The Significance of Environmental Attributes as Indicators of the Life Cycle Environmental Impacts of Packaging and Food Service Ware

Final Report

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State of Oregon
Department of Environmental Quality

The Significance of Environmental Attributes as Indicators of the Life Cycle Environmental Impacts of Packaging and Food Service Ware

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Glossary of Terms

Acidification Potential	The acidifying potential resulting from acid rain caused by inorganic air emissions.
Avoided Burden Allocation (0/100) Method	End-of-life allocation method that assigns the benefits of material recycling to products recycled at end-of-life and does not give credits for use of recycled content.
Biobased	Products derived from plants and other renewable agricultural, marine, and forestry materials (USDA 2018).
Board	Heavy, plant fiber-based, rigid material.
Chemical recycling	The process of reacting plastics to break them down into monomers or other basic chemicals which can be repolymerized into new plastics or used for other purposes.
Comparative Toxic Unit	Estimate of the potentially affected fraction of species integrated over time and volume (Fantke 2017).
Compostable	Materials that degrade by biological processes to yield carbon dioxide, water, inorganic compounds, and biomass at a rate consistent with biodegradation of natural waste while leaving no visually distinguishable remnants or unacceptable levels of toxic residues (ASTM International 2012).
Corrugated Cardboard	Material consisting of a fluted corrugated sheet and one or two flat linerboards.
Cradle-to-cradle	Term used in life cycle assessment to specify a life cycle system boundary where the last step in the life cycle is a recycling process.
Cradle-to-gate	Term used in life cycle assessment to specify the system boundary of a partial life cycle assessment including processes from resource extraction (cradle) to the factory gate, before the product is delivered to a consumer.
Cradle-to-grave	Term used in life cycle assessment to specify a life cycle system boundary where the last step in the life cycle is disposal.
Downcycling	The recycling of waste where the recycled material is of lower quality and has a different function than the original material.

Ecotoxicity	Ecosystem toxicity; the potential of chemicals to cause toxic effects on organisms, often measured in terms of effects on aquatic species. Effects are measured in terms of equivalency to the toxic effects of a benchmark environmental pollutant or in terms of comparative toxic units (CTU_e) when calculated using the USEtox LCIA method.
Endpoint impact	An effect on human health, ecosystem quality, or resource availability directly experienced by a person or organism. Examples of endpoint impacts commonly measured in LCA studies are human health effects measured in disability adjusted life years, ecosystem quality effects measured in partially disappeared species over an area-year, or economic effects measured as the increased cost associated with extracting scarce resources.
Eutrophication Potential	The potential of nutrient releases to cause harmful acceleration of biological productivity in an ecosystem. Calculated at the midpoint level in terms of benchmark nutrient pollutants such as nitrogen or phosphate.
Fossil Energy Depletion	Use of scarce fossil fuel resources measured in megajoules or kilograms oil equivalents.
Global Warming Potential	The heat trapping capacity, or radiative forcing potential, of greenhouse gases and precursors. Calculated in terms of kilograms carbon dioxide equivalents. Abbreviated as GWP in this report.
Particulate Matter Formation Potential	Potential to form particulate matter leading to human respiratory effects.
Human Health, Toxicity	Increase in morbidity in the human population due to exposure to carcinogenic substances or substances resulting in cancer and non-cancer diseases measured in common toxic units.
Impact category	Class representing environmental issues of concern to which life cycle inventory analysis results may be assigned (ISO 2006). Examples include global warming potential, acidification potential, eutrophication potential, human toxicity potential, etc.
Land Use	Area of land occupied over time measured in units of area or equivalency to land area of a certain type determined in terms of biological productivity.
Life cycle assessment	Compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle (ISO 2006). Abbreviated to LCA in this report.

Life cycle inventory	Phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle (ISO 2006). Abbreviated to LCI in this report.
Life cycle impact assessment	Phase of life cycle assessment aimed at understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system throughout the life cycle of the product (ISO 2006). Abbreviated to LCIA in this report.
Midpoint impact	An indicator of an environmental effect at a level of the cause-effect chain between pollutant releases or resource use and the endpoint (Hiederer et al. 2011). Midpoint impacts are often used as the metric for the results of LCA studies because they are suited for comparison between options or scenarios while avoiding much of the uncertainty/variability associated with endpoint metrics. Global warming potential, smog formation potential, acidification potential, and human toxicity potential are all examples of midpoint impacts.
Mineral Depletion Potential	Depletion of minerals measured in terms of relative natural availability compared to iron in ore.
Ozone Depletion Potential	Capacity of substances to deplete the stratospheric ozone, measured for example in kilograms chlorofluorocarbon-11 equivalents.
Recyclable	The potential for a material to be recovered from the solid waste stream to be made into a new product at the end of a product's useful life.
Recycled Content	The portion of materials used in a product that have been diverted from the solid waste stream.
Recycled Content Allocation (100/0) Method	End-of-life allocation method that allocates the entire burden of primary material production to the first product while the burdens of material collection and processing for recycling are attributed to the subsequent product.
Shared Burden Allocation (50/50) Method	End-of-life allocation method that assigns virgin production, recycling, and final disposal burdens equally across all product life cycles.
Smog Formation Potential	The relative reactivity of substances that produce ground-level ozone in the presence of sunlight measured in kilograms ozone equivalents.
Water Consumption	Volume of water removed from a watershed, measured by volume.

Abbreviations

0/100	Avoided Burden Allocation Method
100/0	Recycled Content Allocation Method
50/50	Shared Burden Allocation Method
ASTM	American Section of the International Association for Testing Materials
bio*	Biologically-Based, we have used "bio" preceding the acronyms for various plastic resins to indicate biologically-based resins
DEQ	Department of Environmental Quality
EOL	End-of-life
EPS	Expanded Polystyrene
FSW	Food Service Ware
FTC	Federal Trade Commission
GHG	Greenhouse Gas
GWP	Global Warming Potential
HDPE	High Density Polyethylene
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LDPE	Low Density Polyethylene
LUC	Land Use Change
MOPP	Metallized Oriented Polypropylene
MPLA	Metallized Polylactic Acid
MSW	Municipal Solid Waste
OPP	Oriented Polypropylene
PBAT	Polybutylene Adipate Terephthalate
PE	Polyethylene

PET	Polyethylene Terephthalate
PHA	Polyhydroxyalkanoates
PIA	Poly(itaconic Acid)
PLA	Polylactic Acid
PP	Polypropylene
PTA	Purified Terephthalic Acid
PVC	Polyvinyl Chloride
r*	Recycled, we have used "r" preceding the acronyms for various plastic resins to indicate recycled resins
TPS	Thermoplastic Starch
XPS	Extruded Polystyrene

Introduction

Over the last decade, the Oregon Department of Environmental Quality (Oregon DEQ) has broadened its focus from solid waste management to materials management across the full life cycle of materials and products, including but not limited to the management of materials as wastes. Although Oregon has made significant strides towards conserving resources through recycling and minimizing environmental impacts of disposal through proper waste management practices, overall waste generation, which dropped during the 2008 – 2009 recession, has resumed its historic upward trend (Oregon DEQ 2017).

Packaging is often targeted in sustainable materials management strategies because it is generally disposed after a single use and because of the large quantities of packaging entering the municipal solid waste (MSW) stream each year (Oregon DEQ 2012; U.S. EPA 2015). In 2014, Americans generated 76.7 million tons of packaging waste, comprising 30 percent of total MSW generation by weight (U.S. EPA Office of Resource Conservation and Recovery 2016). Notwithstanding a packaging recycling rate of 51.5 percent, packaging still represents 22 percent of the MSW sent to landfills or incinerated (U.S. EPA Office of Resource Conservation and Recovery 2016). Similarly, packaging makes up 20-30 percent of business and household waste in Oregon (Oregon DEQ 2017).

Yet while public concern and policy often focuses on the impacts of packaging waste, packaging impacts the environment in many other ways as well. Packaging consumes raw materials and energy for production and transport which in turn generates pollution, and disposal of packaging in landfills or by incineration represents a loss of the resources they contain as well as further pollution. While packaging plays an important role in minimizing waste by preventing damage to products, improvements in packaging design and informed choices of packaging material have the potential to considerably lower the environmental impacts of packaging.

This study considers four attributes commonly used as indicators of environmental benefits from the perspective of their effect on the life cycle environmental impacts. Considering the full life cycle is important for sustainable materials management efforts addressing municipal waste.

Research has shown that waste management only contributes a small part of the full life cycle environmental impacts of most goods. For example, waste management contributes only about 2 percent of annual U.S. greenhouse gas (GHG) emissions according to the U.S. Environmental Protection Agency's (U.S. EPA) GHG inventory (U.S. EPA 2017), while the full life cycle impacts associated with the provision of food and consumer goods represents 42 percent of the nation's annual GHG emissions (U.S. EPA Office of Solid Waste and Emergency Response 2009). In Oregon, production, distribution, and disposal of consumer products made up 41 percent of the state's 2015 consumption-related GHG emissions (Oregon DEQ 2012, 2018).

As noted in the "Materials Management in Oregon: 2050 Vision and Framework for Action" report, Oregon DEQ is working to determine the most efficient ways to reduce environmental impacts of products and materials by looking across their full life cycles (Oregon DEQ 2012). DEQ's use of life cycle thinking – considering environmental impacts across the full life cycle of materials, as opposed to focusing narrowly on impacts from just one stage of the life cycle, such as solid waste – dates back to the 1980s. DEQ's first significant use of environmental life cycle assessment (LCA), in 2004, evaluated 26 different options for shipping non-breakable items in an e-commerce order fulfillment center. This was followed by LCAs of drinking water delivery (2009) and single-family housing (2010). All three of these studies used LCA to quantitatively assess the potential environmental impacts of a wide variety of materials. Importantly, the studies describe impacts such as global warming potential (GWP), human toxicity potential, and ecosystem toxicity potential.

Such evaluation of potential environmental and human health impacts stands in contrast to how governments, individuals, and businesses have traditionally evaluated or considered the environmental impacts of materials. Historically, most selection or prioritization of materials for environmental purposes has relied on a series of attributes, which describe the sources or physical characteristics of materials. Common attributes include having *recycled* or *biobased content* or being *recyclable* or *compostable.*

It is widely believed that these attributes align with materials that have reduced overall impact on the environment. However, each of the DEQ studies mentioned above provided examples that challenged this popular wisdom. For example, DEQ's e-commerce packaging assessment identified that lightweight shipping bags – even if made from mixed materials such as paper/plastic blends – often resulted in lower environmental burdens than paperboard boxes, even if the shipping bags were difficult to recycle and contained limited recycled materials. Similar inconsistencies between attributes and impacts were found in other studies, leading DEQ to question whether these popular attributes reliably and consistently point in the direction of materials that have lower impacts to the environment, as is widely assumed.

Goal and Scope

The goal of this work is to review LCA studies of packaging and food service ware (FSW) to determine whether it is appropriate to use the material attributes *recycled content*, *recyclable*, *biobased*, and *compostable* to infer environmental benefits. This literature review assesses the conditions under which each attribute is directly correlated, inversely correlated, or uncorrelated with environmental benefits when viewed across the entire life cycle.

The products and attributes included in the study were selected based on their role in many sustainable materials management strategies and the availability of sufficient LCA studies. A preliminary literature search was performed prior to the formal literature review to assess the availability of LCA studies related to products and attributes commonly discussed in the field of sustainable materials management. While the focus of this study is the United States, LCA studies are included regardless of their geographic scope to provide a broad basis for conclusions. The potential effect of geography on study results is considered in the interpretation of findings. The metrics included in the analysis are energy, water, and land use and a series of commonly used life cycle impact assessment (LCIA) midpoint impact indicators. It should be noted that impacts associated with litter or marine debris are not included since methods to assess these impacts in LCA studies are not available. Also, solid waste mass or volume are not metrics typically reported by LCA studies as the impacts associated with waste management are included in other metrics such as land use, GWP, and ecosystem toxicity.

The Life Cycle of Packaging

The life cycle of packaging, as shown in Figure 1, includes raw material extraction, primary material production, packaging production, distribution, use, and end-of-life consisting of recycling, reuse, or disposal. Litter refers to uncollected releases to the environment produced from packaging, whether on land or water. Often comparative LCAs omit parts of the life cycle that are identical across comparisons. For this reason, the environmental burdens related to the product contained in the package may or may not be included in LCAs examining packaging. This will affect the percent changes in impact metrics associated with packaging and food service ware scenarios. In general, the product itself contributes more to the overall life cycle impacts than the packaging (Hanssen 1998). If the product is included within the system boundary, changes in recycled content or end-of-life treatment of the packaging can have minimal impact on overall impact results. If packaging is examined on its own, raw material production usually contributes the greatest proportion of impacts to the overall life cycle of packaging for all results categories typically included in LCAs (Hanssen 1998). Packaging production may have substantial environmental burdens related to energy use from the manufacturing process. The distribution phase includes transport to the product packaging location and to retail and tends to contribute most to acidification and smog formation impacts. In all cases, the weight of the product being packaged is factored into the calculation of the burdens for distribution to retail locations even if production and processing of the product is excluded. Typically, the process of filling the packaging with the product and the product use phase are not considered in the life cycle of packaging unless there is an aspect of using one type of material for packaging as opposed to another that results in differences in the filling or use phase for the product. End-of-life is an area of significant focus for single-use packaging since it is discarded after delivering the product to the consumer and can involve a substantial amount of material.

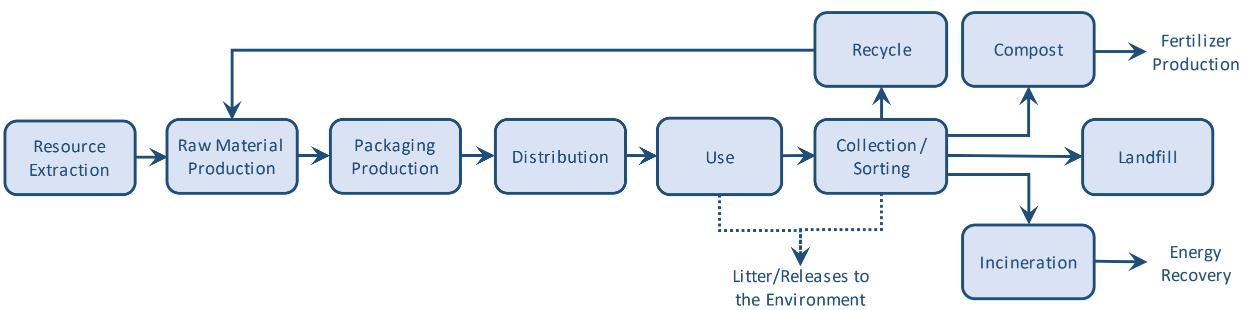

Figure 1.The life cycle of single-use packaging and food service ware.

The Life Cycle of Food Service Ware

The life cycle of FSW follows the same general stages as the life cycle for packaging, from raw material extraction through to disposal, as shown in Figure 1, but each stage can vary considerably based on the specific product. FSW products include items such as knives, forks, spoons, cups, straws, dishes, wrappers, trays and tray liners, and clamshells and similar containers when used for takeaway food consumption. Bottles for liquid consumption are not necessarily takeaway food items, and thus are considered as packaging, not FSW. The FSW products included in this report are made from various plastic resins and wood fiber. Results of the studies included in this review suggest that most of the environmental impacts from FSW products occur during the feedstock production and manufacturing phases, particularly for biobased FSW, though transportation can sometimes produce significant emissions for the GWP and acidification impact categories, depending on the distances traveled (Vercalsteren and colleagues, 2006). Despite a wide range of use cases for FSW products: restaurants, takeout, schools, hospitals, prisons, contract catering, large outdoor events, festivals, employee break rooms, and home use either for everyday use or parties and special events; the system boundary for most LCAs of FSW products do not provide much detail regarding the use case considered. These different uses can affect their environmental footprint by influencing which end-of-life processing a specific FSW product undergoes. For example, the composition of waste streams are more easily controlled in contained venues, such as sporting events, which can contribute to the efficiency of recycling or composting logistics (Hottle and colleagues, 2015). Additionally, some types of FSW products are more likely to be reused than general packaging, as they can be cleaned and reused relatively easily. Because of this, reusable FSW is better represented in the literature than for other types of packaging.

Recycled Content

The U.S. Federal Trade Commission (FTC) defines recycled content as the portion of materials used in a product that have been diverted from the solid waste stream. Recycled content is further distinguished as pre-consumer or post-consumer where post-consumer recycled content is from material recovered after product use by the consumer and pre-consumer recycled content is sourced directly from manufacturing process waste.[1] The FTC notes that in order to claim pre-consumer material as recycled content, it must be shown that the material would have otherwise been managed as solid waste (FTC, 2012). Therefore, scrap that can be reused in the same manufacturing process that produced it should not be counted as recycled content.

When material from one product is recycled and then incorporated into a new product as recycled content, a decision must be made on how to allocate the burdens of virgin materials production, collection and recycling of the material, and final material disposal across the life cycles of the two or more products (Weidema 2000). The recycled content method, avoided burden method, and 50/50 method are three commonly used recycling allocation methods used in attributional LCAs (Toniolo et al. 2017; Johnson, McMillan, and Keoleian 2013). All three of these methods were represented in the studies reviewed in this project.

[1] Pre-consumer recycled content may also be referred to as post-industrial recycled content, external regrind, and prompt scrap.

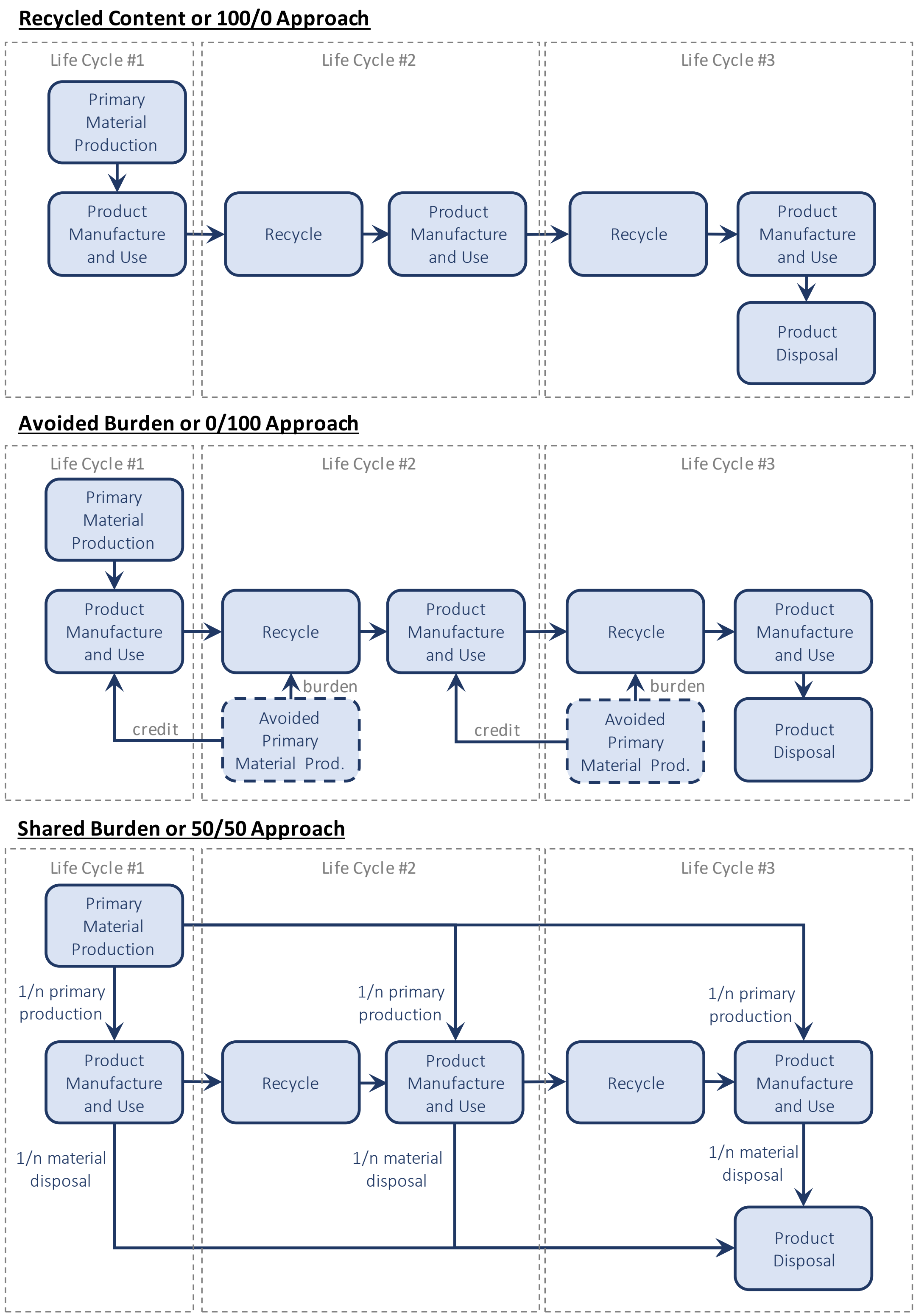

Figure 2. Recycling Allocation Methods Used in Life Cycle Assessment Studies.

The recycled content method, also referred to as the cut-off or 100/0 method allocates the entire burden of primary material production to the first product while the burdens of material collection and processing for recycling are attributed to the subsequent product (see Figure 2) (Toniolo et al. 2017). End-of-life disposal burdens are assigned to the product system from which the material is ultimately disposed, and no credits are granted for energy recovery resulting from disposal of materials.

The avoided burden approach, also known as system expansion, end-of-life-recycling, recyclability substitution, or 0/100 (Toniolo et al. 2017), assigns the benefits of material recycling to products recycled at end-of-life and does not give credits for use of recycled content. The first product, when recycled, is credited for the avoided burdens of virgin material production for the form in which the material is used in the second product. The avoided material credit may not correspond exactly to the form in which the material was used in the first product. For example, polyethylene terephthalate (PET) bottles made of high viscosity food-grade PET may be recycled into amorphous PET and used in the second system without restoring the material to the same viscosity and level of decontamination as in its first use. In this case, the material credit would be based on avoiding the production of amorphous PET, a ‘down-cycling’ of the original high viscosity food-grade PET. The secondary product would then be associated with production impacts equal to the material credit of the first product, and is eligible for a material credit if it is also recycled (FTC, 2012).

The shared burden or 50/50 method assigns virgin material production, recycling, and final disposal burdens equally across the first and second products (Nicholson et al. 2009). When a material is recycled across many product life cycles, the virgin material production, recycling, and final disposal burdens are divided equally across all product life cycles.

The difficulty of selecting a method for assigning impacts associated with recycled materials is compounded by several factors. First, that the number of product life cycles in which a material is used is not well understood and is also highly variable depending on the fate of the product. For example, old corrugated containers, a standard term for recovered corrugated board, can be recycled back into corrugated medium or linerboard (used to make new corrugated containers), but the rate at which old containers are recycled varies depending on whether the containers are used in commercial settings (such as grocery stores, where the recycling rate is high) or in households (where the recycling rate is lower). Second, the wide range of alternate uses for some recycled packaging material can make the material difficult to track. Third, the properties of certain materials change as they are recycled, and this limits their use to different products as they progress from one life cycle to the next. Paper products, for example, can only be recycled a limited number of times because the wood fibers break down and at some point become too short to be useful (Ekvall 2000a). Metals and plastics, while infinitely recyclable in theory, decline in quality as they become contaminated with other materials over time; examples include plasticizers, coatings, dyes, and alloying agents (Johnson, McMillan, and Keoleian 2013).

While statistics are available on recycling rates for certain materials/product types, data are not available to quantify the average number of product lifetimes in which a material is used. Therefore, an LCA practitioner could only reasonably predict a material’s use in two to three product lifetimes – for a product using virgin material that is recycled at the end of the product's useful life, the material will be incorporated into at least one more product before disposal, and for a product with recycled content, the practitioner knows that there was at least one prior use of the material and can assume one more use if the product being modeled is recycled at end-of-life. The recycled content approach favors products with high levels of recycled content because recycled material comes into the system free of virgin production burdens. Conversely, the avoided burden method results in lower impacts for products with high recycling rates. The avoided burden method is preferred by the metals industry as the supply of recycled metals is limited due to the long service life of many products utilizing metals. The metals industry contends that the benefits of material recycling should be attributed to recycling at the end-of-life to encourage recycling to increase the amount of secondary metals available (Atherton 2007; Johnson, McMillan, and Keoleian 2013; World Steel Association 2011). The avoided burden approach is also employed in US EPA's WARM tool, which is widely used by recycling and solid waste professionals in the US.

In addition to these methods addressing post-consumer recycled content, allocation of pre-consumer recycled content, material that has not been used in a prior product life cycle, is a special case. In general, pre-consumer recycled content is either assigned the full impacts of virgin material production or considered a co-product of the original product's manufacturing process where burdens of virgin material production for the scrap could be allocated between the original and subsequent product on a mass or economic basis. Note that this treatment conflicts with the FTC definition of recycled content as being material diverted from a waste stream, as a waste would not generally be considered a co-product.

In the presentation of packaging results below, comparisons between packaging with recycled content and packaging of different material with less or no recycled content are provided using all studies and then again using only studies that use the recycled content (100/0) allocation method.

Recyclable

Recyclability is the potential for a material to be remade into a new product at the end of a prior product's useful life. The FTC defines a recyclable product as one that can be collected, separated, or otherwise recovered from a waste stream through a recycling program for use in the manufacturing of a new item. The FTC also stipulates that a claim of recyclability can only be made in markets where the recycling program is available to at least 60 percent of the communities or consumers (FTC, 2012). This requirement recognizes the importance of collection and reprocessing services in addition to the physical characteristics of the material. Promoting the availability of recycling services is a focus of many waste and sustainable materials management programs. For example, in Oregon all municipalities with at least 4,000 people are required to provide recycling services, which the State strictly defines in terms of various collection programs, drop-offs, and educational programs in addition to material types (Oregon Revised Statute 459A).

When considering the benefits of recyclable packaging and FSW, it is important to represent accurately the portion of the material that is recycled. A few key factors play in to actual recovery rates of materials. The availability of collection services and drop off locations for recyclable material is the first consideration that determines the fraction of the material diverted from landfilling or incineration. Once collected, materials are transported, sorted, cleaned, reprocessed, and finally sold to make new products. Some portion of the collected material can be lost in these steps, decreasing the fraction recycled. For example, mixed recyclables contaminated by food waste, broken glass, or other non-recyclable items are often separated by recycling facilities and sent to landfills or incinerators. Some materials, particularly recovered materials sold offshore, may require additional sorting and cleaning steps to remove contamination to prepare them for various product manufacturing steps. For these reasons, LCA studies addressing the benefits of recyclability should account for the losses in material collection and the inputs required for the recycling process.

The benefits of recyclable packaging and FSW also depend on the market for recycled material. Which materials are likely to be recycled often depends on the cost of collection, separation, processing, and transportation being lower than the market price for the secondary (waste) materials, along with any savings that may be realized because of diverting the materials away from other disposal methods, such as savings in landfill tipping fees. In other words, recycling must be an economically viable undertaking, or it must otherwise be subsidized. Even when collection and recycling services are available, if the market for post-consumer material is saturated, additional materials depress prices and can potentially cause material to go unused and ultimately disposed. In a market consistently constrained by the availability of secondary material, it is more likely for a product to be recyclable than to use recycled content. This is because the secondary material from one product could be used in another product if it were not used in the product at hand, and secondary material is likely to have a higher price than in an unconstrained market. In a market where there is a surplus of secondary material and few products that use it, it is more likely for a product to have recycled content than to be recyclable, as the secondary material is likely to have a lower price than in a constrained market. In LCA studies, market effects are reflected in the allocation of impacts between primary and secondary materials described in the previous section. The recycled content or 100/0 method is appropriate in a market saturated with secondary materials while the avoided burden or 0/100 method is appropriate where there is high unmet demand for secondary material (Frees 2008; Gala, Raugei, and Fullana-i-Palmer 2015; Geyer et al. 2016; Zink, Geyer, and Startz 2016). However, market conditions are often unknown or unavailable to LCA practitioners considering products that are recycled.

Few LCA studies with quantitative results explicitly analyze all the factors mentioned above when considering recyclable packaging. To expand the pool of LCA studies in the review, studies that analyzed one or more of these characteristics, such as quantitative environmental results for varying recycling rates for packaging materials were included. Studies that use the 0/100 and 50/50 allocation methods to evaluate recyclability are also included in this report. The 0/100 or avoided burden approach is included because it is used for allocating the life cycle benefits of avoided production of primary material when materials undergo recycling at the end-of-life (EOL), assuming there is primary material displacement. The 50/50 method is included because it is a recommended approach when secondary material markets are unknown and material displacement data are unavailable for LCA studies (Ekvall 2000b; Zink, Geyer, and Startz 2016). Comparisons that use the recycled content (100/0) allocation method are not included in this section as it assigns no benefit to recyclability at the end-of-life of packaging materials.

Biobased

Biobased packaging is defined as packaging made from renewable feedstocks that can be replenished as they are used. The most common feedstocks are currently plant-based, utilizing fermentation processes via starch or sugars, but can also be a direct product of the metabolism of a plant or be generated from alternative biological sources like fungi, animals, or organic wastes (Gerngross and Slater 2000; Haneef et al. 2017; Hottle, Bilec, and Landis 2013; Kendall 2012; Koller et al. 2013; Kurdikar et al. 2000; Schiffman 2013). Biobased materials are commonly considered beneficial because it is assumed that their feedstocks do not deplete reserves of non-renewable resources such as petroleum, natural gas, and mineral deposits. In addition, plant derived biobased materials are considered beneficial because they offer a greenhouse gas benefit due to the carbon dioxide drawn from the atmosphere during plant growth. This study was conducted in part to test these kinds of assumptions and understand the differences between fossil and biobased production paradigms. We identified studies related to biobased, renewable packaging or FSW and will refer to these materials collectively as biobased throughout this document.

It is worth noting that “biobased” does not necessarily imply biodegradability or compostability and that some biobased materials neither readily degrade nor can be composted. Not all packaging materials included in this review are 100 percent biobased either. Many biobased plastic products require the addition of plasticizers and stabilizers to meet performance requirements and, as in the case of Coca-Cola’s PlantBottle®, there are some products where the majority of feedstocks are not biobased (Tabone et al. 2010; Weiss et al. 2012; Yates and Barlow 2013). Depending on the material type, a biobased product may or may not be compostable, biodegradable, and/or recyclable at the end of the product’s useful life. Some products have been developed specifically to provide compostable options for packaging, single-use, and non-durable goods, including compostable paperboard materials, many biobased plastics, like polylactic acid (PLA) and thermoplastic starch (TPS). Some traditionally fossil-based resins, e.g. polyethylene (PE) and PET, can be manufactured using biobased feedstocks with an identical chemical structure and functional properties which can leverage existing manufacturing and waste infrastructure, but neither the fossil-based nor biobased options are necessarily degradable or compostable.

This literature review considers the environmental impacts of biobased packaging materials and FSW compared to their fossil-based counterparts. The initial search for biobased materials included generic terms like ‘biobased’ and ‘renewable’ to identify any studies that fall within the scope of the review. All the materials evaluated in the relevant studies fall into three distinct categories, those made from wood, paper, and biobased plastics. In addition to the high-level review of biobased options, material- and study-specific examples are highlighted based on an in-depth evaluation of the available literature and are available in the *Supporting Information B.*

Compostable

Compostable materials are those that degrade by biological processes to yield CO_2, water, inorganic compounds, and biomass at a rate consistent with biodegradation of natural waste while leaving no visually distinguishable remnants or unacceptable levels of toxic residues (ASTM International, 2012). In the context of plastic/polymer packaging materials, compostability is governed by international standards (ASTM International, 2012, 2017; European Committee for Standardization, 2000). While organics composting can be accomplished in a backyard setting, nearly all packaging marketed as compostable does not biodegrade in such settings, instead requiring an appropriately scaled composting facility to provide the conditions required to meet the expected levels of degradation in a reasonable time frame. Moreover, these facilities require that materials used as input for the compost process have limited contamination from non-compostable materials. Depending on their use, compostable packaging may be exposed to contaminants that limit its ability to be composted in such facilities without a previous screening or washing process. These requirements differentiate compostable packaging materials from biodegradable ones, which usually do not have facility or device requirements, and may biodegrade in environments that do not meet the specific conditions required to create compost. Unless otherwise noted, the studies included in this review assessed the impacts of industrial scale composting for compostable packaging.

Materials used in the production of compostable packaging include petroleum based polymers (e.g. Polybutylene adipate-co-terephthalate or PBAT); biobased polymers such as PLA and TPS; and blends of petroleum and biobased polymers, such as PBAT+PLA (Ecovio®) (Kijchavengkul and Auras 2008). Additionally, paper and paperboard materials are compostable and can also be used for packaging, as they are flexible and can be easily shaped. However, paperboard is often permeable to liquids and oils and have to be coated, laminated or otherwise amended with barrier materials to ensure they can be used effectively as packaging (Farmer 2013). This may introduce non-compostable materials or additional toxins to paperboard packaging, which may prevent the packaging from being accepted at industrial composting facilities.

It is important to distinguish the composting process, which is an aerobic process, from anaerobic digestion. While composting can sometimes be referred to as “aerobic composting”, and digestion can sometimes yield compost as an output, the processes involved are different enough that they can be used for different purposes, such as biogas production from anaerobic digestion that can be used to offset other energy demand. Anaerobic digestion is not a focus of this review, but it is included in cases where individual studies compared the environmental impacts of both digestion and composting of the same material.

In the LCA studies reviewed, the main benefits of composting as a waste management method are usually associated with the use of the compost as a replacement for agricultural use of fossil-based fertilizers and peat. Most studies assign environmental credits to the compost for the avoided production of these items based on the amount of nutrients included in the compost. However, there is little consensus in the literature regarding how to allocate these credits, with some studies assigning few credits to compost resulting only from packaging due to the relatively low nutrient content of the packaging materials. On the other hand, compostable packaging is usually assigned the burdens associated with the production of their feedstock materials, such as biomass production for biobased packaging, or petroleum extraction for PBAT, as well as the impacts related to the composting process itself.

Life Cycle Assessment

LCA is a rigorous and standardized approach for quantifying the environmental impacts of a product (or service) across its entire life cycle, including the extraction or harvesting of raw materials from the earth through material processing, manufacturing, distribution, use, and end-of-life. When done properly, an LCA should quantify all relevant environmental flows such as releases to the environment as well as the use of resources such as land, water, fossil fuels, and minerals. The International Standards Organization's (ISO) 14040 series of standards govern and provide guidance for those performing LCA studies. LCA studies begin with a clearly defined goal and scope, which includes the definition of a functional unit which serves as the basis for any comparisons made in the LCA and/or for the presentation of results. The functional unit ensures an apples-to-apples comparison. For example, two packaging options might be compared based on their ability to deliver a certain quantity of product. When necessary the system is expanded to include relevant inputs outside the packaging itself. For example, a comparison between a half gallon of ready to serve orange juice and frozen orange juice concentrate would also include the water and reusable pitcher used to prepare the juice from concentrate. And, of course, the impacts of producing the reusable pitcher would be allocated across many uses over its lifetime. Thus, although the basic concept of an LCA is somewhat simple, in practice resolving interactions between systems and comparing across different products is quite complex. Fortunately, a large body of research provides robust methods for performing LCA studies.

The process of performing an LCA study is often divided into the life cycle inventory and life cycle impact assessment stages. In the life cycle inventory (LCI) stage, the practitioner prepares an inventory of releases to the environment and resource inputs from the environment involved in the life cycle of the product or service being assessed. LCA models are constructed of a series of *unit processes*, and releases and resource inputs are tracked for each unit process. Unit processes are also connected to one another when the output of one is used as an input to another. For example, iron ore concentrate from iron mining, coal from coal mining, and train transport would all be inputs to steel production. When creating an LCA model, typically the practitioner creates a series of custom unit process datasets for the unit processes that contribute most significantly to the results for their system of interest. These unit processes are referred to as the *foreground* unit processes. When creating foreground unit processes, the practitioner takes into consideration the location where each process occurs, and the technologies used. For example, the electricity grid in the northwest U.S. would reflect a higher percentage of hydroelectricity than the electricity grid in certain Midwestern states for example which rely heavily on coal. The supply chains for less significant inputs as well as those for common and standardized inputs such as grid electricity, PET resin, or aluminum are often represented by off-the-shelf *background* datasets such as ecoinvent, GaBi, or the U.S. Life Cycle Inventory. The foreground and background datasets are brought together in an LCA tool such as SimaPro, openLCA, GaBi, or Umberto to resolve the connections between unit processes into the inventory of releases to the environment and resources from the environment associated with the provision of the functional unit.

Life cycle impact assessment (LCIA) involves calculating the potential environmental impacts associated with each release to the environment and resource from the environment included in the life cycle inventory. Typically, life cycle impact assessment is performed using generalized LCIA methods developed for characterizing various impacts in a given region. Examples of commonly used LCIA methods include the U.S. EPA's Tool for the Reduction and Assessment of Chemical and Environmental Impacts (TRACI), ReCiPe, and IMPACT 2002/IMPACT World+. These LCIA methods are a collection of models and estimation methods used to characterize various impacts. For example, both the TRACI and IMPACT World+ LCIA methods adopt the USEtox model to characterize human health and ecosystem quality impacts. In practice, LCIA methods are a collection of characterization factors describing the impact of a given release or resource input. For example, GWP is often characterized in terms of kilograms carbon dioxide equivalents, thus the release of one kilogram of carbon dioxide causes an impact of one kilogram carbon dioxide equivalent and the release of one kilogram of methane causes an impact of 28 kilograms of carbon dioxide equivalents (IPCC 2014).

Impacts can be tracked at the midpoint or endpoint level. Endpoint impacts are considered the ultimate impacts of interest, such as human health and ecosystem quality effects, measured in quality/disability adjusted life years (QALY/DALY) and partially disappeared species-square meter-years respectively. In some cases, resource scarcity is also tracked at the endpoint level in terms of the increased cost of providing energy following depletion of more easily available resources. Midpoint impacts are metrics defined along the cause-effect chain between releases and resource inputs and the endpoint impacts. Examples of midpoint impacts include GWP measured in kilograms carbon dioxide equivalents, particulate matter formation potential (leading to human respiratory effects) measured in kilograms particulate matter 2.5 (<2.5 microns) equivalents, acidification potential measured in kilograms sulfur dioxide equivalents, and human toxicity measured in common toxic units, a metric specifically developed for comparing toxicity in the context of LCA studies. While endpoint metrics reflect the actual impacts of interest, midpoint metrics are often reported in LCA studies because they relate more closely to the releases and resource inputs tracked in the life cycle inventory and can be reported with more certainty. Estimating endpoint impacts from midpoint impacts requires additional assumptions regarding uncertain effects and effects occurring over longer periods of time. For example, while estimating the radiative forcing effect of various greenhouse gases can be accomplished using atmospheric modeling, estimating specific human health and ecosystem quality effects of a warming climate involves more uncertainty.

In this study, we consider twelve midpoint impact categories commonly tracked in LCA studies. Although different LCIA methods are used to estimate these impacts across LCA studies, the definition of each midpoint is consistent, and the differences in approaches are reflective of the fact that LCIA methods improve as researchers develop them over time. The twelve midpoint impacts tracked in this review are as follows.

Midpoints calculated based on environmental releases

- **Global Warming Potential** - The heat trapping capacity (radiative forcing potential) of greenhouse gases (GHGs) and their precursors. Calculated in terms of kilograms carbon dioxide equivalents.

- **Eutrophication Potential** - Enrichment of an ecosystem with nutrients that accelerate biological productivity. Eutrophication is characterized separately for freshwater and marine ecosystems as freshwater ecosystems are more frequently phosphorus limited while marine ecosystems are more frequently nitrogen limited. Calculated in terms of the nitrogen or phosphorous content of releases available to ecosystems and measured in kilograms nitrogen equivalents or kilograms phosphorus equivalents.
- **Particulate Matter Formation Potential** - Potential to form particulate matter leading to human respiratory effects.
- **Smog Formation Potential** - The relative reactivity of substances that produce ground-level ozone in the presence of sunlight measured in kilograms ground-level ozone equivalents.
- **Human Toxicity** - Increase in morbidity in the total human population due to exposure to carcinogenic substances or substances resulting in non-cancer diseases measured in common toxic units or toxic chemical equivalency.
- **Ecosystem Toxicity** - Potential of chemicals to cause toxic effects on aquatic species. The models used to estimate ecosystem toxicity typically consider chemical fate, transport, and exposure of organisms. Measured in terms of common toxic units or toxic chemical equivalency.[2]
- **Acidification Potential** - The acidifying potential resulting from acid rain caused by inorganic air emissions measured in sulfur dioxide equivalents.
- **Ozone Depletion Potential** - Capacity of substances to deplete the stratospheric ozone, measured in kilograms chlorofluorocarbon-11 equivalents.

Midpoint indicators calculated based on resource use

- **Land Use/Occupation** - Area of land occupied over time, characterized in terms of biological productivity. Measured in terms of area equivalent to an area of land of specified biological productivity.
- **Water Consumption** – Water withdrawals less water returned to the same watershed, measured by volume.
- **Fossil Energy Depletion** – Amount of fossil energy extracted from the earth, measured in terms of equivalency to a reference fossil energy source, typically oil equivalents, in terms of relative abundance in the earth’s crust.
- **Mineral Depletion** – Amount of scarce minerals extracted from the earth, measured in terms of equivalency to a reference mineral, for example iron equivalents, in terms of relative abundance in the earth’s crust.

The strength of LCA for comparing the relative benefits of packaging and food service ware options, is that it is an approach that has been developed over time for comparing options based on their environmental effects. There are many methodological issues which have been extensively discussed in the LCA literature. The previous section described various approaches that are used for assigning the impacts of recycled materials to their downstream and upstream

[2] Note than the impacts of plastic debris, fragments, and degradation products on marine ecosystems are not explicitly tracked in the ecosystem toxicity impact assessment methods used in LCA studies. Marine debris is discussed in more detail in the following section.

life cycles. More generally, assigning impacts to multiple products of a single unit process is another issue that commonly arises in LCA. Following the ISO guidance, first multi-output unit processes should be further detailed to assign the impacts of subprocesses to specific products wherever possible. When it is not possible to separate impacts, the ISO 14044 standard recommends expanding the system to avoid the need for allocation. For example, in the case of a process using electricity from another process that also produces low-grade steam used for heating, the electricity might be credited with the avoided impact of the combustion of natural gas which would have otherwise been used to meet the heat demand. In other cases, practitioners will allocate impacts across multiple outputs according to their mass, energy content, or monetary value. The ecoinvent and GaBi background datasets used in LCA models have also been refined over time. These datasets provide the building blocks that can be used to represent almost any system. For many common and environmentally intense processes, ecoinvent and GaBi provide representations of various technologies and production in different locations due to differences in technologies, electricity sources, transport modes and distances, etc. Similarly, LCIA methods have been developed and improved over time and are customized to reflect, for example, North American or European conditions. LCIA methods have further benefitted from the activities of the UNEP-SETAC Life Cycle Initiative which have considered various impact assessment methods and produced consensus-based approaches such as the USEtox model for assessing human health and ecosystem quality effects.

As a result, LCA is very good at providing quantitative results for comparisons between products or technologies. For this review, we can compare quantitative results across the twelve impact categories previously described. LCA studies provide a clear description of the functional unit and the scope of the system considered in the study. We can understand something about the quality of the results based on the description of the foreground processes, background datasets, and impact assessment methods. The LCIA methods used in LCA studies have been well thought through and are reflective of the actual environmental impacts of interest.

A challenge in a review such as the one conducted here is the complexity of LCA models and the large amount of data used to populate them. As a result, it is challenging for an LCA practitioner to report and a reader to comprehend all the data and details involved in the calculation of an LCA result. For this reason, this literature review is limited to comparisons presented in a given study. We do not consider comparisons across LCA studies. Similarly, we are limited in our ability to track and attribute all the system details and modeling decisions to the various results. The comparisons between packaging and food service ware options with the attributes considered in this study and their conventional counterparts are affected by the locations where the packaging and FSW are produced, the technologies used, and other factors. Relatively new systems such as biobased plastics and compostable packaging could potentially be improved with new technologies such as cellulosic feedstocks, larger-scale production facilities, improved logistics, and combined collection/composting with nutrient-rich food waste. The findings presented here reflect the quality of the LCA studies included in the review. Presumably peer-reviewed LCA studies should provide reliable results, however in practice there has been some variability in the quality of published LCA studies. To address this, we provide Supporting Information to help the reader understand the underlying studies, data, and comparisons upon which our findings are drawn.

Plastic Pollution in Marine Environments

Plastic pollution in marine environments has received much attention (Cózar et al. 2014; Lebreton et al. 2018; Thompson et al. 2009; Wong, Green, and Cretney 1974). In a recent assessment, Jambeck and colleagues (2015) estimated that 5 to 13 million metric tons of plastic entered oceans in 2010. Plastics are persistent in the marine environment and cause harm to wildlife through ingestion, disrupting ecosystems and potentially endangering species such as sea birds and turtles (Derraik 2002). Plastics also break down over time creating fragments which are more easily ingested and which expose marine species to toxic degradation products (Rochman 2015).

When not properly managed, single-use packaging and food service ware contribute to the marine debris problem (Geyer, Jambeck, and Law 2017). Schmidt and colleagues (2017) assessed the flow of plastic debris from rivers to the ocean and found that 10 large watersheds, 8 in Asia and 2 in Africa, with large populations and insufficient waste management services contribute over 90 percent of the plastic mass released to the ocean annually. This finding supports Jambeck and colleagues' (2015) estimate based on waste generation and management practices that estimates significant quantities of plastic waste entering oceans from countries with large populations, significant consumption, and significant amounts of mismanaged waste. The top 20 countries contributing to waste plastic in oceans are Asian and African countries with the notable exceptions of Brazil (#16) and the U.S. (#20). While most countries earned a spot in the list due to a high percentage of mismanaged waste, the U.S. contributed 40-110 thousand metric tons of plastic marine debris due to a combination of high per capita plastic waste, one-third kilogram per person per day, and a comparatively higher rate of waste management at 98 percent, as opposed to many other countries in the list with much lower waste management rates, generally between 10 and 40 percent.

It is important to recognize that the amount of plastic entering marine environments attributable to U.S. consumption may be larger than indicated by the above estimates reflecting the direct flow of plastic waste to oceans from the U.S. Most of the materials collected via recycling programs in North America and Europe, particularly plastics, are exported. These materials typically undergo additional sorting and processing at the importing country where solid waste management systems are poorly developed and lack controls for leakages into the environment. This step in the global secondary material exchange could indirectly contribute significant plastics to marine debris. Similarly, the acceptance of compostable materials such as food service ware in some organic waste collection systems results in contamination with non-compostable plastics by generators who may not successfully distinguish acceptable from non-acceptable materials. While many industrial composters attempt to screen non-compostable plastics from inputs and/or un-composted plastic fragments from finished compost, plastic contamination of finished compost is a growing concern, and is another vector by which plastics may find their way into freshwater and marine environments. Thus, while Jambeck and colleagues (2015) find that the direct flow of plastic waste to oceans from the U.S. is less than one percent of the annual amount of plastic waste entering oceans, it is possible that leakage of plastic waste exported from the U.S. for recycling and the contamination of compost produced from the U.S.' compostable packaging and food service ware with non-compostable plastic contribute amounts in addition to those reported by Jambeck and colleagues.

For the purposes of this review, the issue of plastic marine debris does not play a significant role. This is primarily because we are focused on packaging and food service ware systems in the U.S., where most of the discarded materials are managed through "conventional" waste management, taken to mean landfilling, incineration, and recycling as currently practiced in the U.S. (The Economist 2018). Plastic becomes marine debris due to mismanagement, through individual behavior or inadequate collection and treatment infrastructure services, rather than because of the choice of management practice. In other words, any of the waste management practices discussed in this report, recycling, landfilling, incineration, or composting, may reduce plastic marine debris by reducing litter or informal management practices. Additionally, plastic marine debris is not a metric that is typically tracked in LCA studies and was not explicitly discussed in the studies reviewed here.

It should be noted that the attributes examined here could marginally influence the release of plastic to oceans. For example, the creation of a market for secondary materials could serve to reduce incentives for the improper management of these materials. Keeping plastic packaging and food service ware out of landfills could reduce unintentional losses to wind from uncovered landfills, although these losses should be low for properly managed landfills. Compostable packaging and food service ware which is biodegradable in the natural environment would reduce the lifetime of packaging and food service ware litter in the environment. While these topics were outside the scope of this review, they offer interesting topics for future research.

Method

Literature Reviews

The literature review was conducted in three phases: (1) literature search, (2) review and identification of potentially relevant search results, and (3) in-depth review of selected relevant articles. Given the rapid development in LCA methodology over the last few decades, a decision was made to limit this literature review to studies published in the years 2000 to 2017.

The first round of the literature search focused on five journals known to publish high quality LCA studies – International Journal of LCA, Journal of Industrial Ecology, Journal of Cleaner Production, Environmental Science & Technology, and Packaging Technology and Science. Additional literature reviewed included dissertations and known packaging LCA studies published by Oregon DEQ and other reputable sources such as the International Reference Center for the Life Cycle of Products, Processes and Services, Quantis, PE Americas (now thinkstep), University of Michigan Center for Sustainable Systems, and Michigan State University Center for Packaging Innovation and Sustainability as well as publicly available studies completed by Franklin Associates.

The initial search terms were determined based on a preliminary search for LCA studies addressing a product type, material, or material attribute. Our initial investigation found that *LCA* and *life cycle* were effective for limiting results to LCA studies and related work while *packaging*, *package*, *container*, *bag*, *box*, *clamshell*, and *bottle* were effective in returning packaging-related studies.

The searches for literature were then expanded in a second round and included other searchable peer-reviewed sources to cover all journals, dissertations, and published technical reports by using a university library search engine.[3] An abstract search was performed using expanded queries. After the second round of search was conducted, the previously identified studies were selected for inclusion in this review based on the following criteria:

- The study focused on packaging, FSW, or materials used to create packaging or FSW.
- The study focused on at least one of the four material attributes: recycled content, recyclability, biobased/renewable, or compostability.
- The study in question was a published, comparative LCA accessible online.
- The study included quantitative results for one or more impact categories for the materials analyzed.

Supporting Information A contains a complete list of the search terms used for each product attribute combination, as well as information on the product and packaging types studied, the functional unit, geography, system boundary, life cycle inventory (LCI) data sources, allocation method(s), LCIA method, result categories, and key findings for the studies selected for an in-depth review and analysis.

Packaging

Recycled Content

Terms used to identify articles for the recycled content attribute include *recycled content*, *recycled material*, *recycled fiber*, *rPET*, *cullet*, and *OCC*, where rPET refers to recycled PET and OCC refers to old corrugated cardboard. Although steel, aluminum, and high density polyethylene (HDPE) are also highly recycled materials (U.S. EPA Office of Resource Conservation and Recovery 2016), search terms for these materials were not explicitly included. Whereas the terms *rPET*, *cullet*, and *OCC* are unique in that they specifically refer to the recycled content of PET, glass, and corrugated boxes and therefore help to identify studies where recycled content is considered.

An initial search was conducted which was limited to journal articles that included the terms *LCA* and *packaging* as well as either *recycled content*, *recycled material*, *recycled fiber*, *rPET*, or *cullet* somewhere within the article. The search yielded 73 results. An initial manual scan of the results looking at the title and abstract identified 10 potentially relevant articles.

After further examination of the 10 articles, seven were found to address the topic of environmental impacts in the context of recycled content as a material attribute.

The literature review for recycled content was expanded, resulting in 59 potentially relevant studies, excluding newspaper and magazine articles. The same query was used to search all fields

[3] University of Cincinnati Library advanced search engine, last accessed September 2017: [http://uc.summon.serialssolutions.com/advanced#!/advanced]

combined with the journal publication title through the advanced search tool across several publishers' websites for Journal of Cleaner Production[4], Environmental Science and Technology[5], Journal of Industrial Ecology[6], and Packaging Technology and Science.[6] These searches identified 389, 75, 132, and 35 journal articles, respectively.

Several articles relating to recycled content or recycling were identified for further review, four of which were determined to examine the life cycle environmental impact of recycled content in packaging. Four publicly available LCA studies carried out by Franklin Associates were found to be relevant to this study, two of which were completed on behalf of Oregon DEQ. Additional studies were identified by Oregon DEQ for inclusion. In total there were 20 studies identified for review, three of which compare environmental impacts of recycled materials to virgin materials and 17 that compare the environmental impacts of two or more functionally equivalent packaging options with different levels of recycled content.

Recyclable

Terms used to identify articles for the recyclable attribute include *recyclable*, *recycling*, *recyclability*, *material recycling facilities*, and *recyclate*. These terms were combined with *LCA* and *life cycle assessment*, as well as with *packaging*, *plastic*, *cardboard*, *aluminum*, and other materials to create complex search terms with two or three components (e.g. *recyclable* and *LCA* and *packaging*). The terms *packaging weight* and *packaging type* were combined with the above terms to expand the search for studies that considered the impacts caused by different packaging materials, such as flexible or lightweight packaging. The terms *market*, *market-based*, and *displacement* were also combined with *LCA* terms to find studies that used market-based methods of allocating displacement of primary products and the related environmental impact. The term *not recyclable* was included in the search to find comparisons between packaging materials that are recyclable and materials that are explicitly not recyclable and used for the same purpose.

The results of the searches varied significantly by search terms. Overall, the search results included many studies that focused on different aspects of recycling, but few that combined recycling with quantitative LCA results focused on packaging. Searches including the terms *recyclable* or *recyclability* returned a low number of records with no studies that fit the above criteria. In contrast, searches including the term *recycling* resulted in more records that were relevant. The complex search term *recycling* and *packaging* and *life cycle assessment* returned the largest number of records, 1,143, yielding 8 studies that met all criteria. The searches for *packaging weight* and *packaging type* and *not recyclable* did not result in any additional relevant studies. One additional relevant study was identified using the *packaging weight* search term. A study previously identified during the recycled content review but not used for that attribute, met the inclusion criteria for the recyclable attribute. The analysis of the recyclable attribute is based

[4] Science Direct advanced search engine, last accessed September 2017: [http://www.sciencedirect.com/science/search]
[5] ACS Publications advanced search engine, last accessed September 2017: [http://pubs.acs.org/search/advanced]
[6] Wiley Online Library advanced search engine, last accessed September 2017: [http://onlinelibrary.wiley.com/advanced/search]

on a total of 18 comparative studies, 10 studies with quantitative environmental comparisons unique to the recyclability section, six identified in the Recycled Content section and two from the Compostable section.

Biobased

Terms used to identify articles for the biobased attribute include *modal*, *tencel*, *wood, paper*, *paperboard*, *kraft paper*, *corrugated cardboard*, *polyethylene terephthalate, PET*, *polylactic acid*, *PLA*, *poly(itaconic acid)*, *PIA*, *polyhydroxyalkanoates, PHA*, *polyethylene,* and *PE*. The scope of the search was initially limited to LCA studies that discussed the materials within the context of packaging and FSW. This yielded relatively few studies. The search was then expanded to include biobased chemicals used to make plastic packaging. These combined searches yielded a total of 41 LCAs for further consideration. Twenty-seven of the LCAs were selected for inclusion in this literature review.

Of the studies considered for inclusion, 10 were comparative reviews of LCA studies, which provided context for the differences between material types, while 15 studies provided quantitative results for environmental impact categories. Those 15 studies yielded a total of 459 comparisons including 238 comparisons of the same material with biobased or fossil feedstocks and 221 comparisons of biobased materials to a conventional material used in similar applications.

The biobased materials assessed by these studies include cellulosic materials like wood and plant fibers, seven plastics: PLA, polyhydroxyalkanoates (PHA), HDPE, low-density polyethylene (LDPE), laminated film (variety of feedstocks), PET, and TPS, as well as three chemical precursors: poly(itaconic acid) (PIA), purified terephthalic acid (PTA), and p-xylene. Note that PIA is a chemical precursor that is proposed for use as a desiccant or in the production of coatings for packaging. Although it is not currently in widespread use in packaging, because it came up in our literature review as a biobased chemical with potential packaging applications, we added it to our search terms.

Compostable

Terms used to identify articles for the compostable attribute include *life cycle assessment*, *compost*, and *packaging*, and variations thereof. Due to a low number of studies found initially, the search criteria were modified to include studies that focused on polymers used to create compostable packaging materials.

Additional search terms used included the combination of *compost*, *LCA*, and types of packaging materials (*boxes*, *films*, *bottles*, *sheets*, etc.). The cumulative search results yielded 10 studies that matched the revised search criteria and provided quantitative comparative results. Additional studies which provide valuable context for compostable packages/materials (e.g. reviews of LCAs on organic waste management) were reviewed in detail.

Food Service Ware

The same procedure employed during the literature review for packaging was used in the search for FSW products for the four attributes. LCA studies of FSW which include quantitative life cycle impact assessment results allowing comparisons between FSW with and without at least one of the four attributes were considered suitable for inclusion in the review. The review includes both peer reviewed journal articles and published reports. Several studies were reviewed but not included (Beauregard and colleagues, 2007; Jishi and colleagues 2013), either because the studies did not clearly define a functional unit or basis of comparison for their results, or because they did not contain sufficient information to understand the scope and quality of the results presented.

Terms used to search for studies include combinations of *life cycle assessment*, *food service ware*, *tableware, takeout,* and *dining*. Search terms for specific FSW products were also used and combined with *life cycle assessment* and specific attributes to create complex terms (see *Supporting Information A* for specific search terms used). The specific FSW product LCAs identified in the literature search are *cutlery, knives, forks, spoons, cups, lids, plates, dishes, napkins,* and *straws.* In addition to studies found using the search terms, additional relevant studies were discovered through inspecting the references cited in the initially identified studies. In total, 11 relevant studies were identified, which provided 654 comparisons for the recyclability attribute, 327 for the biobased/renewable attribute, and 363 for the compostability attribute. No LCA midpoint comparisons were identified for the recycled content attribute.

Creating Comparisons from the Literature

LCIA results were extracted from studies and recorded in a table. The impact categories explored in each study were aggregated into the impact categories previously described in the introduction. The categories include global warming, acidification, eutrophication (freshwater and marine), ozone depletion, smog formation, particulate matter formation (leading to human respiratory effects), human toxicity, ecosystem toxicity (ecotoxicity), fossil energy depletion, mineral depletion, water consumption, and land use. Some LCIA methods aggregate freshwater and marine eutrophication using normalized emissions; these are identifiable in the tables by a single ‘x’ in a combined cell for both forms of eutrophication.

LCIA results were summarized and compared by creating ratios of results for a packaging or FSW option with the scenarios including the environmental attribute of interest (i.e. recycled content, recyclable, biobased, or compostable) divided by the results for the conventional option that provides the same function. Specifically, for each impact category included in a study, the comparisons were calculated using the following equation:

$$\frac{[LCIA\ result\ for\ attribute]}{[LCIA\ result\ for\ conventional]}$$

Comparisons are made across packaging and FSW scenarios examined within a given study and not across studies as it is difficult to harmonize results to account for differences in system boundaries, data sources, allocation methods, and impact assessment methods. The ratios were recorded in a table (see *Supporting Information B*) and are the basis for the figures presented in the results section. Although some studies consider packaging systems of different sizes, all comparisons made are based on the same functional unit.

LCA inherently involves a considerable amount of data and modeling uncertainty. For this review, differences in comparisons are considered meaningful if the ratio of results is less than 0.75 or greater than 1.25. More specifically, we classified the ratios of an option with the attribute to an option without the attribute as follows:

- lower impact (<0.75)
- marginal decrease in impact (≥0.75 and <1.00)
- no difference (1.0)
- marginal increase in impact (>1.00 and ≤1.25)
- higher impact (>1.25)

The larger the ratio value, the greater the environmental impact of the material(s) being evaluated compared to the baseline material. These ranges are presented in the results sections and are used to interpret the comparisons found through the literature for each attribute.

In some cases, the LCIA results are presented as negative values, meaning they represent a net reduction in environmental impacts. This is usually a result of avoided burdens assigned to that scenario. For example, incineration of plastic waste may result in a negative GWP, as the energy recovered from the plastic may offset electricity or heat generated from more carbon intensive fossil-based sources. If a conventional packaging scenario results in a negative LCIA value, the normal ratio equation will not yield a result consistent with the previously introduced classifications. Instead, the percent difference between the two LCIA results is calculated, using the following equation:

$$1 - \frac{[LCIA\ result\ for\ conventional] - [LCIA\ result\ for\ attribute]}{[LCIA\ result\ for\ conventional]}$$

This approach yields the same results as the simpler ratio equation for two positive values and ensures that comparisons where the conventional option has a negative value will yield an outcome consistent with the impact classifications presented.

Each of the attribute result sections below discusses the scope of the literature for that topic, the results of the application of the ratios for comparison, and a discussion of the key findings for studies investigating the relevant attribute. The figures used to depict the findings of the review of quantitative comparisons provided in the literature show the results for all comparisons, allowing the reader to see when comparisons were deemed marginal (due to being less than a 25 percent difference), and break out results for each impact category allowing the reader to easily see which impact categories are more and less frequently studied in the literature. Our discussion takes the number of comparisons for each impact category as well as the marginal comparisons into account to provide a complete picture of the results available in published LCA studies.

Packaging Results

Recycled Content in Packaging

Scope

The analysis of the effect of using recycled content in packaging (not including FSW, which is presented separately) includes three studies that compare environmental impacts of primary and secondary production of the same material. Seventeen studies that compare the environmental impacts of two or more functionally equivalent packaging options (different materials) with different levels of recycled content were also identified and analyzed. A summary of the product type, materials used in packaging, functional unit, geographic scope, recycling allocation used, and results categories considered for each of these studies is documented in Table 1. The studies are organized by decreasing number of impact categories considered; thus, the study by Cleary (2013) is listed at the top of the table as it is the only study that considered all impact categories. Figure 3 illustrates the frequency with which packaging material, life cycle phase, or results category is represented within the studies included. Across all comparative packaging studies analyzed, a total of 136 packaging system scenarios were documented, allowing for a total of 793 distinct comparisons across all results categories.

Table 1. Scope of studies included in the packaging recycled content comparisons.

					Materials								Boundary					Result Categories												
Author and Year	*Packaging Type*	*Functional Unit*	*Geography*	*LCIA Method*	Plastic Resins	Biobased Resins	Laminated Paperboard	Paper / Cardboard	Glass	Steel	Aluminum	Composite / Other	Allocation Method [1]	Incl. Product Life Cycle	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Cleary, 2013	Glass bottles for wine and spirits	1 L of wine; 750 ml of spirits	Toronto	ReCiPe v1.02, IMPACT 2002+, TRACI 2	x		x		x				AB		x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
Zampori, 2014	Plastic and metal trays for poultry	1 tray w/ 1 piece poultry	Italy	ILCD midpoint, GHG protocol, CED	x						x		RC		x	x	x	x	x	x	x	x	x	x	x	x		x	x	x
Amienyo, 2014	Bottles and cartons for wine	0.75 L consumed	UK, Australia	CML 2001			x		x				RC	x	x		x	x	x	x		x	x		x	x	x	x	x	
Amienyo, 2016	Glass bottles and metal cans for beer	1 L consumed	UK	CML 2001					x	x	x		RC	x	x		x	x	x	x		x	x		x	x	x	x	x	
Amienyo, 2013	Glass and plastic bottles, aluminum cans	1 L packaged beverage	UK	CML 2001	x				x		x		AB	x	x		x	x	x	x		x	x		x	x	x	x		
Franklin Assoc., 2011	Resin	1 kg resin	N America	TRACI	x								RC		x			x	x	x		x	x	x	x	x	x		x	
Belley, 2011	Plastic trays for fruits and vegetables	1 tray to consumer	Canada	Impact2002+	x	x		x					RC		x		x	x	x	x		x	x		x	x		x		
Franklin Assoc., 2009	Plastic, aluminum, and glass bottles for water	1,000 gallons to consumer	US	TRACI	x	x							50	x	x		x	x	x	x		x	x	x	x	x	x			
Markwardt, 2016	Variety of plastic, metal, and glass food containers	1,000 L to retail	EU	German Fed. Env. Agency, CML, CED	x		x		x	x			50		x		x	x	x	x		x	x	x				x		
Pricewaterhouse Coopers, 2008	Cork and aluminum/ plastic closures for wine	1,000 0.75L bottle stoppers	UK	IPCC, ETH, WMO, CML	x						x	x	AB		x		x	x	x	x			x				x		x	
Kuczenski, 2012	Plastic bottles	PET to deliver 1L to consumer	US	CML, TRACI 2.0	x								AB		x			x	x	x			x	x			x			

Recycling allocation methods: RC = Recycled Content; AB = Avoided Burden; 50 = 50/50; MB = Market-based.
Studies ordered from most to least coverage of result categories

Table 1. Scope of studies included in the packaging recycled content comparisons, continued.

					Materials								Boundary					Result Categories												
Author and Year	*Packaging Type*	*Functional Unit*	*Geography*	*LCIA Method*	Plastic Resins	Biobased Resins	Laminated Paperboard	Paper / Cardboard	Glass	Steel	Aluminum	Composite / Other	Allocation Method[1]	Incl. Product Life Cycle	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Dhaliwal, 2014	Plastic and glass bottles for medical supplies	1 dose (96 ml) to patient	US, Global	ReCiPe and CED	x				x				MB		x		x	x				x	x	x	x		x			
Franklin Assoc., 2004	Various boxes, bags, and fill for mail-order goods	10,000 packages to consumer	US	none	x			x				x	50		x		x	x									x			
Franklin Assoc., 2008	Various containers and bags for coffee	10,000 oz to consumer	US	IPCC 2005	x			x		x		x	50		x		x	x									x			
Franklin Assoc., 2008	Cans, cups, and pouches for tuna	10,000 oz to consumer	US	IPCC 2005	x					x		x	50		x		x	x									x			
Krystofik, 2014	Multi-material inkjet cartridges	5 use cycles	US	ReCiPe, CED, and GWP	x								AB		x		x	x									x			
Arena, 2003	Plastic containers for liquids	1 kg resin	Italy	Several[3]	x								RC		x												x			
BIER, 2012	Various packaging and bottles for beer	1 L to consumer	N America, UK	GHG Protocol, PAS 2050					x	x	x		RC	x	x	x	x	x												
Dormer, 2013	Plastic trays for mushrooms	14 kg delivered to consumer	UK	PAS 2050	x								RC		x		x	x												
Mattila, 2011	Plastic, paper, cotton, and starch grocery bags	Bags for use for 1 year	Finland	Not specified	x	x		x					AB		x		x	x												
Percent of Total					85	15	15	20	35	25	25	20		25	100	10	85	95	55	55	35	50	60	35	55	40	70	35	30	10

Recycling allocation methods: RC = Recycled Content; AB = Avoided Burden; 50 = 50/50; MB = Market-based.
Studies ordered from most to least coverage of result categories

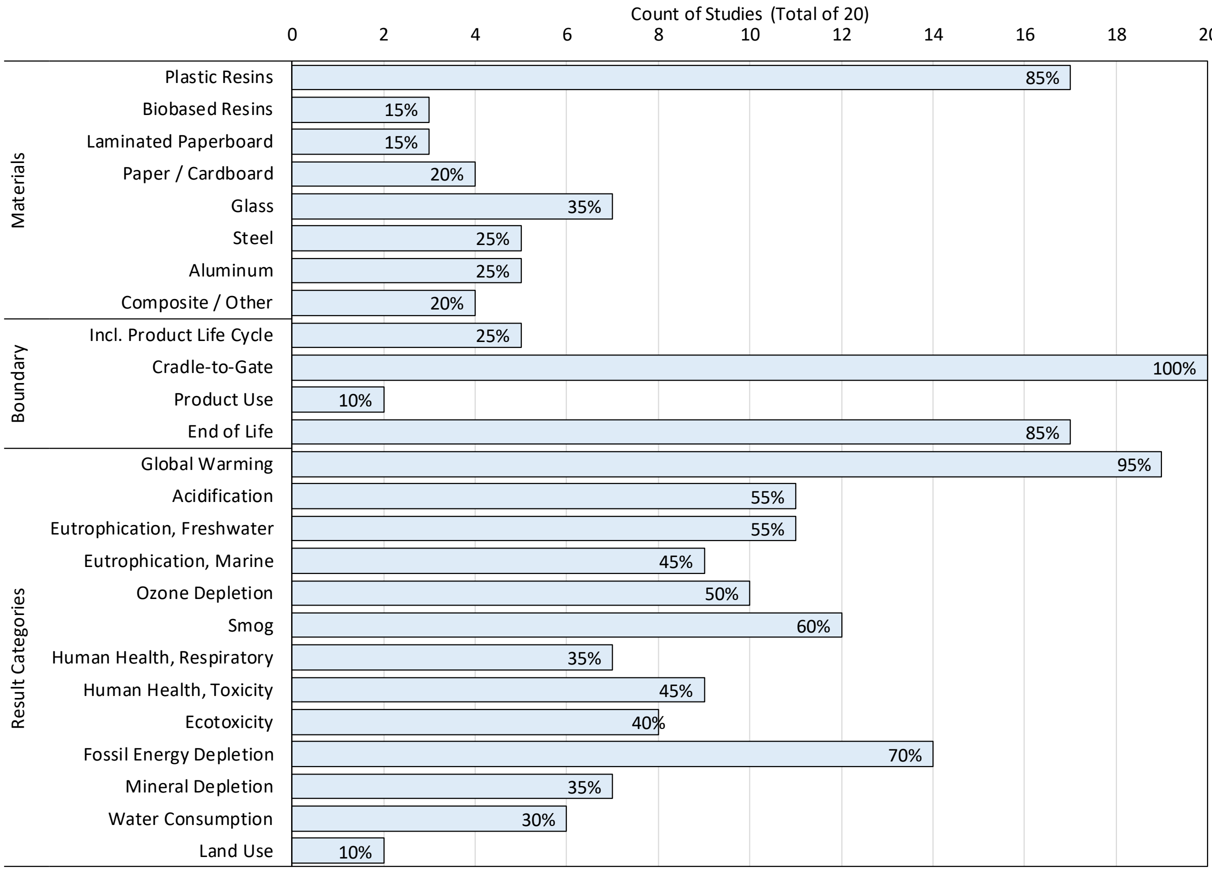

Figure 3. Scope of studies included in the packaging recycled content comparisons.

Materials with recycled content generally have lower environmental impacts than producing the same materials from primary feedstocks

When comparing production of plastic resin with 100 percent recycled content to plastic resin with 100 percent virgin content, three studies were identified that allowed for 5 comparisons across one or more environmental impact categories. A total of 30 comparisons were made. Across all impact categories, the recycled resin had significantly lower impacts in 27 of these comparisons, marginally lower impacts in two comparisons, and marginally higher impacts in one of the comparisons.

The only instances where recycled material was found to result in higher impacts involved eutrophication, and here the results were mixed with higher impacts in only one of three comparisons for eutrophication. The following case studies from the literature illustrate these results.

Kuczenski and Geyer completed an LCA of recycling of PET bottles collected in California's bottle redemption program and subsequent production of rPET pellets. The impacts of producing rPET pellets are compared to those of producing virgin PET pellets in addition to end-of-life scenarios including landfilling post-consumer PET bottles which are recovered for recycling in the rPET production scenario. Net impacts for in-state reclamation of PET bottles were significantly lower for the GWP, particulate matter, primary energy demand, acidification, and smog formation potential categories, ranging from 17 percent to 39 percent of impact levels compared to virgin PET pellets (including landfilling of PET bottles). Eutrophication potential impacts were higher based on the TRACI impact method due to wastewater emissions from the reclamation process. Eutrophication, smog, and acidification potentials as well as particulate matter results were sensitive to the assumptions used to allocate burdens for consumer drop-off collection, fuel efficiency for curbside collection, and the distance recovered bottles are transported to the reclamation facility (in-state vs out-of-state vs East Asia). Primary energy demand and GWP were also sensitive to consumer drop-off collection scenario assumptions.

An impact assessment performed on national average life cycle inventory (LCI) data for virgin and recycled PET and HDPE production that were collected by Franklin Associates and published in the National Renewable Energy Laboratory's US LCI database (NREL 2015) showed that total energy demand, smog formation, human toxicity, and ecotoxicity impacts for rPET and recycled HDPE (rHDPE) range from 2-22 percent of the virgin resin impact levels, 30-65 percent of virgin resin levels for global warming, acidification, eutrophication, ozone depletion, and particulate matter formation impacts, and 76-90 percent of virgin resin levels for water depletion.

A life cycle inventory for the recycling process of PE and PET liquid containers in Italy found that both rPET and recycled PE were energetically favorable in comparison to the virgin resins (Arena, Mastellone, and Perugini 2003). Gross energy consumption per kg of rPET ranged from 42-55 MJ compared with the 77 MJ per kg for the cradle-to-gate gross energy demand of virgin PET. Similarly, they calculated 40-49 MJ per kg recycled PE compared to 80 MJ per kg virgin PE resin.

In addition to the three studies comparing plastic resins, four studies provided comparisons of packaging systems with 100 percent recycled content material with 100 percent virgin material. Krystofik and colleagues (2014) compared inkjet cartridges with 100 percent recycled content PET and 100 percent virgin PET and quantified GWP and primary energy demand. They found that the cartridges with recycled content performed marginally better for both metrics with impact ratios of 0.90 and 0.85 respectively. Dormer and colleagues (2013) compare PET trays for mushroom packaging and found that a 100 percent RC PET tray has a lower GWP than a 100 percent virgin PET tray with an impact ratio of 0.32. Amieyo and colleagues compare 100 percent recycled content glass wine (2014) and beer (2016) with 100 percent virgin glass. They found marginally lower impacts for the recycled content glass for the global warming, acidification, ozone depletion, smog, human toxicity, ecotoxicity, and fossil energy demand categories, and equivalent impacts for the water consumption category.

Packaging with higher recycled content tends to be environmentally preferable to options using the same material with less recycled content

When comparing functionally equivalent packaging systems made of the same material but with different levels of recycled content, 16 studies were found and a total of 259 comparisons were made. Most comparisons (203, 74 percent) showed marginally decreased impacts across all impact categories for the packaging with higher recycled content, significantly had lower impacts in 52 (20 percent) comparisons, and significantly higher impacts in only one of the comparisons.

The only instance where packaging with higher recycled content was found to result in higher impacts involved ozone depletion, and this was a case where a single-serving beverage container including recycled content was compared to a virgin material multi-serve beverage container that would require significantly less material to fulfill the functional unit. These results are summarized in Figure 4 and are illustrated by the following case studies from the literature.

An LCA study on poultry product packaging included an assessment of an aluminum tray with either 30 percent or 100 percent recycled content (Zampori and Dotelli 2014). Shifting from 30 percent to 100 percent recycled content in the aluminum tray decreased most *production* impacts 60-80 percent. Considering the full life cycle, human toxicity potential from carcinogens decreases by roughly two-thirds, eutrophication potential and freshwater ecotoxicity potential each decrease by roughly half, and mineral depletion potential decreased by roughly a third. All other impact categories decrease by 5-20 percent.

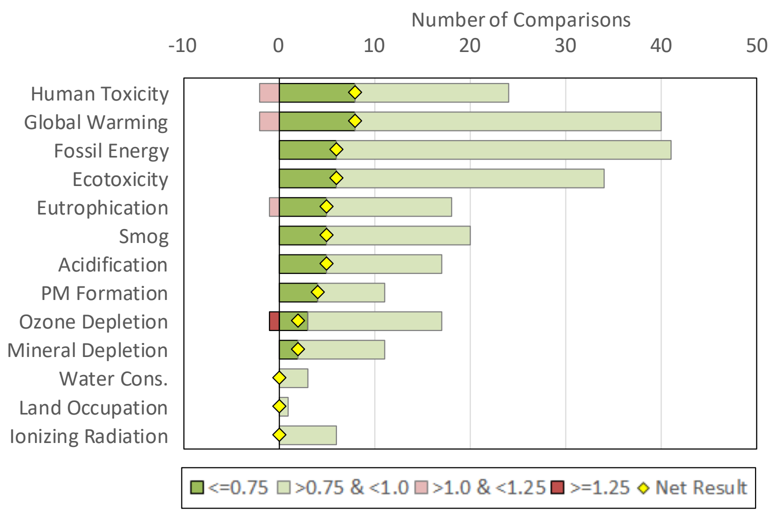

Figure 4. Comparison of packaging with higher recycled content to packaging of the same material with less or no recycled content. *Ratios reflect the result for packaging that has higher recycled content divided by the result for packaging of the same material that has less or no recycled content. Thus ratios <1 indicate packaging with higher recycled content performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicate packaging with higher recycled content performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

Dhaliwal and colleagues (2014) looked at packaging for global distribution of contrast media for x-rays and the effect of using ‘internally reworked glass cullet’ (pre-consumer recycled material) to reduce the impacts of a glass vial. Using a market-based allocation method,[7] the authors found increasing recycled content by 10 percent (from 10 to 20 percent or 20 to 30 percent) resulted in a 3-5 percent decrease in overall results across impact categories. Increasing glass cullet by 30 percentage points (from 30 to 60 percent) reduced total impacts an additional 8-14 percent.

A third study by Krystofik and colleagues (2014) conducted a sensitivity analysis to determine the effect of greater rPET content in inkjet cartridges. In comparison to inkjet cartridges with 100 percent virgin PET, incorporating 30, 50, 70, and 100 percent rPET into the cartridge resulted in 2, 4, 6 and 10 percent reductions in life cycle GWP of the cartridge, respectively. Similarly, life cycle cumulative energy demand for the cartridge was reduced 5, 8, 11, and 15 percent.

Seven additional studies evaluated a variety of packaging including red wine containers (Amienyo, Camilleri, and Azapagic 2014), coffee containers (Franklin Associates 2008a), food trays (Belley 2011; Dormer et al. 2013), shopping bags (Mattila et al. 2011), water bottles (Franklin Associates 2009), and packaging for e-commerce shipping (Franklin Associates 2004) confirm the trend of improved environmental performance with increased recycled content in glass, aluminum, PET, LDPE, expanded polystyrene (EPS), newsprint, kraft paper, corrugated boxes, and fiberboard packaging.

Packaging design, material choice, and weight are more important in determining environmental preference than recycled content

When comparing functionally equivalent packaging systems made of different material and with differing levels of recycled content, 15 studies were identified that allowed for 114 scenarios across one or more environmental impact categories. A total of 534 comparisons were made. Across all impact categories, the packaging with higher recycled content had significantly lower impacts in 113 of these comparisons and significantly higher impacts in 327 of the comparisons. In 93 of the comparisons, the impact results for the packaging with higher recycled content were not deemed to be meaningfully different.

When considering individual impact categories, the results comparing packaging systems made of a material with higher recycled content with a packaging system of different material with lower or no recycled content are mixed; see Figure 5. The top portion of the figure (a) shows all comparisons. The bottom portion of the figure (panel b) explores whether this finding was related to the choice of allocation method used by the studies, and it reveals that even taking only studies which use the recycled content (100/0) allocation method thus favoring the use of recycled content, the results are still mixed. This indicates that material differences and recycled content both significantly influence life cycle impacts. The following case studies from the literature are characteristic of these results.

Markwardt and Wellenreuther (2016) looked at various types of liquid food packaging systems used in the European market including two laminate board cartons, a laminate pouch, a glass jar

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[7] Market-based allocation is a form of economic allocation in which the burdens of inputs and are assigned to the co-product outputs in proportion to their relative market values.

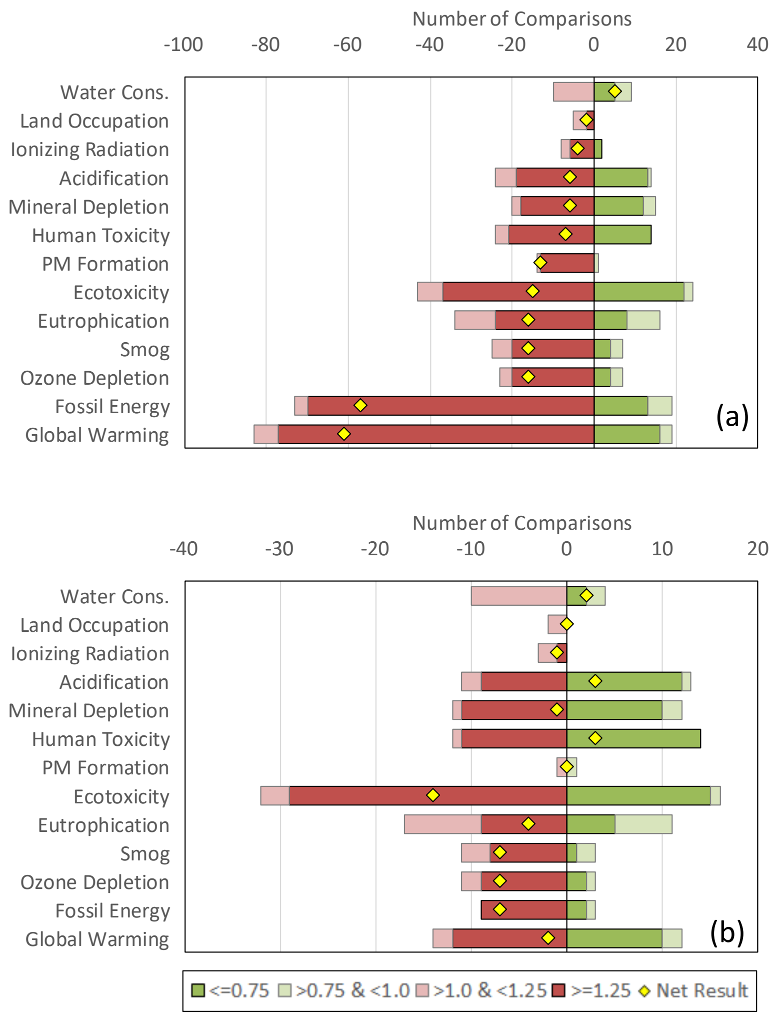

Figure 5. Comparison of packaging with higher recycled content to packaging of different material with less or no recycled content, all comparisons (a) and only comparisons using the recycled content (100/0) allocation method (b). *Ratios reflect the result for packaging that has higher recycled content divided by the result for packaging of a different material that has less or no recycled content. Thus ratios <1 indicate packaging with higher recycled content performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicate packaging with higher recycled content performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons. All panels sorted by net result of comparisons from panel a.*

manufactured with 59 percent recycled content, a steel can, and a plastic container.[8] Despite the recycled content in the glass jar, the glass packaging system had either the highest or second highest results for all the impact categories examined while the lightweight cartons performed the best. As shown in *Supporting Information B*, only the plastic container performed worse than the glass jar in the smog formation, aquatic eutrophication, and abiotic resource depletion categories.

Likewise, the Dhaliwal and colleagues (2014) study mentioned above also looked at the impacts of a virgin polypropylene (PP) vial to deliver contrast media. The baseline analysis showed that the results for the PP vial ranged from 24 to 55 percent of impacts of the glass vial with 20 percent recycled content. Since increasing the recycled cullet content of the glass vial content to 60 percent would only decrease total impacts by about 20 percent, the PP vial impacts still ranged from roughly 29 to 64 percent of the impacts for the 60 percent recycled content glass vial.

Several additional studies comparing heavier packaging such as glass bottles, rigid plastic containers, steel cans, and corrugated boxes with recycled content to lighter-weight virgin packaging such as laminate paperboard cartons, plastic laminate pouches, and paper or film mailing bags, find that the lighter-weight packaging outperforms the heavier packaging in all impact categories examined regardless of recycled content (Amienyo, Camilleri, and Azapagic 2014; Franklin Associates 2004, 2008b, 2008a). This finding is driven by the additional material extraction, processing, and manufacturing for heavier materials rather than differences in transportation-related impacts associated with the packaging options.

Other cases comparing packaging with and without recycled content also show mixed results. Amienyo and colleagues (2013) compared the impacts associated with one liter of carbonated drinks packaged in aluminum cans, glass bottles, and PET bottles in the UK. While the glass bottle with 35 percent recycled content had higher impact results than the virgin PET bottle in all categories except aquatic eutrophication and freshwater ecotoxicity, the 38 percent recycled content aluminum can had lower impact results than the virgin PET bottle for primary energy demand, aquatic eutrophication, ozone depletion potential, smog formation, freshwater ecotoxicity, terrestrial ecotoxicity, and abiotic resource depletion.

In a comparison of food packaging trays by Belley (2011), the virgin extruded polystyrene (XPS) tray impacts ranged from 20 to 87 percent of the PET tray made completely from recycled content. However, the XPS trays had about twice the acidification potential and human toxicity impacts as the molded pulp trays made completely from recycled content, largely due to processes associated with the virgin material production.

In the Zampori and Dotelli (2014) study, the 30 and 100 percent recycled content aluminum trays were compared to a virgin polystyrene (PS) tray. When looking just at tray production impacts, results for PS tray were 75-97.5 percent lower than the 30 percent recycled content aluminum

[8] Note that Markwardt and Wellenreuther (2016) does not state the recycled content of the other packaging options, however conventional steel and plastic often have recycled content. The authors did not respond to our request for more information.

tray. It is important to note that this study includes the use phase of cooking the poultry in an electric oven because the aluminum tray was specifically designed to be used in the oven and reduce cooking time (and therefore energy use) relative to cooking in a typical ceramic pan.[9] Even though the aluminum tray use phase impacts are 13 percent lower than for the PS tray, the high production impacts for the 30 percent recycled content aluminum tray resulted in higher life cycle impacts than the PS tray in six of the 16 impact categories considered. The differences for nine of the 16 categories were not considered meaningful. The impacts for the 30 percent recycled content aluminum tray were only lower than the virgin PS tray in the category of water depletion. When 100 percent recycled content aluminum is modeled, the aluminum impacts are not meaningfully different than the PS tray in seven of the 16 impact categories (global warming, ozone depletion, particulate matter, smog formation, terrestrial and marine eutrophication, and acidification), but considerably (60 percent) lower in water depletion. The 100 percent recycled content aluminum tray was still higher in freshwater eutrophication and carcinogenic human toxicity cancer effects. Zampori and Dotelli's Monte Carlo analysis[10] found the differences between 100 percent recycled content Al tray and virgin PS tray significant except in the case of ionizing radiation (human health and ecosystems), freshwater ecotoxicity, and resource depletion. The authors found that GWP results for these two packaging types would change in countries such as France where more of the electricity is sourced from renewable energy or nuclear sources, which would diminish the contribution of the use phase, possibly causing the PS tray to perform better than the aluminum tray even with 100 percent recycled content.

Summary – recycled content

Increasing the recycled content of a product generally results in a decrease in harmful environmental impacts. Three studies found that when comparing production of 100 percent recycled content material to the same material with 100 percent virgin content, the 100 percent recycled material incurs reduced environmental impacts over the lifetime of the material. However, However, some environmental indicators do not always follow this trend, such as eutrophication impacts which increase in some cases due to the additional processing to clean recycled feedstocks (Kuczenski and Geyer 2013). Other investigations demonstrate the reductions in life cycle impacts associated with using recycled material can vary considerably in magnitude – from 60-80 percent for aluminum packaging down to a few percentage points for inkjet cartridges made of PET (Zampori and Dotelli 2014; Dhaliwal et al. 2014; Krystofik, Babbitt, and Gaustad 2014). A key explanation for this variation is the inherent difference in production of varying types of virgin materials. For example, the production of aluminum is much more energy intensive than that of plastic resins, so replacing virgin aluminum with recycled content results in greater impact reductions than can be achieved through recycled plastics.

The studies examined suggest that it is not possible to infer environmental preference for a packaging of one material type over another solely based on recycled content, with packaging

[9] Use of a ceramic pan and washing was accounted for in the PS tray system but made insignificant contribution to overall results.

[10] A Monte Carlo analysis uses repeated random sampling of values within the uncertainty distribution associated with each input parameter to show the distribution of overall model results.

material and weight being determining factors. As an example, while it is common for glass products to contain recycled cullet, glass packaging tends to have higher impacts than virgin plastic containers (Markwardt and Wellenreuther 2016; Dhaliwal et al. 2014; Amienyo et al. 2013). For some packaging products, other aspects of the life cycle, such as cooking during the use phase for the aluminum tray, or electricity grid mix may have greater influence over which packaging material has lower impacts for a given product than whether the packaging includes recycled content.

Recyclable Packaging

Scope

A total of 18 LCA studies comparing recyclable packaging that met all criteria were found. Six of these studies are also included in the recycled content results of this review; another two are also included in the compostability results; and 10 are only relevant to recyclability, as these studies are either framed in terms of recycling rates or recycling as an end-of-life disposal option, rather than in terms of recycled content in packaging. Table 2 and Figure 6 provide a summary of the studies, which include 153 packaging scenarios and 960 unique comparisons across the impact categories considered. Plastic resins are considered in most studies (78 percent), while cork (for bottle stoppers) is the least represented material, appearing in only one study. Five studies consider more than one recycling rate for the same packaging format. The remaining studies assume various recycling rates for the different materials they consider. Likewise, four studies focus only on the end-of-life for the packaging and assume that other life cycle phases are equivalent. Additional studies were identified that discuss approaches for accurately reflecting the potential benefits of recyclability in LCA studies, but which do not provide quantitative results for packaging systems. These studies are cited in the discussion portion of this section.

The discussion of findings for comparisons between recyclable and non-recyclable packaging in this section is complicated by nuances in the definition of *recyclable*. In the Introduction section we provided a definition for recyclable packaging that requires the existence of infrastructure making it feasible to collect and reprocess post-consumer packaging (FTC, 2012). Thus, it is possible that a packaging format considered recyclable in one area may not be considered recyclable in another due to the lack of adequate collection services and/or reprocessing facilities. This fine point makes generalizing comparisons between recyclable and non-recyclable packaging options challenging. Thus, in the discussion that follows, we refer to packaging options such as laminate packaging as non-recyclable since sufficient infrastructure for economically efficient recycling does not exist in most areas in the U.S. Studies that consider such materials are indicated in the discussion below. Furthermore, most of the packaging options included in the studies we identified are recyclable and therefore most of the comparisons discussed in this section are between packaging that is recycled at its end-of-life and another packaging option that is recyclable, but which is either recycled at a lower rate or not recycled at its end-of-life. This approach to defining recyclable materials is also used for the comparisons discussed in the results for recyclable FSW.

Table 2. Scope of studies included in the recyclable packaging comparisons.

					Materials								Recyclability Considerations		Boundary			Result Categories												
Author and Year	*Packaging Type*	*Functional Unit*	*Geography*	*LCIA Method*	Plastic Resins	Bio-based Plastic Resins	Laminated Paper Board	Paper/Cardboard	Glass	Steel	Aluminum	Composite/Other	Includes multiple recycling rates for same material	Recycling Allocation Method	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Toniolo et al., 2013	Sliced meat	Tray with sealed lid, 0.54 L capacity	Italy	ReCiPe 2008 midpoint	x									RC/AB	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Cleary, 2013[1]	Wine; Spirits	1 L of wine; 750 ml of spirits	Toronto	ReCiPe v1.02, IMPACT 2002+, TRACI	x		x		x					AB	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
Oliveira and Magrini, 2017	Lubricant Oil (Plastic Containers)	1 tonne HDPE	Brazil	ReCiPe 2008 midpoint	x								x	AB	x	x	x	x	x	x	x	x	x	x	x	x	x	x		x
Rossi et al., 2015[2]	Dry packaging	EOL treatment of 1 kg of dry packaging material	Europe	IMPACT 2002+		x								AB			x	x	x	x		x	x	x	x	x	x		x	x
Humbert et al., 2009	Baby food	1 baby food meal	Europe	IMPACT 2002+, CML 2001	x				x					AB	x		x	x	x	x		x	x		x	x	x	x		x
Hottle et al., 2017[2]	Polymers for packaging	1 kg of polymer	United States	TRACI v 2.1		x								AB			x	x	x	x		x	x	x	x	x	x			
Xie et al., 2013	Aseptic packaging	1 ton post-consumption Tetra Pak waste	China	EcoIndicator99	x		x				x			AB			x	x	x	x				x	x	x	x	x		x
Franklin Assoc., 2009[1]	Bottled water	1,000 gallons to consumer	US	TRACI v 1	x	x							x	50	x		x	x	x	x		x	x	x	x	x	x			
Markwardt, 2016[1]	Liquid food	1,000 L to retail	Europe	CML 2002, IPCC 2007	x		x		x	x				50	x		x	x	x	x		x	x	x	x			x		
Xie et al., 2011	Milk (Aseptic packaging)	1000 L of milk	China	EcoIndicator 99	x		x				x			AB	x		x	x	x					x	x	x	x	x		x

[1] Also found in the recycled content section
[2] Also found in Compostability section
[3] Cradle to cradle product system (for 30 loops)
Recycling allocation methods: RC = Recycled Content (100/0); AB = Avoided Burden (0/100); 50 = 50/50.
Studies ordered from most to least coverage of result categories

Table 2. Scope of studies included in the recyclable packaging comparisons, continued.

					Materials								Recyclability Considerations		Boundary			Result Categories												
Author and Year	*Product Packaged*	*Functional Unit*	*Geography*	*LCIA Method*	Plastic Resins	Bio-based Plastic Resins	Laminated Paper Board	Paper/Cardboard	Glass	Steel	Aluminum	Composite/Other	Includes multiple recycling rates for same material	Recycling Allocation Method	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Marion, 2005	Cardboard packaging waste	1 tonne of cardboard packaging waste	Europe	CML 2002				x					x	AB	x		x	x	x	x			x		x	x	x			
PricewaterhouseCoopers, 2008[1]	Wine	1,000 0.75L bottle stoppers	UK	IPCC, ETH, WMO, CML	x						x	x		AB	x		x	x	x	x			x				x		x	
Ferreira et al., 2015	Packaging waste recycling system	Total 2010 packaging waste in Belgium	Belgium	Ecocosts 2012, Stepwise 2006,	x			x	x					AB			x	x	x	x			x		x					
Mourad et al., 2008	Milk (Aseptic packaging)	1000 L of milk	Brazil	Inventory based	x		x				x		x	AB	x		x	x						x			x	x	x	
Niero, M., & Olsen, S., 2016	Beer (Aluminum cans)	Containment of 1 hl of beer for 30 life cycles	UK	ILCD recommended LCIA methods							x		x	AB	x[3]		x	x							x		x			
Pasqualino et al., 2011	Beer, juice, water	1 L of beverage	Spain	Inventory based	x		x		x		x			AB	x		x	x									x			
Franklin Assoc., 2008[1]	Coffee	10,000 oz to consumer	US	IPCC 2005	x			x		x		x		50	x		x	x									x			
Franklin Assoc., 2008[1]	Tuna	10,000 oz to consumer	US	IPCC 2005	x					x		x		50	x		x	x									x			
Percent of Total Number					78	17	33	17	32	17	33	17	28	78	74	11	100	100	72	67	17	44	61	56	72	56	89	44	28	39

[1] Also found in the recycled content section
[2] Also found in Compostability section
[3] Cradle to cradle product system (for 30 loops)
Recycling allocation methods: RC = Recycled Content (100/0); AB = Avoided Burden (0/100); 50 = 50/50.
Studies ordered from most to least coverage of result categories

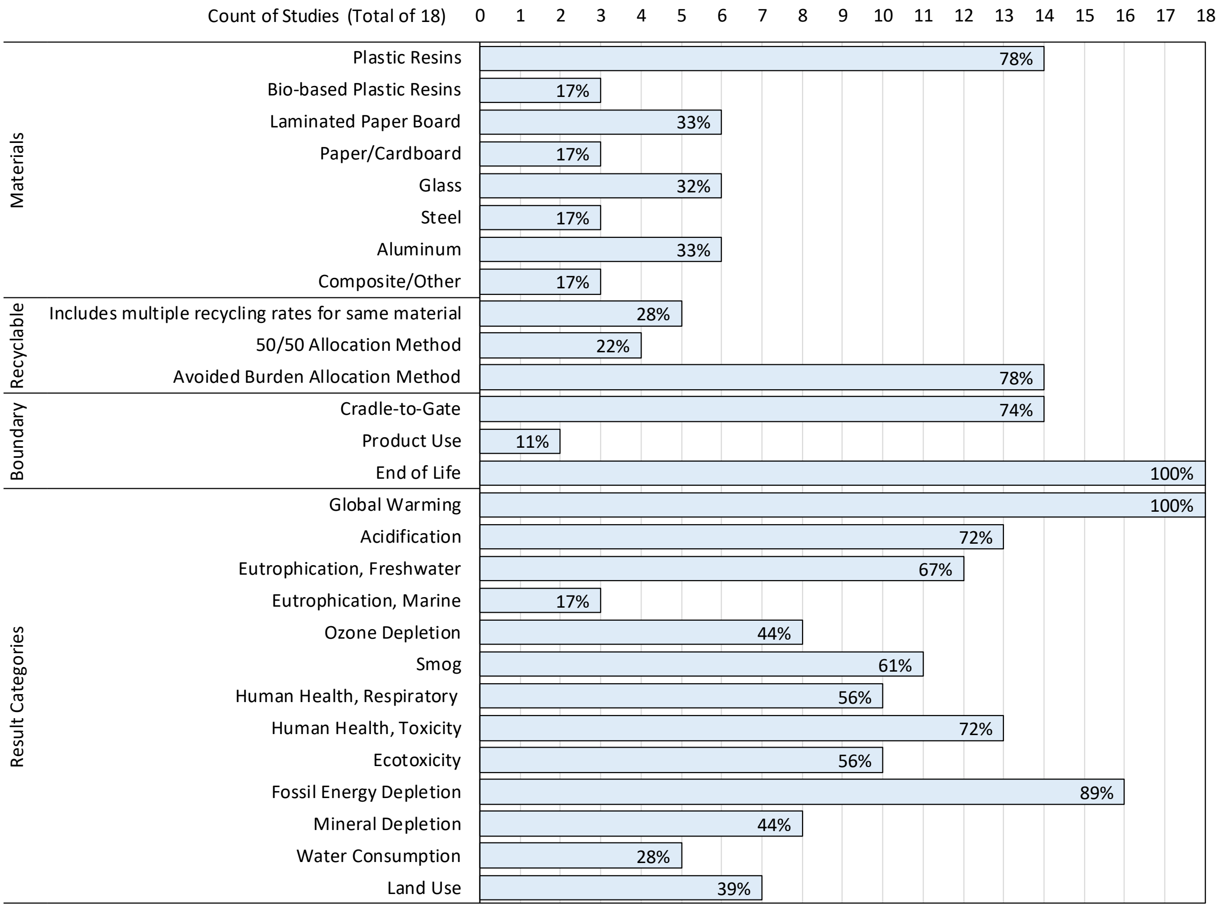

Figure 6. Scope of studies included in the recyclable packaging comparisons.

Material type and weight may be more important than recyclability in determining life cycle environmental burdens

When comparing recyclable packaging that is recycled with packaging of a different material that is not recycled or that is recycled at a lower rate (limited by access to collecting services or recycling services) or is not recyclable (limited by recycling technology), 10 studies were found and a total of 522 comparisons were made across two or more environmental impact categories. The packaging that is recyclable and recycled or has a higher recycling rate at the end-of-life had lower impacts (ratio <0.75) in 203 comparisons (39 percent); marginal decrease in impact (ratio between 0.75 and 1) in 58 comparisons (11 percent); marginal increase in impacts (ratios between 1 and 1.25) in 32 comparisons (6 percent); and higher impacts (ratio > 1.25) in 228 comparisons (44 percent), with one comparison resulting in a ratio of 1.0

Results of all comparisons between different materials for individual impact categories are mixed, as shown in Figure 7 (panel a). Overall, these comparisons suggest that packaging materials may be more important in determining a package's environmental footprint than recyclability. Comparisons of a given recycled material and a different recyclable material that is incinerated or landfilled results in lower GWP and fossil energy impacts. Examples include comparisons between recycled aseptic cartons against landfilled or incinerated HDPE and between refillable glass bottles that are recycled against landfilled PET bottles. In the case of the refillable glass comparisons, it is worth noting that the results depend on the return and utilization rate as indicated by the study conditions. Conversely, comparisons resulting in higher impacts for packaging recycled at higher rates are mostly between glass bottles and steel containers that are recycled and other materials that are recycled at lower rates; the exception is water consumption, where these same comparisons result in lower impacts.

Markwardt and Wellenreuther (2016) and Franklin Associates (2008a, 2008b) analyzed steel cans recycled at rates of 71 and 62 percent, glass containers recycled at a rate of 69 percent, and aseptic cartons and plastic containers (pots and canisters) recycled at rates no higher than 37 percent. Out of the 44 comparisons between the steel containers and the less-recycled materials, 13 result in impact ratios lower than 0.75 for the steel containers. These low ratios were found for comparisons of the eutrophication, ozone depletion, energy demand, and smog categories when comparing steel cans vs. plastic containers, because of emissions during incineration of the non-recycled portions of discarded plastic, and for the acidification, eutrophication, and human toxicity categories when comparing steel vs. glass containers, due to the higher impacts during glass production. Comparisons between steel containers which are recycled at a higher rate and aseptic cartons which are recycled at a lower rate always result in higher impacts for the steel containers.

The two studies mentioned above also analyze laminate packaging, which we consider not recyclable for the purposes of this review as they are not currently recycled in any appreciable amount. There are 56 comparisons from these studies between recyclable materials (steel, glass, aseptic carton, and HDPE packaging) and non-recyclable materials (laminate packaging). Fifteen of these comparisons result in a rate lower than 0.75. However, these low ratios for recyclable packaging are of comparisons between recycled carton-based packaging and non-recyclable laminate pouches, where the carton packaging performs better for all impact categories considered. The rest of the comparisons are between recycled steel, glass, and plastic containers vs. laminate packaging. 32 comparisons result in ratios greater than 1.25, for the global warming, acidification, eutrophication, ozone depletion, human toxicity, energy demand, and mineral depletion categories, despite the high recycling rate of glass and steel. The ratios for the comparisons between recyclable and non-recyclable packaging are shown in Figure 7 (panel c).

It should be noted that Markwardt and Wellenreuther (2016) and Franklin Associates (2008a, 2008b) use the 50/50 allocation method, which assigns lower benefits to recycling than the avoided burden allocation method. To test whether these results are related to the allocation method used, Figure 7 (panel b) shows only results of studies that use the avoided burden allocation method. Results in this part of the figure are still mixed. This suggests that although the avoided burden allocation method tends to assign higher benefits to recyclable materials at the end-of-life than studies that use the 50/50 allocation method, using this method does not guarantee a low ratio for the more recyclable material. Indeed, even if only the comparisons that use the avoided burden method are counted, 39 percent of the comparisons still result in ratios higher than 1.25. The following comparisons from the literature are representative of these results.

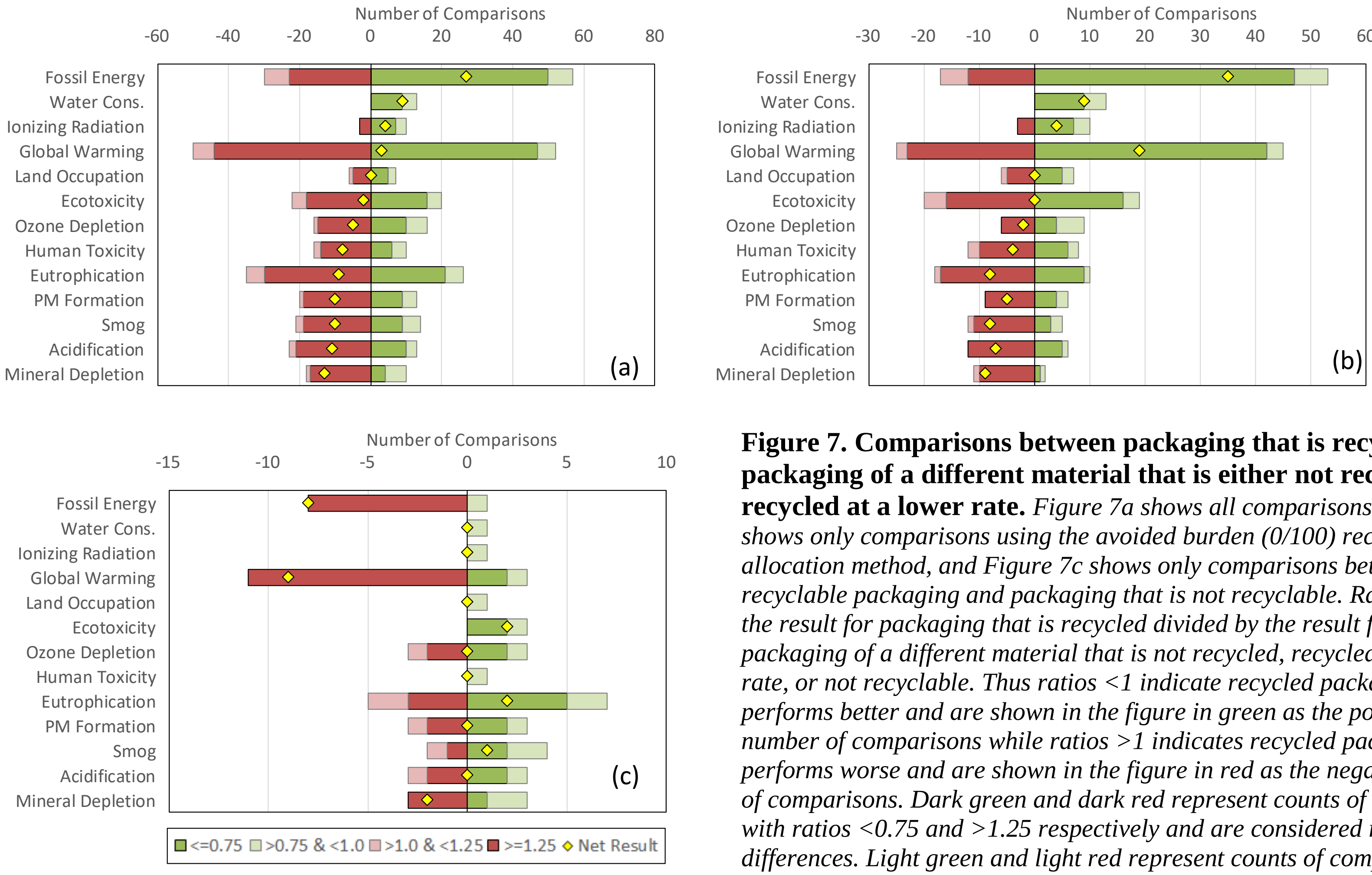

Figure 7. Comparisons between packaging that is recycled and packaging of a different material that is either not recycled or recycled at a lower rate. *Figure 7a shows all comparisons, Figure 7b shows only comparisons using the avoided burden (0/100) recycling allocation method, and Figure 7c shows only comparisons between recyclable packaging and packaging that is not recyclable. Ratios reflect the result for packaging that is recycled divided by the result for packaging of a different material that is not recycled, recycled at a lower rate, or not recyclable. Thus ratios <1 indicate recycled packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates recycled packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-1.0 and 1.0-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons. All panels sorted by net result of comparisons from panel a.*

Toniolo and colleagues (2013) compared similar trays used for packaging sliced meat across the following scenarios: a recyclable tray that is recycled at a rate of 33.5 percent and a tray that is not recyclable. The recyclable tray is composed of a mono-material film, while the non-recyclable tray is composed of a multilayer higher barrier co-extruded film. They find the recyclable tray performs better in marine eutrophication, freshwater ecotoxicity, and marine ecotoxicity potentials and results for other metrics are inconclusive (ratios between 0.88 and 1 with most between 0.95 and 1). These results are driven by the avoided burdens from recycling the single layer tray, as impacts during production and use of both trays are similar.

Cleary (2013) compared five types of one-liter wine packages: single-use conventional glass bottles, lightweight glass bottles, virgin PET bottles, aseptic carton, and refillable glass bottles. The study also compared four types of 750-ml spirit packages, made from the same materials as the wine containers except for aseptic cartons, which are not used for spirits. All the glass bottle options are recycled at 69 percent, PET bottles are recycled at a rate of 34 percent, and the fiber portion of the aseptic carton, which is three-quarters of the carton by mass, are recycled at a rate of 29 percent. They assume the materials not recycled are landfilled. Refillable glass bottles are assumed to be refilled 14 times and have return rates like Ontario, Canada’s deposit return system for refillable beer containers (99 percent). Our review compares the glass containers against the PET bottles and the aseptic cartons which are recycled at a lower rate. Refillable glass bottles resulted in the lowest environmental impacts for all 11 impact categories. Sensitivity analysis on number of reuses indicates that with ten refills, emissions from refillable glass bottles would increase but they would remain lower than for the single use options. For the single use packaging options, recyclable glass bottles have higher impacts than the less-recycled PET bottles and aseptic cartons for all categories except water depletion and freshwater ecotoxicity. In the case of water depletion, the recyclable glass bottle shows a benefit compared to the aseptic carton, while in the case of freshwater ecotoxicity the glass bottle shows a benefit compared to the PET bottle. This is because 57-68 percent of all impacts occur during material production, depending on the material. The avoided burdens from glass bottle recycling are not enough to offset production impacts. For example, the GHG emissions avoided by recycling reduce impacts by approximately 10 percent for the single use glass and PET bottles. In other words, the difference in impacts between recycled and virgin glass production is less significant than the difference between glass production and production of the PET bottles and aseptic cartons. We also noted that impacts were correlated with packaging weight.

PricewaterhouseCoopers (2008) performed an analysis of wine bottle closures made from 65 percent primary and 35 percent secondary aluminum, virgin LDPE/HDPE plastic, and cork stoppers. The analysis considered all the components necessary for sealing 0.75-liter water bottles. Aluminum closures are modeled as recycled at a rate of 32 percent while the plastic closures are recycled at a rate of 19 percent and the cork stoppers are landfilled. Life cycle results show that the aluminum closures perform considerably worse than the cork or plastic for GWP, acidification potential, and smog; similarly, for eutrophication potential and primary energy use; and better for water consumption. Aluminum and plastic closures perform similar during their production phases, while cork requires significantly more water during production. Aluminum closures result lower water consumption impacts during the bottling stage due to the use of polyvinyl chloride (PVC) covers by both cork and plastic closures, which are not used for the aluminum closures.

Humbert and colleagues (2009) compared glass and plastic baby food jars. The authors assumed the glass jars are recycled at a rate of 86 percent, while the plastic jars are recycled at a rate of 40 percent, and the unrecycled material is landfilled. The more highly recycled glass jars perform worse in the global warming, eutrophication, ozone depletion, human toxicity, ecotoxicity, mineral depletion, and land use categories. Results for acidification, smog formation, and energy demand also suggest the glass jars perform worse although the ratios for these comparisons represent marginal differences (1.00 – 1.25). The better performance for plastic jars despite their lower recycling rate is due to lower impacts from producing plastic and reduction of mass which reduces distribution impacts to the lighter packaging.

Pasqualino and colleagues (2011) compared the GWP and energy demand of aluminum cans, glass bottles, PET bottles, HDPE bottles, and aseptic cartons of various sizes for juice, beer, and water. In Figure 7 (panel a) we include 104 comparisons between containers that are recycled at EOL and containers (of the same volume) made of a different material that are assumed to be landfilled or incinerated at EOL. This is possible because the study includes recycling, landfilling, and incineration scenarios for each material. The authors assumed high recycling rates: 93 percent for aluminum, 88 percent for glass, 76 percent for PET and HDPE, and 75 percent for aseptic cartons. The results of these comparisons are mixed. Glass containers that are recycled perform worse when compared with aseptic cartons, PET and HDPE of the same volume that are landfilled or incinerated (ratios > 1.25). Likewise, aluminum cans and HDPE containers that are recycled perform worse than landfilled aseptic cartons (ratios > 1.25) due to the high energy requirements for aluminum reprocessing. However, recycled aluminum performs better than incinerated aseptic carton and considerably better than landfilled or incinerated PET containers (ratios <0.75). In most cases where recycled materials perform worse than materials that are landfilled or incinerated, the recycled packaging is also heavier. On the other hand, PET bottles and aseptic cartons that are recycled always result in significantly reduced environmental impacts when compared to landfilled glass or aluminum containers (80 comparisons with ratios <0.75).

The benefit of recyclable packaging depends on its role in reducing production of primary material

Recyclability is beneficial to the extent it displaces production of more energy intensive primary material. In the recycled content review, we found that for most LCA metrics, secondary (recycled) material generally has lower impacts than the same material produced from primary feedstocks. This is mainly due to the smaller amount of energy required to produce secondary materials compared with primary materials. The exceptions are impacts related to features of the secondary material supply chain, such as eutrophication and water use impacts associated with washing and wastewater emissions from the reclamation process of PET for recycling.

However, an increase in the recycling of a material does not necessarily result in an equal displacement of primary material. An increase in supply of a material may cause the market price of that material to fall – and the lower price can result in increased demand, although not necessarily in an equal amount. Indeed, recycling may serve to increase overall material use by providing industry with lower-cost feedstocks, depending on the price elasticities of the materials involved.

Several LCA studies have demonstrated that the displacement of primary material production is dependent on market factors and that the amount and type of primary material displaced are critical in determining the environmental benefits of recycling, and by extension, recyclability. (Ekvall 2000b; Frees 2008; Gala, Raugei, and Fullana-i-Palmer 2015; Geyer et al. 2016; Zink, Geyer, and Startz 2016). For example, Zink and colleagues (2016) show that different assumptions about displacement rate have the potential to reverse the outcome of an LCA. Ekvall (2000b) proposes a model for open loop recycling that includes the price elasticities of supply and demand of secondary materials to estimate the changes in the market due to the introduction of additional secondary material. Gala and colleagues (2015) propose a formula that calculates the environmental credits of material recovery in waste management by assuming displacement of the market average mix of primary and secondary materials currently in use. This approach is like the way avoided burdens from electricity production are estimated based on the local grid mix. Zink and colleagues (2016) predict changes in primary and secondary material using price elasticities to estimate responses to price changes. They find that when displacement is low, recycling may even increase environmental impacts.

There are relatively few LCA studies that attempt to estimate the displacement of primary material production. This is not surprising, as the ISO 14040:2006 LCA guidelines do not consider market factors within the scope of an LCA study (with the exception using economic allocation when defining the outputs of multi-product processes). The lack of current, regional or market-specific information regarding the substitution of primary with secondary material for different types of packaging materials also make their inclusion in LCA studies difficult. The availability of market mix or price elasticity information varies according to which material is being considered, and even if available may be highly uncertain. For example, elasticities for a single material fluctuate depending on the location or might have changed significantly since the data was collected. Ekvall (2000b) reports that elasticities of old paper and newsprint might range from 0.06 to 1.7, which greatly influences the results of LCA studies of paper products.

Due to these difficulties in using market data, the shared burden (or 50/50) allocation approach may be used when one-to-one displacement of primary material is not a viable assumption. Ekvall (2000b) suggests that the 50/50 allocation method results in a rough approximation of the case when 50 percent of recycled material replaces virgin material, and 50 percent replaces other recycled material. This effectively means that only 50 percent of the recycled material displaces primary production. This suggests that the comparisons that use the 50/50 allocation method included in Figure 7 (panel a), mixed with avoided burden comparisons, may represent a better estimate of market conditions than the purely avoided burden comparisons in Figure 7b.

Results are mixed for study that estimated the displacement of primary material production

We identified only one LCA study that provided quantitative comparative results for packaging and that considered the role of market effects in determining the amount of primary material displaced. This study is discussed below (Meylan and colleagues, 2014), but is not included in the comparative figures as it does not present environmental results in a way that allows a straightforward mapping to conventional impact assessment categories. However, the inclusion of market effects by Meylan and colleagues (2014) allow their results to highlight the tradeoffs between environmental and economic considerations for recycling of packaging. Additionally, two other studies that estimate primary material displacement, but which do not pertain specifically to packaging (Frees, 2008; Muñoz and colleagues, 2004), are also discussed in the following section.

Meylan and colleagues performed a two-part study on MSW in Switzerland and focused on waste glass-packaging. Part I of the study develops several future states to which the current MSW situation could transition to, based on a historical analysis of drivers that shaped the Swiss waste glass-packaging disposal system (Meylan and colleagues, 2013). This analysis resulted in 18 scenarios, which are described as combinations of overarching goals for MSW, policies and fees adopted to achieve those goals, glass cullet collection and processing schemes, internal market conditions, and external constraints. The 18 scenarios can be grouped into three main categories: 1) all cullet is exported to foreign (European) glass packaging factories; 2) all cullet is domestically downcycled into foam glass; and 3) all cullet is either recycled back to glass packaging, or downcycled to foam glass (high grade downcycling) or to sand substitute (low grade downcycling). The scenarios within each category share vary in the specific collection schemes, recycling rates, and policy objectives. In part II Meylan and colleagues (2014) performed a hybrid LCA analysis of the scenarios resulting from part I. The life cycle inventory used is based on a national Swiss input-output (IO) model, which was disaggregated and complemented with additional data sources to create custom economic sectors to describe waste glass-packaging disposal. The IO approach allows modeling of internal Swiss demand, imports and exports of waste glass-packaging, and provides both economic and environmental impact estimates. The impact estimates obtained from this model are compared to the MSW situation in 2009 (base year for the studies).

There are three main takeaways from these studies regarding the recyclability of waste glass-packaging. First, from a policy perspective, part I of the study concluded that financial incentives set by regulations to achieve specific waste disposal goals (such as favoring recycling over downcycling) can be counteracted by constraints outside of the policy makers' control. These constraints can be part of the waste glass-packaging disposal system (e.g., costs of waste glass-packaging collection) or external to it (e.g., commodity prices). Secondly, from an environmental perspective, part II indicates that scenarios where glass cullet undergoes domestic high grade downcycling result in the lowest impacts; foreign low grade downcycling results in the highest impacts; and 100 percent closed-loop recycling of glass-packaging is in between these scenarios, though still better than the Swiss situation at the time of the study. Domestic downcycling of glass-packaging into foam glass results in lower environmental impacts than recycling back into glass-packaging due to the lower energy required for downcycling, as well as the displacement of domestic extruded polystyrene (XPS) insulation production. Finally, although 100 percent recycling of the glass cullet does not produce the best environmental results, it produces the highest gross value-added scenarios, as recycling produces higher value-added than downcycling and no XPS production is displaced. Indeed, only the scenarios with recycling and high grade downcycling achieve higher value-added results than the baseline situation. These results illustrate the interconnections between environmental and economic considerations for recycling waste glass-packaging, as well as the limits of policy in shaping MSW transitions in Switzerland.

Results are mixed for studies that consider recyclable materials, but which do not focus specifically on packaging

Frees (2008) used a method similar to the one proposed by Ekvall (2000b) to evaluate the displacement of primary aluminum due to aluminum scrap. This study concludes that price of aluminum scrap is inelastic given that secondary material covers only 30-40 percent of aluminum demand and thus there must be primary production. As the market is constrained by the availability of secondary material, the study suggests that the avoided burden method with full displacement of primary material is appropriate for aluminum recycling. This is a noteworthy finding since it bolsters the results reported by LCA studies that use the avoided burden method for analyzing the environmental impacts of aluminum production and recycling without including a market analysis, such as Nieron and Olsen (2016).

Muñoz and colleagues (2004) analyzed different integrated waste management system scenarios for the Gipuzkoa department in Spain, which assumed a decrease in landfilling and increase in recycling of the disposed materials. This management system includes recycling of different types of materials for which they assumed different displacement ratios. For example, they assumed a 1:0.75 ratio of recovered paper to displaced virgin pulp, and ratios of 1:0.078, 1:0.031, and 1:0.022 for displacement of mineral nitrogen, phosphorous, and potassium in fertilizers. For materials where information was lacking, such as plastic and metal wastes, they used a 1:1 displacement ratios. For all waste management system scenarios that included recycling, a net environmental benefit was observed, despite high energy consumption for collection and transport of some materials. However, and in contrast with the finding by Frees (2008) for aluminum, the authors explicitly note that environmental benefits may be overestimated for those materials which assumed 1:1 displacement ratios.

When assuming one to one displacement of primary material, recycling is generally preferable to landfilling and incineration for a given packaging material

When comparing recyclable packaging that is recycled at its end-of-life with packaging of the same material that is not recycled or packaging of the same material that is recycled at a lower rate, 11 studies were found and a total of 438 comparisons were made across two or more environmental impact categories. The packaging that is recycled or has a higher recycling rate at the end-of-life had lower impacts (ratio <0.75) in 299 comparisons (68 percent); marginal decrease in impact (ratio between 0.75 and 1) in 71 comparisons (16 percent); marginal increase in impacts (ratios between 1 and 1.25) in 30 comparisons (7 percent); and higher impacts (ratio > 1.25) in 32 comparisons (7 percent), with six comparisons resulting in a ratio equal to 1.0.

Six studies included in this section of the review contained comparisons between materials that are recycled, and the same materials being exclusively landfilled or incinerated (Ferreira et al., 2017; Hottle, Bilec, & Landis, 2017; Rossi et al., 2015; Xie, Qiao, Sun, & Zhang, 2013; Pasqualino, Meneses, & Castells, 2011; Marion, 2005). These comparisons encompass various types of packaging materials, but overall results indicate that recycling generally performs better than either landfilling or incineration for the same packaging material for most impact categories. These results are shown in Figure 8. In most cases, this is because the credits from avoided primary production due to the recycling of the material are often greater than the energy credits assigned to incineration of the material and greater than the zero credits assigned to landfilling. A few examples of these patterns are discussed below, as well as some exceptions.

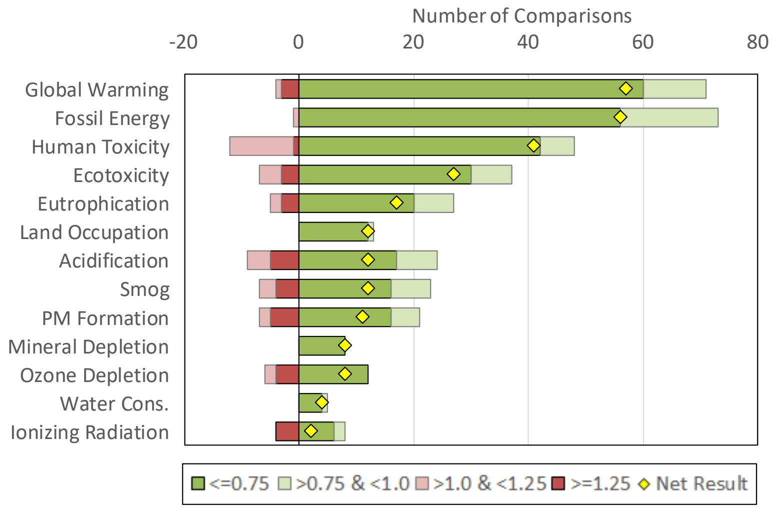

Figure 8. Recyclable packaging that is recycled vs. recyclable packaging of the same material that is not recycled or recycled at a lower rate. *Ratios reflect the result for the packaging that is recycled divided by the result for the packaging of the same material that is not recycled or recycled at a lower rate. Thus ratios <1 indicate more highly recycled packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates more highly recycled packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

Hottle and colleagues (2017) compare recycling to landfilling of six different packaging materials: PET, HDPE, LDPE, bioPET, bioHDPE, and bioLDPE. For all materials, GWP and energy impacts are lower for recycling over landfilling, due to the offsets of virgin material production assigned to recycling. However, environmental impact ratios are greater than one for the acidification, eutrophication, smog, and ozone depletion categories for recycling as compared to landfilling of these materials. In this study, the recycling of these materials is modeled as being shipped to China; this represents 610 km and 11,600 km of ground and ocean freighter transport, respectively, as that this was the actual path taken for these materials given they were produced and used in Arizona. The transportation of these packaging materials overseas is the main cause of the high impacts in the previously mentioned categories as compared to landfilling. Sensitivity analysis on shipping distance, which assumes there is no ocean transport and materials are recycled within the ground transportation distance, results in significant reductions in emissions for recycling, such that it is either comparable to or better than landfilling in the previously mentioned impact categories.

Marion (2005) compares mixed disposal of paperboard packaging waste (43 percent landfill, 4 percent incineration with energy recovery, 53 percent recycling, which represents current disposal methods at the time of study publication) with both 100 percent landfilling and 100 percent incineration disposal. The comparisons for this study show that the disposal with recycling is slightly preferable to landfilling (ratio <1) for the smog and energy demand categories, highly preferable for GWP (ratio <0.75), and comparable in the acidification, eutrophication and ecotoxicity categories (ratios ~1). However, recycling does not perform better than incineration in this study. The mixed recycling option performs slightly worse (ratio >1) than the incineration option for global warming, acidification, eutrophication, smog, and energy demand categories, and considerably worse for the ecotoxicity category (ratio ~1.8). These favorable results for incineration are due to the credits generated through energy recovery from the incineration process. These credits result in avoided burdens from other electricity generation using the average grid mix. A sensitivity analysis where renewable energy sources are modeled results in the mixed disposal option that includes recycling being preferable in all categories to the 100 percent incineration option.

Xie and colleagues (2013) consider two scenarios for recycling of aseptic packaging composed of 75 percent fiber, 20 percent PE, and 5 percent aluminum foil. In the first scenario, only the fiber portion is recycled, while the rest is landfilled; the second scenario considers separation and recycling all three components. These two recycling scenarios are compared against landfilling and incineration of the entire packaging. The recycling scenarios perform better than landfill in most categories except acidification and GWP where the energy required for recycling process for the carton leads to higher emissions than the landfilling of the entire material. In contrast, recycling performs better than incineration of the packaging only in the land use, mineral depletion, and primary energy demand categories, while performing worse in the global warming, acidification, human toxicity, and ecotoxicity categories, because of the higher credits assigned to incineration for avoided energy consumption than to recycling for avoided primary material displacement.

When assuming one to one displacement of primary material, a higher recycling rate is generally preferable for a given packaging material unless the secondary material recovery and recycling becomes very inefficient at higher recovery rates.

Five studies quantitatively analyzed the effects of different recycling rates for the same materials (Marion 2005; Mourad et al. 2008; Franklin Associates 2009; Niero and Olsen 2016; Oliveira and Magrini 2017). These studies were used to compare the environmental impacts of higher versus lower recycling rates presented by each study, for example Marion (2005) compares a recycling rate of 80 percent against lower recycling rates of 35, 53, 60, and 70 percent, one at a time. In general, higher recycling rates are preferable to lower recycling rates for the same packaging material.

Mourad and colleagues (2008) and Marion (2005) focus their analysis on packaging made mostly from paper (aseptic packaging and paperboard packaging waste, respectively). Mourad and colleagues compare aseptic packaging at recycling rates of 2 and 22 percent for the global warming, human toxicity, energy demand, water depletion, and land occupation categories. Marion has scenarios for 35, 53, 60, 70, and 80 percent recycling and results for the global warming, acidification, eutrophication, smog, human toxicity, ecotoxicity, and energy demand categories. In both studies, the portion of waste not recycled is landfilled. The comparisons based on both these studies resulted in ratios between 0.75 and 1 for all impact categories and recycling rates. The only exception was for GWP for the comparison in Marion between 80 percent and 35 percent recycled paperboard waste, where the ratio was 0.7. Ratios vary both between recycling rates and categories, but in broad terms, every 20 percent increase in paper recycling rate corresponds to approximately 10 percent improvement in most impact categories, with diminishing returns as recycling rates increase.

Oliveria and colleagues (2017) analyzed various disposal options for HDPE based lubricant oil plastic containers in Brazil which combined landfilling, incineration, and recycling in different amounts. Four scenarios were compared: one where 50 percent of the material was recycled and the remaining 50 percent landfilled; another where 50 percent of the material was recycled and the remaining 50 percent incinerated with energy recovery; a third scenario with a mix of 16 percent recycling, 68 percent landfilling, and 16 percent incineration; and a fourth scenario with a mix of 16 percent recycling and 84 percent landfilling. This study considered all the impact categories except water consumption. The two disposal options with 50 percent recycling performed considerably better than the disposal scenarios with lower recycling rates (ratios <0.75). As with previous studies, these results are driven by avoided burdens in the recycling scenarios; particularly for global warming and ecotoxicity in the 50/50 recycling/landfilling scenario, and for land occupation, human toxicity, and eutrophication categories for the 50/50 recycling/incineration scenario.

Finally, Niero and Olsen (2016) compared various recycling rates for aluminum beverage cans, using both closed-loop (i.e., can-to-can) and open-loop (i.e., mixed aluminum packaging-to-can) recycling pathways for the cans for a span of 30 life cycle loops. For each pathway, rates of 75 percent recycling were compared against rates of 55 and 65 percent. The comparisons between the 75 and 65 percent recycling rates result in ratios between 1.0 and 0.75 for the GWP and energy depletion categories, while the comparison between the 75 and 55 percent recycling rates result in lower impacts for global warming, energy depletion, and human toxicity (ratios <0.75). Additionally, impacts from closed-loop recycling are slightly lower for most categories, with GWP being considerably lower. However, while overall impacts decrease at higher recycling rates, additional materials (such as manganese) need to be added in the recycling process at higher amounts per kilogram of material recycled, which imposes a limit to the rate that aluminum cans can be recycled.

Taken in aggregate, the results from the comparisons of different recycling rates show that higher recycling rates result in lower environmental impacts for the material recycled. However, the various materials' environmental impacts seem to benefit differently from the increased rates, with aluminum and HDPE showing greater improvements than paperboard packaging for similar recycling rate increases. The main reason for these results are the avoided burdens for recycling, with higher recycling rates credited with higher avoided burdens. It is important to note that these comparisons usually assume that the material is transported to adequate recycling facilities and that enough of these facilities are available for the different materials to be recycled at different rates. Additionally, not all the studies consider material losses during collection or the possibility of contamination of the recyclate. These issues are considered below.

One-to-one displacement of primary material production is not a realistic assumption for recyclable packaging materials.

Studies summarized in the preceding discussion about the benefits of recyclable packaging often assume one-to-one displacement of primary material via recycling activities. One-to-one displacement of primary material by recycled material is likely not a realistic assumption, yet as demonstrated by the studies presented in this section, this assumption is common in the literature. Results of studies that assume 100 percent displacement of primary material by recycled material should in most cases be considered a best-case scenario for the materials analyzed. While optimistic, these results can provide an estimate of the best possible reductions in environmental impacts that recycling of different materials could achieve. Besides market dynamics, there are other obstacles that often prevent recycled materials from achieving 100 percent displacement of primary material. Some of these limitations are discussed below.

Collection of recyclable material. For secondary material production to occur, used and recyclable material must be collected, sorted, and transported to material recycling or reprocessing facilities. Many of these items are collected through municipal recycling programs: about 94 percent of the population in the U.S. has access to some type of recycling program, which can be curb side pickup, drop-off, or both (Resource Recycling Systems 2016). Collection schemes vary by the location and material type which can make estimates of total recovered and reprocessed materials challenging to obtain. For example, 93 percent of communities in urban areas (population of 250,000 or greater) have access to such programs, while only 65 percent of communities with a population of less than 65,000 do. Limited access to recycling programs for smaller communities suggests that a greater share of materials that could be recycled go uncollected in rural areas, limiting the amount of secondary material production.

Processing losses, recycling capacity and quality of recyclable materials. After collection, materials are transported to material recycling facilities. However, facilities that sort mixed recyclables typically misdirect a fraction of materials, even if properly sorted by the generator. For example, flattened plastic bottles may hide within layers of paper, and inadvertently be sent to a paper mill, where they will subsequently be screened and sent to disposal. Further, not all facilities are able to process all materials. For example, recycling programs in the U.S. are largely available for corrugated boxes, various types of plastic containers, as well as steel, aluminum, and glass beverage containers, with coverage for these materials exceeding 60 percent of the population. On the other hand, recycling programs for expanded PS are available to less than 20 percent of the population, depending on the specific product (Resource Recycling Systems 2016). The overall recycling rates in the U.S. for selected recyclable materials is shown in Figure 9. These are the rates estimated at the end-user or export market, meaning the materials are actually recycled. It is also important to note that the U.S. exports materials to other countries, so that material not recycled domestically due to low prices or a lack of recycling capacity can be processed elsewhere. Indeed, over 2 million metric tons of waste plastics were exported in 2011 for processing in other countries, with over 80 percent of these exports going to China (Velis 2014). This can affect the impacts of recycling such that other disposal methods are environmentally preferable, as shown by the comparisons from Hottle and colleagues (2017). In addition, the practice of exporting recovered materials for possible secondary sorting prior to using them for new products may have implications to impacts areas such as plastic marine debris as discussed in the Plastic Pollution in Marine Environments section.

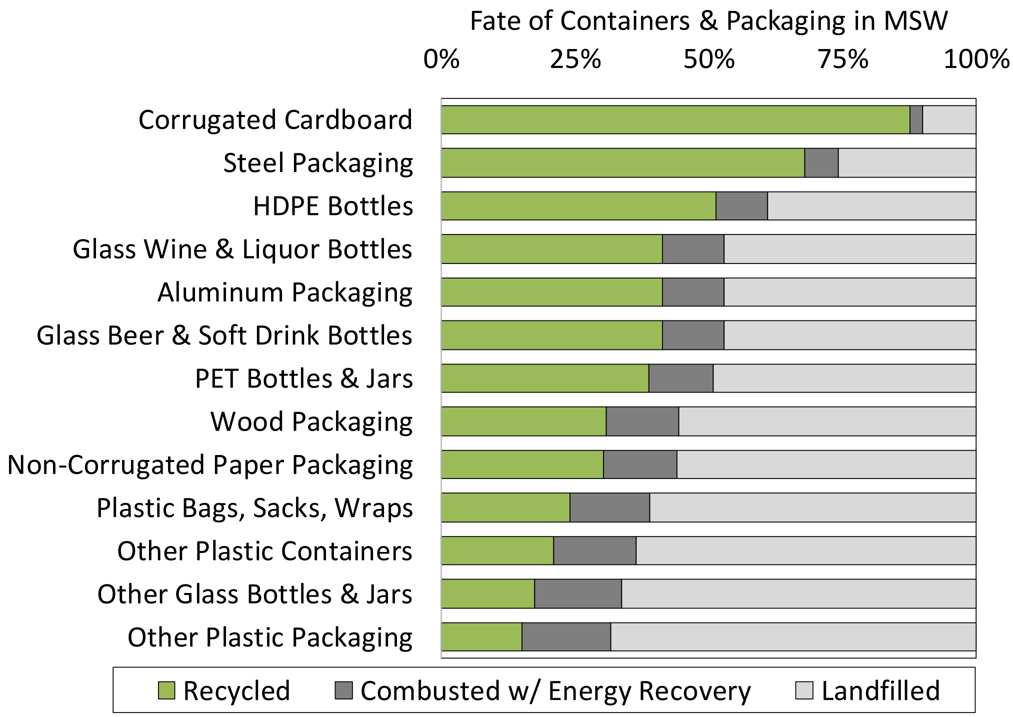

Figure 9. Recycling rates for packaging materials, 2014 U.S. average (U.S. EPA Office of Resource Conservation and Recovery 2016).

Another factor that influences the market for recyclable materials is the quality of the materials themselves. Recycling facilities strictly control the amount of contamination in a secondary feedstock. Thus, the level of contamination introduced during collection affects the market for secondary feedstocks. The markets for secondary feedstocks are also affected by national regulations and trade policies. This is currently the case for plastic wastes, where new Chinese import policies have limited imports of post-consumer plastic, considerably reducing the market for these materials. As a result, plastic packaging in the U.S. that was previously exported to be reprocessed could instead be landfilled or incinerated, at least for the short term, while new export markets or additional domestic reprocessing capacities may take some time to be established.

Summary – recyclable packaging

Results of comparing packaging made from different materials suggest that packaging weight and material type considerations are a better predictor of environmental impacts than the attribute of recyclability. Furthermore, the environmental benefits of recycled materials are dependent on the amount of primary product they displace. Market factors such as price elasticities and material constraints play important roles in determining how much primary material is displaced, as do non-market factors such as collection schemes and the quality of recycled material. While several methods have recently been described to more accurately estimate primary material displacement, historically their use in LCA has been limited, and particularly limited in LCA studies of packaging. However, the studies that explicitly included market-based displacement support the notion that one-to-one avoided burden assumption is usually not accurate. Rather, the one-to-one avoided burden assumption should be considered a best-case scenario given that materials deteriorate functionally over repeated reprocessing cycles. Thus, while the studies identified indicate recycling generally results in fewer environmental impacts than landfilling or incineration, and that higher recycling rates are generally preferable to lower recycling rates when comparing different recycling rates for the same material, at present the LCA literature is inconclusive regarding the benefits of recyclability given differences in upstream impacts for functionally equivalent materials, market conditions and primary material displacement rates.

It should be noted that recyclability has the potential to be environmentally beneficial in other aspects in addition to the impact categories compared in this section. For example, in communities with inadequate systems for waste disposal, recycling may reduce the amount of plastics that end up in marine environments. However, these aspects are not traditionally measured by LCA studies and as such are not included in the scope of this review.

Biobased Packaging

Scope

Biobased packaging made from renewable feedstocks can be compostable or recyclable independent from the fact that they are biobased. The following section is focused on comparing biobased and fossil-based materials rather than their ability to be recycled, composted, or contain recycled materials. Among the 17 LCA studies identified which included life cycle impact results, the number of impact categories assessed varied widely between one and 17 within the literature, with the median number of impact categories addressed being five. These studies yielded a total of 102 comparisons between biobased and fossil materials.

Together Table 3 and Figure 10 provide a summary of the scope of the studies offering life cycle comparisons between biobased and other packaging options including coverage of environmental impact categories as well as the system boundaries and materials evaluated.

Most of the studies are limited in scope to cradle-to-gate impacts, only four of the 17 studies include end-of-life materials management (Hottle, Bilec, and Landis 2017; Liptow and Tillman 2012; Madival et al. 2009; Shen, Worrell, and Patel 2012). The omission of end-of-life management of biobased materials is important as many of these packaging materials are also compostable or involve different recycling considerations than conventional materials, and highlights a key area for future investigation (Flanigan, Frischknecht, and Montalbo 2013).

Table 3. Scope of studies included in the biobased packaging comparisons.

					Materials													Boundary			Result Categories												
Author and Year	*Packaging Type*	*Functional Unit*	*Geography*	*LCIA Method*	PE	PET	HDPE	LDPE	PHA	PLA	PP	PS	TPS	Plastic Precursors	Rayon	Paper	Polyester	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Hottle, 2017	Polymers/biopolymers used for packaging	1 kg	US	TRACI v 2.1		x	x	x		x			x					x		x	x	x	x		x	x	x	x	x	x			
Madival, 2009	Strawberries	1,000 x 1lb containers	US, Europe, Middle East	IMPACT 2002+		x				x		x						x	x	x	x	x	x			x	x	x	x				x
Chen, 2016	Beverage bottles	1 kg	US	TRACI v 2.1		x												x			x	x	x			x	x	x					x
Lin, 2015	Chemical precursor	1,000 kg	US	ReCiPe v1.0										x				x			x		x			x	x	x	x				x
Günkaya, 2016	Packaging film	$1 m^2$	Turkey	CML-IA baseline v 3				x										x			x	x	x			x		x	x				
Groot, 2010	Polymers/biopolymers used for packaging	1,000 kg	Thailand	CML, IPCC						x								x			x	x	x			x		x		x			
Hermann, 2010	Food packaging film	$1 m^2$	Europe	CML 2 baseline	x					x	x					x	x	x			x	x	x			x				x		x	
Detzel, 2013	Various	30 g / 15 g	Germany	Several[1]	x					x		x						x			x	x	x			x	x	x					
Liptow, 2012	Polymers/biopolymers used for packaging	1 kg LDPE	Brazil	Inventory based				x										x		x	x	x	x*			x				x			

Studies ordered from most to least coverage of result categories

Table 3. Scope of studies included in the biobased packaging comparisons, continued.

					Materials													Boundary			Result Categories												
Author and Year	*Packaging Type*	*Functional Unit*	*Geography*	*LCIA Method*	PE	PET	HDPE	LDPE	PHA	PLA	PP	PS	TPS	Plastic Precursors	Rayon	Paper	Polyester	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Nuss, 2013	Chemical precursor	1 kg	US	ReCiPe 2008, TRACI										x				x			x	x	x									x	x
Tsiropoulos, 2015	Polymers/biopolymers used for packaging	1 kg	Europe, India, Brazil	IMPACT 2002+		x	x											x			x							x	x	x		x	
Akanuma, 2014	Chemical precursor	1 kg	US	IMPACT 2002+		x												x			x	x	x							x			
Kim, 2005	Polymers/biopolymers used for packaging	1 kg resin	US	TRACI					x			x						x			x	x	x							x			
Papong, 2014	Water bottles	1,000 x 250ml drinking water	Thailand	CML 2 baseline		x				x								x				x	x					x		x			
Suwanmanee, 2012	Food box	10,000 boxes	Thailand	CML 2 baseline						x		x						x			x	x				x							
Shen, 2012	Beverage bottles	1 kg	Europe	Several[1]		x				x		x			x			x		x	x									x			
Bohlman, 2004	Yogurt cups	1,000 kg	US	Inventory based						x	x							x			x												
Percent of Total					12	41	12	18	6	53	12	29	6	12	6	6	6	100	6	24	94	76	76	71	6	59	29	53	29	53	0	18	24

Studies ordered from most to least coverage of result categories

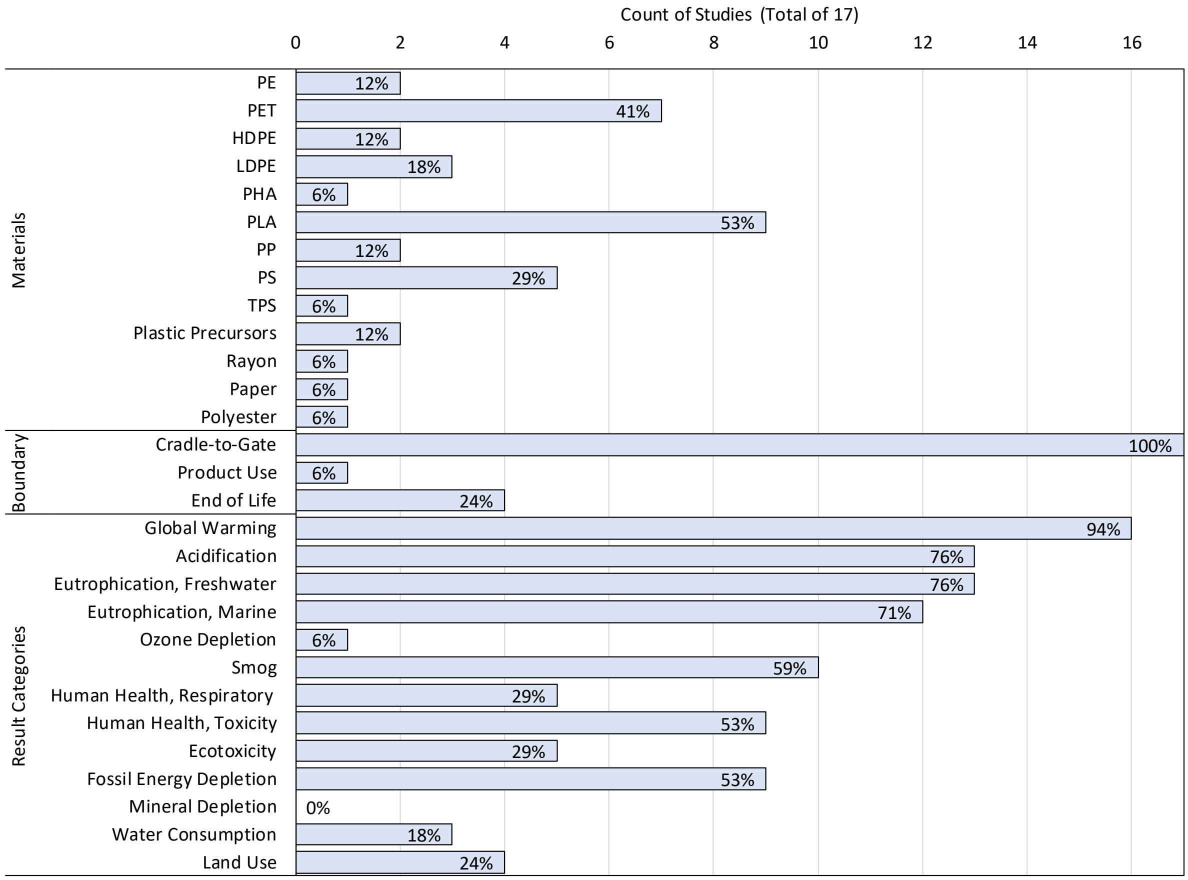

Figure 10. Scope of studies included in the biobased packaging comparisons.

Biobased materials do not always provide life cycle impact reductions in categories generally associated with chemical industries

Biobased packaging is often believed to be associated with benefits for GWP and fossil fuel depletion but, as seen in Figure 11, this is not always the case. The comparisons which lead to meaningful differences for these categories are mixed and a large portion of the comparisons were inconclusive. This range of results can be associated with factors like the crop, climate and geography, and processing technologies used to convert agricultural feedstocks into packaging materials. This was highlighted by Yates and Barlow (2013) who found that producing biopolymer feedstocks requires a significant amount of fossil fuel for agricultural operations and inputs such as fertilizers and pesticides as well as milling, fermentation, and other conversion processes.

Suwanmanee and colleagues (2013) found that land use change associated with corn and cassava, drove the GWP and AP of PLA and PLA/starch boxes to exceed those of polystyrene. However, they did not include the flow of CO_2 that is sequestered during the growing of biobased feedstocks used in PLA or released during decomposition or incineration. As a counter example, Papong and colleagues (2014) found that replacing fossil fuels, used for heating in cassava-based PLA production, with biogas decreased the eutrophication potential and further reduced GWP which already favored the biobased material in their study.

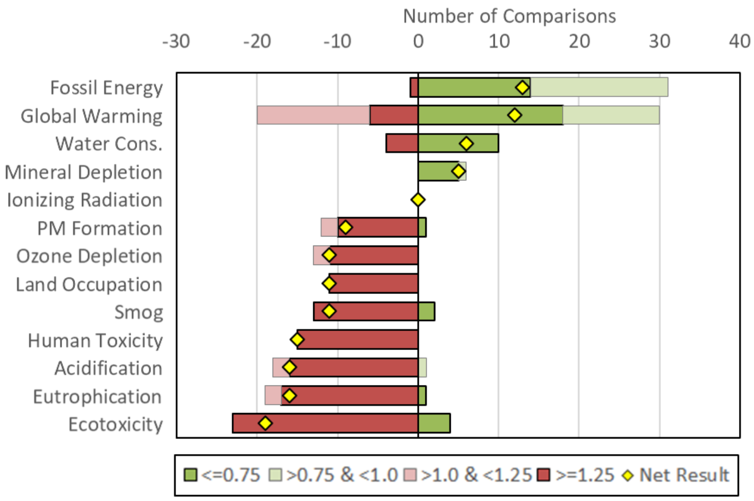

Figure 11. Comparisons of biobased packaging to conventional packaging of the same polymer. *Ratios reflect the result for the biobased packaging divided by the result for the conventional packaging. Thus ratios <1 indicate biobased packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates biobased packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

More difficult to interpret are the tradeoffs associated with shifting to a biobased production pathway which results in different potential waste management pathways during end-of-life. Figure 12 shows the mixed results associated with comparisons of conventional packaging and dissimilar biobased alternatives some of which enable composting. The graph highlights the divergent results associated with specific materials and across comparisons. Although the comparison of production for biobased and conventional materials can be independent from end-of-life pathways for those materials, many biobased products are explicitly advertised for features associated with biodegradability or compostability. If biobased packaging and or food service ware enables new end-of-life options or eliminates other options, like recycling, it is worth consideration.

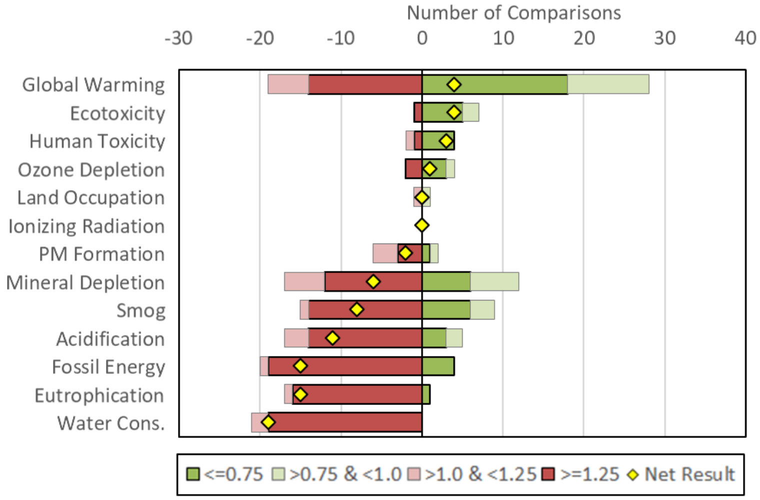

Figure 12. Comparisons of biobased packaging to conventional packaging of a different polymer. *Ratios reflect the result for the biobased packaging divided by the result for the conventional packaging. Thus ratios <1 indicate biobased packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates biobased packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons. Results are sorted in the same order as Figure 11 to facilitate comparison.*

Although few LCA studies have addressed waste management for biopolymers, the choice of waste management can shift impacts of biobased products significantly enough to tip the findings in favor of one type of material over another (Hottle, Bilec, and Landis 2013, 2017; Yates and Barlow 2013) as discussed below. Of the studies that model the end-of-life for biobased packaging materials, four waste pathways were analyzed: recycling, composting, incineration, and landfilling.

Recycling results in significant life cycle impact reductions for fossil-based resins. These benefits could extend to biopolymers if they are identical to polymers that can be recycled in existing collection systems or, in the case of biopolymers like PLA, recycling technologies can be scaled to warrant the collection and processing efforts. For example, Hottle and colleagues (2017) found that biobased PET, HDPE, and LDPE achieved negative fossil fuel impacts, assuming recycling of these polymers offsets virgin fossil-based resins used to manufacture new products.

Rossi and colleagues (2015) found that mechanical recycling leads to the greatest reductions of environmental impacts, whereas composting was less favorable in some categories when comparing end-of-life options for TPS and PLA. The environmental benefits of mechanical recycling are a result of offsetting the impacts associated with production using virgin materials. Composting, anaerobic digestion, and incineration resulted in similar generation of GWP; in the composting scenario, TPS, PLA, and the subsequent compost material degrade almost completely (~93 percent) over a one-hundred-year time frame, though this assumption is not applied in all LCAs evaluating composting. The compost material was found to have low levels of nitrogen, phosphorous, and potassium, and are therefore was considered inadequate replacements for fertilizer while the other waste management options generate offsets due to energy production (Rossi et al. 2015).

Growing and processing agricultural feedstocks required for biobased manufacturing drives increased acidification, eutrophication, ecotoxicity, and land use impacts

Agricultural inputs for the biobased material feedstocks appear to be the primary driver of increased environmental impacts, with the additional production-related energy and conversion technologies as a secondary driver. Trends across studies indicate increased cradle-to-gate impacts for categories such as acidification, eutrophication, ecotoxicity, and land use that are linked to agricultural practices for biobased feedstocks. This is illustrated by Figure 11, which compares studies that evaluated the same material across biobased and fossil-based production pathways.

Hermann and colleagues found that land and agricultural inputs used to produce biobased materials can often be identified as the drivers of elevated environmental impacts (2013). Findings from Chen and colleagues (2016) in a comparative LCA of PET and bioPET reinforce this concept, finding that “biomass feedstock extraction and preprocessing are likely more emission-intensive than corresponding fossil refinery processes, either due to the extra energy required for agricultural operations, or because of the production and application of required chemicals.”

Some assessments included scenarios evaluating different biobased feedstocks or different conversion technologies for biobased materials. The studies evaluating cellulosic feedstocks, associated with lower impact waste materials or perennial crops, found that although the impacts associated with growing the feedstocks was lower, the additional energy required to convert these feedstocks to functional technical materials outweighed the benefits of using starch- or sugar-based feedstocks like corn, wheat, cassava, or sugarcane (Akanuma, Selke, and Auras 2014; Chen, Pelton, and Smith 2016; Kim and Dale 2005). For example, PTA made from switchgrass-based ethylene glycol and wood-based ethylene glycol had higher (sometimes double) environmental impact comparative ratios for acidification, eutrophication, smog, and particulate matter formation when compared to corn- and wheat-based ethylene glycol due to the significantly higher level of energy required to process cellulosic-based feedstocks (Chen, Pelton, and Smith 2016). Detzel and colleagues (2013) found that a lignocellulose-based PLA under a future production scenario which anticipates improved fermentation processes may have slightly lower impacts than sugar beets, highlighting the potential for technological advancements in biobased production.

Kim and Dale (2005) also found that the level of environmental impacts resulting from feedstock processing can depend upon the conversion technologies used. They found that the fermentation process used to produce PHA from corn grain had greater smog, acidification, and eutrophication potentials than polystyrene. Conversely, these impacts were less than those of polystyrene when a combination of corn grain and stover-based PHA was assessed under an anticipated near future scenario.

Environmental performance of biobased materials depends on the manufacturing and feedstock requirements for specific materials

In addition to the broad findings across material-types, the review of literature also generated findings specific to certain biobased packaging materials. These findings are presented below.

Biobased Polylactic Acid (PLA)

As shown in Table 3, nine studies (with LCA data) compared the impacts of biobased PLA to fossil-based products, including PS, PE, including HDPE and LDPE, PET, and PP. Six studies evaluated PET, four evaluated PS, four evaluated PET, and two evaluated PP (Bohlmann 2004; Detzel, Kauertz, and Derreza-Greeven 2013; Groot and Borén 2010; Hermann, Blok, and Patel 2010; Hottle, Bilec, and Landis 2017; Madival et al. 2009; Papong et al. 2014; Shen, Worrell, and Patel 2012; Suwanmanee et al. 2013).

Table 4 shows ratios resulting from the comparison of PLA and PS materials for select environmental impact categories. The table shows the variation in impacts across studies and PLA feedstocks. Acidification, eutrophication, and human health impacts are higher for PLA, consistent with its reliance on the use of fertilizer and pesticides.

Table 4. Comparative LCA results for polylactic acid vs. polystyrene.

Author	*Materials*	Global Warming	Acidification	Eutrophication	Ozone Depletion	Smog	Human Health Impacts
Detzel, 2013	PLA (sugar beet) vs PS	0.88	2.3	1.8		0.23	1.9
	PLA (ligno-cellulose) vs PS	0.69	2.1	1.4		0.15	1.7
Suwanmanee, 2013	PLA (corn grain) vs PS	35	3.0			1.5	
Madival, 2009	PLA (corn grain) vs PS	1.0	1.7	1.7	1.4		1.4-1.5
Groot, 2010	PLA (sugarcane) vs PS	0.65					

Red cells indicate PLA has higher impacts than PS, while green cells indicate PLA has lower impacts than PS. The calculation of ratios is described in the Methods section.

While the study methodologies differ to the extent that LCA results could not be compared across studies, within studies, the type of biobased feedstock used in the PLA, for example corn, beets, or sugarcane, affected the level of environmental impacts for a given category. For example, when comparing PLA made of beet and ligno-cellulose, Detzel and colleagues (2013) found that the lignocellulose PLA had slightly lower environmental impacts than PLA made of beets. Groot and Borén (2010) found that PLA manufactured with sugarcane feedstocks have lower GHG impacts than fossil PS due in part to renewable energy derived from combustion of the biomass remaining after sugar extraction. Papong and colleagues (2014) determined that corn-based PLA has higher fossil fuel impacts than cassava or sugarcane-based PLA due to the increased use of fertilizers and pesticides in corn production.

Hottle and colleagues (2013) also identified tradeoffs across impact categories for PLA. When comparing data for the bioplastics TPS and PLA with the fossil-based plastics HDPE, LDPE, PET, PP, and PS, they found higher eutrophication and ozone depletion potentials for TPS and PLA than any of the fossil-based alternatives. Both the eutrophication and non-carcinogenic human health impacts for the biobased polymers were attributed to agricultural emissions of phosphorus- and nitrogen-substances primarily due to the application of fertilizers, with the effluent from the fermentation and distillation processes being a secondary factor for eutrophication. The transport of fossil fuels needed to make the biobased plastic accounts for higher rates of ozone depletion. The results in the remaining impact categories were mixed for the biobased and fossil-based materials.

When comparing PLA and fossil-based PET, findings vary by impact category (*Supporting Information B*). Madival and colleagues (2009) found PLA has lower environmental impacts than PET across most impact categories including GWP, aquatic eutrophication, aquatic ecotoxicity, ozone depletion, and land use but PLA had greater impacts in acidification and human health. Papong and colleagues (2014) found that cassava-based PLA bottles performed better than PET bottles where global warming, fossil fuel depletion, and human toxicity were concerned but PLA was worse in acidification and eutrophication.

Packaging Films

Hermann and colleagues (2010) showed that laminated packaging films containing one or more biobased packaging materials largely have greater impacts compared to fossil-based films. The study showed that films made with various forms of PLA had generally greater environmental impacts for eutrophication, water consumption, mineral depletion, and energy depletion, with mixed results for GWP. Similarly, Günkaya and Banar (2016) found that biocomposite film made from orange peel-derived pectin jelly and corn starch increased environmental impacts of GWP, acidification, eutrophication, smog, and human toxicity compared to film made from LDPE.

Polyhydroxyalkanoates (PHA)

Kim and Dale (2005) found that the fermentation and recovery processes used in producing PHA from corn grain resulted in increased environmental impacts compared to fossil-based polystyrene (PS) for the categories of smog, acidification, and eutrophication.[11] The study showed that estimating environmental impacts using expected future technologies for processing cellulosic feedstocks and a combination of corn grain and stover, as opposed to just grain, led to fewer impacts than polystyrene except for eutrophication potential. PHA made from a combination of corn grain and stover was found to have a lower GWP than PHA made from corn grain alone. Conversely, PHA made from corn grain and stover had slightly higher impacts for acidification, eutrophication, and smog (see *Supporting Information B* for details).

[11] The comparison with PS was based on external inventories but quantitative data was not included in the Kim and Dale (2005) so comparative ratios were not included for PS

High-Density Polyethylene (HDPE)

The relative impacts of biobased HDPE and fossil HDPE vary by impact category. Tsiropoulos and colleagues (2015) and Hottle and colleagues (2017) found that biobased HDPE had generally lower greenhouse gas emissions and non-renewable energy use impacts than fossil HDPE; however, Tsiropoulos found that greenhouse gas emissions decreased from 140 percent lower than fossil HDPE (net negative due to CO_2 storage) to only 20 percent lower when indirect land use change was considered.[12] In contrast, the biobased HDPE had increased impacts of human health, ecotoxicity, and water consumption were concerned.

Low-Density Polyethylene (LDPE)

Liptow and Tillman (2012) showed that the production of sugarcane-based LDPE compared to its fossil-based counterpart reduced the impacts of GWP and photochemical oxidation potential; increased the eutrophication potential and required more process energy (although the major fuel source for this energy is renewable); and marginally decreased acidification potential.. Major contributors to the environmental impact of sugarcane LDPE are ethanol production, polymerization, and long-distance sea transport. The GWP of the sugarcane-based LDPE was roughly half that of fossil-based LDPE. The addition of land use change associated with sugarcane production, which has a high degree of uncertainty, resulted in no meaningful difference in GWP between the bio- and fossil-based LDPE.

Polyethylene Terephthalate (PET)

When comparing bioPET to fossil-based counterparts, Shen and colleagues (2012) found that bioPET had lower impacts for GWP and non-renewable energy use compared to fossil PET. The bioPET is not entirely derived from biobased feedstocks. The ethylene component of the bioPET polymer is bio-based, while the terephthalate remains a fossil-based feedstock. The bioPET derived from sugarcane had slightly lower environmental impacts than that made of corn for these same impact categories (Shen, Worrell, and Patel 2012). Tsiropoulos and colleagues (2015) found that bioPET production had slightly higher non-renewable energy use impacts than fossil PET when including direct and indirect land use change, with comparative LCA ratios of approximately 1.0 for each impact category. For the impact categories of human health and ecotoxicity, the bioPET had significantly higher impacts, with comparative LCA ratios ranging from 8 (human health) to 13 (ecotoxicity).

Purified terephthalic acid (PTA)

Two studies evaluated several purified terephthalic acid (PTA) pathways, including those made with corn, wheat, wood, and switchgrass (Akanuma, Selke, and Auras 2014; Chen, Pelton, and Smith 2016). PTA is combined with ethylene glycol to produce PET. The published results from Akanuma and colleagues (2014) show increased impacts associated with biobased PTA for

[12] Indirect land use change encompasses the unintended consequences of releasing more carbon emissions due to land-use changes, such as those made for agricultural purposes to cultivate feedstocks for use in a biobased material.

acidification, eutrophication, and GWP, while fossil fuel depletion and mineral resource depletion potentials were decreased when compared to fossil-based PTA.

p-xylene

p-xylene is used to make PTA and dimethyl-terephthalate, both of which are used to make PET. When comparing p-xylene made from fossil resources to biobased options of corn and oak, Lin (2015) found that the corn-based version of the chemical had significantly higher environmental impacts due to the agricultural practices associated with cultivation and harvesting, regardless of allocation method.[13] At least 75 percent of the environmental impacts can be attributed to agricultural practices for the following impact categories: eutrophication, ecotoxicity, particulate matter formation, and land occupation. The lignocellulose-based oak feedstock was found to be comparable with the fossil-based p-xylene based on a single score comparison.

Summary – biobased packaging

The results of the comparative environmental impact ratios developed show that biobased packaging materials have environmental tradeoffs compared to non-biobased counterparts.[14] In the comparisons, biobased materials performed best for GWP, although this finding is not consistent across studies/comparisons. More than half the comparisons resulted in ratios above 1.0, marginal increase to higher impacts. This is especially important in impact categories such as acidification and eutrophication, which saw greater impacts for nearly all comparisons; see Figure 11 and Figure 12. This pattern of impact category tradeoffs has been identified in previous reviews of the literature and can typically be associated with the shift from fossil to agricultural-based resources (Hottle, Bilec, and Landis 2013; Miller et al. 2007; Zhang et al. 2016).

Agricultural production drove consistently meaningful differences in the acidification, eutrophication, ecotoxicity, and land use categories when comparing packaging that has biobased and conventional fossil-based production pathways. The potential significance of agricultural production to these impact categories depends on the location of production as well as on geospatial contexts such as proximity to bodies of water or baseline land use practices to understand the potential significance of these impacts. Several studies indicated that different biobased feedstocks or conversion technologies have the potential to tip the findings in favor of biobased packaging materials. The variations in results comparing biobased options highlights the importance of assessing the specific material options for the desired application. The conventional industries associated with packaging production are characterized by mature, developed pathways for feedstock acquisition and material production. However, much of the research concerning biobased materials focuses on the pursuit of improved environmental performance through feedstock development and process improvements with potential for process advancements as technologies for biobased pathways mature.

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[13] LCA data was only provided for p-xylene made from corn and oak. Impacts for petroleum p-xylene were only indicated in a single score summary.
[14] The exception was the study performed by Hermann et al. (2010) on laminated, printed film.

Understanding the scope and system boundaries considered for each study is critical so any environmental tradeoffs s can be understood in context. The difference between biobased and conventional packaging materials is not simply a function of differing carbon pathways. The shift from fossil to biobased alters both the upstream dynamics of feedstock acquisition and the downstream processing required to create the final marketable product. Additionally, the material itself dictates the potential for packaging to be recycled or composted. Any decisions to shift to biobased materials should also consider the end-of-life options available for that specific material in the context of its intended use. A shift away from recyclable materials in a system where recycling is a feasible end-of-life pathway could eliminate a potentially better end-of-life management option from an environmental standpoint. Only four studies evaluated included waste management options in their impact assessment which may have significant influence on the total life cycle impacts of these materials, so more research on this topic is needed.

Compostable Packaging

Scope

Compostable packaging degrades biologically to yield CO_2, water, inorganic compounds, and biomass while leaving no visually distinguishable remnants or unacceptable levels of toxic residues. Our search yielded 10 LCA studies of compostable packaging, which allowed a total of 1,287 comparisons. These studies allowed for comparisons between compostable and non-compostable materials, as well as between compostable materials that are composted vs. landfilled, incinerated, or recycled. The most common EOL disposition for all types of materials was landfilling (included in 9 of 10 studies) followed by incineration with energy recovery (7 of 10) and recycling 5of 10). The packaging types included in the studies were cushioning/expanded packaging, sheets, wrapping films, thermoformed boxes, water bottles, clamshell packaging, and various polymers used to make compostable packaging. The compostable materials considered were PLA, a starch-based expanded polystyrene (EPS), TPS, and trademarked materials such as Mater-Bi™, starch-based biopolymers, and Ingeo™ (PLA-based). Biobased PLA was the most common material, included in seven of the ten studies. All the studies considered GWP; 70 percent included energy demand; 40 percent considered acidification, eutrophication, and smog formation potential; and 30 percent or fewer presented results for ozone depletion, human toxicity, ecosystem toxicity, resource depletion, water use, and land use.

There are some additional characteristics for compostable materials, which are important to consider when interpreting results. For example, seven of the studies state that the materials considered meet ASTM, ISO, or certified compostable guidelines; four studies considered impacts from land use change associated with the production of agricultural feedstocks, one of which only includes land use change in the sensitivity analysis; and two studies provide the nutrient content of the compost made from packaging; . Three studies include transportation of the packaged product and the collection of packaging waste. Table 5 and Figure 13 provide a summary of the studies included in the comparisons.

The environmental results for compostable packaging are affected by the assumptions that studies make when analyzing the different materials, such as system boundaries and treatment of biogenic CO_2. It also depends on the end-of-life modeling of the packaging, since not all scenarios involving compostable packaging assumed composting as the only waste management strategy. The most influential factors are discussed below and summarized in Table 5 under the heading Compost-Related Sope.

System Boundaries and Life Cycle Stages

Not all studies considered the entire life cycle of the packaging product/materials in their analyses. Hermann and colleagues (2011) focused their analysis on the impacts of end-of-life stage for compostable packaging exclusively. Additionally, while most studies included the production stages, most also excluded the use phase (all except for Leejarkpai and colleagues, 2016 and Papong and colleagues, 2014). Differences in the modeling of specific life cycle stages also affect the impact estimates. For example, waste collection and transportation distances vary by study, and their related emissions can account for as much as 11 percent of total GWP emissions for composted PLA (Hottle and colleagues 2017) to as low as 1.5 percent (Leejarkpai and colleagues, 2016). Another factor that was not treated consistently between the studies is the accounting of carbon uptake during biomass growth and release during composting (i.e. biogenic carbon) in the life cycle inventory of biobased compostable products. Three studies omitted accounting of biogenic carbon entirely, while the rest had different assumptions on how to treat it. While these differences in approach do not allow for comparisons between studies, they do not affect intra-study comparisons.

Land Use Change

The inclusion of land use change impacts has the potential to significantly influence environmental impact results, especially GWP. Land use change impacts vary considerably, depending on accounting methods and land type used for feedstock growth (e.g., change in crops grown or clearing new land for crop production). Some studies mentioned that land use change was not included due to lack of data, while the studies that included land use change in their analysis accounted for it differently. Leejarkpai and colleagues (2016) included land use change emissions directly in the system boundary of the main comparisons. This study compared the results of composting to landfilling, both with and without the inclusion of land use change. When land use change impacts from production of the feedstock for PLA are not included, the total GWP of composted PLA is comparable to landfilled PET. However, when the impacts of land use change are included, the total GWP of composted PLA is shown to almost double the total emissions from landfilled PET. Krüger and colleagues (2009) and Rossi and colleagues (2015) included land use change in the life cycle inventory of the feedstock for the compostable products they analyzed. Razza and colleagues (2015) did not include land use change as part of the primary analysis, but included it as part of their sensitivity analysis, and found that impacts for GWP could increase by as much as 61 percent for their composting scenario.

Environmental Credits for Compost

Like the decisions regarding system boundaries, the way that credits are assigned for compost resulting from compostable packaging affects the overall environmental performance of compostability. For the reviewed studies, Rossi and colleagues (2015) and Hermann and colleagues (2011) explicitly mention the low nutrient content of compostable packaging polymers, and assigned fewer credits for avoided use of fertilizers to the resulting compost when compared to anaerobic digestion of the same material. The rest of the studies generally assumed that composting took place in industrial facilities where the packaging would be composted with other organic material and did not specify further on credits assigned due to low nutrient content. This discrepancy in the interactions of the compostable product with the avoided processes considered in the LCI of the different studies is common. Schott and colleagues (2016) performed an analytical review of over 100 treatment scenarios of food and organic waste, disposed through incineration, landfill, anaerobic digestion and compost. Their results found large differences for GWP both between and within the different disposal methods, which can be explained, to a large extent, by the choices made in the energy and bio-system process substitutions for compost.

Table 5. Scope of studies included in the compostable packaging comparisons.

					Materials			Compost-Related Scope				Boundary			Result Categories												
Author and Year	*Packaging Type*	*Functional Unit*	*Geography*	*LCIA Method*	Biobased Plastic	Paper	Cellulose	Accounts for Biogenic CO2	Compost Certification	Considers Land Use Change	Mentions Low Nutrient Content	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Rossi, 2015	Dry packaging (e.g. non-food packaging, secondary packaging)	EOL treatment of 1 kg of dry packaging	Europe	IMPACT 2002+	x				x[1]	x		x		x	x	x	x		x	x	x	x	x	x		x	x
Hottle, 2017	Polymers/biopolymers used for packaging production	1 kg of polymer[3]	United States	TRACI v 2.1	x			x	x[1]		x	x		x	x	x	x		x	x	x	x	x	x			
Kruger et al, 2009	Clamshell food packaging made from Polylactide (Ingeo ™)	1000 clamshells	Europe	Inventory based				x	x	x	x	x		x	x	x	x			x	x	x		x			x
Razza, 2015	Cushioning/expanded packaging	100 port-hole spacers	Europe	IMPACT2002+; IPCC 2007 100y; CML 2001; & custom	x			x		x[2]		x		x	x	x	x		x	x				x	x		
Girgenti, 2014	Strawberry wrapping films/sheets	250 gram flow pack	Italy	IPCC 2007, Inventory based	x				x[1]			x		x	x									x			
Hermann, 2011	Various biodegradable polymers	1 kg of polymer[3]	Europe	IPCC 2007, Inventory based	x	x	x	x	x					x	x									x			
Quantis, 2011	Espresso capsule and packaging system	40 ml espresso cup	France, Switzerland	IMPACT 2002+, ReCiPe	x			x				x	x	x	x									x			
Hermann, 2010	Food	1 m^2 of packaging film	Europe	CML 2 baseline	x	x	x	x	x[1]			x		x	x												
Leejarkpai, 2016	Thermoform boxes (single use)	1000 boxes, 0.2 L, 100 g capacity	Thailand	IPCC 2007	x			x	x	x		x	x	x	x												
Papong, 2014	Water bottles	1000 drinking water bottles, 250 ml each.	Thailand	CML 2 baseline	x							x	x	x	x												
Percent of Total Number					90	20	20	70	70	40	20	90	30	100	100	40	40	0	30	40	30	30	20	70	10	10	20

[1] Composting scenario uses data for trademarked material (MaterBi ™) or primary data from manufacturer with composting certification.
[2] Mentions land use change (LUC) as part of sensitivity analysis, not main scenario comparisons. Inclusion of LUC in sensitivity increases GWP of composting scenarios.
[3] Comparison of feedstock materials used to produce packaging
Studies ordered from most to least coverage of result categories

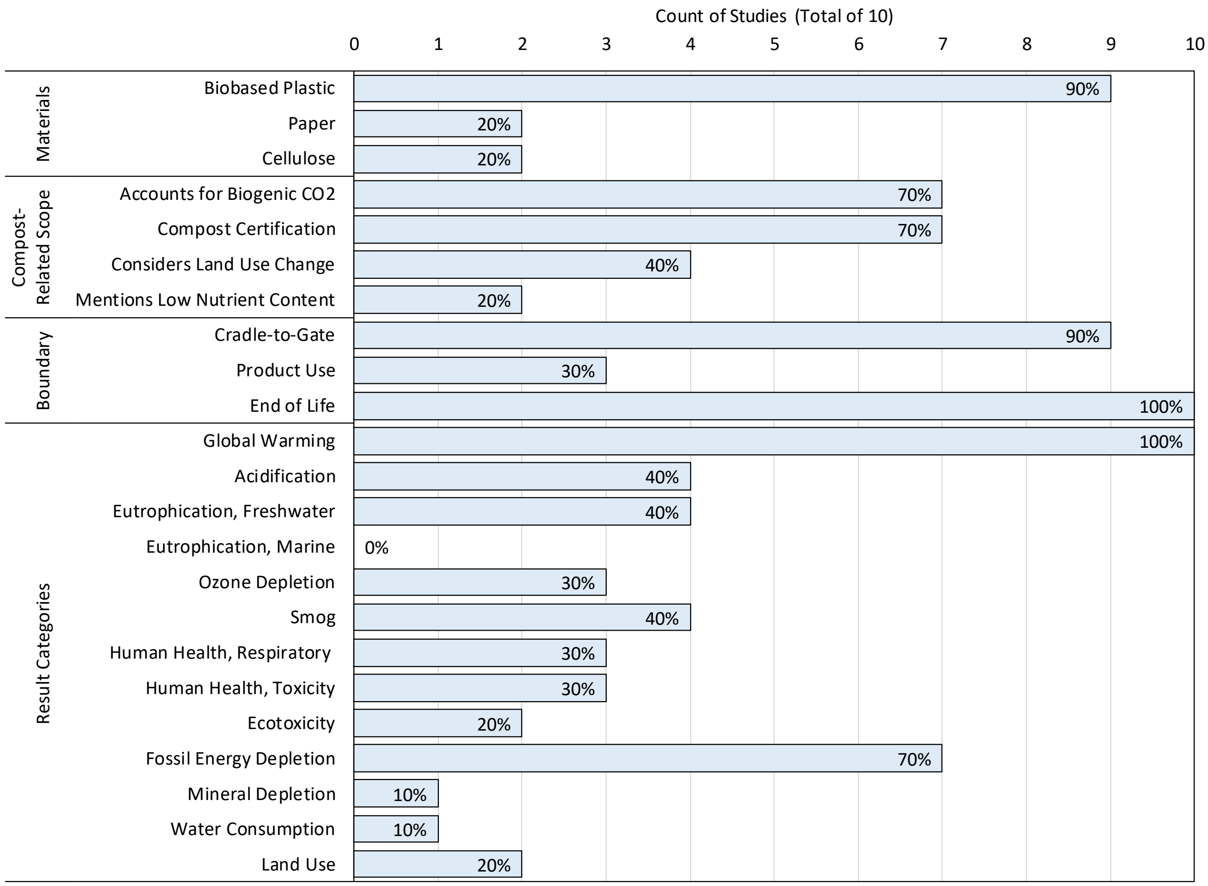
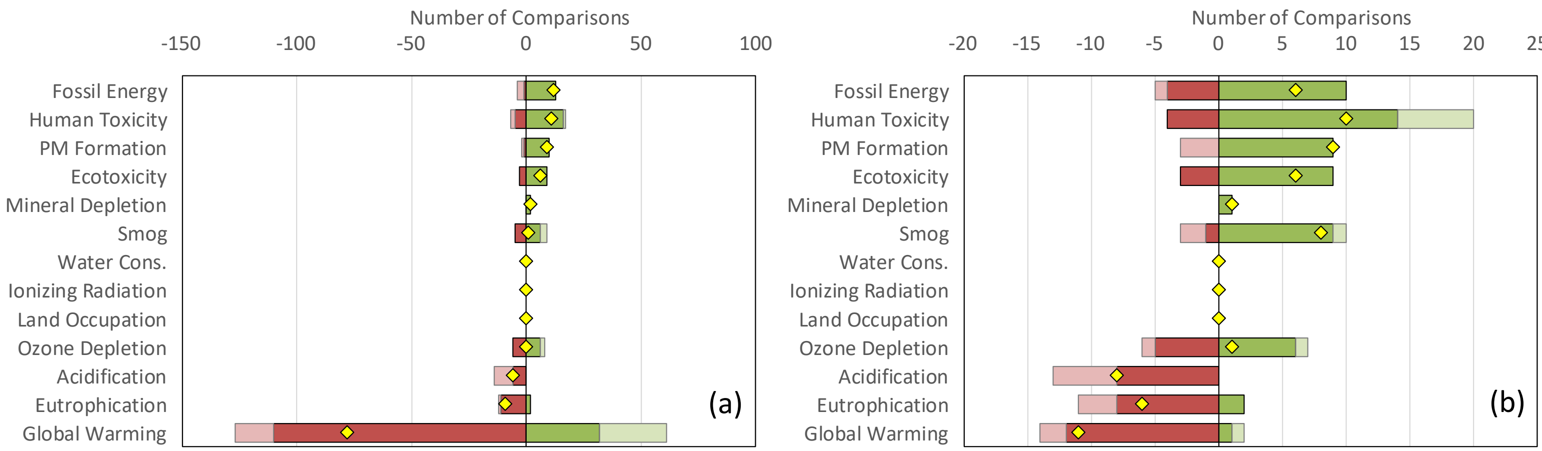


Figure 13. Scope of studies included in the compostable packaging comparisons.

Comparisons of composted packaging with non-compostable packaging that is landfilled, incinerated, or recycled have mixed results

Seven of the reviewed studies allowed for comparisons between compostable materials that are composted at end-of-life and non-compostable materials that are landfilled, incinerated, or recycled. Of these, all seven studies allowed comparisons for GWP, while only three studies allowed for comparisons for other impact categories (Girgenti and colleagues, 2014; Hottle and colleagues, 2017; Quantis, 2011; Razza and colleagues, 2015). Note that comparisons of compostable packaging materials that are not composted are discussed in the next subheading.

Figure 14 is divided into three parts and shows the results for the comparisons between composted materials and non-compostable materials that are landfilled (panel a), recycled (panel b), and incinerated with energy recovery (panel c).



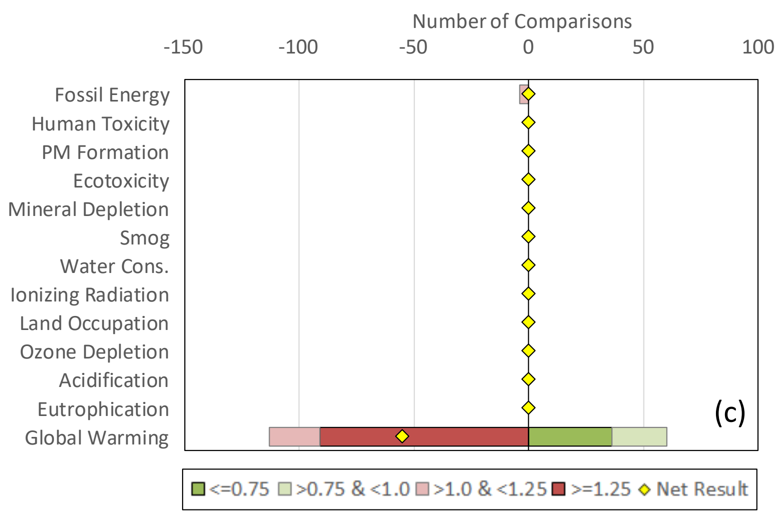


Figure 14. Comparisons of the life cycle impacts of compostable packaging that is composted with non-compostable packaging landfilled (a), recycled (b), and incinerated for energy recovery (c). *Ratios reflect the result for the composted packaging divided by the result for the non-compostable packaging. Thus ratios <1 indicate composted packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates composted packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons. All panels sorted by net result of comparisons from panel a.*

Composted Packaging vs. Landfilled Non-Compostable Packaging

Global Warming Impacts. Figure 14 (panel a) shows the comparisons between composted vs. landfilled, non-compostable packaging for GWP. 168 (89 percent) of these comparisons are obtained from the study by Hermann and colleagues (2010). Most comparisons where compostable materials perform poorly compared to non-compostable landfilled materials come from this study, due to its comparatively large sample size. There are three cases where composting results in greater GWP: 1) the landfill scenarios for Hermann and colleagues (2010) include gas recovery, and most of the materials in this study are bio-based, allowing for energy offsets; 2) most of the compostable packaging is in the form of laminates that contain non-compostable materials, decreasing the amount of compostable material available per functional unit, and 3) the study models some of the compostable materials (e.g. PLA, paper-based laminate) as releasing more GHG emissions from composting than from landfilling. The higher ratios for composting scenarios in the second GWP row in (panel a), excluding Hermann and colleagues (2010) are due to the effects of including land use change in the system boundaries of the composted PLA (Leejarkpai and colleagues, 2016), and comparisons between composted TPS and landfilled bio-HDPE and bio-LDPE, as these materials have lower cradle to gate GHG emissions than TPS (Hottle and colleagues 2017).

On the other hand, the comparisons that produced lower impacts for GWP for compostable materials for the Hermann and colleagues (2010) study are mostly caused by the comparisons of laminates with only compostable materials (e.g., PLA) against complex laminates with the compostable materials that are modeled as releasing higher gas yields during landfilling (e.g., containing cellulose). Girgenti and colleagues (2014), Papong and colleagues (2014), and Razza and colleagues (2015) also indicate that compostable packaging materials (Mater-Bi, PLA, and starch-based expanded plastic) produce lower GHG impacts than the landfilled conventional materials they were compared against (PE, PET, fossil-based EPS). Hottle and colleagues (2017) also found favorable results for some comparisons, with cradle-to-grave GHG emissions of compostable polymers generally lower than conventional polymers when they were composted. For example, the study indicates that GHG emissions from PLA production are lower than those from PET, LDPE, and HDPE production, while emissions from TPS production are lower than PET and LDPE, but not HDPE, which has similar GHG emissions for production. Both cradle-to-gate and cradle-to-grave emissions are sensitive to study assumptions, as discussed above.

Other Impact Categories. Comparative results from the review indicate that both life cycle and cradle-to-gate energy impacts for compostable materials are lower than those from conventional landfilled materials (Girgenti et al. 2014; Hottle, Bilec, and Landis 2017; Razza et al. 2015). Unlike GHG emissions, this holds true for energy use across all the materials compared in the individual studies. Of the reviewed studies, only two presented results comparing compostable packaging and conventional landfilled materials for impact categories other than GWP and fossil energy use. Results for compostable materials were mixed, with specific materials performing better or worse on specific impact categories.

Razza and colleagues (2015) indicate that the compostable, starch-based expanded bioplastic included in their analysis performs better than landfilled expanded polystyrene (EPS) for smog formation potential and mineral depletion, and performs similarly for ozone depletion. These results are consistent for both cradle-to-gate and cradle-to-grave impacts. Hottle and colleagues (2017) show mixed results depending on the particular impact category and materials being compared. For example, PLA performs better than landfilled PET for ecotoxicity and ozone depletion, while TPS does not. All the ratios in Figure 14 (panel a) for the human toxicity category are obtained from this study. Results indicate that composted PLA and TPS generally lead to lower human toxicity impacts than landfilled PET, HDPE, LDPE, bioPET, bioHDPE, and bioLDPE. In contrast to the results from Razza and colleagues (2015), these results are not consistent between production and total life cycle impacts, as the study shows that waste management strategies play an important role for these categories. Both studies, however, show that all compostable materials considered generally result in significantly greater impacts (meaning ratios >1.25) for both acidification and eutrophication categories than conventional materials. These ratios are driven by fertilizer and water use in the production of the biobased materials as well as heavy machinery use during the composting process (Hottle, Bilec, and Landis 2013). However, it should be noted that the magnitude increase of these categories is lower in absolute terms than the magnitude of reductions that composting achieved for GWP.

Composted Materials vs. Non-Compostable Materials Recycled and Incinerated.

Four studies included comparisons of composted and non-compostable, recycled materials, shown in Figure 14 (panel b). The biobased starch studied by Razza and colleagues (2015) results in lower production impacts for the compostable material, leading to the positive ratios for the global warming, smog formation, fossil energy depletion, and mineral depletion impact categories, regardless of whether the non-compostable material is recycled or landfilled. Papong and colleagues (2015) founds that composted PLA and chemically recycled PET perform similarly, with both materials producing similar life cycle GHG emissions despite the different end-of-life treatment. Quantis (2011) found that composting PLA based espresso capsules results in marginally higher impacts for GWP and energy use than recycling of aluminum based capsules.

Most of the comparisons for composting vs. recycling of non-compostable materials come from Hottle and colleagues (2017). The composted materials, PLA and TPS, are compared to recycled PET, HDPE, LDPE, bio-PET, bio-HDPE, and bio-LDPE. For GWP, the non-compostable recycled materials perform better due to the credits assigned to avoided production of virgin material by recycling. Additionally, the recycled bio-based polymers are credited with the offset of fossil-based plastics due to the large scale of existing plastics recovery infrastructure, resulting in lower impacts for the fuel depletion category. Conversely, the composted materials generally perform better than the recycled materials in the human toxicity and smog impacts, with mixed results for ozone depletion, and ecotoxicity categories. The lower impacts of the composted materials for these categories are due to the small magnitude of emissions related to the production, transportation, and composting of the materials, while the recyclable materials include the emissions of transportation from the U.S. to China, where they are recycled. The emissions related to the long-distance transportation are a main cause for the higher impacts in these categories for the recycled materials, though the magnitude of the impacts is considerably lower than the GWP and energy categories. Additionally, sensitivity analysis for the recycled materials where the transportation distance is reduced, and materials are modeled as recycled within the U.S., results in the impacts for non-compostable recycled materials being more comparable to the emissions from composting of the materials.

Three studies compared composted materials vs. non-compostable materials that are incinerated with energy recovery, shown in Figure 14 (panel c). A total of 177 comparisons were made, four of which were for the fossil energy demand category and the rest for GWP, and 168 of which came from Hermann and colleagues (2010). From this study, results with ratios less than one for compostable materials are mostly observed in the comparisons of laminates made from only one compostable material, such as PLA or cellulose, against complex laminates where made from multiple materials, not all of which allow for efficient energy recovery (e.g. aluminum). The lower amount of energy recovered in these scenarios results in lower electricity and heat offsets, which in turn drives higher GHG emissions for the non-compostable material than the GHG emissions stored in the compost. Papong and colleagues (2014) compared composted PLA bottles against incinerated PET bottles, with the lone comparison yielding a ratio of 0.65. This result was driven by the higher anthropogenic GHG emissions obtained from the incineration of the PET bottle when compared to the composting process of PLA bottles. The biogenic emissions for the composting of PLA were considered carbon neutral in this study. Quantis (2011) found that composting PLA based espresso capsules results in marginally higher impacts for GWP and energy use than incineration of aluminum based capsules.

Considering most impact metrics, composting was not found to consistently result in significantly lower impacts for a given compostable packaging material

While all the studies identified contain scenarios for compostable materials, eight studies provide comparisons between composting of the materials under review and impacts of other waste management options for the same material. Seven provide comparisons with landfilling of the same material; six provide comparisons with incineration; two studies provide comparisons for recycling and anaerobic digestion; and one for direct fuel substitution and home composting (as opposed to industrial composting, which is the default assumption in the other scenarios). Overall results from these comparisons suggest that composting is rarely the most favorable waste management option of compostable packaging materials for the different impact categories, as shown in Figure 15.

Composting vs. landfilling of compostable materials

Whether composting of compostable packaging materials results in lower GWP than landfilling the same materials depends on the rate of degradation of the materials in a landfill. Assumptions regarding degradation in landfills vary across studies. Leejarkpai and colleagues (2016) performed an experiment to test the degradation rate of PLA in a landfill, and found that PLA degrades anaerobically, causing significantly more GWP in a landfill than in an industrial composter. A similar assumption of high anaerobic degradation for PLA under landfill conditions was used by Quantis (2011), resulting in marginally lower GWP for composted PLA. The literature reviewed by Rossi and colleagues (2015) suggests that PLA does not degrade fully in landfill conditions in 100-year timeframe, while TPS does. Hottle and colleagues (2017) included both high and low degradation scenarios for PLA to model different literature assumptions, with the high degradation scenario having higher emissions than composting, while the low scenario does not. Krüger and colleagues (2009) assumed no degradation in landfill, resulting in fewer emissions for this waste management option. These mixed results are similar to those found by previous reviews of waste management options for organics (Morris, Matthews, and Morawski 2013) and food waste (Schott, Wenzel, and la Cour Jansen 2016), who also found the degree of degradation assumed by different LCAs to be a major contributing factor in the differences between results. For Hermann and colleagues (2010), the main factor is the fact that the study assumes landfill gas recovery for many compostable materials in landfill conditions, which grants energy credits to the landfilling scenarios that assume displacement of grid electricity consumption and thus lower total emissions from landfilling.

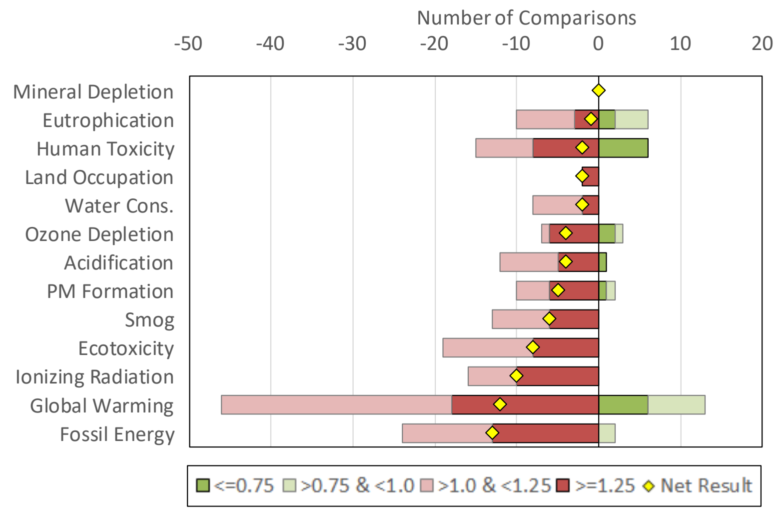


Figure 15. Comparison of life cycle impacts of composting of compostable packaging materials vs. other waste management options for the same compostable materials. *Ratios reflect the result for the composted packaging divided by the result for the same compostable packaging that is not composted. Thus ratios <1 indicate composted packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates composted packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

When comparing composting and landfilling of compostable materials for impact categories other than GWP, composting shows mixed results for eutrophication and ozone depletion, and performs worse for acidification, smog, ecotoxicity, and fossil energy depletion. However, the results for these categories are dependent upon the specific system boundary assumptions and system expansion credits assigned to the composting scenarios, which makes drawing more specific conclusions difficult.

Composting vs. Recycling and Incineration of compostable materials

Results are also mixed when comparing composting and other, non-landfilling dispositions like recycling or incineration. Due to the differences in scope, materials, and number of comparisons provided by each study, it is not possible to define with certainty a waste management hierarchy for compostable materials. However, some general trends can be observed.

Incineration and direct fuel substitution generally result in lower impacts for most categories due to energy recovery credits and displacement of other energy sources/processes (Krüger et al., 2009; Papong et al., 2014; Quantis, 2011; Rossi et al., 2015). The exception is for human health impacts, where composting avoids the particulate matter and human toxicity emissions from burning of the material. Recycling of biobased compostable polymers is shown to be better, or at the very least, comparable to composting those same materials, due to the avoided impacts of producing virgin material especially with regards to GHG and fossil resource use (Hottle, Bilec, and Landis 2017; Rossi et al. 2015). However, unless biobased compostable materials can be recycled using established infrastructure for more commonly recycled, fossil-based materials, this waste management option for compostable materials is limited.

Compostable packaging that is landfilled, recycled, or incinerated can be more impactful than non-compostable packaging that is landfilled, recycled, or incinerated

Four studies provide comparisons of compostable packaging which is landfilled, recycled, or incinerated versus non-compostable packaging that is landfilled, recycled, or incinerated, as shown in Figure 16. Note that these comparisons can be between different materials and management methods, such as incinerated PLA vs landfilled PET (see *Supporting Information B).* For GWP, Leejarkpai and colleagues (2016), Papong and colleagues (2016), and Hottle and colleagues (2017) provide comparisons of landfilled PLA vs. landfilled and recycled PET, with landfilled PLA performing considerably worse (most ratios > 1.4). This is predominantly due to the high degradation rates assumed for PLA under landfill conditions, coupled with the cradle to gate production emissions from land use change in the case of Leejarkpai and colleagues (2016). Comparisons with ratios lower than one for GWP for the compostable materials are those where low degradation rates are assumed under landfill conditions (Hottle and colleagues, 2017). When compared to non-compostable materials incinerated with energy recovery, landfilled PLA performs worse when landfilled without energy recovery, and similarly when landfilled or incinerated with energy recovery. Additionally, Quantis (2011) provides comparisons between landfilled and incinerated PLA espresso capsules vs. landfilled, incinerated, and recycled aluminum capsules for the GWP and energy depletion categories, with 47 out of 48 comparisons resulting in either marginally or significantly higher impacts for the composted PLA capsules.

Most the comparisons for categories other than GWP are obtained from Hottle and colleagues (2017). Most of the ratios below 0.75 for the human toxicity, ecotoxicity, fossil energy depletion, ozone depletion, and smog categories in Figure 16 come from comparisons of compostable PLA and TPS against non-compostable, fossil-based PET, HDPE, and LDPE, as well as bio-based versions of these polymers, made from blends of renewable and conventional feedstocks. These comparisons are between landfilled PLA and TPS vs. both landfilled and recycled conventional and bio-based polymers. The low ratios for these categories result from the fewer emissions produced by the compostable materials during the production phase than the emissions produced by fossil-based polymers and blended bio-polymers, regardless of which EOL treatment the non-compostable polymers undergo. The exceptions are the acidification and eutrophication categories due to the fertilizers used in the production of compostable PLA and TPS.

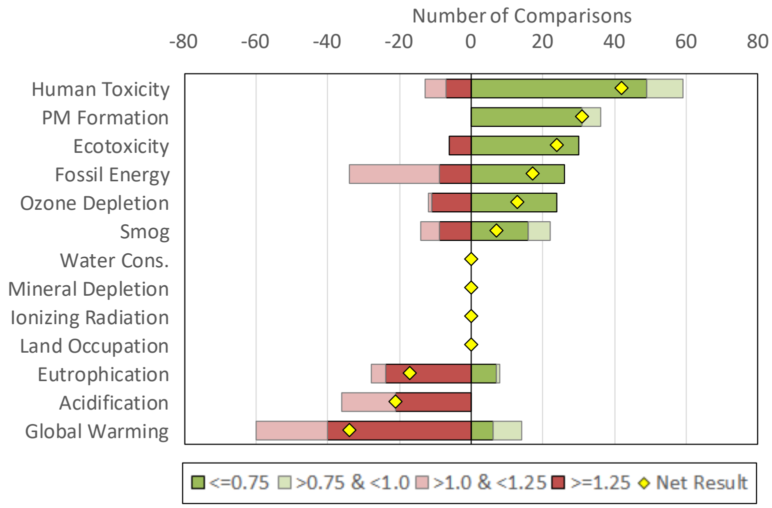


Figure 16. Comparisons of compostable packaging that is landfilled, incinerated, or recycled disposed with non-compostable packaging that is landfilled, incinerated, or recycled. *Ratios reflect the result for compostable packaging that is landfilled, incinerated, or recycled divided by the result for the non-compostable packaging that is landfilled, incinerated, or recycled. Thus ratios <1 indicate compostable packaging performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates compostable packaging performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

Other considerations related to compostable packaging

While the results discussed above provide valuable insight into the potential environmental tradeoffs of compostable packaging, there are several points worth noting that are not well-addressed in the literature. The following subsections discuss some of these issues.

Availability of Industrial-scale Composting Facilities. While composting may be the intended waste management strategy for most of these packaging materials, it is likely that a significant fraction of such materials will be landfilled. This is because landfilling is the end case scenario for about half of Municipal Solid Waste (MSW) in the U.S., while composting accounts for just under 9 percent (U.S. EPA Office of Resource Conservation and Recovery 2016).

Most of the studies reviewed assumed adequate facilities for composting (and the other waste management options they considered), but in reality the majority of dedicated composting facilities in the U.S. accept only yard trimmings and similar organic refuse (Platt et al. 2014). Facilities have little incentive to include compostable packaging products in their operations, for various reasons: increased risk of contamination to the compost product (e.g., microchemicals and metals, non compostable materials comingled with compostable packaging); increased operational costs from additional screening and labor; potential increases in compost production time as a result of longer material breakdown periods; and inadequate standards for compostable materials which result in such materials not composting under real operatin conditions. This lack of consistently accessible infrastucture for compostable packaging often means that these materials end up in landfills instead (Meeks et al. 2015).

Another difficulty for adequate disposal of compostable packaging is that often consumers do not have a clear understanding of the differences between biodegradable and compostable packaging, and between home and industrial composting. For commercial consumers, such as cafeterias and catering services, important factors that contributed to appropriate disposal of compostable products were the presence of a dedicated sustainability manager who understood the differences between biodegradable and compostable products and who could implement strategic, measureable sustainability goals, as well as the existence of a robust compost market (Meeks et al. 2015). For residential consumers, it has been shown that education of the public and adequate product labeling can help improve source separation by households so that compostable packaging can be collected with organic waste and thus sent to the appropriate facilities. Though such a scheme requires extensive planning, the compost resulting from streams with an increased amount of compostable packaging showed no difference in quality to compost wihtout the compostable packaging (Davis and Song 2006). In addition, significant amount of compostable packaging is marketed and used for takeout food items which due to its distributed nature contribute to contamination or be diverted to landfill.

Certification of compostability for packaging materials

It is important to distinguish between the LCA studies of compostable packaging and those of potentially compostable packaging materials or packaging precursors. Certification is often done on a product by product basis, and while resin manufacturers can provide certification for compostable resins they produce, the packaging manufacturers can modify the compostability of the resins by using additives and forming the packaging. Of the studies included in this review, five used certified compostable products or followed ISO or ASTM guidelines for identifying compostable products. Three studies used MaterBi ™, a material certified by the European Standard EN 13432 which defines the characteristics for a material to be compostable (Girgenti et al., 2014; Rossi et al., 2015; Hottle et al., 2017); Leejarkpai and colleagues (2016) followed ISO guidelines when setting up their degradability experiment; all of the materials reviewed by Hermann and colleagues (2010; 2011) meet the European standard EN 13432 as well as the ASTM D64004 and ASTM D68685 standards; and the polymer considered by Kruger and colleagues (2009) is certified by the Biodegradable Products Institute . Razza and colleagues (2015) and Papong and colleagues (2014) did not specify compostability certification of the materials used in their analyses.

Biodegradation of compostable plastic in the environment

Although it was included as part of the goal of the literature review, no literature was found with quantitive comparative results regarding the biodegradation of compostable packaging under ambient exposure conditions. However, Emadian and colleagues (2017) studied different types of compostable packaging under different environments, and found that the pH, moisture, oxygen content, and temperature of the environment, as well as the structure composition of the biopolymer play a significant role in the degree of biodegradation. Compostable bioplastics generally showed high degradability in soil environments, but generally do not degrade in fresh water and marine environments, making these materials comparable to conventional plastics in terms of their potential to harm freshwater and marine animals.

Opportunities for joint collection of compostable packaging and food waste
Promoting the joint collection of food waste and compostable packaging in a single waste stream has the potential to improve the collection rates of the latter, as suggested by Davis & Song (2006). However, none of the compostable packaging studies included in this review considered this dynamic, and during the literature search no studies were found that provided quantitative results for it, which reflects the fact that the single waste stream approach for organic waste and compostable packaging is still in the early stages of adoption. Additionally, while compostable packaging may deliver residual food scraps to the composter, the environmental impacts of composting food waste (without including compostable packaging) vary considerably from study to study, based on modeling assumptions (Schott, Wenzel, and la Cour Jansen 2016). Thus, while joint collection of compostable packaging and food waste could provide beneficial nutrient value to the compost made from packaging, the uncertain benefits of composting of food waste suggest more research is needed to determine how the environmental impacts of compostable packaging would change under such a scheme. Additionally, two studies were found that analyzed composting of tableware in conjunction with food waste. These are discussed in the food service ware section.

Summary – compostable packaging

Most of the studies included in this review for compostable packaging focused on GWP. Results show that use and composting of compostable packaging often results in higher GWP when compared to non-compostable materials and other waste management options for the compostable materials. Compostable materials also tend to be biobased, and accordingly produce higher acidification and eutrophication impacts than fossil-based materials due to the fertilizers and machinery needed in their production. Conversely, compostable packaging tended to perform better in the human toxicity, ecotoxicity, smog, and ozone depletion categories when compared to other materials such as PET, HDPE, and LDPE, due to the low emissions for the impact categories during their production, though this depends to a large extent on the specific materials being compared. Additionally, while the results are based over 1,200 individual comparisons, these come from a limited number of studies (ten), which makes the results particularly representative of the conditions and assumptions from those studies.

While compostable packaging represents a small share of the packaging market, consumer interest in sustainability has increased demand for it (Meeks et al. 2015). Unfortunately, the studies included in this review suggest that the use of compostable packaging has signficant environmental tradeoffs when compared with non-compostable materials and other end-of-life packaging management practices. Results suggest that other waste recovery strategies, such as recycling, may be preferable when considering the disposal of a specific compostable packaging material. Still, there are considerable uncertainties in the study findings due to different treatment of system boundaries, biogenic carbon accounting, and system expansion credits, indicating that more researchis needed to better understand the environmental impacts of compostable packaging.

In addition to mixed environmental perfomance, compostable packaging faces other obstacles to becoming more widely used, such as difficulties integrating compostable packaging into existing manufacturing processes; consumers’ confusion regarding the differences between what is biobased, biodegradable, and compostable, as this increases the difficulty of compostable packaging being properly collected and sorted by composting facilities; consistent availability of industrial composting facilities nationwide; and the lack of adequate markets for compost and facilities that accept compostable packaging (Meeks et al. 2015).

Food Service Ware Results

This section describes the results for food service ware (FSW) related to the four attributes previously presented for packaging. While the emphasis is on the impacts of the attributes on FSW, the similarities and differences of the findings for FSW with respect to packaging will be highligted. The search identified 11 relevant studies across all attributes for FSW, which included 106 different scenarios for FSW products and 1,344 comparisons. Cups were the most commonly studied product , included in seven of the studies and 783 comparisons. Other FSW products studied include lids, plates, dishes, napkins, straws, and cutlery, such as knives, forks, and spoons. GWP was included in all studies, while land occupation was the least common impact category included, only present in four studies. The biobased/renewable and compostable attributes were most commonly represented in the studies, being included in seven studies. The recycled content attribute was not adequately represented in the literature search for food service ware to draw broadly meaningful conclusions.

During the literature search for FSW for the four attributes, several of the studies found included analysis on the impacts of reusability of FSW in addition to one or more of the four main attributes. In addition, several studies were identified that focused exclusively on the environmental impacts of reusable FSW. While reusability as an attribute is out of the scope of this work for FSW and packaging, its prevalence in the literature indicates it is an active area of research. Accordingly, a short summary of the analysis of reusable FSW for the studies included in this report is found in the *Supporting Information A*.

Recycled Content in Food Service Ware

The effect of recycled content on the life cycle impacts of food service ware has not been well examined; aggregate results from one study suggest benefits from recycled content

We were not able to identify any studies that provided midpoint impact results for the recycled content attribute. Pladerer and colleagues’ (2008) study includes some comparisons of single use cups with and without recycled content as part of a sensitivity analysis, however these results are only provided in terms of an aggregate indicator based on the Environmental Burden Point (EBP) 2006 economic scarcity LCIA method. This LCIA method is reported as a single indicator score which aggregates weighted midpoint impact results and does not break out results by impact category, as indicated in Table 6.

Pladerer and colleagues (2008) analyzed cups made from PS and BELLAND ® material. BELLAND ® is an experimental plastic designed to be easily recycled. Results from the aggregate rating indicate that cups with recycled content are preferable to those without recycled content. More specifically, PS cups with 80 percent recycled content and BELLAND cups with 50 percent recycled content yielded 25 percent to 35 percent lower scores on the aggregate EBP

2006 indicator than virgin PS cups and BELLAND. These improvements are consistent with the findings for packaging with recycled content. However, Pladerer and colleagues also found that paperboard cups with no recycled content yielded lower scores than the plastic cup options with recycled content, a finding that echoes other findings that product design and material choice can have more influence on environmental performance than recycled content as an attribute without further context.

It is possible that many suppliers include some amount of post-consumer recycled content (less than 20 percent) in FSW products such as PET cold cups, paper cups, and paperboard clamshells.[15] While some research suggests that particular materials with recycled content may not be suitable for food contact due to chemical contamination (Binderup and colleagues, 2002; Biedermann and Grob, 2010), it is worth noting that the U.S. Food and Drug Administration (FDA) does not prohibit recycled content in packaging that is in contact with food. The FDA also issues letters of no objection to qualifying post-consumer plastic reclaimers (U.S. Food and Drug Administration 2018). Additionally, some organizations maintain rigorous testing protocols as a means for voluntary provision of food safety assurance for recycled materials used in the manufacture of FSW, such as the Recycled Paperboard Technical Association's "Comprehensive Program for Food-Contact Paperboard Produced From Recycled Fiber".

Recyclable Food Service Ware

Scope

We identified two studies of recyclable FSW that provide comparisons between FSW recycled at the EOL with FSW not recycled or recycled at a lower rate at EOL. Materials included in both studies were PS, PP, PLA, and paper. Table 7 and Figure 17 provide a summary of these studies, which provide 654 comparisons between FSW recycled at EOL and FSW that is not recycled or recycled at a lower rate. Of these, 192 comparisons were between FSW of the same material and 462 comparisons were between FSW of different materials. All the comparisons are between recyclable FSW that is recycled at EOL and other recyclable FSW that is not recycled or recycled at a lower rate at EOL, with recycling rates ranging from 100 to 0 percent depending on the specific comparisons. We did not find any comparisons between recyclable FSW and non-recyclable FSW that were suitable for inclusion in this review. Both studies included most impact categories tracked in this review.

As with the results for recyclable packaging, the definition for recyclable FSW requires the existence of infrastructure making it feasible to collect and reprocess post-consumer materials. Thus, it is possible that a packaging format considered recyclable in one area may not be considered recyclable in another due to the lack of adequate collection services and/or reprocessing facilities. This fine point makes generalizing comparisons between recyclable and non-recyclable FSW challenging. Therefore, most of the comparisons discussed in this section are between packaging that is recycled at its end-of-life and another packaging option that is recyclable, but which is either recycled at a lower rate or not recycled at its end-of-life.

[15] Adam Gendell, Associate Director, Sustainable Packaging Coalition, personal communication, March 2, 2018.

Table 6. Scope of Pladerer and colleagues' study described in this section.

					Materials			Boundary			Result Categories												
Author and Year	*FSW Product*	*Functional Unit*	*Geography*	*LCIA Method*	Fossil-based Plastic Resins	Bio-based Plastic Resins	Paper/Cardboard	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use	Aggregare Indicator
Pladerer et al., 2008[1]	Cups	0.5 L drink	Germany, Austria, Switzerland	EBP 2006	x	x	x	x	x	x													x
Percent of Total Number					100	100	100	100	100	100	0	0	0	0	0	0	0	0	0	0	0	0	100

[1] Included an aggregate impact score comparing cups with and without recycled content as part of sensitivity analysis, but results are not provided for individual impact categories.

Table 7. Scope of studies included in the recyclable FSW comparisons.

					Materials			Recyclability	Boundary			Result Categories												
Author and Year	*FSW Product*	*Functional Unit*	*Geography*	*LCIA Method*	Fossil-based Plastic Resins	Bio-based Plastic Resins	Paper/Cardboard	Recycling Allocation Method	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Pro.Mo, 2015	Cups	1,000 meals/drinks	Italy	ILCD 2011 midpoint	x	x	x	AB	x	x	x	x	x	x	x	x	x	x	x	x		x	x	x
Potting and van der Harst, 2015	Cups	Serving of one hot beverage from vending machine	The Netherlands	CML 2001 baseline, Ecoinvent CED	x	x	x	AB	x		x	x	x	x		x	x		x	x	x	x		
Percent of Total Number					100	100	100	100	100	50	100	100	100	100	50	100	100	50	100	100	50	100	50	50

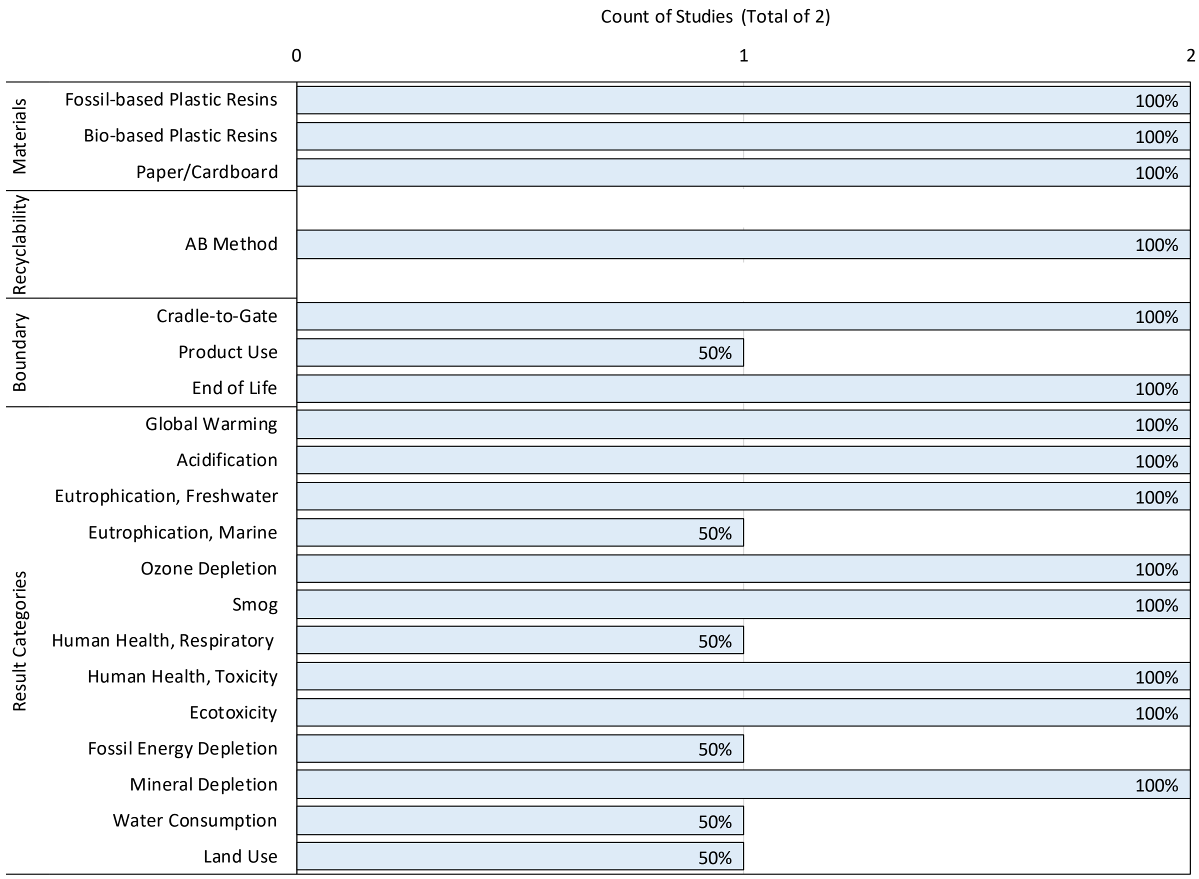


Figure 17. Scope of studies included in the recyclable FSW comparisons.

The two studies identified suggest recyclable food service ware is preferable to non-recyclable (or not recycled) food service ware

For comparisons between FSW of different materials, shown in Figure 18 (panel a), recyclable FSW that is recycled or which has a higher recycling rate at EOL is frequently found to have lower impacts across all impact categories. More specifically, when comparing FSW of different materials, recycled or more highly recycled FSW has significantly lower impacts in 258 comparisons, marginally lower impacts in 99 comparisons, marginally higher impacts in 58 comparisons, and significantly higher impacts in 44 comparisons. The ecotoxicity, human toxicity, and mineral depletion impact categories are responsible for 30 of the significantly higher impact comparisons.

Figure 18 (panel b) shows comparisons between recyclable FSW recycled at the EOL with FSW of the same material that is not recycled or recycled at a lower rate at EOL. Recyclable FSW that is recycled or has a higher recycling rate at EOL is usually found to have lower impacts, although for many comparisons the difference is marginal. More specifically, recyclable FSW that is recycled or has a higher recycling rate at EOL has significantly lower impacts in 44 comparisons (23 percent), marginally lower impacts in 145 comparisons (76 percent), and marginally higher impacts in three comparisons (2 percent). Recyclable products compared include cups and dishes made of PS and PP. The scenarios that result in marginally higher impacts (ratios between 1.01 and 1.05 for GWP) are when PP dishes are disposed of with a mix of recycling and incineration vs. when they are disposed via landfill. These are the only ratios > 1 in Figure 18 (panel b), which are caused by the higher GHG emissions caused from burning of the PP dishes for incineration with energy recovery as opposed to landfilling them.

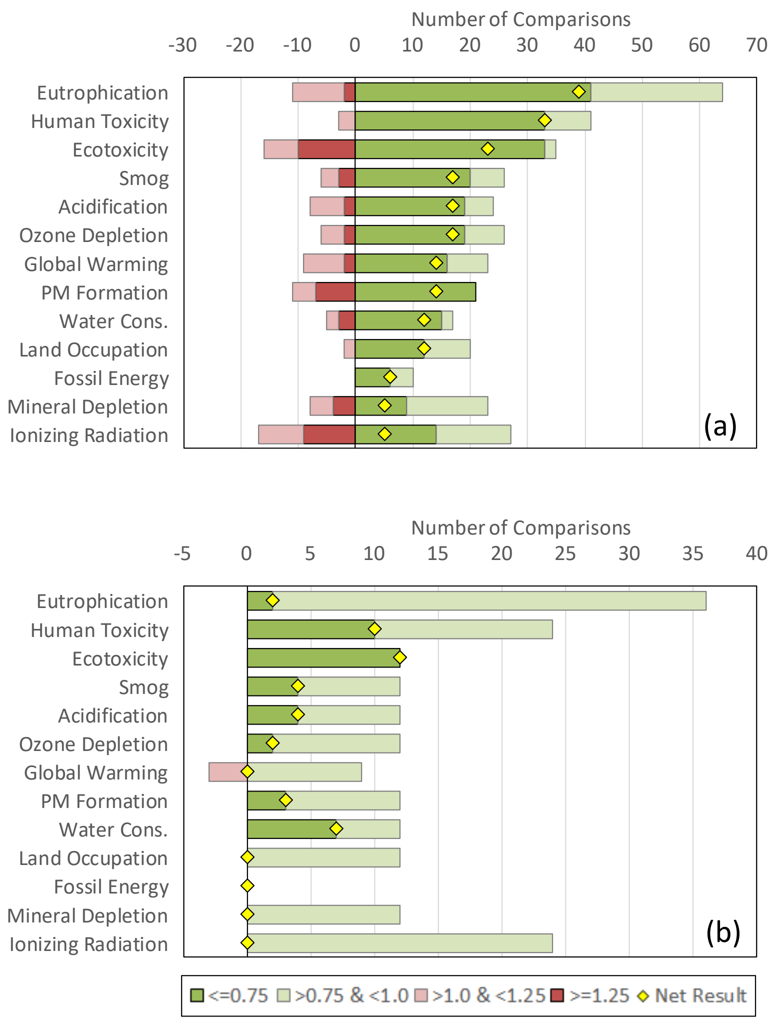


Figure 18. Comparisons of FSW products that are recycled to FSW products of a different material (a) and same material (b) that is not recycled or recycled at a lower rate.
Ratios reflect the result for the recyclable FSW products that are recycled divided by the result for the FSW products that are not recycled or recycled at a lower rate. Thus ratios <1 indicate recycled FSW performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates recycled FSW performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons. All panels sorted by net result of comparisons from panel a.

It should be noted that the studies included in Figure 18 (panels a and b) do not explicitly mention food contamination when considering the recycling rates of the analyzed products. Since contamination from food can impact the rate at which some FSW items are accepted for recycling, these estimates should be considered a best-case scenario for recycling these items. Specific studies included in the comparisons for recyclable FSW are described below.

Environmental impacts of recyclable FSW can depend on the specific product being analyzed. For example, a study by Pro.mo, an industry group consisting of six Italian companies that produce about 80 percent of the plastic tableware in the country, analyzed the impacts of disposable cups and dishes made from PS, PP, PLA, and cellulose pulp (Pro.mo Industry Group, 2015). Comparisons between PS and PP based products that are 50 percent recycled and 50 percent incinerated are made against the same type of product made from a different material (e.g., PS vs. PP, PLA, or paperboard) that is disposed via landfill or a mixture of landfill and incineration. The recycled PP and PS cups performed better in 47 percent of the comparisons, while the recycled PP and PS dishes performed better in 58 percent (128 comparisons each). The difference in performance between cups and dishes is likely caused by the differences in the amount of materials used for the manufacturing of the two FSW products.

Potting and van der Harst (2015) analyzed PS, PLA, and paperboard cups. The study suggests that while impacts from recycling are generally lower than incineration and composting for these materials, results are sensitive to the specific life cycle production data used and the modeling assumptions of the various end-of-life disposal options. For example, results from this study show that recycling of PS cups result in lower impacts when compared to incineration and composting of PLA and paperboard cups for all categories included in Table 7 except for mineral depletion. In contrast, incineration of PS cups results in lower emissions than recycling of PLA and paperboard cups only for the GWP, energy depletion, and mineral depletion categories. This is because incineration at EOL is assigned higher avoided burden credits than recycling due to displacement of electricity from the Dutch grid, which is highly dependent of fossil fuels. Sensitivity analysis for different grid mixes reverse this result, as the use of a cleaner grid results in fewer credits for avoided burdens from incineration than from displaced material production from recycling.

The results for the comparisons for recyclable FSW products differ from the results for recyclable packaging, which showed mixed results overall (see Figure 7). This is primarily due to differences in the materials used for food service ware versus packaging. The unfavorable results for recyclable packaging were mainly comparisons of glass and metals with lighter materials such as plastics and aseptic cartons. Glass and metal are generally not used for disposable FSW products, and thus there is less variability in the production impacts and avoided burdens from recycling in the FSW comparisons.

Biobased Food Service Ware

Scope

We identified seven studies with LCA comparisons between biobased and non-biobased FSW. Table 8 and Figure 19 provide an overview of the scope of each of these studies. Biobased materials include PLA, cellulose and molded fiber, paper, and board. None of the studies included considered biobased versions of conventional polymers, such as bioPET. These studies provided 327 comparisons between biobased and non-biobased FSW products. All seven studies included the cradle to gate and EOL life cycle stages, while only four included the use phase.

GWP was the most commonly analyzed impact category, included in all studies, followed by acidification and fossil energy depletion, included in five each. Even though land use can be a significant source of emissions for biobased products due to impacts of feedstock growth, it was one of the least analyzed categories, included only in two studies.

The comparisons of biobased FSW presented in this section are limited to comparisons between biobased and non-biobased FSW that undergo the same EOL treatment. Due to the mostly negative results of compostable, biobased FSW comparisons (see next section), we wanted to ensure that the results for the biobased comparisons are not mostly driven by the composting of the FSW products. Thus, the comparisons in this section focus on the impacts of the biobased attribute, without differences in waste management methods confounding the results.

Table 8. Scope of studies included in the biobased FSW comparisons.

					Materials				Boundary			Result Categories												
Author and Year	*FSW Product*	*Functional Unit*	*Geography*	*LCIA Method*	Fossil-based Plastic Resins	Bio-based Plastic Resins	Paper/Cardboard	Composite/Other	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Pro.Mo, 2015	Dishes and cups	1,000 meals/drinks	Italy	ILCD 2011 midpoint	x	x	x		x	x	x	x	x	x	x	x	x	x	x	x		x	x	x
Potting and van der Harst, 2015	Cups	Serving of one hot beverage from vending machine	The Netherlands	CML 2001 baseline, Ecoinvent CED	x	x	x		x		x	x	x	x		x	x		x	x	x	x		
Broca, 2008	Plates	Dishwasher load, 2,960 plates	United States	Inventory based, EcoIndicator 99		x		x	x	x	x	x	x			x		x	x	x	x	x		x
Pladerer et al., 2008	Cups	0.5 L drink	Germany, Austria, Switzerland	UBA (German Ministry of the Environment) Method	x	x	x		x	x	x	x	x	x			x	x			x			
PE Americas, 2009	Drinking cups and flat lids	16-ounce single use cold beverage cup with flat lid	United States	CML	x	x			x		x	x	x	x			x				x		x	
Franklin Associates, 2011[1]	Hot and cold cups, plates, clamshells	10,000 items of each FSW product	United States	IPCC 2007	x	x			x		x	x									x		x	
Hakkinen and Vares, 2010	Cups	100,000 cups	Europe	Not specified	x		x		x	x	x	x												
Percent of Total Number					86	86	57	14	100	57	100	100	71	57	14	43	57	43	43	43	71	43	43	29

[1] Update of previous publication

Studies ordered from most to least coverage of result categories

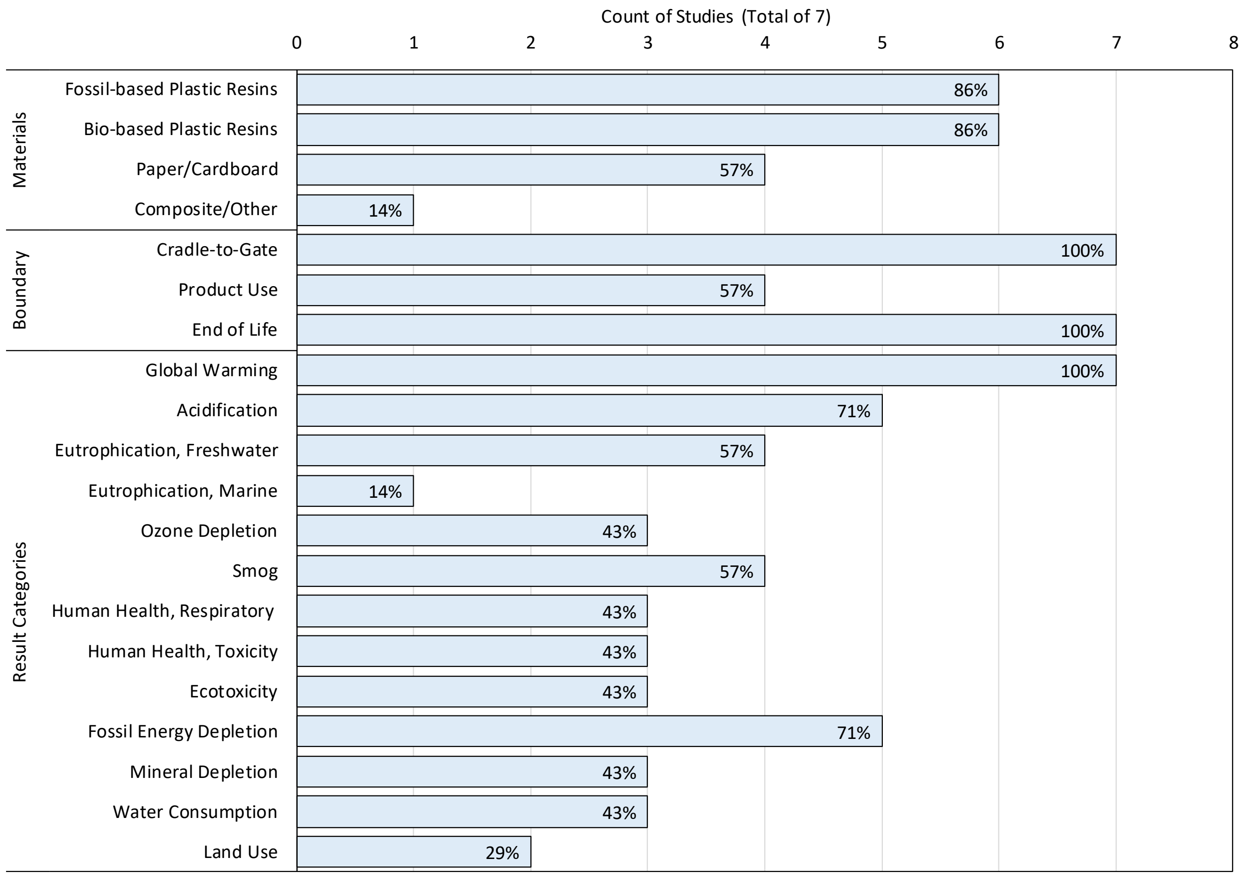


Figure 19. Scope of studies included in the biobased FSW comparisons.

Biobased FSW tends to perform worse for eutrophication, water use, acidification, land use, and toxicity potentials and slightly better for global warming potential

Biobased FSW was found to have significantly higher impacts in 191 comparisons and significantly lower impacts in 83 comparisons. Figure 20 shows the results for all such comparisons.

The results of comparing biobased and non-biobased FSW are like those for biobased and non-biobased packaging, previously presented in Figure 12. Results are driven by the higher production impacts for biobased products for most categories, though impacts vary by specific material and EOL modeling assumptions. As with packaging, biobased FSW shows mixed results tending toward improved performance for GWP and energy demand and tending toward worse performance for eutrophication, water use, acidification, ozone depletion, land use, and toxicity potentials impact categories. Out of 47 comparisons of the GWP of biobased FSW products, 25 are significantly lower, while 13 are significantly higher than their non-biobased counterparts. On the other hand, at least 61 percent of comparisons for the acidification, eutrophication, human toxicity, and ecotoxicity categories result in higher impacts. The results are even more extreme for the ozone depletion, water use, and land use categories, where at least 89 percent of comparisons result in significantly higher impacts. These impacts are mostly caused during the growing and processing of the feedstocks for biobased materials, as discussed in the biobased packaging section.

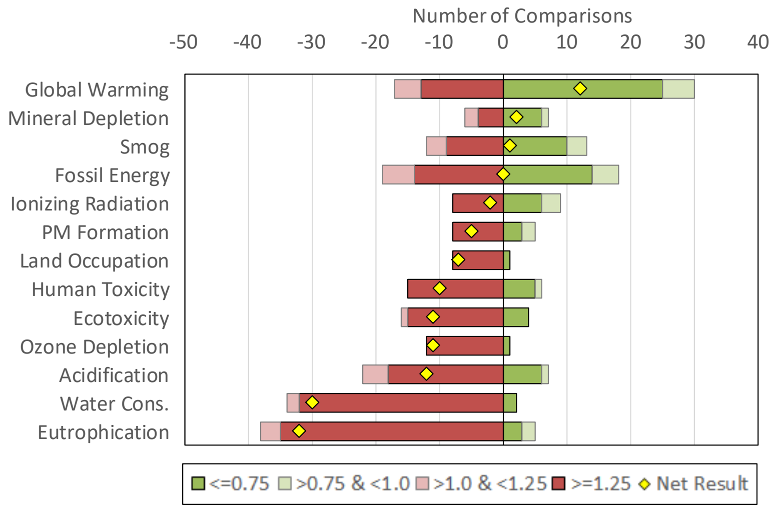


Figure 20. Comparisons of biobased FSW products to non-biobased FSW products.
Ratios reflect the result for biobased FSW divided by the result for the non-biobased FSW. Thus ratios <1 indicate biobased FSW performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates biobased FSW performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons

PLA is most studied biobased polymer used in FSW, and most comparisons between biobased PLA and non-biobased FSW show significantly higher impacts for the FSW made of PLA. These high impacts are caused by the emissions during feedstock production and resin manufacturing for PLA. For example, Pladerer and colleagues (2008), Potting and van der Harst (2015), and Häkinnen and Vares (2010) compared incineration of PLA vs. PS and PET cups; PE Americas (2009) and the Pro.mo industry group (2015) compared landfilling of PLA vs. PS cups and dishes; and Franklin Associates (2011) compared cups made from PLA vs. EPS and general purpose polystyrene (GPPS), 80 percent landfilled and 20 percent incinerated at EOL. From these studies, a total of 211 comparisons were performed for FSW products made from PLA. Across all impact categories, 129 comparisons show significantly higher impacts for PLA while 47 comparisons show significantly lower impacts for PLA.

Despite the overall higher impacts for PLA based FSW, the results vary depending on which materials PLA is being compared against. Comparisons between PLA and PS/PP products consistently resulted in ratios > 1.25 for PLA in most impact categories (Pro.mo industry group, 2015; Potting and van der Harst, 2015). Comparisons between PLA and PET yielded mixed results: PLA generally resulted in lower impacts (ratios <0.75) for GWP and fossil energy depletion categories, while resulting in significantly higher impacts for the acidification, eutrophication, and water depletion categories (PE Americas, 2009). PLA only compared favorably for most impact categories against ceramic plates that are assumed to be used once and then landfilled (Broca 2008), which is an unusual scenario. This is due to the higher production impacts from ceramics (specifically, energy used in kilns during the glazing of ceramic plates).

Like the recyclability attribute, the specific FSW products can influence environmental impacts. For example, 19 out of 64 comparisons between landfilled PLA and paperboard cups against PS/PP cups, all landfilled, resulted in significantly lower emissions for the biobased product. In contrast, only six of 64 similar comparisons between PLA and paperboard dishes against PP/PS dishes resulted in significantly lower impacts for the biobased product (Pro.mo Industry Group, 2015). The lower number of low ratio comparisons for dishes is likely a result of the higher weight of the dishes when compared to cups, and the fact that biobased material production produces more emissions than production of PP/PS resin.

GWP for paper and paperboard FSW were also found to be sensitive to landfill degradation rates. Franklin Associates (2011) studied paperboard cups, dishes, and clamshells with various coatings that are landfilled at EOL and found that a low degradation rate lead to significantly lower GWP estimates for paper based products while a high degradation rate lead to significantly higher GWP. These results comprise 12 comparisons shown in Figure 20, six for each assumption.

Compostable Food Service Ware

Scope

Seven studies provided 363 comparisons between compostable and non-compostable FSW. Compostable FSW products studied include cups, plates, clamshells, and cutlery while the materials studied include PLA, paperboard, cellulose pulp, and paper and board. All seven studies included the cradle to gate and EOL stages, while only four included the product use stage. GWP was the most common impact category analyzed, as it was included in all seven studies. The least represented impact category is land use, present in only two studies. Since compostable materials included in the comparisons in this section are biobased, land use can potentially represent a significant source of emissions due to feedstock growth, as mentioned in the biobased FSW section.

Table 9. Scope of studies included in the compostable FSW comparisons

					Materials				Boundary			Result Categories												
Author and Year	*FSW Product*	*Functional Unit*	*Geography*	*LCIA Method*	Fossil-based Plastic Resins	Bio-based Plastic Resins	Laminated Paper Board	Paper/Cardboard	Cradle-to-Gate	Product Use	End of Life	Global Warming	Acidification	Eutrophication, Freshwater	Eutrophication, Marine	Ozone Depletion	Smog	Human Health, Respiratory	Human Health, Toxicity	Ecotoxicity	Fossil Energy Depletion	Mineral Depletion	Water Consumption	Land Use
Fieschi and Pretato, 2017	Tableware	1,000 single use tableware	Italy	Impact 2002+	x	x		x	x		x	x	x	x	x	x	x	x	x	x		x	x	x
Pro.Mo, 2015	Dishes and cups	1,000 meals/drinks	Italy	ILCD 2011 midpoint	x	x		x	x	x	x	x	x	x	x	x	x	x	x	x		x	x	x
Potting and van der Harst, 2015	Cups	Serving of one hot beverage from vending machine	The Netherlands	CML 2001 baseline, Ecoinvent CED	x	x		x	x		x	x	x	x		x	x		x	x	x	x		
Vercalsteren et al, 2010[1]	Cups	100 L of beverage	Belgium	Eco-Indicator 99	x	x	x		x	x	x	x	x			x		x	x	x	x	x		
Pladerer et al., 2008[2]	Cups	0.5 L drink	Germany, Austria, Switzerland	UBA (German Ministry of the Environment) Method	x	x		x	x	x	x	x	x	x			x	x			x			
Razza et al., 2009	Cutlery	Serving 1000 meals	Italy	Impact 2002+	x	x			x		x	x	x	x							x			
Harnoto, 2013	Clamshells	360 uses	United States	Inventory based	x	x			x	x	x	x									x		x	
Percent of Total Number					100	100	14	57	100	57	100	100	86	71	29	57	57	57	57	57	71	57	43	29

[1] Update of previous publication
[2] Included as part of sensitivity analysis, but quantitative results are not available for individual impact categories.
Studies ordered from most to least coverage of result categories

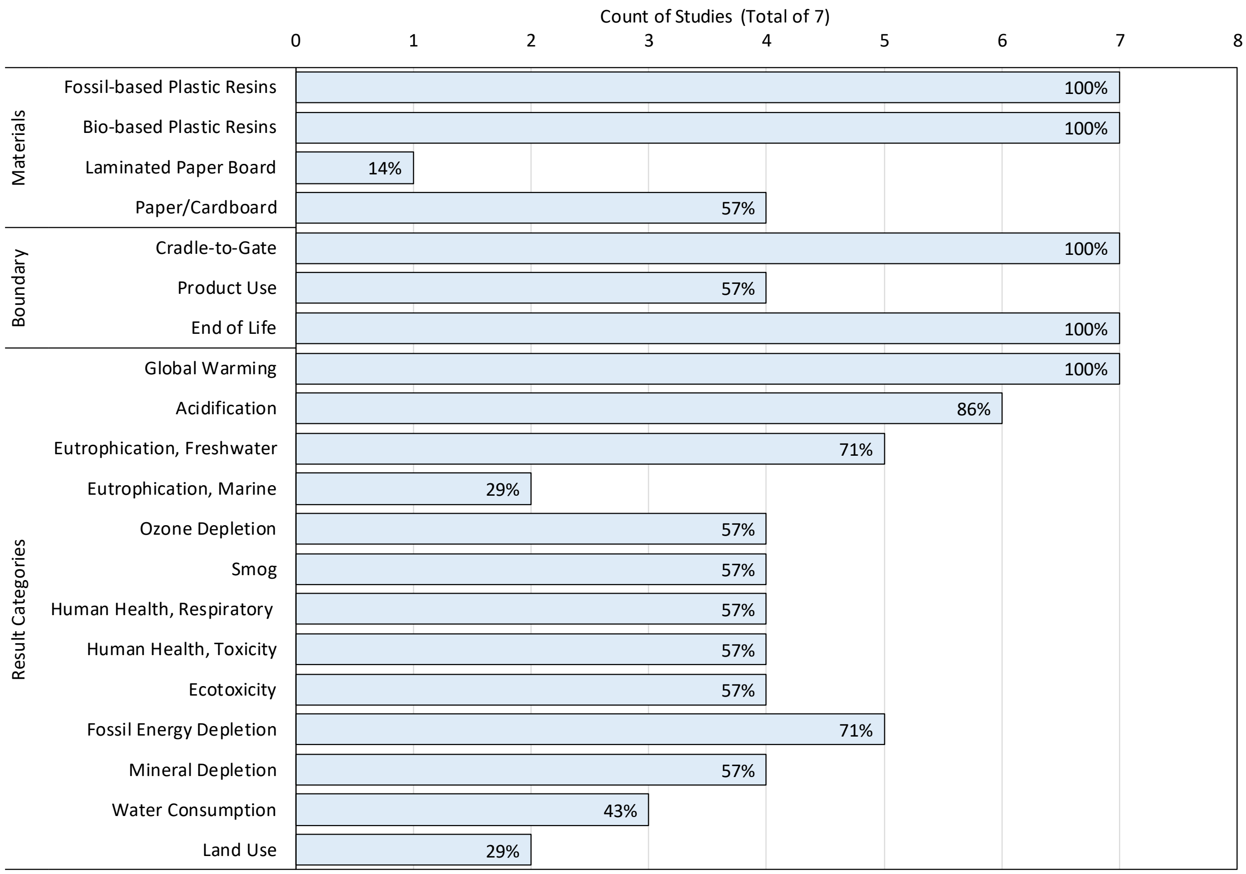


Figure 21. Scope of studies included in the compostable FSW comparisons.

Composting FSW products generally performs worse than landfilling, incinerating, or recycling

Figure 22 shows the comparisons of composted FSW against non-compostable FSW that is landfilled, incinerated, or recycled. Figure 23 shows the comparisons of composted FSW against compostable FSW that is incinerated or recycled. In both figures, composting compostable FSW tends to result in increased impact potentials for most categories when compared with landfilling, incinerating, or recycling it. 261 comparisons showed significantly higher impacts for compostable FSW while 37 comparisons showed significantly lower impacts across both figures. For the comparisons between compostable and non-compostable FSW, the primary reason for these results are the higher production impacts of compostable materials, which are mostly biobased PLA and fiber-based products in the LCA studies identified. For the comparisons between composted FSW and compostable FSW that is incinerated or recycled, the higher impacts for composted FSW are driven by the relatively low avoided burdens credited to compost vs. the avoided burdens credited to incineration or recycling of the compostable materials (Potting and van der Harst 2015). These findings for compostable FSW are consistent with those for compostable packaging previously presented in Figure 14. Select comparisons are described in more detail below.

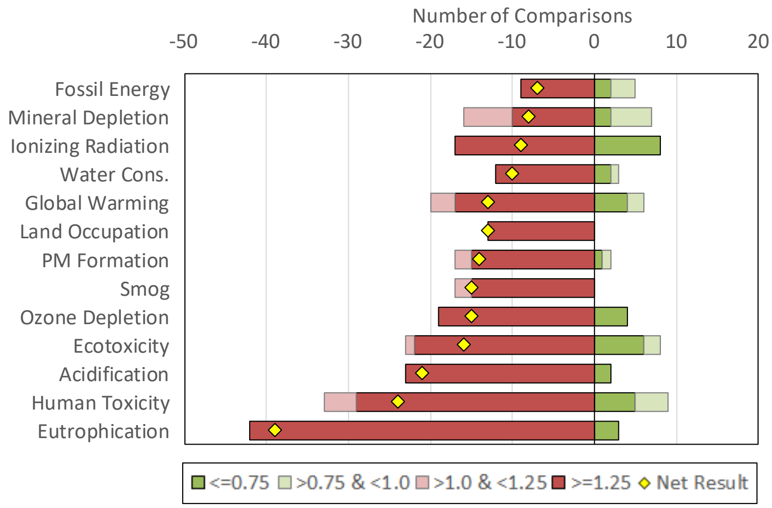


Figure 22. Comparisons of the life cycle impacts of composted FSW with non-compostable FSW. *Ratios reflect the result for composted FSW divided by the result for the non-compostable FSW. Thus ratios <1 indicate compostable FSW performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates compostable FSW performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

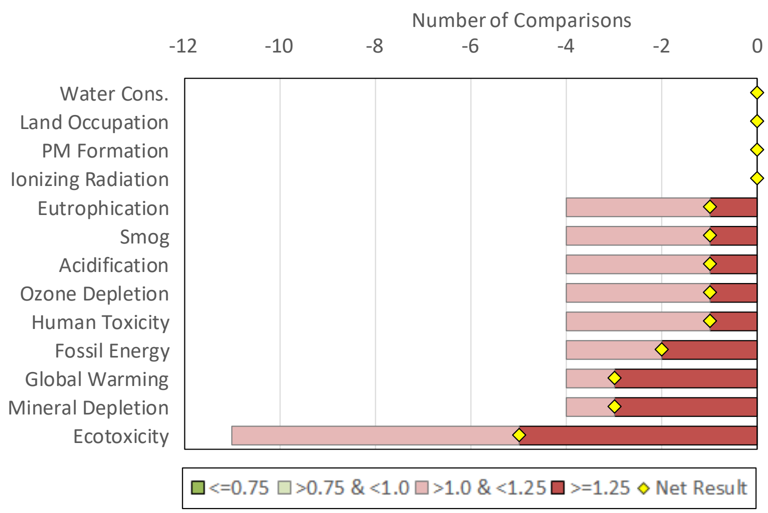


Figure 23. Comparisons of the life cycle impacts of composted FSW with compostable FSW that is not composted. *Ratios reflect the result for composted FSW divided by the result for compostable FSW that is not composted. Thus ratios <1 indicate composted FSW performs better and are shown in the figure in green as the positive number of comparisons while ratios >1 indicates composted FSW performs worse and are shown in the figure in red as the negative number of comparisons. Dark green and dark red represent counts of comparisons with ratios <0.75 and >1.25 respectively and are considered meaningful differences. Light green and light red represent counts of comparisons with ratios 0.75-0.99 and 1.01-1.25 respectively. The figure is sorted by the net result of comparisons representing meaningful differences, i.e. the number of dark green minus dark red comparisons.*

Potting and van der Harst (2015) compared compostable PLA and paperboard cups, assumed to be 100 percent composted to cups of the same materials assumed to be 100 percent incinerated or 100 percent recycled. All the comparisons showed at least marginally worse performance for composting across all impact categories, and 18 of 44 comparisons demonstrated significantly worse performance, as shown in Figure 23. These results are consistent with those for compostable packaging previously presented in Figure 15.

An interesting finding was that most of the comparisons between compostable and non-compostable tableware resulted in ratios lower than 1, which is not the case when only considering compostable cups, dishes, or clamshells. Razza and colleagues (2009) and Fieschi and Pretato (2018) analyzed the impacts of single-use compostable tableware products. The items compared are knives, forks, and spoons in both studies, and in addition to biodegradable napkins and tray liners made from single ply paper in Fieschi and Pretato (2018). Both studies showed significantly GWP for compostable tableware when compared to non-compostable tableware. Additionally, Razza and colleagues showed significantly lower impacts in the energy depletion category, while Fieschi and Pretato showed marginally lower impacts in the mineral and water depletion categories. The results from these studies are driven by the collection of the compostable tableware and the food waste in a single waste stream, unlike the other studies that did not explicitly assume that the FSW is co-collected along with other organic waste. Combining both types of waste reduces the impacts that would otherwise be assigned to the different waste collection and different EOL impacts for the two waste streams, as additional compostable material is diverted to be composted, where the treatment of wet bio-waste generates less impacts compared to incineration or landfilling. Additionally, the added organic materials from food waste in the compost provide a greater amount of nutrients and aid the composting of FSW, which depending on its composition might not happen in isolation. It should be noted that co-collection of food waste and compostable packaging/FSW is restricted in some areas, such as Portland Oregon.

Summary – Food Service Ware

Fewer LCA studies have been performed for FSW than for packaging, therefore our findings for the life cycle environmental impacts of FSW with recycled content and recyclable, compostable, and biobased FSW are based on fewer comparisons than those for packaging presented in the previous section. The available studies indicate that some, but not all, of the conclusions drawn from the review of the various attributes for packaging also apply to FSW. No LCA studies providing midpoint impact results for FSW with recycled content were identified. We found that recycling of FSW products generally results in lower impact potentials when compared with landfilling or incineration. Biobased FSW is generally not preferable to fossil-based FSW. This is because production impacts for biobased materials tend to be higher than for conventional materials. Compostable FSW is generally not preferable to non-compostable FSW, as they are generally biobased, resulting in higher production impacts than fossil-based materials, and receiving less credits at end-of-life than other waste management options. A possible exception is a case where FSW is collected and composted with food waste due to improvements in collection efficiency and the increased nutrient content of the compost resulting from the increased amount of organic material composted along with the FSW. However, the only studies found that explored this option focused on compostable tableware and cutlery, which suggests more research is needed to fully ascertain the benefits of co-collection of compostable FSW and other organic waste.

References

Akanuma, Yasuhiko, Susan E. M. Selke, and Rafael Auras. 2014. "A Preliminary LCA Case Study: Comparison of Different Pathways to Produce Purified Terephthalic Acid Suitable for Synthesis of 100 % Bio-Based PET." *The International Journal of Life Cycle Assessment* 19 (6): 1238–46. https://doi.org/10.1007/s11367-014-0725-2.

Amienyo, David, and Adisa Azapagic. 2016. "Life Cycle Environmental Impacts and Costs of Beer Production and Consumption in the UK." *The International Journal of Life Cycle Assessment* 21 (4): 492–509. https://doi.org/10.1007/s11367-016-1028-6.

Amienyo, David, Cecil Camilleri, and Adisa Azapagic. 2014. "Environmental Impacts of Consumption of Australian Red Wine in the UK." *Journal of Cleaner Production* 72 (June): 110–19. https://doi.org/10.1016/j.jclepro.2014.02.044.

Amienyo, David, Haruna Gujba, Heinz Stichnothe, and Adisa Azapagic. 2013. "Life Cycle Environmental Impacts of Carbonated Soft Drinks." *The International Journal of Life Cycle Assessment* 18 (1): 77–92. https://doi.org/10.1007/s11367-012-0459-y.

Arena, Umberto, Maria Laura Mastellone, and Floriana Perugini. 2003. "Life Cycle Assessment of a Plastic Packaging Recycling System." *The International Journal of Life Cycle Assessment* 8 (2): 92–98. https://doi.org/10.1007/BF02978432.

ASTM International. 2012. "ASTM Standard D6400, 'Specification for Labeling of Plastics Designed to Be Aerobically Composted in Municipal or Industrial Facilities.'" ASTM D6400-12. West Conshohocken, PA.

———. 2017. "ASTM Standard D6868, 'Specification for Labeling of End Items That Incorporate Plastics and Polymers as Coatings or Additives with Paper and Other Substrates Designed to Be Aerobically Composted in Municipal or Industrial Facilities.'" ASTM D6868-17. West Conshohocken, PA.

Atherton, John. 2007. "Declaration by the Metals Industry on Recycling Principles." *The International Journal of Life Cycle Assessment* 12 (1): 59–60. https://doi.org/10.1065/lca2006.11.283.

Beauregard, Sandy, Richard Bi, Hillary Price, and Mike Zargham. 2007. "Lifecycle Analysis of Chinese Food Take-Out Containers." Dartmouth University. https://engineering.dartmouth.edu/~d30345d/courses/engs171/ChineseFoodContainers.pdf.

Belley, C. 2011. "Comparative Life Cycle Assessment Report of Food Packaging Products." Montreal, QC.

Biedermann, Maurus, and Koni Grob. 2010. "Is Recycled Newspaper Suitable for Food Contact Materials? Technical Grade Mineral Oils from Printing Inks." *European Food Research and Technology* 230 (5): 785–96. https://doi.org/10.1007/s00217-010-1223-9.

Binderup, M. L., G. A. Pedersen, A. M. Vinggaard, E. S. Rasmussen, H. Rosenquist, and T. Cederberg. 2002. "Toxicity Testing and Chemical Analyses of Recycled Fibre-Based Paper for Food Contact." *Food Additives & Contaminants* 19 (sup1): 13–28. https://doi.org/10.1080/02652030110089878.

Bohlmann, Gregory M. 2004. "Biodegradable Packaging Life-Cycle Assessment." *Environmental Progress* 23 (4): 342–46. https://doi.org/10.1002/ep.10053.

Broca, Mita. 2008. "A Comparative Analysis of the Environmental Impacts of Ceramic Plates and Biodegradable Plates (Made of Corn Starch) Using the Life Cycle Assessment Tool." Masters, New Delhi, India: TERI School of

Advanced Studies. http://sustainability.tufts.edu/wp-content/uploads/LifeCycleAnalysisPlasticPlatevsCeramic.pdf.

Chen, Luyi, Rylie E.O. Pelton, and Timothy M. Smith. 2016. "Comparative Life Cycle Assessment of Fossil and Bio-Based Polyethylene Terephthalate (PET) Bottles." *Journal of Cleaner Production* 137 (November): 667–76. https://doi.org/10.1016/j.jclepro.2016.07.094.

Cleary, Julian. 2013. "Life Cycle Assessments of Wine and Spirit Packaging at the Product and the Municipal Scale: A Toronto, Canada Case Study." *Journal of Cleaner Production* 44: 143–151.

Cózar, A., F. Echevarria, J. I. Gonzalez-Gordillo, X. Irigoien, B. Ubeda, S. Hernandez-Leon, A. T. Palma, et al. 2014. "Plastic Debris in the Open Ocean." *Proceedings of the National Academy of Sciences* 111 (28): 10239–44. https://doi.org/10.1073/pnas.1314705111.

Davis, G., and J.H. Song. 2006. "Biodegradable Packaging Based on Raw Materials from Crops and Their Impact on Waste Management." *Industrial Crops and Products* 23 (2): 147–61. https://doi.org/10.1016/j.indcrop.2005.05.004.

Derraik, José G.B. 2002. "The Pollution of the Marine Environment by Plastic Debris: A Review." *Marine Pollution Bulletin* 44 (9): 842–52. https://doi.org/10.1016/S0025-326X(02)00220-5.

Detzel, A, Benedikt Kauertz, and C Derreza-Greeven. 2013. "Study of the Environmental Impacts of Packagings Made of Biodegradable Plastics." *Study for the Federal Environment Agency (Germany) by Umweltbundesamt (Https://Tinyurl. Com/Zt9j2x2)*.

Dhaliwal, Harnoor, Martin Browne, William Flanagan, Lise Laurin, and Melissa Hamilton. 2014. "A Life Cycle Assessment of Packaging Options for Contrast Media Delivery: Comparing Polymer Bottle vs. Glass Bottle." *The International Journal of Life Cycle Assessment* 19 (12): 1965–73. https://doi.org/10.1007/s11367-014-0795-1.

Dormer, Aaron, Donal P Finn, Patrick Ward, and John Cullen. 2013. "Carbon Footprint Analysis in Plastics Manufacturing." *Journal of Cleaner Production* 51 (July): 133–41. https://doi.org/10.1016/j.jclepro.2013.01.014.

Ekvall, Tomas. 2000a. "A Market-Based Approach to Allocation at Open-Loop Recycling." *Resources, Conservation and Recycling* 29 (1–2): 91–109. https://doi.org/10.1016/S0921-3449(99)00057-9.

———. 2000b. "A Market-Based Approach to Allocation at Open-Loop Recycling." *Resources, Conservation and Recycling* 29 (1–2): 91–109. https://doi.org/10.1016/S0921-3449(99)00057-9.

Emadian, S. Mehdi, Turgut T. Onay, and Burak Demirel. 2017. "Biodegradation of Bioplastics in Natural Environments." *Waste Management* 59 (January): 526–36. https://doi.org/10.1016/j.wasman.2016.10.006.

European Committee for Standardization. 2000. "European Standard EU 13432, 'Requirements for Packaging Recoverable through Composting and Biodegradation – Test Scheme and Evaluation Criteria for the Final Acceptance of Packaging.'" EN 13432.

Farmer, Neil, ed. 2013. *Trends in Packaging of Food, Beverages and Other Fast-Moving Consumer Goods (FMCG): Markets, Materials and Technologies*. Woodhead Publishing Series in Food Science, Technology and Nutrition 244. Oxford: Woodhead Publ.

Federal Trade Comission. 2012. "Part VII, 16 CFR Part 260: Guides for the Use of Environmental Marketing Claims; Final Rule." *Federal Register* 77 (197): 62122–32.

Federal Trade Commission. 2012. "Part VII, 16 CFR Part 260: Guides for the Use of Environmental Marketing Claims; Final Rule." *Federal Register* 77 (197): 62122–62132.

Ferreira, S., M. Cabral, Simon De Jaeger, Nuno F. Da Cruz, P. Simões, and R. C. Marques. 2017. "Life Cycle Assessment and Valuation of the Packaging Waste Recycling System in Belgium." *Journal of Material Cycles and Waste Management* 19 (1): 144–54. https://doi.org/10.1007/s10163-015-0383-x.

Fieschi, Maurizio, and Ugo Pretato. 2018. "Role of Compostable Tableware in Food Service and Waste Management. A Life Cycle Assessment Study." *Waste Management* 73 (March): 14–25. https://doi.org/10.1016/j.wasman.2017.11.036.

Flanigan, L, R Frischknecht, and T Montalbo. 2013. "An Analysis of Life Cycle Assessment in Packaging for Food & Beverage Applications." UNEP/SETAC Life Cycle Initiative. http://www.lifecycleinitiative.org/wp-content/uploads/2013/11/food_packaging_11.11.13_web.pdf.

Franklin Associates. 2004. "Life Cycle Inventory of Packaging Options for Shipment of Retail Mail-Order Soft Goods: Final Peer-Reviewed Report." Prairie Village, KS.

———. 2008a. "LCI Summary for Eight Coffee Packaging Systems: Peer Reviewed Final Report." Prairie Village, KS.

———. 2008b. "LCI Summary for Six Tuna Packaging Systems: Peer Reviewed Final Report." Prairie Village, KS.

———. 2009. "Life Cycle Assessment of Drinking Water Systems: Bottle Water, Tap Water, and Home/Office Delivery Water."

———. 2011. "Life Cycle Inventory of Foam Polystyrene, Paper-Based, and PLA Foodserivce Products." Prepared for the Plastic Food Service Packaging Group. Prairie Village, KS. https://www.plasticfoodservicefacts.com/wp-content/uploads/2017/12/Peer_Reviewed_Foodservice_LCA_Study-2011.pdf.

Frees, Niels. 2008. "Crediting Aluminium Recycling in LCA by Demand or by Disposal." *The International Journal of Life Cycle Assessment* 13 (3): 212–18. https://doi.org/10.1065/lca2007.06.348.

Gala, Alba Bala, Marco Raugei, and Pere Fullana-i-Palmer. 2015. "Introducing a New Method for Calculating the Environmental Credits of End-of-Life Material Recovery in Attributional LCA." *The International Journal of Life Cycle Assessment* 20 (5): 645–54. https://doi.org/10.1007/s11367-015-0861-3.

Gerngross, Tillman U., and Steven C. Slater. 2000. "How Green Are Green Plastics?" *Scientific American* 283 (2): 36–41.

Geyer, Roland, Jenna R. Jambeck, and Kara Lavender Law. 2017. "Production, Use, and Fate of All Plastics Ever Made." *Science Advances* 3 (7). https://doi.org/10.1126/sciadv.1700782.

Geyer, Roland, Brandon Kuczenski, Trevor Zink, and Ashley Henderson. 2016. "Common Misconceptions about Recycling: Common Misconceptions about Recycling." *Journal of Industrial Ecology* 20 (5): 1010–17. https://doi.org/10.1111/jiec.12355.

Girgenti, Vincenzo, Cristiana Peano, Claudio Baudino, and Nadia Tecco. 2014. "From 'Farm to Fork' Strawberry System: Current Realities and Potential Innovative Scenarios from Life Cycle Assessment of Non-Renewable Energy Use and Green House Gas Emissions." *Science of The Total Environment* 473–474 (March): 48–53. https://doi.org/10.1016/j.scitotenv.2013.11.133.

Groot, Wim J., and Tobias Borén. 2010. "Life Cycle Assessment of the Manufacture of Lactide and PLA Biopolymers from Sugarcane in Thailand." *The International Journal of Life Cycle Assessment* 15 (9): 970–84. https://doi.org/10.1007/s11367-010-0225-y.

Günkaya, Zerrin, and Müfide Banar. 2016. "An Environmental Comparison of Biocomposite Film Based on Orange Peel-Derived Pectin Jelly-Corn Starch and LDPE Film: LCA and Biodegradability." *The International Journal of Life Cycle Assessment* 21 (4): 465–75. https://doi.org/10.1007/s11367-016-1042-8.

Häkkinen, Tarja, and Sirje Vares. 2010. "Environmental Impacts of Disposable Cups with Special Focus on the Effect of Material Choices and End-of-life." *Journal of Cleaner Production* 18 (14): 1458–63. https://doi.org/10.1016/j.jclepro.2010.05.005.

Haneef, Muhammad, Luca Ceseracciu, Claudio Canale, Ilker S. Bayer, José A. Heredia-Guerrero, and Athanassia Athanassiou. 2017. "Advanced Materials From Fungal Mycelium: Fabrication and Tuning of Physical Properties." *Scientific Reports* 7 (January): 41292. https://doi.org/10.1038/srep41292.

Hanssen, Ole Jørgen. 1998. "Environmental Impacts of Product Systems in a Life Cycle Perspective." *Journal of Cleaner Production* 6 (3–4): 299–311. https://doi.org/10.1016/S0959-6526(98)00031-6.

Hermann, Barbara G., Kornelis Blok, and Martin K. Patel. 2010. "Twisting Biomaterials around Your Little Finger: Environmental Impacts of Bio-Based Wrappings." *The International Journal of Life Cycle Assessment* 15 (4): 346–58. https://doi.org/10.1007/s11367-010-0155-8.

Hermann, Barbara G., L. Debeer, B. De Wilde, K. Blok, and M.K. Patel. 2011. "To Compost or Not to Compost: Carbon and Energy Footprints of Biodegradable Materials' Waste Treatment." *Polymer Degradation and Stability* 96 (6): 1159–71. https://doi.org/10.1016/j.polymdegradstab.2010.12.026.

Hiederer, Roland, European Commission, Joint Research Centre, and Institute for Environment and Sustainability. 2011. *International Reference Life Cycle Data System (ILCD) Handbook: General Guide for Life Cycle Assessment: Provisions and Action Steps.* Luxembourg: Publications Office.

Hottle, Troy A., Melissa M. Bilec, Nicholas R. Brown, and Amy E. Landis. 2015. "Toward Zero Waste: Composting and Recycling for Sustainable Venue Based Events." *Waste Management* 38 (April): 86–94. https://doi.org/10.1016/j.wasman.2015.01.019.

Hottle, Troy A., Melissa M. Bilec, and Amy E. Landis. 2013. "Sustainability Assessments of Bio-Based Polymers." *Polymer Degradation and Stability* 98 (9): 1898–1907. https://doi.org/10.1016/j.polymdegradstab.2013.06.016.

———. 2017. "Biopolymer Production and End-of-life Comparisons Using Life Cycle Assessment." *Resources, Conservation and Recycling* 122 (July): 295–306. https://doi.org/10.1016/j.resconrec.2017.03.002.

Humbert, Sebastien, Vincent Rossi, Manuele Margni, Olivier Jolliet, and Yves Loerincik. 2009. "Life Cycle Assessment of Two Baby Food Packaging Alternatives: Glass Jars vs. Plastic Pots." *The International Journal of Life Cycle Assessment* 14 (2): 95–106. https://doi.org/10.1007/s11367-008-0052-6.

IPCC. 2014. *Climate Change 2014: Synthesis Report*. Edited by Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. Geneva, Switzerland: Intergovernmental Panel on Climate Change.

ISO. 2006. "ISO 14040:2006 Environmental Management -- Life Cycle Assessment -- Principles and Framework."

Jambeck, Jenna R., Roland Geyer, Chris Wilcox, Theodore R. Siegler, Miriam Perryman, Anthony Andrady, Ramani Narayan, and Kara Lavender Law. 2015. "Plastic Waste Inputs from Land into the Ocean." *Science* 347 (6223): 768. https://doi.org/10.1126/science.1260352.

Jishi, Sarah, Corinne Krupp, and Sean Rowe. 2013. "Compostable Utensil Evaluation and Feasibility Report for Humboldt State University." Humboldt State University. http://www2.humboldt.edu/sustainability/sites/default/ files/envs410_121213-3.pdf.

Johnson, Jeremiah X., Colin A. McMillan, and Gregory A. Keoleian. 2013. "Evaluation of Life Cycle Assessment Recycling Allocation Methods." *Journal of Industrial Ecology*, September, n/a-n/a. https://doi.org/10.1111/jiec.12050.

Kendall, Alissa. 2012. "A Life Cycle Assessment of Biopolymer Production from Material Recovery Facility Residuals." *Resources, Conservation and Recycling* 61 (Supplement C): 69–74. https://doi.org/10.1016/j.resconrec.2012.01.008.

Kijchavengkul, Thitisilp, and Rafael Auras. 2008. "Compostability of Polymers." *Polymer International* 57 (6): 793–804. https://doi.org/10.1002/pi.2420.

Kim, Seungdo, and Bruce Dale. 2005. "Life Cycle Assessment Study of Biopolymers (Polyhydroxyalkanoates) - Derived from No-Tilled Corn (11 Pp)." *The International Journal of Life Cycle Assessment* 10 (3): 200–210. https://doi.org/10.1065/lca2004.08.171.

Koller, Martin, Daniel Sandholzer, Anna Salerno, Gerhart Braunegg, and Michael Narodoslawsky. 2013. "Biopolymer from Industrial Residues: Life Cycle Assessment of Poly (Hydroxyalkanoates) from Whey." *Resources, Conservation and Recycling* 73: 64–71.

Krüger, Martina, Benedikt Kauertz, and Andreas Detzel. 2009. "Life Cycle Assessment of Food Packaging Made of IngeoTM Biopolymer and (r)PET: Addendum to the LCA Study on Food Packaging Made of NatureWorks® Biopolymer and Alternative Materials [2006]." Heidelberg GmbH: Institut für Energieund Umweltforschung (IFEU).

Krystofik, Mark, Callie W. Babbitt, and Gabrielle Gaustad. 2014. "When Consumer Behavior Dictates Life Cycle Performance beyond the Use Phase: Case Study of Inkjet Cartridge End-of-Life Management." *The International Journal of Life Cycle Assessment* 19 (5): 1129–45. https://doi.org/10.1007/s11367-014-0713-6.

Kuczenski, Brandon, and Roland Geyer. 2013. "PET Bottle Reverse Logistics—environmental Performance of California's CRV Program." *The International Journal of Life Cycle Assessment* 18 (2): 456–71. https://doi.org/10.1007/s11367-012-0495-7.

Kurdikar, Devdatt, Laurence Fournet, Steven C. Slater, Mark Paster, Kenneth J. Gruys, Tillman U. Gerngross, and Remi Coulon. 2000. "Greenhouse Gas Profile of a Plastic Material Derived from a Genetically Modified Plant." *Journal of Industrial Ecology* 4 (3): 107–22. https://doi.org/10.1162/108819800300106410.

Lebreton, L., B. Slat, F. Ferrari, B. Sainte-Rose, J. Aitken, R. Marthouse, S. Hajbane, et al. 2018. "Evidence That the Great Pacific Garbage Patch Is Rapidly Accumulating Plastic." *Scientific Reports* 8 (1). https://doi.org/10.1038/s41598-018-22939-w.

Leejarkpai, Thanawadee, Thumrongrut Mungcharoen, and Unchalee Suwanmanee. 2016. "Comparative Assessment of Global Warming Impact and Eco-Efficiency of PS (Polystyrene), PET (Polyethylene Terephthalate) and PLA (Polylactic Acid) Boxes."

Journal of Cleaner Production 125 (July): 95–107. https://doi.org/10.1016/j.jclepro.2016.03.029.

Lin, Zhaojia, Vladimiros Nikolakis, and Marianthi Ierapetritou. 2015. “Life Cycle Assessment of Biobased *p* -Xylene Production.” *Industrial & Engineering Chemistry Research* 54 (8): 2366–78. https://doi.org/10.1021/ie5037287.

Liptow, Christin, and Anne-Marie Tillman. 2012. “A Comparative Life Cycle Assessment Study of Polyethylene Based on Sugarcane and Crude Oil.” *Journal of Industrial Ecology* 16 (3): 420–435.

Madival, Santosh, Rafael Auras, Sher Paul Singh, and Ramani Narayan. 2009. “Assessment of the Environmental Profile of PLA, PET and PS Clamshell Containers Using LCA Methodology.” *Journal of Cleaner Production* 17 (13): 1183–94. https://doi.org/10.1016/j.jclepro.2009.03.015.

Marion, Stephanie. 2005. “Environmental Costs and Environmental Benefits Analysis of Packaging Waste Recovery and Recycling Targets.” Ph.D. Thesis, United Kingdom: University of Sheffield. http://etheses.whiterose.ac.uk/3576/.

Markwardt, Stefanie, and Frank Wellenreuther. 2016. “Sensitivity Analysis as a Tool to Extend the Applicability of LCA Findings.” *The International Journal of Life Cycle Assessment* 21 (8): 1148–58. https://doi.org/10.1007/s11367-016-1094-9.

Mattila, Tuomas, Marjukka Kujanpää, Helena Dahlbo, Risto Soukka, and Tuuli Myllymaa. 2011. “Uncertainty and Sensitivity in the Carbon Footprint of Shopping Bags.” *Journal of Industrial Ecology* 15 (2): 217–27. https://doi.org/10.1111/j.1530-9290.2010.00326.x.

Meeks, Diana, Troy Hottle, M.M. Bilec, and A.E. Landis. 2015. “Compostable Biopolymer Use in the Real World: Stakeholder Interviews to Better Understand the Motivations and Realities of Use and Disposal in the US.” *Resources, Conservation and Recycling* 105 (December): 134–42. https://doi.org/10.1016/j.resconrec.2015.10.022.

Meylan, Grégoire, Helen Ami, and Andy Spoerri. 2014. “Transitions of Municipal Solid Waste Management. Part II: Hybrid Life Cycle Assessment of Swiss Glass-Packaging Disposal.” *Resources, Conservation and Recycling* 86 (May): 16–27. https://doi.org/10.1016/j.resconrec.2014.01.005.

Meylan, Grégoire, Roman Seidl, and Andy Spoerri. 2013. “Transitions of Municipal Solid Waste Management. Part I: Scenarios of Swiss Waste Glass-Packaging Disposal.” *Resources, Conservation and Recycling* 74 (May): 8–19. https://doi.org/10.1016/j.resconrec.2013.02.011.

Miller, Shelie A., Amy E. Landis, Thomas L. Theis, and Ronald A. Reich. 2007. “A Comparative Life Cycle Assessment of Petroleum and Soybean-Based Lubricants.” *Environmental Science & Technology* 41 (11): 4143–4149.

Morris, Jeffrey, H. Scott Matthews, and Clarissa Morawski. 2013. “Review and Meta-Analysis of 82 Studies on End-of-Life Management Methods for Source Separated Organics.” *Waste Management* 33 (3): 545–51. https://doi.org/10.1016/j.wasman.2012.08.004.

Mourad, Anna Lúcia, Eloisa EC Garcia, Gustavo Braz Vilela, and Fernando von Zuben. 2008. “Environmental Effects from a Recycling Rate Increase of Cardboard of Aseptic Packaging System for Milk Using Life Cycle Approach.” *The International Journal of Life Cycle Assessment* 13 (2): 140.

Muñoz, Ivan, Joan Rieradevall, Xavier Doménech, and Llorenç Milà. 2004. “LCA Application to Integrated Waste Management Planning in Gipuzkoa (Spain).” *The International Journal of Life Cycle Assessment* 9 (4): 272–80. https://doi.org/10.1007/BF02978603.

Nicholson, Anna L., Elsa A. Olivetti, Jeremy R. Gregory, Frank R. Field, and Randolph E. Kirchain. 2009. “End-of-Life LCA Allocation Methods: Open Loop Recycling Impacts on Robustness of Material Selection Decisions.” In *2009 IEEE International Symposium on Sustainable Systems and Technology*, 1–6. IEEE. https://doi.org/10.1109/ISSST.2009.5156769.

Niero, Monia, and Stig Irving Olsen. 2016. “Circular Economy: To Be or Not to Be in a Closed Product Loop? A Life Cycle Assessment of Aluminium Cans with Inclusion of Alloying Elements.” *Resources, Conservation and Recycling* 114 (November): 18–31. https://doi.org/10.1016/j.resconrec.2016.06.023.

NREL. 2015. “US Life Cycle Inventory Database.” U.S. Department of Energy National Renewable Energy Laboratory. 2015. https://uslci.lcacommons.gov/uslci/search.

Oliveira, Maria, and Alessandra Magrini. 2017. “Life Cycle Assessment of Lubricant Oil Plastic Containers in Brazil.” *Sustainability* 9 (4): 576. https://doi.org/10.3390/su9040576.

Oregon DEQ. 2012. “Materials Management in Oregon: 2050 Vision and Framework for Action.” 12-LQ-036. http://www.oregon.gov/deq/FilterDocs/MManagementOR.pdf.

———. 2017. “Materials Management: Packaging.” 2017. http://www.oregon.gov/deq/mm/Pages/Packaging.aspx.

Papong, Seksan, Pomthong Malakul, Ruethai Trungkavashirakun, Pechda Wenunun, Tassaneewan Chom-in, Manit Nithitanakul, and Ed Sarobol. 2014. “Comparative Assessment of the Environmental Profile of PLA and PET Drinking Water Bottles from a Life Cycle Perspective.” *Journal of Cleaner Production* 65 (February): 539–50. https://doi.org/10.1016/j.jclepro.2013.09.030.

Pasqualino, Jorgelina, Montse Meneses, and Francesc Castells. 2011. “The Carbon Footprint and Energy Consumption of Beverage Packaging Selection and Disposal.” *Journal of Food Engineering* 103 (4): 357–65. https://doi.org/10.1016/j.jfoodeng.2010.11.005.

PE Americas. 2009. “Comparative Life Cycle Assessment Ingeo™ Biopolymer, PET, and PP Drinking Cups.” Boston, MA: Prepared for Starbucks Coffee Company and NatureWorks LLC. https://www.natureworksllc.com/~/media/Files/NatureWorks/What-is-Ingeo/Why-it-Matters/LCA/PEA_Cup_Lid_LCA_FullReport_ReviewStatement_121209_pdf.pdf.

Pladerer, Christian, Markus Meissner, Fredy Dinkel, Mischa Zschokke, Günter Dehoust, and Schüler, Doris. 2008. “Comparative Life Cycle Assessment of Various Cup Systems for the Selling of Drinks at Events.” Prepared for Austrian Federal Ministry of Agriculture and Forestry, Environment and Water Management; Swiss Federeal Environment Authority. http://www.meucopoeco.com.br/environmental_study.pdf.

Platt, Brenda, Nora Goldstein, Craig Coker, and Sally Brown. 2014. “State of State of Composting in the U.S: What, Why, Where, & How.” Institute for Local Self Reliance. http://ilsr.org/wp-content/uploads/2014/07/state-of-composting-in-us.pdf.

Potting, José, and Eugenie van der Harst. 2015. “Facility Arrangements and the Environmental Performance of Disposable and Reusable Cups.” *The International Journal of Life Cycle Assessment* 20 (8): 1143–54. https://doi.org/10.1007/s11367-015-0914-7.

PricewaterhouseCoopers. 2008. “Evaluation of the Environmental Impacts of Cork Stoppers versus Aluminium and Plastic Closures.” Prepared for Corticeira Amorim, SGPS, SA. https://www.amorimcork.com/media/cms_page_media/228/Amorim_LCA_Final_Report.pdf.

Pro.mo/Unionplast. 2015. “Comparative Life Cycle Assessment Study of Tableware for Alimentary Use.” Milan, Italy. http://www.pro-mo.it/data/uploads/report-lca-eng.pdf.

Quantis. 2011. "Comparative Full Life Cycle Assessment of B2C Cup of Espresso Made Using a Packaging and Distribution System from Nespresso Espresso and Three Generic Products." Prepared by Quantis Suisse for Nestlé Nespresso.

Razza, Francesco, Francesco Degli Innocenti, Antonio Dobon, Cesar Aliaga, Carmen Sanchez, and Mercedes Hortal. 2015. "Environmental Profile of a Bio-Based and Biodegradable Foamed Packaging Prototype in Comparison with the Current Benchmark." *Journal of Cleaner Production* 102 (September): 493–500. https://doi.org/10.1016/j.jclepro.2015.04.033.

Razza, Francesco, Maurizio Fieschi, Francesco Degli Innocenti, and Catia Bastioli. 2009. "Compostable Cutlery and Waste Management: An LCA Approach." *Waste Management* 29 (4): 1424–33. https://doi.org/10.1016/j.wasman.2008.08.021.

Resource Recycling Systems. 2016. "2015 - 2016 Centralized Study on Availability of Recycling." Sustainable Packaging Coalition.

Rochman, Chelsea M. 2015. "The Complex Mixture, Fate and Toxicity of Chemicals Associated with Plastic Debris in the Marine Environment." In *Marine Anthropogenic Litter*, edited by Melanie Bergmann, Lars Gutow, and Michael Klages, 117–40. Cham: Springer International Publishing. https://doi.org/10.1007/978-3-319-16510-3_5.

Rossi, Vincent, Nina Cleeve-Edwards, Lars Lundquist, Urs Schenker, Carole Dubois, Sebastien Humbert, and Olivier Jolliet. 2015. "Life Cycle Assessment of End-of-Life Options for Two Biodegradable Packaging Materials: Sound Application of the European Waste Hierarchy." *Journal of Cleaner Production* 86 (January): 132–45. https://doi.org/10.1016/j.jclepro.2014.08.049.

Schiffman, Richard. 2013. "Packing Materials Grown from Mushrooms." *New Scientist* 218 (2921): 29. https://doi.org/10.1016/S0262-4079(13)61485-2.

Schmidt, Christian, Tobias Krauth, and Stephan Wagner. 2017. "Export of Plastic Debris by Rivers into the Sea." *Environmental Science & Technology* 51 (21): 12246–53. https://doi.org/10.1021/acs.est.7b02368.

Schott, Anna, Henrik Wenzel, and Jes la Cour Jansen. 2016. "Identification of Decisive Factors for Greenhouse Gas Emissions in Comparative Life Cycle Assessments of Food Waste Management – an Analytical Review." *Journal of Cleaner Production* 119 (April): 13–24. https://doi.org/10.1016/j.jclepro.2016.01.079.

Shen, Li, Ernst Worrell, and Martin K Patel. 2012. "Comparing Life Cycle Energy and GHG Emissions of Bio-Based PET, Recycled PET, PLA, and Man-Made Cellulosics." *Biofuels, Bioproducts and Biorefining* 6 (6): 625–39. https://doi.org/10.1002/bbb.1368.

Suwanmanee, Unchalee, Viganda Varabuntoonvit, Phasawat Chaiwutthinan, Monchai Tajan, Thumrongrut Mungcharoen, and Thanawadee Leejarkpai. 2013. "Life Cycle Assessment of Single Use Thermoform Boxes Made from Polystyrene (PS), Polylactic Acid,(PLA), and PLA/Starch: Cradle to Consumer Gate." *The International Journal of Life Cycle Assessment* 18 (2): 401–417.

Tabone, Michaelangelo D., James J. Cregg, Eric J. Beckman, and Amy E. Landis. 2010. "Sustainability Metrics: Life Cycle Assessment and Green Design in Polymers." *Environmental Science & Technology* 44 (21): 8264–69. https://doi.org/10.1021/es101640n.

The Economist. 2018. "The Known Unknowns of Plastic Pollution." *March 3, 2018*, March.

Thompson, R. C., S. H. Swan, C. J. Moore, and F. S. vom Saal. 2009. “Our Plastic Age.” *Philosophical Transactions of the Royal Society B: Biological Sciences* 364 (1526): 1973–76. https://doi.org/10.1098/rstb.2009.0054.

Toniolo, Sara, Anna Mazzi, Monia Niero, Filippo Zuliani, and Antonio Scipioni. 2013. “Comparative LCA to Evaluate How Much Recycling Is Environmentally Favourable for Food Packaging.” *Resources, Conservation and Recycling* 77 (August): 61–68. https://doi.org/10.1016/j.resconrec.2013.06.003.

Toniolo, Sara, Anna Mazzi, Chiara Pieretto, and Antonio Scipioni. 2017. “Allocation Strategies in Comparative Life Cycle Assessment for Recycling: Considerations from Case Studies.” *Resources, Conservation and Recycling* 117 (February): 249–61. https://doi.org/10.1016/j.resconrec.2016.10.011.

Tsiropoulos, I., A.P.C. Faaij, L. Lundquist, U. Schenker, J.F. Briois, and M.K. Patel. 2015. “Life Cycle Impact Assessment of Bio-Based Plastics from Sugarcane Ethanol.” *Journal of Cleaner Production* 90 (March): 114–27. https://doi.org/10.1016/j.jclepro.2014.11.071.

U.S. EPA. 2015. “U.S. EPA Sustainable Materials Management Program Strategic Plan.” https://www.epa.gov/sites/production/files/2016-03/documents/smm_strategic_plan_october_2015.pdf.

———. 2017. “Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2015.” 430-P-17–001. U.S. EPA. https://www.epa.gov/sites/production/files/2017-02/documents/2017_complete_report.pdf.

U.S. EPA Office of Resource Conservation and Recovery. 2016. “Advancing Sustainable Materials Management: Facts and Figures 2014.” EPA530-R-17–01. U.S. EPA Office of Resource Conservation and Recovery. https://www.epa.gov/sites/production/files/2015-09/documents/2013_advncng_smm_fs.pdf.

U.S. EPA Office of Solid Waste and Emergency Response. 2009. “Opportunities to Reduce Greenhouse Gas Emissions through Materials and Land Management Practices.” 530-R-09–017. U.S. EPA Office of Solid Waste and Emergency Response. https://www.epa.gov/sites/production/files/documents/ghg-land-materials-management.pdf.

U.S. Food and Drug Administration. 2018. *Submissions on Post-Consumer Recycled (PCR) Plastics for Food-Contact Articles Database*. https://www.accessdata.fda.gov/scripts/fdcc/?set=RecycledPlastics.

USDA. 2018. “BioPreferred|Site Map.” 2018. https://www.biopreferred.gov/BioPreferred/faces/pages/AboutBioPreferred.xhtml.

USEtox International Center. 2018. “Frequently Asked Questions - How to Use USEtox Characterization Factors | USEtox®.” 2018. http://www.usetox.org/faq-page/23-0#t23n78.

Velis, Costas. 2014. “Global Recycling Markets - Plastic Waste: A Story for One Player – China.” Report Prepared by FUELogy and Formatted by D-Waste on Behalf of International Solid Waste Association - Globalisation and Waste Management Task Force. Vienna: International Solid Waste Association. https://www.iswa.org/fileadmin/galleries/Task_Forces/TFGWM_Report_GRM_Plastic_China_LR.pdf.

Vercalsteren, An, Carolin Spirinckx, Theo Geerken, and Vito Claeys. 2006. “Comparative LCA of 4 Types of Drinking Cups Used at Events.” Commissioned by Public Waste Agency for the Flemish Region (OVAM): Flemish Institute for Technological Research (VITO).

https://www.natureworksllc.com/~/media/Files/NatureWorks/What-is-Ingeo/Why-it-Matters/LCA/OVAM_Cup_ComparativeLCA_FullReport_0206_pdf.pdf.

Weidema, Bo. 2000. "Avoiding Co-Product Allocation in Life-Cycle Assessment." *Journal of Industrial Ecology* 4 (3): 11–33. https://doi.org/10.1162/108819800300106366.

Weiss, Martin, Juliane Haufe, Michael Carus, Miguel Brandão, Stefan Bringezu, Barbara Hermann, and Martin K. Patel. 2012. "A Review of the Environmental Impacts of Biobased Materials." *Journal of Industrial Ecology* 16 (April): S169–81. https://doi.org/10.1111/j.1530-9290.2012.00468.x.

Wong, C. S., David R. Green, and Walter J. Cretney. 1974. "Quantitative Tar and Plastic Waste Distributions in the Pacific Ocean." *Nature* 247 (5435): 30–32. https://doi.org/10.1038/247030a0.

World Steel Association. 2011. "Life Cycle Assessment Methodology Report." Brussels, Belgium: World Steel Association. https://www.worldsteel.org/en/dam/jcr:6a222ba2-e35a-4126-83ab-5ae5a79e6e46/LCA+Methodology+Report.pdf.

Xie, Minghui, Qi Qiao, Qihong Sun, and Linlin Zhang. 2013. "Life Cycle Assessment of Composite Packaging Waste Management—a Chinese Case Study on Aseptic Packaging." *The International Journal of Life Cycle Assessment* 18 (3): 626–35. https://doi.org/10.1007/s11367-012-0516-6.

Yates, Madeleine R., and Claire Y. Barlow. 2013. "Life Cycle Assessments of Biodegradable, Commercial Biopolymers—A Critical Review." *Resources, Conservation and Recycling* 78 (September): 54–66. https://doi.org/10.1016/j.resconrec.2013.06.010.

Zampori, Luca, and Giovanni Dotelli. 2014. "Design of a Sustainable Packaging in the Food Sector by Applying LCA." *The International Journal of Life Cycle Assessment* 19 (1): 206–17. https://doi.org/10.1007/s11367-013-0618-9.

Zhang, Yimin, Garvin Heath, Alberta Carpenter, and Noah Fisher. 2016. "Air Pollutant Emissions Inventory of Large-Scale Production of Selected Biofuels Feedstocks in 2022: Air Pollutant Emissions Inventory of Large-Scale Production of Selected Biofuels Feedstocks in 2022." *Biofuels, Bioproducts and Biorefining* 10 (1): 56–69. https://doi.org/10.1002/bbb.1620.

Zink, Trevor, Roland Geyer, and Richard Startz. 2016. "A Market-Based Framework for Quantifying Displaced Production from Recycling or Reuse: Market-Based Framework for Quantifying Displaced Production." *Journal of Industrial Ecology* 20 (4): 719–29. https://doi.org/10.1111/jiec.12317.