burden values such as the amount of VOCs as well as the energy in packaging. Also, detail was left out around the composition of the waste leaving the manufacturing facility. As far as the life cycle inventory data available for the battery materials, it was of reasonable quality as well. The most critical inventories were that for MnO_2 where a proxy from the ecoinvent database developed very recently for lithium-ion batteries was used (different chemistry). Also of importance were the inventories for steel and zinc. The ecoinvent documentation around the primary zinc inventory describes some limitations around missing emissions factors to air and water, but describes its data quality as sufficient for a background database. The electrical grid within manufacturing was also important and that was deemed pretty reliable. The other major data quality issues stem from the toxicity and human health impacts associated with, in particular, zinc present in the battery. These numbers are highly uncertain.

For section two, the data were of varying quality and the conclusions presented here may be impacted by the data gaps and deficiencies. Directionally the results are satisfactory, but the absolute numbers should not be considered reliable as there are significant data gaps. First of all, as described above the vast majority of life cycle inventories used here were from a European context. The most critical inventory from an ecosystem and human health perspective was that for zinc and the data quality around that inventory should be further improved especially as it relates to issues of toxicity. However, zinc is energy intensive to produce and in large quantities seems to have some environmental impact on ecosystems, so assigning some credit where actual offset of material is occurring seems useful. More primary data should be gathered on zinc production itself, including how the zinc recycling market functions within the US. More specifically resolving the following data quality issues would provide the most benefit to this work:

1. Recycling process emissions – although they were not critical to the impact of the recycling process (that impact was dominated by electricity), they provided a large gap in the available data
 - a. Recycling metal yields and energy use, although data were provided, would also have a critical impact and should be verified. Those numbers are believed to be reasonably reliable, but the conclusions would be impacted if they were found to be substantially off.
2. Truck capacities for leg three of the battery recycling scenarios. – where this leg had significant impact it would be of value to know the load factors of the trucks used as that would impact the total burden associated with the batteries
3. Further information from battery recyclers on the flows of their recovered materials and what impacts those flows in terms of economics, volumes, etc.
4. As mentioned above further information on the state of zinc production and the toxicity associated with zinc as found in the US. That life cycle inventory was particularly critical to the study.

The results of this study may be viewed from two different perspectives: a total life cycle perspective, found in section 1, and a focus on end of life, found in section 2. The significant outcomes of the alkaline battery life cycle assessment, section 1, include:

- The production of raw materials dominates the life cycle environmental impact. Therefore, the greatest burden lies far upstream of the manufacturing facility itself.
- Manganese dioxide and zinc represent the largest impacts within the raw materials production.
- Of the phases of the alkaline battery life cycle that fall directly within control of the battery manufacturing industry, the manufacturing facility has the largest impact. Electricity use within this facility drives the environmental impact.

For the collection burden associated with battery recycling, the greatest burden was found to be municipal drop-off because of assumptions around the increased likelihood of a more dedicated trip based on the results in the literature. The distances traveled were also greater. Municipal drop-off was on average 3-4 MJ, 0.2 kg CO₂ eq, 1.3×10^{-7} DALY, 0.015 pdf*m2yr, and 0.25 MJ surplus /kg greater than retail drop-off (ranging from 15% to on the order of the same magnitude compared with the materials processing impact depending on the scenario under consideration). In addition, for the municipal drop-off it was assumed that the consolidation transportation step, leg two, required some small amount of the overall burden. Curbside pickup (both MSW and Recycle co-collection) for the recycling scenarios was determined to be lower in impact than both of the drop-off scenarios. As explored in the parameter analyses, the crossover with the MSW 1 (landfilling) impact varies depending on the allocation and distance assumed for the consumer trip to the drop-off facility

The recycling scenario-specific conclusions for the end-of-life focus are described below. It is important to note that the end-of-life scenarios in this work were based on specific sites including particular electrical grid and distances traveled, therefore these conclusions are not meant to compare the recycling technologies rather to investigate the contexts in which these technologies are employed. Generally for metrics of energy and global warming potential the recycling scenarios tend towards environmentally burdensome while for metrics of toxicity and human health they tend towards environmentally beneficial. When the energy used in recycling is only just offset by the materials recovered, the full burden of transportation drives an environmentally burdensome final impact.

- Scenario A was modeled after a slag fuming furnace using coal and hydro-produced electricity to recover Zn to metal value with steel and manganese dioxides reporting to slag which is sold to cement market. This scenario exhibits an environmental benefit as compared to the baseline MSW landfilling scenario for the metric of ecosystem quality for both municipal and retail drop off. However, this scenario results in a more significant environmental burden than landfilling as measured by Cumulative Energy Demand, Global Warming Potential, human health and resources using any modeled collection method.
- Scenario B was modeled after a process using natural gas, coal and a US electrical grid to heat rotary hearth and electric arc furnaces to recover Zn and steel to metal value with some metal value from manganese dioxides and the remainder reporting to slag which is sold to road construction. This scenario exhibits an environmental benefit or neutrality as compared to the

baseline MSW landfilling scenario for metrics of ecosystem quality and human health for both municipal and retail drop off. However, as with Scenario A, Scenario B also results in a more significant environmental burden than landfilling as measured by Cumulative Energy Demand, Global Warming Potential, and resources using any modeled collection method.

- Scenario C was modeled after a process using a Canadian average electric grid and a natural gas-powered heating process lower temperatures than the previous two scenarios recovering steel to metal value and Zn and manganese dioxides sold for micronutrient replacement exhibits an environmental benefit or neutrality as compared to the baseline MSW landfilling scenario for metrics of ecosystem quality and human health. For CED, GWP, and resources, this scenario is generally environmentally burdensome.

The remaining two scenarios are not currently used in the US and the results are as follows:

- Scenario D which recovers Zn and steel (with volume of manganese dioxide) for metal value exhibits an environmental benefit compared to a baseline landfilling scenario for all metrics used in this study save for neutrality in the case of CED and GWP for municipal drop off. This scenario is modeled after using the EAF infrastructure in the US using a US electrical grid and distributed collection. Because of the copper poisoning to steel production, this scenario is limited by capacity (although based on the number of batteries recovered, this value would not be exceeded). This scenario is sensitive to zinc recovery as shown in the parameter analysis. The sensitivity analyses indicate that recovering less than 32-40 wt% of zinc may lead to environmental burden exceeding the benefit.
- Scenario E recovers zinc, steel and manganese for metal value based on European recycling facilities. The grid is based on the average of French and Swiss electrical grids. The transportation scenario assumes transport by road and ship to the EU. The results indicate for the majority of cases, there is an environmental benefit to this scenario, except for CED and GWP (GWP burdensome for municipal drop off).

Finally, the parameter analysis that employed the US electrical grid for all of the recycling scenarios and an average transportation burden for leg 3 indicates that Scenario D may demonstrate some environmental benefit of recycling compared to landfilling (when considering uncertainty) while Scenario A, C and E are more environmentally burdensome. Scenario B flips depending on the metric of interest, with GWP and resources being environmentally burdensome.

- In terms of energy and global warming potential the benefits of recycling don't often fully outweigh the burdens associated with increased transport and processing (beyond that of landfilling, which requires limited transport and less energy-intensive processing). Only when enough materials by weight are recovered from the batteries to a state where they offset metals production and when the batteries aren't transported long distances will there be environmental benefit according to GWP or CED.
- In terms of human health and ecotoxicity, recycling benefits outweigh the burdens for the majority of scenarios explored driven, in most cases by the recovery of zinc. Therefore, the recycling scenarios should be examined in more detail to determine the impact of increased amount of recycled zinc in the overall zinc system and the long term trend of zinc demand. Also,

there is significant uncertainty in the impact assessment methods associated with these environmental metrics.

RECOMMENDATIONS FOR ACTIONS TO REDUCE THE ENVIRONMENTAL IMPACT

LIFE CYCLE IMPACT

- Within raw materials production the top few materials (manganese dioxide, steel and zinc) have the highest impact. Therefore, involving upstream suppliers of those materials in follow-up actions to reduce environmental impact could lead to burden reduction. Another possible action would be reducing the material thickness in the battery can, for example. Increasing the amount of recycled material used may not reduce the environmental burden as increasing demand for these metals does not directly lead to increased amounts of metals recycled.
- Outside of raw material production, manufacturing plays a significant role in the environmental burden. Improvements in the manufacturing operations, particularly those focused on reducing electricity use reduce the overall burden.

END OF LIFE IMPACT

- The primary recommendation of the end-of-life focus of this study would be to explore the use of distributed EAF facilities within the US. The distributed location of these facilities reduces the overall transport burden, the non-dedicated process reduces the environmental burden of recycling and the ability to recover some metal value from steel and perhaps manganese (to be incorporated into steel) provides an environmental benefit. The viability of this hypothetical scenario rests on the relevant legislation enabling transport and handling by these facilities and the batteries need to be tested through pilots to determine if they can be added without contamination to the EAF process and product quality.
- Within the collection scenarios, the transportation impact was driven by the degree of dedication of consumer trips to drop-off facilities (both municipal and retail). Therefore, education should accompany collection programs to promote non-dedicated trips and consolidated trips. While the mechanics of making this possible would need to be investigated, co-collection with curbside transport was the lowest transport impact.

FUTURE WORK

There are several elements that could be considered in more detail to further refine this analysis. A few of these are highlighted in this section:

- There is not one obvious scenario for battery recycling across all of North America because of differences in population density, the carbon intensity of the electrical grid, and facility location.

Mapping the specific context for the part of the country where recycling will occur will provide the greatest opportunity for the most environmentally beneficial scenario to take place.

- Deepening the understanding of the impacts and potential of Scenario D, the EAF scenario. Because of its comparatively low transportation burden and apparent beneficial outcome, this scenario should be investigated in more detail. Further work might investigate which types of steel minimills in the US would be most conducive to battery recycling and the permitting issues surrounding that option. As mentioned above, copper is a poison in the production of steel. While it was understood in the course of conversations occurring over the course of this work, that the copper present in batteries could be managed in the addition to EAFs, the recovered steel may have less value because of the copper-content that would need to be managed. This should be further investigated.
- The environmental impact of collection sites and collection vessels (due to the extraction and processing of raw materials in collection vessels and electricity burden associated with collection sites) should be added to the analysis to ensure they are not a significant portion of the impact. Previous work has not indicated they should be, but for completeness, this could be added to the analysis [9]. Another scenario to explore would be if batteries are sent to a recycling facility and are not processed therefore the transportation burden would double if they are then sent to landfill.
- Explore more transportation scenarios, such as using the mail service or associating primary battery collection programs with rechargeable where sorting processes are effective and further explorations into the impact of population density. If more environmentally intensive transport modes are employed, such as air, the burden will be higher. However due to the high cost, it is perhaps unreasonable to assume that air would be used as a dominant method to transport spent batteries.
- Comparing the infrastructure of alkaline battery collection to other waste materials such as waste paint and other electronics to investigate the possibility of co-collection would be of interest. Economic implications of the collection scenarios could also be added.
- Another potential area of interest to experimental researchers would be to develop technologies that enable closed loop recycling of materials for alkaline batteries. In other words, enabling the use of manganese oxide or zinc recovered from spent batteries to be recycled back into primary batteries.
- The processes to tackle the challenging question of how to develop optimized end of life systems for alkaline batteries may be best identified through a multistakeholder processes drawing in participants from the entire battery recycling value chain. Since the initial publication of this report, such a process has begun involving the battery manufacturers, recyclers, government and nongovernmental organizations, retailers, collection program operators and others.

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Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

Average Annual Inputs (Production per million batteries***; Consumption in a representative facility)					
*Information drawn from representative facility that makes AAAA, AA, D, and 9Vs					
	<i>Input</i>	<i>Units</i>			
Electricity					
Fuel					
Natural Gas					
Fuel Oil					
Water					
Average Annual Outputs (Production per million batteries)					
	<i>Output</i>	<i>Units</i>			
Effluent Waste					
Waste Water					
Facility Waste					
for Recycling					
for disposal (solvents)					
for Landfill					
Air Emissions					
VOC					
***Please note what size batteries are considered in the per million batteries					

Average Annual Inputs (per one million battery units*** - not blister packs)					
	<i>Input</i>	<i>Units</i>			
Materials for packaging					
AAAA 4 Pack Blisters					
C/D/9V 2 Pack Blisters					
Electricity					
Fuel					
Natural Gas					
etc					
Water					
Average Annual Outputs (per one million battery units*** - not blister packs)					
	<i>Input</i>	<i>Units</i>			
Effluent Waste					
Waste Water					
Facility Waste					
for recycling					
for disposal (solvents)					
for landfill					
Air Emissions					
VOC					
***Please note what size batteries are considered in the per million batteries					

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

Average distance traveled from (for the five battery types):								
Manufacturing facility to Packaging facility:					AA			
					AAA			
					C			
					D			
					9V			
Packaging facility to Distribution center:					AA			
					AAA			
					C			
					D			
					9V			
Distribution center to Retail distribution center:					AA			
					AAA			
					C			
					D			
					9V			

Sales of alkaline batteries			
Annual Sales (# of battery units)	2006	2007	
AA			
AAA			
C			
D			
9V			

APPENDIX B: DETAIL AROUND LCI

Functional unit for study 1 kg batteries ~ 30 Weighted-average alkaline batteries.

Numbers below for single battery.

Materials

No.	Material	Ecoinvent 2.2 Database representation in SimaPro	Quantity w/scrap (g)	Quantity in battery (g)
1	Manganese dioxide	Manganese oxide (Mn ₂ O ₃), at plant /CN U used as proxy	13.4 (12)*	13
2	Potassium hydroxide (~35 wt% aq.)	Potassium hydroxide, at regional storage/RER U (35 wt%)	1.3	1.3
		Water, deionised, at plant/CH U (65 wt%)	2.4	2.4
3	Graphite	Graphite, at plant/RER U	1.2	1.2
4	Nickel-plated Steel	Steel, low-alloyed, at plant/RER U, (98 wt%)	6.9	5.9
		Nickel, 99.5%, at plant/GLO U (2 wt%)	0.14	0.12
5	Zinc	Zinc, primary, at regional storage/RER U (70%) Zinc from combined metal production (30%)	5.8	5.8
6	Brass	Zinc, primary, at regional storage/RER U, (35 wt%)	0.36	0.36
		Copper, at regional storage/RER U, (65 wt%)	0.67	0.67
7	Galvanized steel	Steel, low-alloyed, at plant/RER U, (95 wt%)	0.53	0.49
		Zinc, primary, at regional storage/RER U, (5 wt%)	0.028	0.026
8	Plastic	Polyvinylchloride, at regional storage/RER U	0.55	0.51
9	Paper	Kraft paper, unbleached, at plant/RER U	0.55	0.51
10	Nylon	Nylon 66, glass-filled, at plant/RER U	0.55	0.51
Road Transport		Transport, lorry, 16-32t, EURO3/RER U		
Ocean Transport		Transport, transoceanic freight ship/OCE U		

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

* Based on % manganese between MnO₂ and Mn₂O₃

Manufacturing

No.	Inventory	Quantity for battery	Unit
1	Water, deionised, at plant/CH U	32	g
2	Electricity, medium voltage, at grid/US U	0.02	kWh
3	Natural gas, burned in industrial furnace > 100kW/RER U	25	kJ
4	Light fuel oil, burned in industrial furnace 1 MW, non-modulating/RER U	9.3	kJ
5	VOC, volatile organic compounds; high pop.	0.02	g
6	Treatment, sewage, to wastewater treatment, class 3/CH U	32	g
7	Disposal, municipal solid waste, 22.9% water to municipal incineration/CH U	0.5	g
8	Recycling steel and iron/RER U (assuming cut-off approach, no burden or benefit associated)	1.1	g

Packaging materials

No.	Material	Ecoinvent 2.2 Database representation in SimaPro	Quantity (g)
1	Cardboard	Packaging, corrugated board, mixed fibre, single wall, at plant/RER U	1
2	Coreboard	Solid bleached board, at plant/RER U	0.8
3	PVC	Polyvinylchloride, at regional storage/RER U	0.4

Packaging manufacturing

No.	Inventory	Quantity for battery	Unit
1	Water, deionised, at plant/CH U	1.7	g
2	Electricity, medium voltage, at grid/US U	5.5	Wh
3	Natural gas, burned in industrial furnace > 100kW/RER U	6.1	kJ
4	Treatment, sewage, to wastewater treatment, class 3/CH U	1.7	g

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

End of life for batteries – Section 1

No.	Inventory	Quantity for battery	Unit
1	Transport, municipal waste collection, lorry 21t/CH U 33 g; 100 km	0.0033	tkm
2	Incineration/CH U	13	%
3	Landfill/CH U	87	%

Inventories used: Landfill	Waste type
Disposal, steel, 0% waster, to inert material landfill/CH U	Steel
Disposal, paper, 11.2% water to sanitary landfill/CH U	Paper
Disposal, polyvinylchloride, 0.2% water, to sanitary landfill/CH U	PVC
Disposal, municipal solid waste, 22.9% water, to sanitary landfill/CH U*	All waste types

Inventories used: Incineration	Waste type
Disposal, steel, 0% water, to municipal incineration/CH U	Steel
Disposal, paper, 11.2% water, to municipal incineration/CH U	Paper
Disposal, polyvinylchloride, 0.2% water, to municipal incineration/CH U	PVC
Disposal, municipal solid waste, 22.9% water, to municipal incineration/CH U	All waste types
Disposal, copper, 0% water, to municipal incineration/CH U	Copper

End of life for packaging

No.	Inventory	Quantity for battery	Unit
1	Transport, municipal waste collection, lorry 21t/CH U 33 g; 100 km	0.000154	tkm
2	Transport, lorry, 16-32t, EURO3/RER U	0.000264	tkm
5	Recycling paper/RER U (assuming cut-off approach, no burden or benefit associated)	30	%
4	Landfill/CH U (of remainder)	87	%

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

No.	Inventory	Quantity for battery	Unit
3	Incineration/CH U (of remainder)	13	%

Section 2: Inventories on recycling processes

Quantity for treatment of 1 kg alkaline batteries

Scenario A		Subcompartment	Quantity	Unit
Electricity, hydropower, at run-of-river power plant/RER U*		confidential	From company	
Hard coal mix, at regional storage/UCTE U		confidential		
Air emissions**				
Carbon dioxide		high pop	Calculated based on fuel usage	
Carbon monoxide		high pop		
Particulates, > 2.5 um, and < 10 um		high pop		
Nitrogen oxides		high pop		
Sulfur dioxide		high pop		
Mercury		high pop		
Dioxins		high pop	4.5E-12	kg
Cadmium		high pop	1.67E-7	kg
Copper		high pop	5.6E-7	kg
Nickel		high pop	7E-7	kg
Zinc		high pop	0.00002	kg
Water emissions				
Zinc	groundwater		3E-10	kg
Potassium	groundwater		0.05	kg
Waste				
Disposal, inert waste, 5% water, to inert material landfill/CH U			mass balance track with yield range	kg

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

* There is a hydro dam right at the facility as opposed to it coming from offsite, thereby there are not associated transmission losses.

** Other sources of information for the numbers associated with direct emissions were found in the literature

Zn Credit: Weighted mix between: Zinc (primary), at regional storage/RER U (70%), and Zinc from combined metal production, at refinery/SE U (30%)

Mn/Steel Credit: Blast furnace slag cement, at plant/CH U as an upper bound of what the material could receive credit for. This material is not an active substitute for cement, therefore the credit assigned could be as low as zero if we assume it is substituting for something like fly ash or cement kiln dust, potentially close to no credit.

Scenario B	Subcompartment	Quantity	Unit
Electricity, medium voltage, at grid/US U			
Natural gas, at long distance pipeline/RER U			
Hard coal coke, at plant/GLO U			
Transport, lorry 16/32t, EURO3/RER U			
Air emissions**			
Carbon dioxide	high pop	Calculated based on fuel usage	
Carbon monoxide	high pop		
Nitrogen oxides	high pop		
Particulates	high pop		
Sulfur dioxide	high pop		
Cadmium	high pop	3.7E-8	kg
Dioxins	high pop	4.5E-12	kg
Copper	high pop	2.3E-7	kg
Mercury	high pop	1E-9	kg
Nickel	high pop	7E-7	kg
Zinc	high pop	0.0000002	kg
Water emissions			
Zinc	groundwater	3E-10	kg
Potassium	groundwater	0.05	kg
Waste			

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

Disposal, inert waste, 5% water, to inert material landfill/CH U	mass balance track with yield range	kg
--	-------------------------------------	----

Zn Credit: Weighted mix between: Zinc (primary), at regional storage/RER U (70%), and Zinc from combined metal production, at refinery/SE U (30%)

Steel Credit: Weighted mix between: (60%) Steel, converter, un-alloyed, at plant/RER U, (40%) Steel, electric, un and low allowed at plant/RER U (40% weight of Mn to steel)

Mn Credit: Blast furnace slag cement, at plant/CH U as an upper bound of what the material could receive credit for. This material is not an active substitute for cement, therefore the credit assigned could be as low as zero if we assume it is substituting for something like fly ash or cement kiln dust. (50% weight of Mn to steel)

Non-fuel emissions based on EAF ecoinvent process as proxy

Scenario C	Subcompartment	Quantity	Unit
Electricity, medium voltage, at grid/Canadian average		From company	
Natural gas, at long distance pipeline/RER U			
Air emissions			
Carbon dioxide	high pop	Calculated based on fuel	
Carbon monoxide	high pop		
Particulates, > 2.5 um, and < 10 um	high pop		
Nitrogen oxides	high pop		
Sulfur dioxide	high pop		
Copper*	high pop	0.00000042	kg
Iron	high pop	0.00000048	kg
Nickel	high pop	0.00000016	kg
Zinc	high pop	0.00000013	kg
Waste			
Disposal, inert waste, 5% water, to inert material landfill/CH U	mass balance track with yield range		kg

* the remaining emissions are from the econivent shredding process as described in the text.

Steel Credit: Market mix: Steel, converter, un and low-alloyed, at plant/RER U, Steel, electric, un and low allowed at plant/RER U

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

Credit: Average between: Potassium chloride, as K₂O, at regional storehouse/RER U, Potassium nitrate, as K₂O, at regional storehouse/RER with U, Thomas meal, as P₂O₅, at regional storehouse/RER U, Phosphorous fertilizer, production mix, at plant/US, While this fertilizer replacement is not ideal as a micronutrient process from discussions with the company this has some similarities in production process. Because of the great uncertainty a credit of 0 to the average described above was used.

EAF scenario

Using Steel, electric, un and low alloyed at plant/RER U with US electricity grid.

Modify relevant emissions as described in text according to AP32 EPA: <http://www.epa.gov/ttnchie1/ap42>

Credit: Weighted mix between: Zinc (primary), at regional storage/RER U (70%), and Zinc from combined metal production, at refinery/SE U (30%)

Credit: Weighted mix between: (60%) Steel, converter, un and low-alloyed, at plant/RER U, (40%) Steel, electric, un and low allowed at plant/RER U; Ferromanganese, high-coal 75.4% Mn, at regional storage/RER U

Scenario E		Subcompartment	Quantity	Unit
Electricity, medium voltage, at grid/CH U			1.7	kWh
Light fuel oil, at refinery/CH U			0.058	kg
Propane/butane, at refinery/CH U			0.006	kg
Air emissions				
Carbon dioxide	high pop		0.17	kg
Carbon monoxide	high pop		5.2E-4	kg
Nitrogen oxides	high pop		4E-4	kg
Particulates	high pop		9.36E-5	kg
Sulfur dioxide	high pop		4.8E-5	kg
Cadmium	high pop		6E-9	kg
Mercury	high pop		1E-9	kg
Zinc	high pop		2E-7	kg
Water emissions				
Zinc	groundwater		3E-10	kg

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

Potassium	groundwater	0.05	kg
Waste			
Disposal, inert waste, 5% water, to inert material landfill/CH U		mass balance track with yield range	kg

Credit: Weighted mix between: Zinc (primary), at regional storage/RER U (70%), and Zinc from combined metal production, at refinery/SE U (30%) Credit: Ferromanganese, high-coal 75.4% Mn, at regional storage/RER U. Due to lack of data only primary ferromanganese was used rather than a mix as above.

Landfill

Inventories used	Waste type
Disposal, steel, 0% water, to inert material landfill/CH U	Steel
Disposal, paper, 11.2% water to sanitary landfill/CH U	Paper
Disposal, polyvinylchloride, 0.2% water, to sanitary landfill/CH U	PVC
Disposal, municipal solid waste, 22.9% water, to sanitary landfill/CH U*	All waste types
Process-specific burdens, residual material landfill/CH U	All waste types
Process-specific burdens, sanitary landfill/CH U	All waste types
Process-specific burdens, inert material landfill/CH U	All waste types

* The emissions associated with this inventory were modified in the “leach” scenario for the following substances related to potential battery impacts within landfills: zinc, potassium, nickel, and copper as described in the text.

Transport used:

Passenger vehicle: Transport, passenger car, petrol, EURO5/CH U

Trucks: Delivery truck: Transport, lorry 7.5-16t, EURO3/RER U ecoinvent load factor

Truck: Transport, lorry 16-32t, EURO3/RER U ecoinvent load factor

MSW: Transport, municipal waste collection, lorry 21t/CH U

Ocean transport: Transport, transoceanic freight ship/OCE U

Uncertainty ranges and assumptions

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

Item	Form of distribution	Characteristics
Allocation	Triangle	Min: 0% Most likely: 5% Max 10%
Leg 1 Mileage	Triangle	Min: 2 Most likely: 8 Max 16
Leg 2 Mileage	Triangle	Min: 120 Most likely: 220 Max: 400
LCI transportation	Lognormal	DQI based uncertainty altered geographic distribution from that found in ecoinvent because using inventory for US study
LCI zinc&steel	Lognormal	DQI based uncertainty altered geographic distribution from that found in ecoinvent because using inventory for US study
LCI slag credit	Uniform	0 – blast furnace slag
LCI micronutrient credit	Uniform	0 – average of fertilizers mentioned in LCI
Recovery	Uniform	range as indicated

APPENDIX C: NUMERIC RESULTS OF CHAPTER 6

CED (MJ)	Impact			
	Retail drop-off	std	Municipal drop-off	std
MSW 1	1.9			1.1
Scenario A	4.7	1.6	7.9	2.9
Scenario B	7.5	2.4	10.7	3.5
Scenario C	5.0	2.2	9.6	3.2
Scenario D	-1.4	1.8	3.2	3.0
Scenario E	5.0	2.5	9.6	3.4

GWP (kg CO ₂ eq)	Impact			
	Retail drop-off	std	Municipal drop-off	std
MSW 1	0.11			0.08
MSW 2	0.29			0.05
Scenario A	0.65	0.12	0.9	0.2
Scenario B	0.62	0.14	0.85	0.2
Scenario C	0.35	0.12	0.6	0.2
Scenario D	-0.09	0.10	0.2	0.1
Scenario E	-0.11	0.10	0.1	0.1

Ecosystem Quality (pdf*m2yr)	Impact			
	Retail drop-off		Municipal drop-off	std
MSW 1	0.005			0.004
MSW Leach	0.17			0.004
Scenario A	-0.67	0.39	-0.65	0.41
Scenario B	-0.87	0.41	-0.85	0.43
Scenario C	-0.01	0.01	0.00	0.02
Scenario D	-0.76	0.38	-0.75	0.41
Scenario E	-0.88	0.46	-0.86	0.45

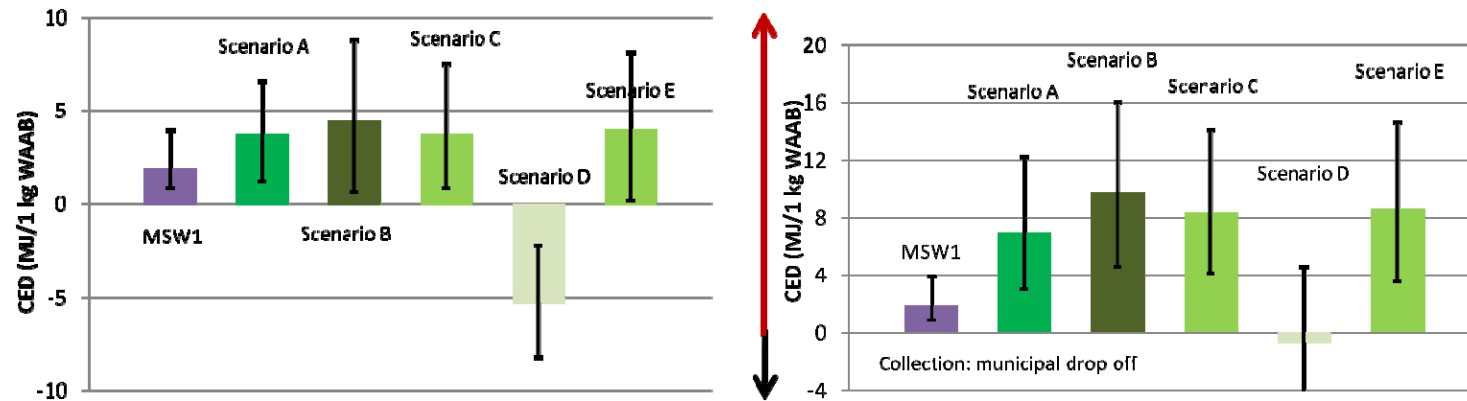
Human health (DALY)	Impact			
	Retail drop-off	std	Municipal drop-off	std
MSW 1	8.52E-07			9.36E-08
Scenario A	1.9E-06	1.4E-06	2.05E-06	1.35E-06
Scenario B	-1.8E-06	1.4E-06	-1.7E-06	1.36E-06
Scenario C	4.2E-08	1.8E-07	1.77E-07	2.14E-07
Scenario D	-2.0E-06	1.3E-06	-1.8E-06	1.27E-06
Scenario E	-5.5E-06	1.5E-06	-5.4E-06	1.51E-06

Resources (MJ Surplus)	Impact				
		Retail drop-off	std	Municipal drop-off	std
MSW 1	0.13				1.1
Scenario A		0.51	0.13	0.76	0.22
Scenario B		0.60	0.20	0.86	0.27
Scenario C		0.39	0.19	0.64	0.26
Scenario D		-0.26	0.12	-0.01	0.22
Scenario E		-0.02	0.14	0.23	0.23

APPENDIX D: FIGURES FOR 100% PRIMARY OFFSET FOR ZINC AND STEEL.

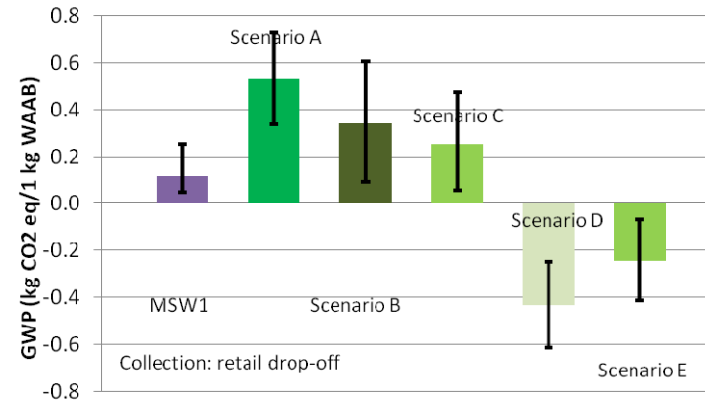
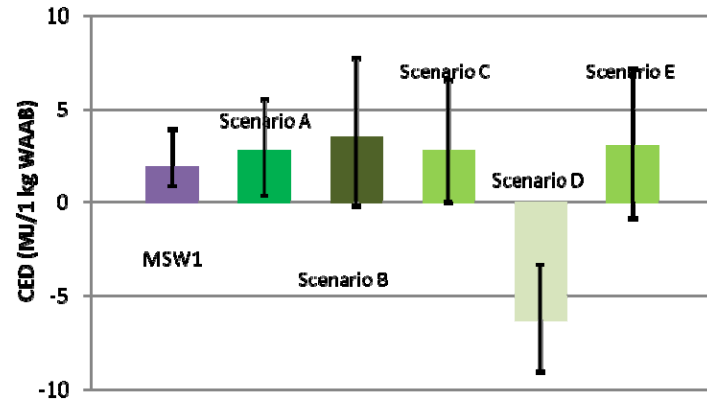
Each of the figures below corresponds to a figure in the section 2 text above but with a 100% primary offset for the zinc and steel as described in the text, for a longer term perspective. For the analysis presented in the text the “market mix” assumption was used consisting of 40% EAF steel and 30% combined production zinc.

The figures below corresponds to Figure 20 & 21 in the text

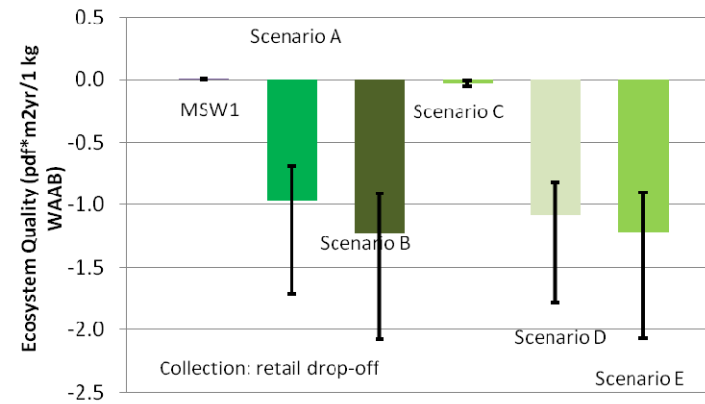
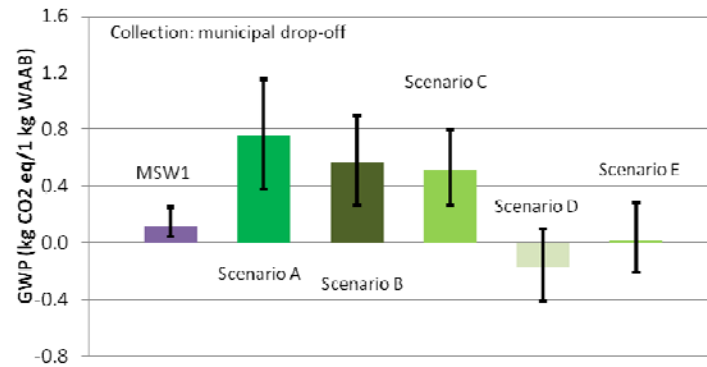


The figures below corresponds to Figure 23& 24 in the text

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

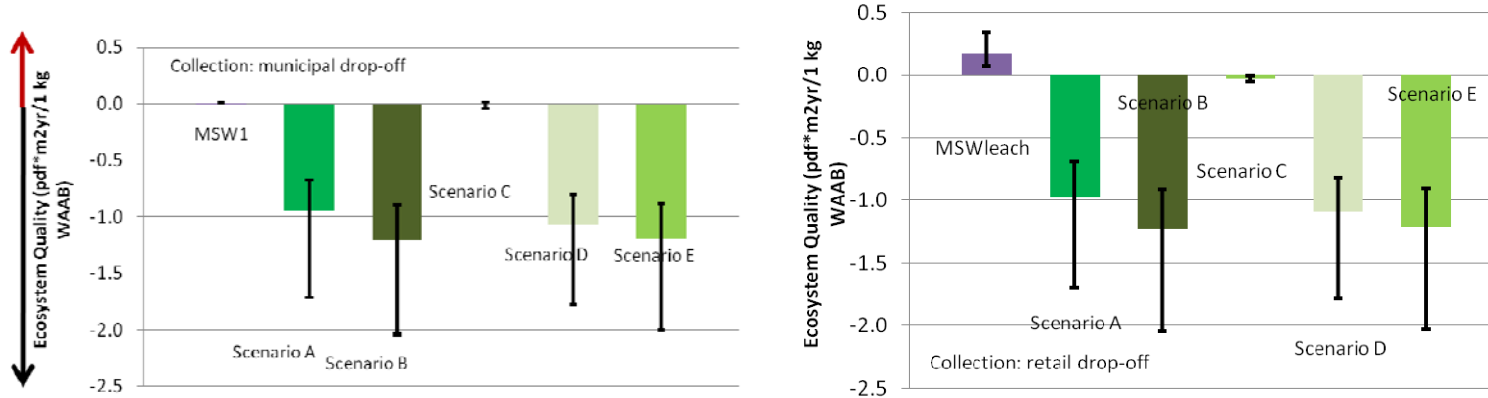


The figures below corresponds to Figure 25& 26 in the text

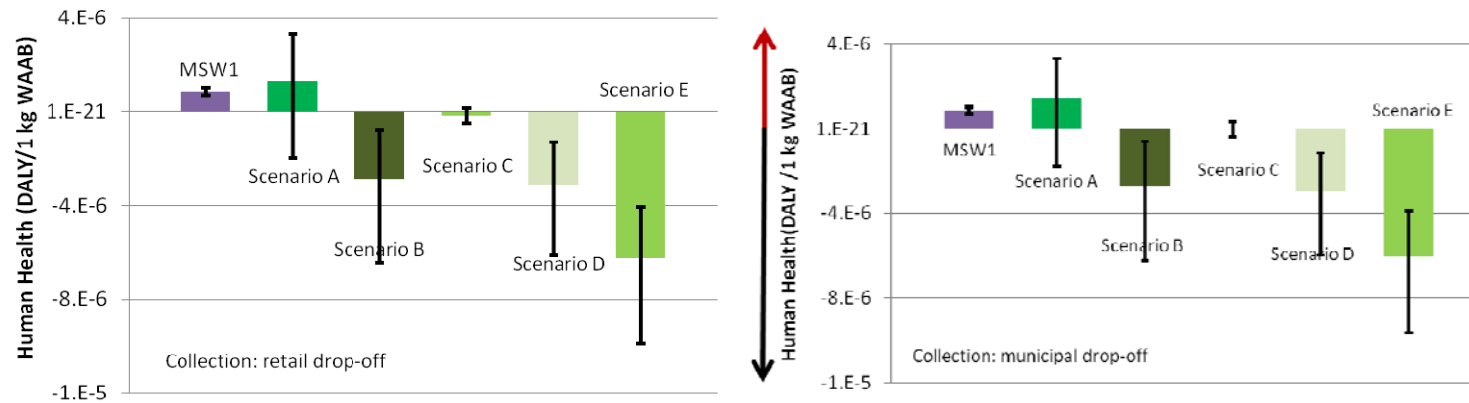


The figures below corresponds to Figure 27& 28 in the text

Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

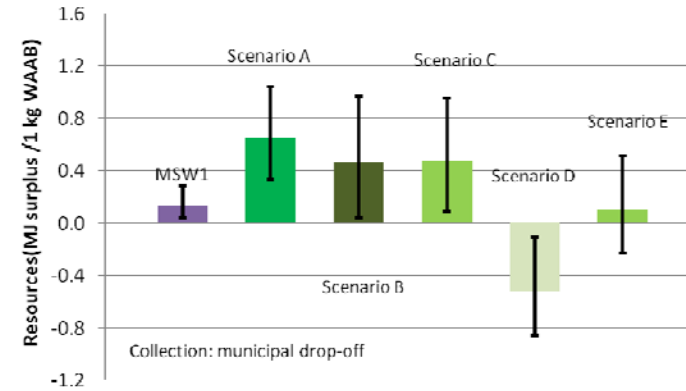
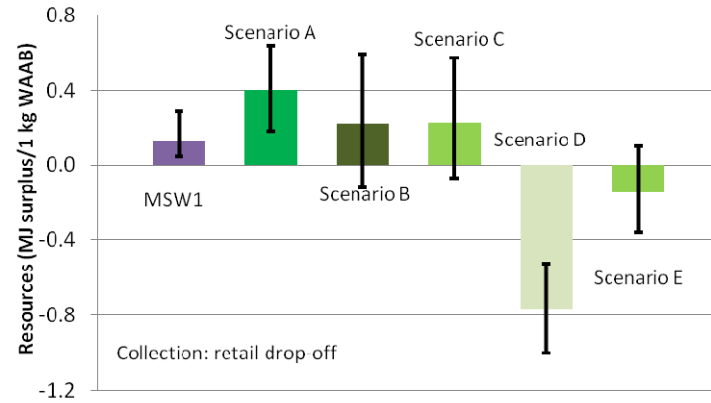


The figures below corresponds to Figure 29& 30 in the text

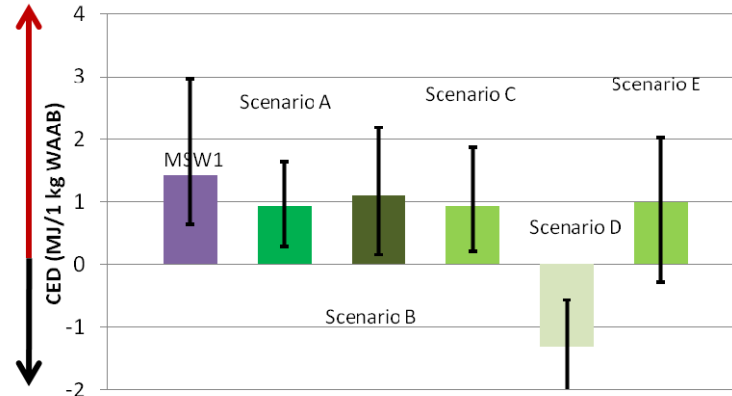
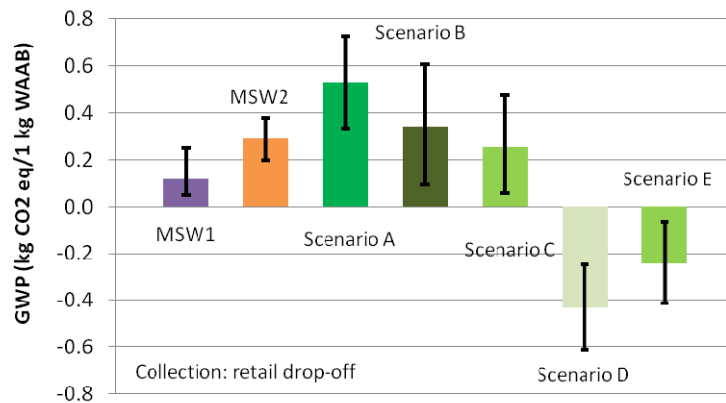


Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

The figures below corresponds to Figure 31& 32 in the text

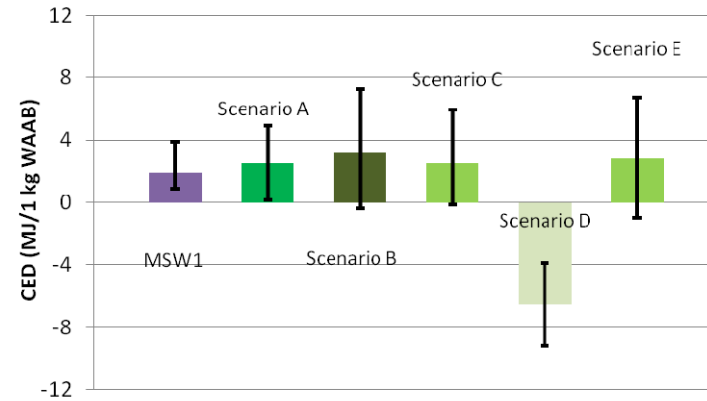
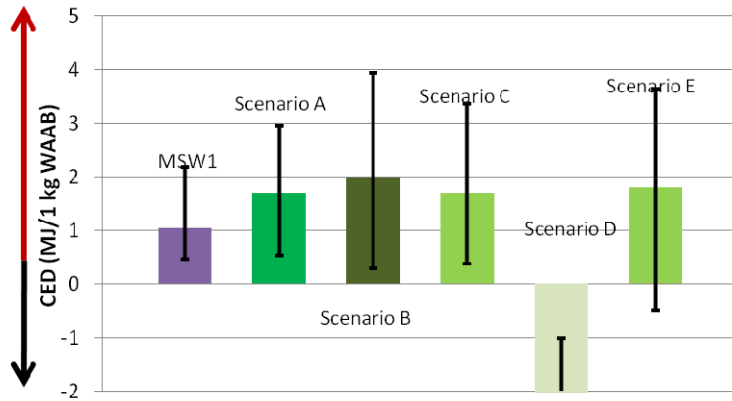


The figures below corresponds to Figure 33& 34 in the text

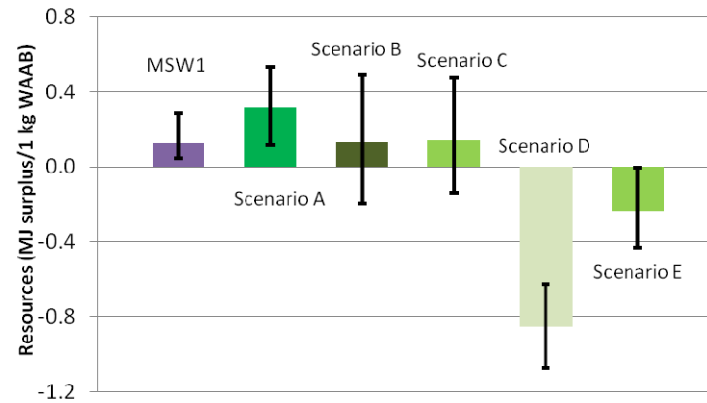
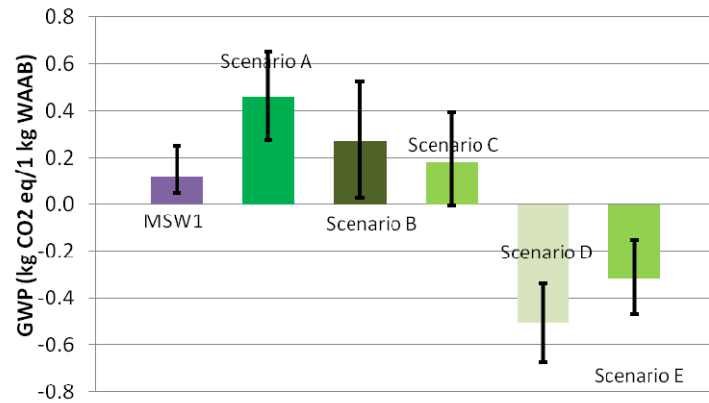


Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

The figures below corresponds to Figure 35& 36 in the text

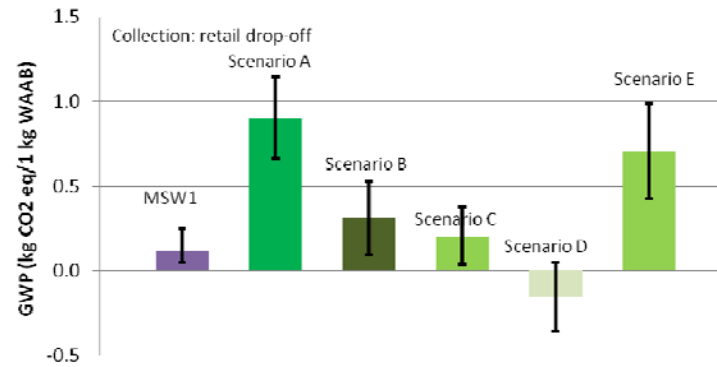
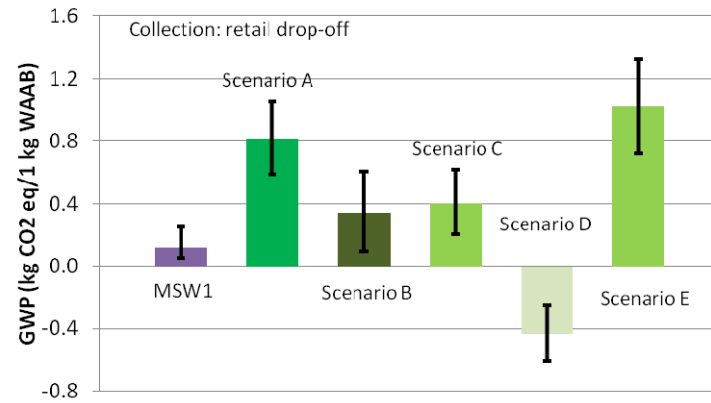


The figures below corresponds to Figure 37& 38 in the text

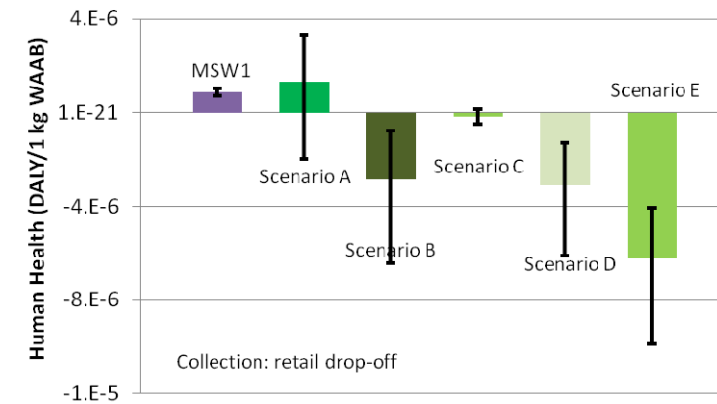
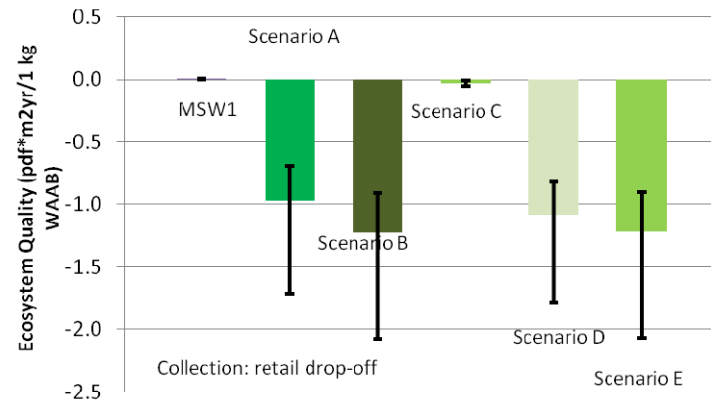


Life cycle assessment of alkaline batteries with focus on end-of-life disposal scenarios

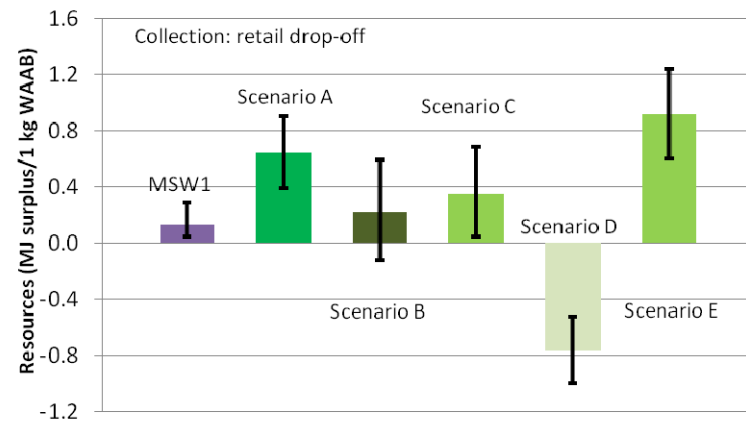
The figures below corresponds to Figure 41& 42 in the text



The figures below corresponds to Figure 43& 44 in the text



The figures below corresponds to Figure 45 in the text



APPENDIX E: EXTERNAL REVIEWS

This appendix provides three external reviews performed for this study and outlined in red are some responses by the author. For reviewer 2 and 3 several drafts were iterated on between the authors and the reviewers.

REVIEWER 1

Review of 'Draft – Life cycle Impact of Alkaline Batteries with a Focus on End-Of-Life.' Draft release: June 2010

Francis McMichael, Professor Emeritus
Carnegie Mellon University
Reviewer #1

Overview

The report is of high quality in terms of documentation, modeling, and inclusion of important references. It is detailed with numerical results and calculations. The report carefully defines many assumptions for its analysis. It may help if these assumptions could be organized on a chapter-by-chapter basis and presented in an appendix for technical readers.

I think that the style and level of technical detail in the draft is more suitable to a technical report than an LCA. Style depends on the background of the intended audience. I think the draft will be better LCA with less technical detail in the main body. If a manuscript is prepared for a refereed technical journal, the more technical detail is essential.

“The intended audience of the study is NEMA, the California EPA and eventual publication (p.12, Chapter 3).”

It is likely that NEMA and California EPA will use this document for public policy discussions. To this end, where sides may differ, this analysis in open peer reviewed literature will ensure that the key findings are put in context to show your work is clearly represented.

The authors considered moving some sections of the report to an appendix however it was concluded that this report would contain the full details in the main body and smaller reports will be created from this larger document.

Methodology and Goal/Scope

Is the methodology consistent with ISO 14040/14044?

The study is consistent with the ISO LCA standard. It addresses issues from cradle-to-grave for alkaline batteries and appropriately emphasizes the production and end-of-life phases in detail.

Are the objectives, scope and boundaries of the study clearly stated?

Yes. More consideration should be given to illustrate the objectives of the audiences for the report. The emphasis of the draft is alkaline battery production and end-of-life disposal of spent batteries.

More detail was added to clarify the goal of the study. This can be found on pages 13 and 14 as well as in the executive summary and conclusions.

Reasons for carrying out the study and audience of the study.

Three audiences are identified (NEMA, local and regional government as well as general publication).

The website of the client NEMA (National Electric Manufacturers Association) shows its interests in many areas including environmental policy. It is a source of both general information and explanations of how battery manufacturers have cooperated and achieved compliance with regulations. At present, there is no battery LCA listed on the website. With some reformatting, the NEMA website may be appropriate for the LCA.

The NEMA website says that all alkaline batteries do not have the same composition. Button size alkaline batteries contain mercury that has been deliberately added for performance. The larger sizes of alkaline batteries are labeled 'no added mercury.' There is a difference between 'no added mercury' and 'contains no mercury.' The paper by Almeida et al, (2009) in the draft list of references clarifies this point from its study of incineration of AA alkaline batteries.

Sentences were added to the report describing the "no added mercury" situation in more detail. These can be found on page 7 and 45/46 of the document.

The function and system boundaries of the product system.

There is no discussion of the function (and benefits) of alkaline batteries. The benefits are partially identified by the number of worldwide sales of alkaline batteries and the perceived value of alkaline batteries reflected in the higher cost of kilowatt-hours of energy the public is willing to pay in contrast from kWh from a wall outlet.

The functional unit.

The draft identifies two functional units. The main functional unit is a single hypothetical battery whose size is based on a 'sales weighted mass for AAA, AA, C, D, and the 9-volt batteries.' The second functional unit is 1 kilogram of the sales weighted alkaline battery (SWAB). The one-kilogram functional unit is equivalent to about 30 batteries with a SWAB mass of 33 grams.

Since AA sales make up about 60 percent of the alkaline batteries sold, it may be appropriate to choose the AA battery as the first functional unit. This would require less explanation for a lay audience. Some audiences are concerned with larger numbers or masses of alkaline batteries. Manufacturers, transporters of product and waste products, and designers of containment and disposal plants deal with more than a single battery. A kilogram may be too small to illustrate LCA issues for these groups. Tonnes may be more appropriate for manufacturers, transporters and end-of-life handlers.

Global annual production of alkaline batteries exceeds 10 billion units. One million batteries is about one hour of global production. I recommend a larger unit than one kilogram for the second functional unit.

The authors considered altering the functional unit used, however, upon extensive discussions with the study commissioners the functional unit was left as is but further clarification for the choice was made on page 14.

The data quality requirements.

Attention is given in Chapter 7 to sensitivity analysis. I think more attention to uncertainty in data quality and/or data variability should be given. This is partially addressed when the draft discusses the selection of comparative production processes in SimaPro software and Ecocontent databases. Some graphs (figures) include ranges and forms of stem –and-leaf diagrams. In many cases, more clarification could be added to figure captions.

The impact categories need more explanation. The use of energy units like megajoules (MJ) and carbon dioxide equivalents (CO₂ equiv) are familiar to technical audiences. Ecosystem and Health Impact units like DALY and PDF/m²/year are less familiar to technical and lay audiences. The uncertainty in assigning and how to do arithmetic with these units needs more explanation and discussion. Reporting these units in several significant figures often is unjustified. Advice on what numbers are large and significant merits attention; and what numbers are small and difficult to interpret is always helpful to most audiences. Justification for the selection ecosystem impacts is needed. If the audience is familiar with impacts, it may help to explain why choice is limited.

The number of reported significant figures was reduced to 2 throughout the report. Data quality tables were added. Further explanation of Ecosystem quality and human health were also added.

Study Assumptions and Conclusion Comments.

The report is well written and the majority of the assumptions are thoughtful and reasonable. It is state-of-the-art.

The assumptions and uncertainty in the SimaPro analysis and Ecocontent databases are not as clearly presented. Uncertainty can limit severely our confidence in the apparent precision from a rigorous analysis. Some assumptions are heavily value laden.

Summaries and abbreviations of LCAs are often made available to the general public. I think that your client should provide public access to the full text of the alkaline battery LCA.

REVIEWER 2

Bob Boughton (see bio below)

1 Introduction

The authors of the study aim to comply with the international standards ISO 14040 series. To meet this standard, critical review according to ISO 14040 needs to be performed by an external LCA-expert.

The review process started in October, 2010 following receipt of an initial draft report. All issues concerning goal and scope of the LCA study as well as methodological and data related issues were discussed between the practitioner and the reviewer at a face to face meeting in November. Review of a revised draft took place in December with the aim to address details and, in particular, to fully support the conclusions of the study in terms of the goal of the study.

Overall, the practitioners were responsive to every concern and comment made during the reviews. The review process was interactive, which lead to clarity, consistency and high quality assurance. The

statements made in this critical review report represent the key concerns that were addressed and any residual considerations for the final report received January 12, 2011.

2 Review criteria

The critical review was performed according to the ISO standard 14040, taking into account the ISO standards 14041 to 14043. The review criteria laid down in ISO 14040 section 7 requires that the critical review process shall ensure that:

- the methods used to carry out the LCA are consistent with the International Standard;
- the methods used to carry out the LCA are scientifically and technically valid;
- the data used are appropriate and reasonable in relation to the goal of the study;
- the interpretations reflect the limitations identified and the goal of the study;
- the study report is transparent and consistent.

The review included checking whether all requirements and constraints, especially those from ISO 14041 and 14042, were met and whether the methods used were consistent and state of the art. The data used were checked for appropriateness and those with a particularly high influence on the result were scrutinized. For processes that seemed to be both relevant and uncertain, sensitivity analyses were proposed.

3 Results of the Critical Review

3.1 General Comments

The LCA study deals in many respects with novel technologies or systems that are available in one location outside the US. In the course of the review, several practical, methodological, and data-related issues emerged. Through a conjoint effort of the practitioners conducting the study and the reviewer, these issues were resolved in a satisfying manner. The report contains not only all formal elements required by ISO 14040, but provides information concerning assumptions, constraints, calculations, and results in a transparent and comprehensible way. There is concern about reproducibility due to undisclosed client private data as well as some proxy data used for battery manufacture. Hence, while the report is not completely transparent for the attributional assessment of the total battery life cycle impacts, this does not limit the conclusions relevant to the end of life treatment option comparison.

The study is titled “Life Cycle Impacts of Alkaline Batteries with a Focus on End of Life”. A reader may infer that the study compares different end of life management techniques, and that it perhaps comes to the conclusion that one *method* is environmentally advantageous compared to the others. However, this report outlines the comparison of different methods in specific locations as options for waste battery management. Hence, the results reported should not be considered representative of those specific techniques in other locations for comparative purposes (this caveat is noted several times in the report).

It should be pointed out that, as a prospective study it is clear which options are favored considering the collection and transportation assumptions made. Further study would need to be done to compare each management technique on a level playing field in order to compare or identify a favored technology. Some assumptions regarding battery management and recycling are of prospective nature and cannot be validated at present. The resulting uncertainty has been taken into account by several sensitivity analyses or acknowledged in the report.

3.2 Method

The ISO standard 14040 requires a four-step procedure for LCA studies: goal and scope, inventory, impact assessment, interpretation. All four steps were carried out in this study. Where necessary, methodological explanations are given (e.g., impact categories, allocation procedure). In the first step “goal and scope definition” the definition of the systems and the system boundaries were carried out as well as assumptions and methodological determinations. The second step “life cycle inventory” (LCI) basically comprises the data gathering, setting up of the network model and the calculation of the inventory. In the course of the critical review the appropriateness of the data used was discussed and procedures in regards to data uncertainties were considered.

The system model was checked and no mistakes were found. The overall boundary was plausibility checked concerning mass balance. A thorough review of the data sets and inventory calculations was not feasible. However, the inventories used (primarily Ecoinvent) have proved suitable in previous LCAs. Finally, the results of the LCI were plausibility-checked and no inconsistencies were found. The method of the third step “life cycle impact assessment” (LCIA) and the impact categories chosen are described in the study. The characterization factors used are documented. Both the method and values are in line with the state of the art. The choice of impacts reported and those not used could be better described.

The aspects of the last step of an LCA, “life cycle interpretation”, are handled appropriately and the report content meets the requirements of ISO 14043 with regard to interpretation (the ISO standard demands for conclusions, limitations, and recommendations are stated in the study). In conclusion, it can be stated overall that the methods used are consistent with the international standards and represent the state of the art.

3.3 Assumptions

Since the locations of four of the end of life management methods under study differ considerably, a direct comparison of the methods would have been inappropriate. Hence, specific scenarios for the systems had to be defined, with related transportation needs and specific inputs like local electrical grids. Hence, the results reflect both the technologies and local geographical conditions. In conclusion, this LCA study is, not a comparison between four defined, specific end of life management systems.

Considering the fact that waste alkaline battery recycling is not commonplace, and that any converting/recovery process is not technically mature yet, it should be assumed that, any recovered materials are returned to commerce and not stored in stock. Additionally it is assumed that no market affects occur from the volume of materials recovered. This assumption may have only a minor influence on the results; however, it should be taken into consideration when interpreting the results. This point has not been adequately reflected in the summary of the report.

3.4 Data

The results show that the most significant life-cycle contributions stem from manufacturing. Unfortunately, the datasets for battery manufacture are aggregated and were not reviewable, and proxy information was used for the major component. In general, data were not scrutinized in detail but appear to be appropriate and reasonable in relation to the goals of the study. It was assumed that the methodological backgrounds of the datasets used do not differ substantially. Further discussion of the applicability of Ecoinvent inventories to the US situation could have been made. Even though it is disappointing that the most important data sets for battery manufacturing systems under study do not allow a detailed review, the conclusions drawn in the report concerning the overall battery life cycle

impacts are supported. The data quality assessment could be improved to fully meet ISO 14044:206 section 4.2.3.6 data quality indicator requirements.

3.5 Interpretation

The interpretation portion of the report deals with the uncertainties of data and assumptions and explains the limitations of results and conclusions. All relevant uncertainties appear to be adequately addressed. It can be stated that the choice of the sensitivity scenarios as well as the conclusions drawn are appropriate and sufficient.

It is the task of the interpretation phase of an LCA to combine the manifold single results of the different scenarios reflecting the uncertainties and limitations identified and to derive conclusions and recommendations from these facts. All relevant aspects are sufficiently considered. It can be stated that the authors of the report succeeded in the difficult task to merge the different findings of the study to only impartial and unbiased conclusions and to derive recommendations to decision makers as well as to the commissioner and the stakeholder community. In general, the interpretations reflect the limitations identified and the goals of the study.

3.6 Report

The requirements of the standard for LCA reporting are stipulated in ISO 14040 and ISO 14041. This report was examined with respect to these requirements. All mandatory elements are well documented in a transparent, consistent, and comprehensible way, or otherwise described above. The report is well-balanced and illuminates both advantages and disadvantages of the observed systems.

4 Review Summary and recommendations

The observed LCA study can be said to be complete, in compliance with the standards, and methodologically proper. The study deals in many respects with novel technologies and markets. In the course of responding to review comments, several practical, methodological, and data-related issues were appropriately addressed. Consequently, the general impression of the LCA study is excellent. It contains all formal elements required in ISO 14040 and provides all information concerning assumptions, constraints, calculations, and results in a transparent and comprehensible way.

In subsequent reports which expand on the topics here, the reviewer sees the following points for further development:

- Expanding the current study to assess each management technique on an equivalent comparative basis,
- A provision of disaggregated datasets for battery manufacture, which would foster transparency and reproducibility of the study
- Refinement of key data sets that lead to the highest uncertainty or range in outcomes.

Short information about the critical reviewer

Bob Boughton

Bob has over 25 years of professional experience in environmental engineering and is currently employed by the State of California EPA as a Senior Engineer where, during the past 10 years, he has been a project leader in the area of LCA and LCA-based end of life assessment. He serves on the advisory council for the American Center for Life Cycle Assessment, and is a Certified Life Cycle Assessment Professional. He is an author of several publications applying Life Cycle Assessment and conducts

journal, peer and ISO reviews. He received a MS in Chemical Engineering from the University of California Santa Barbara.

REVIEWER 3

Critical review report according to ISO 14'040 and 14'044

Author: Hans-Joerg Althaus, Empa

Date: 2011.02.21

Document: DRAFT – Life Cycle Impacts of Alkaline Batteries with a focus on end-of-life
A study conducted for the National Electrical Manufacturers Association
Draft Release to internal group: June 2010
External review: Dec 2010
Elsa Olivetti & Jeremy Gregory, MIT

Background and Assignment

Elsa Olivetti and Jeremy Gregory (MIT) did an LCA study on Battery production and end of life (EOL) for the National Electrical Manufacturers Association, for which they commissioned a critical review with Hans-Joerg Althaus (Empa, Switzerland). The study was under review by two other experts before but the review to be performed by Hans-Joerg Althaus was to be independent of them.

The goal of the critical review is to ensure that:

- The LCA was conducted in accordance to the ISO standards on LCA (ISO 14'040 / 44 2006)
- The methods used in the LCA study are scientifically sound
- The data used are appropriate to the goal of the study and of sufficient quality.
- The discussion reflects the constraints resulting from the goal and scope definition and from the data used.
- The report is consistent and transparent

Procedure

The document was iteratively changed according to comments made in the review process. During this process, the reviewer was given access to all confidential data which are not documented in the final version of the report.

Content of the study

The study consists of two sections: Section 1 assesses the life cycle of alkaline batteries, mainly to provide a context for Section 2, which aims at analysing various (existing and hypothetical) EOL options for spent alkaline batteries.

ISO conformity and scientific soundness of methods applied

Section 1:

The goal and scope definition is short but clear. Parts of what, according to ISO, belongs to the scope definition is documented under different headings in specific chapters. This makes sense since they belong to both sections of the study and since redundancy can be avoided by placing them in a separate chapter. Other elements of the scope definition according to ISO are not explicitly stated (e.g. interpretation to be used, value choices, data quality requirements, type of critical review). The definitions nevertheless are sufficient to understand the study and its results.

A section was added to the goal and scope section describing the interpretation, data quality concerns and the type of critical review for both section 1 and 2.

Section 2:

What was said to the goal and scope definition of section 1 applies as well for section 2. The way how allocation problems are modelled for EOL recycling is different in Section 2 compared to section 1. However, since no recycling of the main products is considered in section 1, this difference leads to only slightly higher burdens for the production phase in section 1 and in the overall context is acceptable.

Applicability and consistency of LCI data and models

Section 1:

Material and energy demand and direct emissions and waste flows for battery production are collected in a survey with several producers. The flows are correctly linked to ecoinvent datasets which is used as background database. The same applies for transports and packaging of the batteries. The EOL model in this section is rather crude. Generic ecoinvent data is used for landfill and incineration. All the metals seem to be represented by a proxy for steel disposal. Thus, potentially harmful emissions of zinc are not considered. The data is consistent and its quality suits the purpose of section 1.

Section 2:

A material inventory of spent batteries was established based on production data. Collection of spent batteries is modelled very carefully and uncertainties are taken into account. The various EOL options

are modelled based on partly confidential information from the operators. Primary data is available for the energy demand of the processes but the process emissions are mainly based on calculations from fuel consumed and on estimates. The material yield of the existing recycling processes are estimated by the process operators and therefore of the best achievable quality. Recycled metals are assumed to substitute for a mix of primary and secondary metals (according to market shares). This makes sense in a short-term perspective. In consequential LCA, one usually has a long-term perspective and thus, for the system under consideration, would expect a substitution of primary materials. A scenario considering this substitution is calculated. Some explanation which scenario to use for answering which questions would be an asset.

A short explanation was added to the text about the longer term perspective.

Correctness and consistency of results

Section 1:

Results are comprehensively presented and seem plausible. Checking the correctness of all results was out of scope.

Section 2:

Results are comprehensively presented and seem plausible. Checking the correctness of all results was out of scope. The transparency of the results is somewhat hampered by the restrictions from data confidentiality but the discussion tries to compensate for that.

Consistency of conclusions and recommendations with goal and scope and results of the LCA

Conclusions are in line with the results presented. Also the recommendations are consistent with the results but they mainly focus on reducing the overall burdens of the battery life cycle. Since this was not one of the goals defined for the study, the recommendations are rather vague and demand for future work.

Elements were added to the future work section to add specificity around the specific needs around end of life.

Conclusion:

The report is well structured and gives a good introduction into the topic. The study is set up in a reasonable way to answer the key questions and the presentation of data, assumptions and results is generally good but, due to confidentiality, not always sufficient for reproducing the results. Even though the LCA does not fulfil all requirements of ISO 14'040 / 14'044 (e.g. parts of scope definition and some reporting requirements), it is carefully made and the main requirements are fulfilled.